Autores (p. o. de firma): Eugenia Gimeno-García*, Vicente Andreu and José Luis Rubio

Título: Influence of vegetation recovery on water erosion at short and medium-term after experimental fires in a Mediterranean shrubland

Revista: Catena

Volumen: 69

Página inicial: 150 **Página final:** 160

Fecha: 2007

**Influence of vegetation recovery on water erosion at short and medium-term after
experimental fires in a Mediterranean shrubland**

Eugenia Gimeno-García*, Vicente Andreu and José Luis Rubio

Centro de Investigaciones sobre Desertificación-CIDE (CSIC; UV, GV)

Camí de la Marjal, s/n. 46470-Albal, Valencia. Spain

Abstract

This paper reports the influence that vegetation recovery has exerted on the soil behaviour to erosion by water during both the first and eight years after experimental fires. The work was carried out at La Concordia Experimental Station (Valencia, Spain), which includes nine plots (4 m wide x 20 m long) installed on a calcareous hillside representative of Mediterranean shrubland areas. In June 1995 a set of experimental fires were carry out at two intensity levels (high and moderate) with three plots replication for each treatment. The remaining three plots were used as the control. Rain events between June 1995 to June 1996, and from June 2002 to June 2003 were monitored and its effect on soil erosion processes determined. The vegetation changes (biomass amount and plant cover) for each studied period were also assessed.

Total runoff and sediment yield measured during the first post-fire year was 19.43 L m⁻² and 561 g m⁻² in the intense fire, and 14.72 L m⁻² and 326 g m⁻² in the moderate one, which contrast with the very low runoff (3.82 L m⁻²) and soil loss (8.56 g m⁻²) in control plots.

*Correspondence to: E. Gimeno-García. Fax: +34(9)6 127 09 67. E-mail:

eugenia.gimeno@uv.es

Eight years after fire, the amount of vegetation on the burned plots represents between 63 and 69% compared to the biomass present before the fire in 1995. The regeneration of plant cover, up to 30-40% eight years after fire, facilitated a decrease in the difference of soil losses between fire treatments and between burned and unburned plots. However, runoff generation still remains greater in burned plots than in the control ones eight years after the fire.

Key words: fire intensity, runoff, soil loss, erosion plots, Mediterranean shrubland

1. Introduction

In the last two decades forest fires in European Mediterranean countries (Portugal, Spain, France, Italy and Greece) have increased in number and frequency (European Communities, 2002). The consequential damage, which may result in high erosion rates after them, shows the need to study the short and medium term soil and vegetation response to this impact in natural semiarid Mediterranean areas prone to wildfires.

Forest fires not only reduce plant cover, leaving soil surfaces unprotected against raindrop impact, but also have important consequences for soil hydrology and soil losses. There are reports about the increase in runoff and consequent erosion (Rubio *et al.*, 1997; Inbar *et al.*, 1998; Moody and Martin, 2001), about decrease in the water infiltration capacity (Cerdá, 1998; Martin and Moody, 2001) and about increase in soil hydrophobicity (DeBano, 2000; Robichaud and Hungerford, 2000), which can affect several hydrologic processes like raindrop splash, rill formation and total watershed

responses (DeBano, 2000). The effects of fire on water erosion and the subsequent soil response depend on fire behaviour, mainly related to fire intensity (that is, maximum temperature on the soil surface and their duration) (Whelan, 1997), as well as to the characteristics of any subsequent rain event (Rubio *et al.*, 1996). Complete vegetation removal may also lead to irreversible soil degradation in semi-arid areas (Castillo *et al.*, 1997).

Vegetation cover has long been recognized as a key factor in runoff production and protection against erosion, as vegetation increases infiltration and surface roughness, and reduces the kinetic impact of raindrops (Morgan, 1995). Plant cover is the most important vegetation parameter for splash and interrill erosion, whereas for rill and ephemeral gully erosion plant roots are at least as important as vegetation cover (Gyssels *et al.*, 2005). Plant recoveries after fire and vegetation resilience are key factors in the reduction of soil erosion in Mediterranean ecosystems. This depends on plant regeneration strategies (seeder or resprouter) (Keely and Zedler, 1978), on the fire intensity and fire recurrence (Whelan 1997), on the capacity of vegetation in using post-fire nutrient pools (Romanyà *et al.*, 2001) and also on the climatic conditions, specially the timing of intense rainfall events (De Luis *et al.*, 2001).

Increases in runoff and soil loss after shrub fires have been measured in different Mediterranean type shrublands (Sánchez *et al.*, 1994; Cerdá *et al.*, 1995; Rubio *et al.*, 1997; Cerdá, 1998; Gimeno-García *et al.*, 2000; Andreu *et al.*, 2001), but generally, they are focused in the few months or years after fires. There are few studies assessing the erosional response during the whole recovery period after fire (Moody and Martin, 2001; Cerdá and Lasanta, 2005). The present paper compares the soil response to erosion processes by water at plot scale (20m long by 4 m wide) during the first year

(short-term) after experimental fires of different intensities, and eight years later (medium-term), related to the effect of vegetation recovery on erosion control in a Mediterranean shrubland.

