Effects of 30-year-old Aleppo pine plantations on runoff, soil erosion, and plant diversity in a semi-arid landscape in south eastern Spain

E. Chirino a,*, A. Bonet b, J. Bellot b, J.R. Sánchez b

* Fundación Centro de Estudios Ambientales del Mediterráneo (CEAM), Spain
b Department of Ecology, University of Alicante, Apdo. 99, 03080, Alicante, Spain

Received 9 March 2004; revised 31 August 2005; accepted 13 September 2005

Abstract

Forest management policies in Mediterranean areas have traditionally encouraged land cover changes, with the establishment of tree cover (Aleppo pine) in natural or degraded ecosystems for soil conservation purposes: to reduce soil erosion and to increase the vegetation structure. In order to evaluate the usefulness of these management policies on reduced erosion in semi-arid landscapes, we compared 5 vegetation cover types (bare soil, dry grassland, shrublands, afforested dry grasslands and afforested thorn shrublands), monitored in 15 hydrological plots (8 x 2 m²), in the Ventos catchment (Alicante, SE Spain), over 4 hydrological years (1995-1996-1998-1999). Each cover type represented a different dominant patch of the vegetation mosaic on the north-facing slopes of this catchment. The results showed that runoff coefficients of vegetated plots were less than 1% of the precipitation volume; whereas runoff in denuded areas was nearly 4%. Soil losses in vegetation plots averaged 0.04 Mg ha⁻¹ year⁻¹ and increased 40-fold in open-land plots. The evaluation of these forest management policies, in contrast with the natural vegetation communities, suggests that: (1) thorn shrublands and dry grassland communities with vegetation cover could control runoff and sediment yield as effectively as Aleppo pine afforestation in these communities, and (2) afforestation with a pine stratum improved the stand's vertical structure resulting in pluri-stratified communities, but reduced the species richness and plant diversity in the understory of the plantations.

Keywords: Runoff; Erosion rate; Afforestation; Semi-arid; Soil erosion

1. Introduction

During the 1950s and 1960s, Aleppo pine (Pinus halepensis Miller) plantation over low cover formations was the most widespread forest administration management policy in most areas of Spain (Valle and Bocio, 1996; Vallejo, 1997; Cortina et al., 2004). The medium and long-term purpose of these afforestation policies was to recover the vegetation cover for soil conservation and to reduce the effects of soil erosion. With these aims, about 2.5 million hectares in Spain, have been afforested over the last 50 years; that is approximately 5% of the national territory (Vilagrosa et al., 1997). The II National Forest Inventory (IN) of the Ministry of the Environment in Spain (1998) reported that Aleppo pine forest had increased the woodland surface over all in Spain by 75%. Aleppo pine forest occupies practically all of the woodland surface of the Spanish semi-arid region. Aleppo pine forest constitutes the dominant forest species in the Alicante province covering more than 90% of the forested area. As a consequence of this forest policy the semi-arid lands have become a patch-mosaic landscape of a mixed formation of grasslands and shrublands with afforested woodlots, occupying the north-facing slopes, where water availability is less restricted. Whereas Alpha grass steppes of Stipa tenacissima L. are widespread in south-facing slopes and the presence of pine trees and other shrub species is less significant (Bonet et al., 2001). Recently, the increase of abandoned land has changed this patch-mosaic as described by Reynolds et al. 2005.
The performance of Aleppo pine forest has been strongly related to water balances and the disturbance regime in Mediterranean conditions (Zavala et al., 2000). Moreover, it has also been suggested that Aleppo pine plantations in extreme water-stressed arid conditions could not develop self-persistent populations, mature and multi-layer communities, unlike the situation in other wetter Mediterranean areas (Rivas-Martínez, 1987; De la Torre, 1988; De la Torre and Alias, 1996; Costa, 1987), in spite of their drought-resistant characteristics in dryer regions (Schiller and Cohen, 1998; Atzmon et al., 2004). This is probably due to slow growth and unsuccessful regeneration dynamics in extreme water-stressed events (Raventós et al., 2001).

The objective of the work presented here is to evaluate afforestation using Aleppo pine as the appropriate recovery strategy in semi-arid regions, in contrast with the natural recovery of these areas, considering the effects on the magnitude of runoff, soil losses, plant cover, species richness and plant diversity. This management strategy is based on the hypotheses that erosion decreases exponentially with vegetation cover (Elwell and Stocking, 1976), despite the work of Francis and Thomas, which indicates that this hypothesis is true but more-or-less independent of cover type (Francis and Thomas, 1990a,b), and that the advantages of Aleppo pine afforestations are only a little above those of natural and anthropogenically degraded shrubland and grassland (Francis and Thomas, 1990b).