2. Materials and methods

2.1. Study area

La Concordia Experimental Station (latitude 39°45' N and longitude 0°43' W) is located in the municipality of Liria (Valencia, Spain), 50 km NW of Valencia city on land ceded by the Forestry Services of the Valencia Government (Generalitat Valenciana). The Experimental Station is located at 575 m a.s.l. on a South-South East facing hillslope on a 30% slope. The soil is a Rendzic Leptosol (FAO-UNESCO, 1988) developed on Jurassic limestone, with variable depth always less than 40 cm, sandy-loam texture, alkaline pH (7.4), and mean organic matter content of 9.8% (Table 1). The dominant vegetation type in the area is the *Rhamno lycioidis-Quercetum cocciferae* association, which is typical of semi-arid Mediterranean areas. Climatically the area belongs to the dry ombroclimate of the lower mesomediterranean belt (Rivas-Martínez, 1981). Mean monthly temperatures range from 13.3°C in January to 25.8°C in August. The mean annual precipitation is 379 mm with two maxima, autumn and spring (182 and 99 mm, respectively), and a dry period from June to September (71 mm of rain). Data recorded by the meteorological station placed at La Concordia also show the high inter-annual variability in the Mediterranean climate, especially in rainfall volume and

intensity. From 1995 to 2004, the years with lowest volume of rainfall were 1998 and 1999, with only 204 and 238 mm, respectively. Those values contrast with the 556 mm fallen in 2002. Maximum 30 minutes rainfall intensity (I_{30}) for that period ranges from 8.6 in 2002 to 40.4 mm h⁻¹ in 1999.

La Concordia mountain area is a forestry land that has undergone a long history of land uses. The Forest Management Plan of La Concordia (Ministerio de Agricultura, 1961) reflects the previous uses of this land since 15th century, which include intensive grazing and esparto harvesting from 1438 to 1961, pinewood cutting (*Pinus halepensis*) and shrubs exploitation to supply ovens and pottery industry and to enhance grasses development for grazing. From this document it could be inferred that the above mentioned land uses prevailed until the decade 1950s. Although no references have been found about wildfire and fire frequency in the past centuries, it is well documented that the last wildfire at La Concordia (including the area where plots are settled) take place in September 1978, affecting 1058 hectares mainly covered by *Pinus halepensis*. It was also well-known that at the Experimental Station place there was a pine forest between 67 and 87 years old, with a tree height ranged from 6.6 and 10.4 m and a diameter between 24 and 54 cm. The reported shrub floristic composition by the Forest Management Plan of La Concordia (1961) included *Sitpa tenacísima*, *Brachypodium ramosum*, *Rosmarinus officinalis*, *Globularia alypum*, *Ulex parviflorus*, *Rhamnus lycioides*, *Satureja montana*, *Fumana ericoides*, *Quercus coccifera*, *Juniperus oxycedrus*, *Erica multiflora*, *Cistus albidus*, *Thymus vulgaris*, *Tymelaea hirsuta*, which is quite similar to the described in 1995 when the experimental plots were established. Pines did not regenerate after the fire in 1978 at La Concordia plots and the standing vegetation at the beginning of this study is 17 years old.

2.2. Erosion plots

The experimental set-up consists of nine plots (20 m long by 4 m wide) with similar slope gradient, rock outcrops, soil type and vegetation cover. The plots are oriented parallel to the slope and bounded by bricks. At the foot of each plot a 2-m wide collector runs into a 1500 L tank to collect all the runoff and sediment produced during each rainfall event. Inside this tank a 30 L tank facilitates the collection of runoff and sediments produced by small rainfall events.

The thermopluviometric features of La Concordia Experimental Station are registered by an automatic meteorological station placed inside its limits. The equipment provides digitized values at 5-min intervals. These data are used to determine the duration, total volume and intensity of rainfall events. Rainfall intensity is measured as I_{30} , the maximum amount of rain during a 30-min period and expressed as millimetres per hour.

The study period covers the first year after the experimental fires (from 20 June 1995 to 19 June 1996) and eight years later (from 20 June 2002 to 19 June 2003). Runoff and sediment produced were collected and measured for all plots after each rain event. Those events that show at least runoff production were considered as the erosive rainfalls. The erosion parameters: runoff yield (L m^{-2}), runoff coefficient (%), sediment yield (g m^{-2}), and sediment concentration (g L^{-1}), were studied. The runoff coefficient is calculated as the ratio of runoff yield/rainfall volume, expressed as percentage.

2.3. Natural biomass quantification

Biomass amount and percentage of plant cover were measured and estimated, respectively, in each plot. Composition and spatial distribution of vegetation in each plot were determined by identifying the species, counting individual plants and measuring their size (height, maximum and minimum canopy diameter in cm), as well as the percentage of soil covered by plants on a 1m x 1m grid basis. This information was used to map dry biomass (Gimeno-García *et al.*, 2004) and plant cover percentage and to calculate the mean dry biomass present in the plots. These observations and measurements were made before experimental fires in 1995 and eight years after them in 2003. In the first post-fire year only visual estimation of the plant cover percentage was made.

Dry biomass amount was determined by using a non-destructive methodology similar to those proposed by Etienne (1989). Different algorithms and equations based on this methodology, which related dimensions and dry weight for the dominant species, were used to calculate the dry weight of each species, where the independent variable was the weight and the dependent variable was the volume. A known geometrical form was assigned to each species based on visual observations of their architecture (for example, a cylinder for *Ulex parviflorus*). Each individual inside the plots was measured in height and canopy diameter. To quantify the dry weight, several individuals of the dominant species (8 individuals of *Rosmarinus officinalis*, 8 individuals of *Ulex parviflorus*, 3 individuals of *Stipa tenacissima*, 2 individuals of *Quercus coccifera*) were selected in the surrounding area and their height and diameters measured. Afterwards, they were cut and the samples were carried to the laboratory, where biomass were weighted and placed in an oven 48 h at 70°C and, finally volumes

and dry weights were linearly regressed. Moreover, under the three species that cover the highest percentage of soil surface (*Ulex parviflorus*, *Rosmarinus officinalis* and *Quercus coccifera*), 9 litter samples on a grid of 25 cm x 25 cm were collected, weighted and placed in an oven 48 h at 70°C and the dry biomass was also directly measured.

2.4. Fire treatments

A design of two different fire treatments, with three plots each, was used. Fire treatments were based on the addition of contrasted amounts of biomass with the aim to obtain different fire intensities. One consisted in the addition of 4 kg biomass m⁻² (H treatment). The second consisted in the addition of 2 kg biomass m⁻² (M treatment). These amounts were established based on laboratory experiences on heat capacity of different shrubs species, as well as on the studies by Paipó and Trabaud (1990 and 1991). The extra biomass, which was obtained from the surrounding shrub vegetation, was spread uniformly on the plots just before fire. The remaining three plots were used as control (C treatment, no fire). The assignation of fire treatment to each plot was made completely at random without blocking.

To measure the temperatures on the soil surface and their duration, thermosensitive paints and thermocouples were used. Six thermocouples (type K Inconel 600-insulated) per plot were installed at ground level along parallel lines running downslope and separated from one another by 3 m. Each thermocouple was connected to a Unidata data logger, recording temperatures every 5 seconds. From these measurements direct estimates were made of the duration that temperature exceeded the threshold value of

100°C. This value was selected because beyond this temperature changes in soil properties can occur. The second system used to measure soil temperatures was a set of twenty-four thermosensitive paints (Omega Stick Crayons), ranging between 100°C and 677°C. They were applied on iron rods each covered with another identical rod, but not painted, to protect them from ashes and flames. Just before the experimental fire one iron rod per square metre was placed (a total of 80 iron rods per plot) with the painted side in contact with soil. Immediately after the passage of fire the iron rods were collected and read.

Experimental fires were carried out under field conditions on 20 and 21 June 1995. When the thermocouples and the thermosensitive paints were in place and the extra biomass was spread on the plots, a small amount of fuel oil was applied at the bottom of the plots and fire started. The fires progressed upslope and their patterns were uniform in all the plots, except in plot 6 that suffered repeated changes in wind direction.

The duration for temperatures greater than 100°C was 36.3 minutes and 17.6 minutes in the H and M treatments, respectively. Assuming that the temperature measured with one iron rod corresponds to 1 m² of the plot surface, we found that on the H plots more than 50% of the surface had temperatures between 400°C and 600°C (mean value 439°C), whereas on the M plots, 50% of their surface had temperatures between 200°C and 400°C (mean value 232°C). The post-fire appearances of charred vegetation, litter consumption and ash colour (the darker coloured the lower intensity), are other indicators of the fire intensity. Field observations showed great vegetation and litter consumption in both fire treatments, but ash colour was clearly different, being white coloured for H treatment and dark grey for M treatment. If the mean values of soil temperature are taken into account, as well as the duration of temperatures greater than

100°C, results show the more biomass the higher temperatures in soil surface and their longer duration above 100°C. Mean temperature values and temperature duration beyond 100°C for H plots are twice times greater than for M plots. Taking into account these results and the fire intensity classes established in the literature (Ulrey and Graham, 1993, DeBano *et al.*, 1998; Robichaud *et al.*, 2000), it could be assumed that H plots suffered high fire intensity and the M plots suffered a fire of moderate intensity. The complete description of the experimental fires and the main results of soil surface temperatures are given in Gimeno-García *et al.* (2004).

2.5. Statistical analysis

Vegetation data (biomass amount and percentage of soil cover) and erosion data (runoff yield, runoff coefficient, sediment yield and concentration of sediment) were analysed by analysis of variance (ANOVA) to test significant differences between fire intensity treatments and the two studied periods as the main effects. When significant differences were detected among means, the minimum significant difference for fire intensity treatments were calculated using Tukey's test ($p < 0.05$). A single regression analysis was used to examine the influence of rainfall volume and intensity, as well as the plant cover percentage on runoff and sediment yield at the first post-fire year and eight years later. Correlations between the erosion data and rainfall parameters were also tested.

3. Results

3.1. Vegetation recovery after fire

In 1995, the measured amount of biomass in the plots, before experimental fires, was between 0.5 and 0.8 kg m⁻² and the percentage of plant cover varied from 19 and 29% (Figure 1). The most abundant species at that time were *Rosmarinus officinalis*, *Ulex parviflorus* and *Globularia alypum*, which represented 37, 15 and 27% of the number of the plants, respectively.

The biomass was completely consumed by the flames in both fire treatments and the soil surface was covered with ashes and charred material. This layer remained partially on the soil surface until autumn of 1995, when it was progressively removed by heavy rainfalls and the intermittent effect of wind. Qualitative observations, in the first post-fire year, showed that germination of herbaceous species took place in the early spring. In our case, the plant cover in spring 1996 was mainly dominated by a resprouter and native perennial grass (*Brachypodium retusum*), in accordance to the observations of other authors (Sanroque *et al.*, 1985; Abad *et al.*, 2000). Smaller numbers of other regenerating species such as *Globularia alypum*, *Rhamnus lycioides*, *Quercus coccifera* and *Stipa tenacissima*, were also found. The mean percentage of plant cover at the first post-fire year was 12% for the intense fire treatment and 9% for the moderate one.

The regenerated biomass amount in 2003 was 0.5 kg m⁻² for the intense fire treatment, and 0.4 kg m⁻² for the moderate one. This means that biomass in the intense and moderate fire treatments were 69% and 63% of that in 1995, respectively. The regenerated biomass amount of that eight-year-old shrubland was significantly lower than that of similar burned shrubland ecosystems with three, nine and twelve years old

(1.2, 3.8 and 4.4 kg m⁻², respectively) reported by Baeza *et al.* (2002). On the other hand, the percentage of soil cover eight years after fire, increases by 16% for the intense fire plots and 42.5% for the moderate ones compared to the status in 1995 (Figure 1B). Similar values of soil cover are also reported by Cerdá *et al.* (1995), in a similar ecosystem, ten years after a fire in the South of the Valencia province (Spain).

The natural development of vegetation on the plots not affected by fire, showed a notable increase in biomass during the eight years period, from 0.45 kg m⁻² in 1995 to 0.90 kg m⁻² in 2003 (Figure 1A). This amount of biomass on the control plots in 2003 was significantly different to the values attended for both fire treatments in the same year. Moreover, the percentage of vegetation cover on the control plots also showed a notable increase during the eight years of study, from 26% in 1995 to a 45% in 2003 (Figure 1B).

3.2. Rainfall variables influencing water erosion responses

The two studied periods (1995-1996 and 2002-2003), showed different precipitation patterns, not only in the number of erosive rainfall events, but also in the rainfall volume, duration and intensity. In the two periods, 36 erosive rainfall events were registered; 24 in the first post-fire year and 12 in the period from June 2002 to June 2003 (Table 2). The first period after fire was characterised by most aggressive rains, with maximum I₃₀ of 35.4 mm h⁻¹, and by shorter duration (482 minutes on average) and lower volume (321 mm) than in the second studied period (Table 2).

Runoff yield showed a significant and positive relationship ($p < 0.01$) with rainfall volume and I₃₀ throughout the first post-fire period for all fire treatments (Table 3).

However, sediment yield from burned plots was only significantly correlated ($p < 0.01$) with the I_{30} for the intense and moderate fire treatments during this period. The duration of the rainfall events did not show significant correlation neither with runoff nor sediment yield. Eight years after fire, no significant relationship between rainfall variables (volume, I_{30} and duration) and sediment yield was observed. Only runoff yield from burned plots was significantly and positively correlated with I_{30} in that period (Table 3). The relationships between I_{30} and runoff yield, and I_{30} and sediment yield for the three different fire treatments during the first post-fire year (1995-1996) showed a generally wide scatter, but significant linear relationships ($p < 0.05$) were found (Figure 2).

During the first post-fire year, high levels of runoff and sediment yield were generally given when rainfall intensity was greater than 20 mm h^{-1} (Figure 3). These highly erosive rainfall events, which were concentrated in autumn and spring, produced mean runoff yield of 3.15 L m^{-2} and mean soil losses of 134 g m^{-2} for the intense fire treatment. However, the two rainfall events occurred immediately after the fire impact, in despite of I_{30} values of 20.8 and 14.5 mm h^{-1} , produced very low runoff and sediment yields in burned plots (between 0.1 and 0.35 L m^{-2} , 1 and 11 g m^{-2} respectively) (Figure 3). Those low erosion rates may be attributable to the protective effect of the ash layer deposited on soil surface after fire that plays an important role in absorbing the rain drop impact, lowering the runoff production, and controlling the detachment and transport of sediment. Three months after fire, important soil losses were produced when one autumn rainfall event occurs with I_{30} of 35 mm h^{-1} (Figure 3). That single event produced mean values of soil losses of 186.7 and 157 g m^{-2} in intense and moderate fire treatments, respectively, which contrast to the 0.98 g m^{-2} measured in the

control plots. Eight years after fire, although erosive rainfall events with $I_{30} > 20 \text{ mm h}^{-1}$ are given, the values for the erosive parameters in burned plots are significantly lower than those observed in the first post-fire year (runoff yield of 1.5 L m^{-2} and sediment yield 0.2 g m^{-2}) (Figure 3).

3.3. Fire treatments and vegetation recovery influencing water erosion responses

Runoff yield, runoff coefficient, sediment yield and sediment concentration show that in the first year after fire, the high intensity treatment resulted in larger values than the moderate one, and both of them show higher values than those measured in control plots. Statistical significant differences between burned and unburned plots (Table 4) were also observed during that period.

Total runoff yield in the first post-fire year was $19.4 \text{ L m}^{-2} \text{ yr}^{-1}$ for the intense fire, $14.7 \text{ L m}^{-2} \text{ yr}^{-1}$ for the moderate fire and $3.8 \text{ L m}^{-2} \text{ yr}^{-1}$ for control plots. Total soil loss in that period was almost twice in the high intensity treatments than in the moderate one (561 g m^{-2} and 326 g m^{-2} , respectively), and both values were clearly higher than the sediment yield measured in control plots (8.5 g m^{-2}).

Eight years after fire, the biomass amount has been regenerated close to 66% of the pre-fire status, and the soil covered by plants was between 30 and 40%. In this conditions it is quantified a decrease in soil loss around 99% compared with 1995-1996 period, in all fire treatments (Table 4). The reduction in the concentration of sediment in runoff from 1995 to 2003 is more than 98% in all burned plots (Table 4). The development of the vegetation cover is also reflected in the 50% decrease of mean runoff yield from 1995 to 2003 in the intense fire treatment and 30% in the moderate

one (Table 4), but in this case, there are no statistical differences between both studied periods. Runoff coefficient in the high intensity fire treatment also shows significant differences between the two studied periods. The increment of biomass in control plots, from 0.47 to 0.94 kg m⁻² (Figure 1A), is also reflected in the decrease of the runoff coefficient by 70% (Table 4).

The relationships between plant cover with runoff yield and plant cover with soil loss, for the first post-fire year and eight years later were negative and exponential (Figure 4). Results showed that, eight years after fire, there is a strong effect of plant cover on reducing runoff when its percentage is close to 30%. Sediment yield showed the same trend, and an exponential decrease in soil loss occurred above 20% of plant cover, while with lower covers, soil loss showed an exponential increase in the first post-fire year (Figure 4). Considering the biomass amount regenerated during the eight years period after fire, results also showed an exponential decrease for runoff and sediment yield (Figure 4) when plots had more than 0.6 kg m⁻² of biomass.

4. Discussion

Several studies, on similar conditions, indicate that the period of strong erosion in burned slopes is relatively short and occurred primarily in the first years after fire (Robichaud and Waldrop, 1994; DeBano, 2000; Moody and Martin, 2001). In Mediterranean ecosystems, where wildfires occur mainly in summer and torrential rainfall events are frequent in autumn, the critical period is 4-6 months after fire, when the highest soil susceptibility to water erosion processes is manifested (Soler *et al.*, 1994; Andreu *et al.*, 2001). This general pattern depends upon the sensitivity of the

system to erosion, on the fire intensity and on the intrinsic characteristics and distribution of the rainfall events (Rubio and Calvo, 1996).

In our case the hydrologic and erosional soil response after experimental fires is related to fire intensity during the first post-fire year. This fact agrees with the findings of Vega *et al.* (2005) for the annual runoff and soil losses. Nevertheless, in our experimental layout, the similar values for runoff and sediment yields in the first section of the Figures 3B and 3C, in the intense and moderate fire treatments, led us to think that in a short period of time after fire (3 months) erosion processes are strongly intensified but the intrinsic characteristics of the erosive rainfall events have more influence on the erosional soil response than the intensity of fire, mainly due to the high I_{30} values accounted in that period (35 mm h^{-1}). It is after that autumn rainy period when the differences in runoff and soil loss between both fire treatments are more significant. De Luis *et al.* (2003) also found a good relationship between fire intensity and soil losses when torrential rains occur after fires.

Maximum soil losses were measured in the first year after fire, when I_{30} values were higher than 20 mm h^{-1} . This parameter showed the best correlation coefficient with runoff production and soil loss. In both fire treatments, five rainfalls, all with $I_{30} \geq 20 \text{ mm h}^{-1}$, produced 96% of the soil losses during the first post-fire year. These same rainfalls caused 66 to 68% of the runoff measured in that period. Martin and Moody (2001) found soil loss acceleration in burned areas with values of I_{30} higher than 10 mm h^{-1} . Castillo *et al.* (1997), in plots without vegetation in semiarid areas, also measured the 80% of soil losses when $I_{30} > 20 \text{ mm h}^{-1}$.

The erosion rates measured during the first post-fire year at La Concordia Experimental Station ($5610 \text{ kg ha}^{-1} \text{ yr}^{-1}$ for the intense fire treatment and 3260 kg ha^{-1}

yr⁻¹ for the moderate one) were higher than those measured in other Mediterranean environments. Abad *et al.* (2000) found maximum soil loss of 2000 kg ha⁻¹ yr⁻¹. Andreu *et al* (2001) measured soil losses of 235 kg ha⁻¹ and 475 kg ha⁻¹ in burned soils with north and south aspects, respectively, during a three year post-fire period, and Cerdá and Lasanta (2005) reported values between 83 and 1041 kg ha⁻¹ yr⁻¹.

The period of time for soil to recover the pre-fire conditions is highly variable owing to the many factors influencing the processes. Some authors indicate a period of 3-4 years (Morgan, 1995) and that period is in between 7 and 12 years to the establishment of a vegetative cover similar to the previously existing one in areas under Mediterranean climate (Naveh, 1975; Trabaud and Lepart, 1980). In Mediterranean areas of Israel, Inbar *et al.* (1998) have suggested a period of 5-10 years to return to the background levels of sediment yield, meanwhile Martin and Moody (2001), mentioned a period between 3 and 4 years for burned areas in Colorado. Cerdá and Lasanta (2005), during an eight years post-fire period, measured the higher runoff, solute and suspended sediment concentrations, and erosion rates during the first two years following fires.

Those authors measured a fast vegetation recovery in the first year after fire, with more than 50% of plant cover. In our case, the herbaceous and shrub recovery in the first post-fire year, which offered a plant cover between 9 and 12%, is not enough to reduce post-fire erosion. Eight years after fires, it has been produced a clear reduction in sediment yield, as well as in the concentration of sediments in runoff water, in both fire treatments, showing similar behaviour than the control plots (Figure 3). However, total runoff yield eight years after fire is still 4 times greater than that measured in control plots, which their shrub vegetation regenerated after the last wildfire in the area is 25 years old.

It has been suggested that below 30% vegetation cover, plants may not be able to give an effective soil protection, and as a result, the soil-plant system may gradually be degraded (Thornes, 1988; De Luis *et al.*, 2001). As Figure 4 shows, the negative and exponential relationships between plant cover with runoff yield and plant cover with soil loss, for the first post-fire year and eight years later, is in accordance with several models used to investigate the competitive behaviour between vegetation cover and soil erosion (Elwell and Stoking, 1976; Thornes, 1988), and also with the observations of other authors in semi-arid Mediterranean environments (Francis and Thornes, 1990; Vega *et al.*, 2005). In our case, the regenerated vegetation cover is between 30-40% eight years after the fire impact and the biomass amount is between 0.4 and 0.5 kg m⁻², which is a 35% lower than the biomass presented before fire in 1995, and its effectiveness in reducing runoff production and soil loss is reflected by the results obtained (Figure 4).

5. Conclusions

In the first year after the experimental fires, the intense fire treatment resulted in greater runoff yield, runoff coefficient, sediment yield and sediment concentration than the moderate one, although the differences between them are not statistically significant. In this year, average runoff and sediment yield in burned plots are 75% and 98% greater than the values measured in the control ones. Burned plots show statistical significant differences for all the erosive parameters considered respect to the control ones. Eight

years after fire, in 2003, these differences disappear, but still remain clear differences between burned and control plots for the runoff coefficient.

Biomass amount in 2003 was still 31% and 37% lower than in 1995 before fires, for treatments of high and moderate fire intensity, respectively. The regenerated vegetation during the eight years after fire offered a percentage of cover between 30-40% in both fire treatments. On the other hand, in the unburned plots there has been measured an increment of biomass from 0.45 kg m^{-2} to 0.90 kg m^{-2} in that period.

The effectiveness of these vegetation recoveries on erosion control in the eight years period is reflected in the reduction of runoff and sediment yield in all fire treatments. Total runoff yield measured in the high and moderate fire intensity treatments during the first post-fire year were 4 and 3 times greater than the measured for the same fire treatments eight years after fire. Total soil loss was 289 times greater in the intense fire during the first post-fire year than eight years later, and in the moderate fire intensity treatment, soil loss was 150 times greater than the measured in the last studied period.

In the immediate post-fire period, soil is highly susceptible to water erosion, especially when this period coincides with the torrential rainy season. During the first year after fire, soil erosion responds both to the amount of rainfall and to rainfall intensities (I_{30}), however, the dominant variable for sediment yield appears to be I_{30} rather than rainfall amount alone. Maximum soil losses in that first post-fire year occur when I_{30} values were higher than 20 mm h^{-1} . Eight years after fire, the regenerated shrub cover (between 30-40%) have a much greater effect in reducing both runoff and sediment yield than the highly erosive rain events in producing soil erosion. This fact is of great importance in considering the appropriate management strategies against water erosion process in burned Mediterranean slopes.

Acknowledgements

This study was partially supported by funding from Programa Ramón y Cajal from the Spanish Ministerio de Educación y Ciencia. We also thank the EU project “Eco-engineering and conservation of slopes for long term protection from erosion, landslides and storms (ECO-SLOPES)”, QLRT-2000-00289 (LIFE/992/0676) and to the Generalitat Valenciana (GV-CSIC 01807) for the financial support. We wish to thank Julián Campo, Oscar González and Eva López for their help with the field vegetation inventories in the 2003 campaign.

References

- Abad, N., Bautista, S., Blade, C., Caturla, R.N. 2000 Seeding and mulching as erosion control techniques after wildfires in the Valencia region. In: P. Balabanis, D. Peter, A. Ghazi, M. Tsogas (Editors), Mediterranean Desertification Research Results and Policy Implications, Vol 2., Directorate-General Research, European Commission, Brussels, pp. 419-429.
- Andreu, V., Imeson, A.C., Rubio, J.L. 2001. Temporal changes in soil aggregates and water erosion after a wildfire in a Mediterranean pine forest. *Catena*, 44: 69-84.
- Baeza, M.J., De Luís, M., Raventós, J., Escarré, A. 2002. Factors influencing fire behaviour in shrublands of different stand ages and the implications for using

479 prescribed burning to reduce wildfire risk. *Journal of Environmental Management*,
480 65:199-208.

481 Castillo, V.M., Martínez-Mena, M., Albaladejo, J. 1997. Runoff and soil loss response
482 to vegetation removal in a semiarid environment. *Soil Science Society of American*
483 *Journal*, 61: 1116-1121.

484 Cerdá, A. 1998. Changes in overland flow and infiltration after a rangeland fire in a
485 Mediterranean shrubland. *Hydrological Processes*, 12: 1031-1042.

486 Cerdá, A., Lasanta, T. 2005. Long-term erosional responses after fire in the Central
487 Spanish Pyrenees 1. Water and sediment yield. *Catena*, 60: 59-80.

488 Cerdá, A., Imeson, A.C., Calvo, A. 1995. Fire and aspect induced differences on the
489 erodibility and hydrology of soils at La Costera, Valencia, Southeast Spain. *Catena*,
490 24: 289-304.

491 DeBano, L.F. 2000. The role of fire and soil heating on water repellency in wildland
492 environments: a review. *Journal of Hydrology*, 231-232: 195-206.

493 DeBano, L.F., Neary, D.G., Ffolliott, P.F. 1998. Fire's effects on ecosystems. John
494 Wiley & Sons, New York.

495 De Luís, M., García-Cano, M.F., Cortina, J., Raventós, J., González-Hidalgo, J.C.,
496 Sánchez, J.R. 2001. Climatic trends, disturbances and short-term vegetation
497 dynamics in a Mediterranean shrubland. *Forest Ecology and Management*, 147: 25-
498 37.

499 De Luís, M., González-Hidalgo, J.C., Raventós, J. 2003. Effects of fire and torrential
500 rainfall on erosion in a Mediterranean gorse community. *Land Degradation &*
501 *Development* 14: 203-213.

502 Elwell, C.F., Stocking, M.A. 1976. Vegetal cover to estimate soil erosion hazard in
 503 Rhodesia. *Geoderma*, 15: 61-70.

504 Etienne, M. 1989. Non destructive methods for evaluating shrub biomass: a review. *Acta*
 505 *Oecologica-Oecologia Applicata*, 10: 115-128.

506 European Communities. 2002. Forest fires in Europe – 2001 fire campaign. Report No
 507 2. S.P.I. 02.72 EN. European Communities, Italy.

508 FAO-UNESCO. 1988. Soil Map of the World. Revised Legend 1:5.000.000. FAO,
 509 Roma.

510 Francis, C.F., Thornes, J.B. 1990. Runoff hydrographs from three Mediterranean
 511 vegetation cover types. In: Thornes, J.B. (Ed.), *Vegetation and erosion. Processes*
 512 *and environments*. John Wiley & Sons Ltd, Chichester, pp. 363-384.

513 Gimeno-García, E., Andreu, V., Rubio, J.L. 2000. Changes in organic matter, nitrogen,
 514 phosphorous and cations in soil as a result of fire and water erosion in a
 515 Mediterranean landscape. *European Journal of Soil Science*, 51: 201-210.

516 Gimeno-García, E., Andreu, V., Rubio, J.L. 2004. Spatial patterns of soil temperatures
 517 during experimental fires. *Geoderma*, 118: 17-38.

518 Gyssels, G., Poesen, J., Bochet, E., Li, Y. 2005. Impact of plant roots on the resistance
 519 of soils to erosion by water: a review. *Progress in Physical Geography*, 29: 198-217.

520 Inbar, M., Tamir, M., Wittenberg, L. 1998. Runoff and erosion processes after a forest
 521 fire in Mount Carmel, a Mediterranean area. *Geomorphology*, 24: 17-33.

522 Keely, J.E., Zedler, P.J. 1978. Reproduction of chaparral shrubs after fire: a comparison
 523 of sprouting and seeding strategies. *American Midland Naturalist*, 99: 142-161.

524 Martin, D.A., Moody, J.A. 2001. Comparison of soil infiltration rates in burned and
 525 unburned mountainous watersheds. *Hydrological Processes*, 15: 2893-29003.

526 Misterio de Agricultura. Dirección General de Montes, Caza y Pesca Fluvial.
 527 Subdirección del Patrimonio Forestal del Estado. 1961. Proyecto de ordenación del
 528 Monte denominado La Concordia nº 87 del Catálogo de los de utilidad pública
 529 perteneciente al propio de Liria. Technical report.
 530 Moody, J.A., Martin, D.A. 2001. Initial hydrologic and geomorphic response following
 531 a wildfire in the Colorado Front Range. *Earth Surface Processes and Landforms*, 26:
 532 1049-1070.
 533 Morgan, R.P.C. 1995. Soil erosion and conservation. Longman Group Limited, London.
 534 Naveh, Z. 1975. The evolutionary sequence of fire in the Mediterranean region.
 535 *Vegetation*, 9: 199-206.
 536 Paipó, C., Trabaud, L. 1990. Structural characteristics of combustible components of
 537 five Mediterranean shrubs. *Forest Ecology and Management*, 35: 249-259.
 538 Paipó, C., Trabaud, L. 1991. Comparative study of the aerial structure of five shrubs of
 539 Mediterranean shrublands. *Forest Science*, 37: 146-159.
 540 Rivas-Martínez, S. 1981. Etages bioclimatiques, secteurs chronologiques et séries de
 541 végétation de l'Espagne Méditerranée. *Ecología Mediterránea*, 8: 275-288.
 542 Robichaud, P.R., Waldrop, T.A. 1994. A comparison of surface runoff and sediment
 543 yields from low-and-high severity site preparation burns. *Water Resources Bulletin*,
 544 30: 27-34.
 545 Robichaud, P.R., Hungerford, R.D. 2000. Water repellency by laboratory burning of
 546 four northern Rocky Mountain forest soils. *Journal of Hydrology*. 231-232: 207-
 547 219.
 548 Robichaud, P.R., Beyers, J.L., Neary, D.G. 2000. Evaluating the effectiveness of
 549 postfire rehabilitation treatments. USDA Forest Service. General Technical Report

550 RMRS-GTR-63. Fort Collins, U.S. Department of Agriculture, Forest Service,
 551 Rocky Mountain Research Station.

552 Romanyà, J., Casal, P., Vallejo, V.R. 2001. Short-term effects of fire on soil nitrogen
 553 availability in Mediterranean grasslands and shrublands growing in old fields. *Forest
 554 Ecology and Management*, 147: 39-53.

555 Rubio, J.L., Calvo, A. 1996. Mechanisms and processes of soil erosion by water in
 556 Mediterranean Spain. In: Rubio, J.L., Calvo, A. (Editors), *Soil degradation and
 557 desertification in Mediterranean environments*. Geoforma Ediciones, Logroño, pp.
 558 37-48.

559 Rubio, J.L., Forteza, J., Andreu, V., Cerni, R. 1996. Effects of forest fires on runoff and
 560 soil erosion. In: Plá Sentís, I., López Falcón, R., Lobo Luján, D. (Editors), *Soil
 561 Erosion Processes on Steep Lands: Evaluation and Modelling*. CIDIAT, Mérida, pp.
 562 41-53.

563 Rubio, J.L., Forteza, J., Andreu, V., Cerni, R. 1997. Soil profile characteristics
 564 influencing runoff and soil erosion after forest fire: a case study (Valencia, Spain).
 565 *Soil Technology*, 11: 67-78.

566 Sánchez, J.R., Mangas, V.J., Ortiz, C., Bellot, J. 1994. Forest fire effect on soil
 567 chemical properties and runoff. In: Sala, M., Rubio, J.L. (Editors), *Soil erosion and
 568 degradation as consequence of forest fires*. Geoforma Ediciones, Logroño, pp. 53-
 569 65.

570 Sanroque, P., Rubio, J.L., Mansanet, J. 1985. Efectos de los incendios forestales en las
 571 propiedades del suelo, en la composición florística y en la erosión hídrica de las
 572 zonas forestales de Valencia (España). *Revue d'Écologie et Biologie du Sol*, 22:
 573 131-147.

574 Soler, M., Sala, M., Gallart, F. 1994. Post fire evolution on runoff and erosion during an
 575 eighteen month period. In: Sala, M., Rubio, J.L. (Editors), Soil Erosion as a
 576 Consequence of Forest Fires. Geoforma Ediciones, Logroño, pp. 149-161.
 577 Thornes, J.B. 1988. Erosional equilibria under grazing. In: Bintliff, J., Davidson, D.,
 578 Grant, E. (Editors). Conceptual Issues in Environmental Archaeology. University
 579 Press, Edinburgh, pp. 193-210.
 580 Trabaud, L., Lepart, J. 1980. Diversity and stability in garrigue ecosystems after fire.
 581 Vegetation, 43: 49-57.
 582 Ulrey, A.L., Graham, R.C. 1993. Forest fire effects on soil color and texture. Soil
 583 Science Society of American Journal, 57: 135-140.
 584 Vega, J.A., Fernández, C., Fonturbel, T. 2005. Throughfall, runoff and soil erosion alter
 585 prescribed burning in gorse shrubland in Galicia (NW Spain). Land Degradation &
 586 Development, 16: 37-51.
 587 Whelan, R.J. 1997. The ecology of fire. Cambridge University Press, Cambridge.
 588

589 Table 1. Some soil physical and chemical properties of La Concordia soil

590

	Horizons		
	Ah1	Ah2	Ck
Depth (cm)	0-12	12-30	30-40
% Sand (2-0.05 mm)	60.84	-	-
% Silt (0.05-0.002 mm)	27.88	-	-
% Clay (< 0.002 mm)	7.52	-	-
Texture	Sandy loam	-	-
Water retention at field capacity (%)	30.83	28.54	29.55
Water stability of aggregates (%)	32.95	39.70	-
Particle density (g cm ⁻³)	1.87	-	-
Organic matter (%)	9.81	6.22	4.72
pH	7.17	7.30	7.21
Electric conductivity (dS.m ⁻¹)	0.71	0.59	0.99
Total carbonate (%)	43.01	56.72	69.89
Total Nitrogen (%)	0.41	0.29	0.24
Mineral Nitrogen (mg kg ⁻¹)	15.67	9.63	17.66
Available Phosphorus (mg kg ⁻¹)	3.50	3.30	3.30
CEC (cmol _c . kg ⁻¹)	29.46	29.02	27.28

591

592

593 Table 2. Characteristics of the erosive rainfall events during each post-fire period considered at La Concordia Experimental Station

594

	One year after fire June 1995 - June 1996	Eight years after fire June 2002 - June 2003
Annual total volume of rainfall (mm)	385.44	446.40
Number of erosive rainfall events	24	12
Annual total volume of erosive rainfall (mm)	321	390.4
Range of variation of erosive rainfall volume (mm)	2.6-33.54	5.7-67.10
Annual mean I_{30} (mm h ⁻¹)	10.37	6.18
Range of variation of I_{30} (mm h ⁻¹)	1.4-35.4	1.6-21.2
Lowest I_{30} to produce runoff (mm h ⁻¹)	1.4-2.6	1.6-2.6
Lowest I_{30} to produce sediment (mm h ⁻¹)	1.6-2.6	3.8
Annual mean duration of rainfall (minutes)	482	1515
Range of variation of rainfall duration (minutes)	30-1850	185-5450

595

Table 3. Correlation matrix between rainfall volume, rainfall intensity (I_{30}), rainfall duration, runoff yield and sediment yield for each post-fire period and for each fire intensity treatment

	Rainfall volume	I_{30}	Duration
1995-1996 (n=24)			
Runoff yield			
High	0.632 **	0.811 **	-0.177 (ns)
Moderate	0.597 **	0.821 **	-0.188 (ns)
Control	0.881 **	0.570 **	0.030 (ns)
Sediment yield			
High	0.321 (ns)	0.772 **	-0.254 (ns)
Moderate	0.326 (ns)	0.846 **	-0.287 (ns)
Control	0.553 **	0.737 **	-0.321 (ns)
2002-2003 (n=12)			
Runoff yield			
High	0.473 (ns)	0.745 **	0.062 (ns)
Moderate	0.492 (ns)	0.742 **	0.001 (ns)
Control	0.601 *	0.501 (ns)	0.195 (ns)
Sediment yield			
High	0.182 (ns)	0.292 (ns)	-0.288 (ns)
Moderate	0.194 (ns)	0.262 (ns)	-0.291 (ns)
Control	0.055 (ns)	0.140 (ns)	-0.246 (ns)

** Significant correlation at level 0.01; * significant correlation at level 0.05; ns: not significant

Table 4. Mean values of the erosive parameters for the two studied periods at La Concordia Experimental Station

Parameter ^a	Fire treatment	June 1995-June1996 ^b	June 2002-June 2003 ^b	Differences ^c
RY (L m ⁻²)	High	0.81 a (0.11) ^d	0.41 a (0.14)	
	Moderate	0.61 ab (0.11)	0.45 a (0.14)	
	Control	0.16 b (0.11)	0.11 a (0.14)	
RC (%)	High	4.29 a (0.58)	1.17 a (0.41)	*
	Moderate	3.26 a (0.58)	1.28 a (0.41)	
	Control	0.93 b (0.58)	0.30 b (0.41)	*
SY (g m ⁻²)	High	23.39 a (5.03)	0.16 a (0.08)	*
	Moderate	13.59 a (5.03)	0.18 a (0.08)	*
	Control	0.36 b (5.03)	0.0004 a (0.08)	*
CON (g L ⁻¹)	High	9.67 a (2.06)	0.16 a (0.08)	*
	Moderate	8.73 a (2.06)	0.18 a (0.08)	*
	Control	2.09 b (2.06)	0.0007 a (0.08)	*

^a RY: Runoff yield; RC: Runoff coefficient; SY: Sediment yield; CON: Concentration of sediments in runoff

^b Means not sharing the same letter in a column show there are statistical significant differences between fire treatments (p<0.05)

^c The * indicates there are statistical significant differences between the two studied periods (p<0.05)

^d Standard error in brackets

Figure captions

Figure 1. (A) Estimated amount of biomass (kg m^{-2}) and (B) percentage of soil covered by plants for each fire intensity treatment in the two studied years. Different letters show significant differences ($p < 0.05$) in amount of biomass between control and burned treatments in 2003. Vertical bars represent standard error

Figure 2. Statistical relationships between (A) runoff yield in burned and control treatments and rainfall intensity (I_{30}) in the first post-fire year (June 1995- June 1996); (B) sediment yield in burned and control treatments and rainfall intensity (I_{30}) in the first post-fire year (June 1995- June 1996)

Figure 3. (A) Rainfall volume and rainfall intensity of the erosive rain events for both studied periods; (B) accumulated values of runoff yield; (C) accumulated values of sediment yield and (D) accumulated values of concentration of sediments for each fire intensity treatment during the first year after experimental fires 1995-1996 and during 2002-2003

Figure 4. Statistical relationships between percentage of plant cover in both fire intensity and control treatments and total runoff yield (A) and total sediment yield (B) for the whole data set; (C) statistical relationship between biomass amount in both fire intensity and control treatments and runoff yield and sediment yield eight years after fire. H1, M1 and C1 stand for high, moderate and control fire treatments, respectively in the first post-fire year; H8, M8 and C8 stand for high, moderate and control fire treatments, respectively, eight years after the experimental fires. RY and SY stand for runoff and sediment yield, respectively.

637
638

639

(A)

(B)

(A)

(B)

(C)