From this perspective it was expected that the effects of an Aleppo pine stratum planted 30 years ago, would significantly reduce erosion, in contrast with the areas without active management, where the regeneration of the plant cover only took place slowly through natural vegetation dynamics (Bonet, 2004). To evaluate the previous hypotheses, we analysed five land-cover types that represented the composition of the patch-mosaic landscape in north-facing semi-arid environments (Bonet et al., 2001, 2004). In the study area, north-facing afforestations with Aleppo pine tree had been more successful in extent, growth and survival than in south-facing aspects (Cortina et al., 2004; Bonet et al., 2004). These cover types included a gradient from more degraded (open areas without vegetation) to less degraded (grassland and shrubland) landscape units and the result of afforestation practices. The aim of this paper is to evaluate the effectiveness of Aleppo pine afforestation as a common technique to restore plant cover, facilitate the establishment of other understory species, and reduce the effects of erosion in a semi-arid landscape.

2. Area description

The research was carried out in the Ventós-Agost Catchment Experimental Station (University of Alicante), located in the Municipality of Agost, Alicante Province, SE Spain (38°28'N, 0°37'W). The study area covers approximately 537 ha, with altitudes ranging around 600 m a.s.l., and slopes between 23° and 26°. The climate is semi-arid Mediterranean, with a very high inter-annual variability. Mean annual temperature is 18.2 °C and annual rainfall is of 291.7 mm (mean of the 1961–1999 period), most of which falls in autumn (Pérez Cuevas, 1994). Lithology is loam and loamy limestone supporting calcareous Cambisol (FAO-UNESCO). The soils are characterised as shallow (0–30 cm), with a medium content of organic matter (5.7%), between 48% and 58% of clay content, soil bulk density of 1246.0 kg m⁻³, and a soil particle density of 2352.0 kg m⁻³. The hydraulic performance of the soil is determined by a total porosity of 46.0%, field capacity of 25.0%, a saturated soil hydraulic conductivity of 3.37 cm s⁻¹, and average infiltration rate of 512.6 L m⁻² h⁻¹. Resistance to the penetration of a cone of 0–30 cm of depth (18% humidity) varies between 31 and 97 kPa (Derouiche et al., 1996; Bellet et al., 2001).

The plantation period lasted from 1960 to 1983, but was especially intense during the first decade. The Forest Service afforested the area with P. halepensis Miller woodlots over land degraded grasslands and shrublands, maintaining 30% of the territory managed with no intervention. Afforestation was carried out by planting in holes made by hand, with a mean plantation density of 5280±692 trees ha⁻¹ over grasslands and 3733±1097 trees ha⁻¹ over shrublands. In these areas forest management was restricted to a minimum selective pruning, and no thinning was carried out after the treatments.

The present vegetation landscape in the study area is composed of different land cover types (patch-mosaic) that can be situated in a successional and structural vegetation complexity gradient: degraded open land or bare soil (B), sometimes occupied by microphytic crusts (Maestre et al., 2002); dry grassland formations (G) of Brachypodium reptatum Pers. Beauv., with dwarf shrubs (Anthyllis cytisoides L., Helianthemum syriacum (Jacq.) Dum.-Cours. and Thymus vulgaris L.); and more mature landscape patches composed of scattered thorn and xerophyllum shrublands (S) with Quercus coccifera L., Pistacia lentiscus L., Erica multiflora L., Rhamnus lycioides L. and Rosmarinus officinalis L. (Bonet et al., 2001). Other anthropic components of the present landscape mosaic are afforested dry grasslands (A) and afforested thorn shrublands (AS).

3. Methods and experimental design

The research was conducted over a 4-year period (1996–1999) and 30 years after the plantation work. In each land-cover type (B, G, S, AG and AS) we installed 3 hydrological plots of 16 m² (2×8 m) randomly distributed in each landscape patch. After each rain event, the volumes of runoff and soil loss were collected and measured in the Gerlach box of 2 m width, installed at the bottom of each.
plot. The precipitation and other climatic variables (temperature, radiation, wind, and air humidity) were measured and registered continuously by means of an automatic meteorological station (Campbell CR 110). The analysis of annual runoff coefficient and erosion rates were calculated considering 4 years (1996–1999), and the one-way ANOVA was used for statistical analyses of vegetation type responses during 1998 and 1999. This analysis also was carried out in two groups of rainfall events, those exceeding 10 mm and those below 10 mm.

In order to determine the afforestation effect on vegetation structure, the vegetation survey was performed in two ways to characterise the hydrological plots and the landscape patches, respectively. First, each hydrological plot was divided into eight subplots (2 m²) to determine the cover (%) of each plant species, stoniness, litter cover and bare soil. As a functional indicator of vegetation community structure, we also recorded the leaf area index (LAI), which was estimated by means of a destructive analysis in 0.25 m² plots close to hydrological plots. The second vegetation survey was carried out on landscape patches over a wider scale during the spring of 1998, using 5 stands (as replicates) of 100 m² in each land cover type. On each stand, 10 squares (1 m x 1 m) were placed in a random procedure. A complete species list was recorded and the species cover (%) was estimated in each through the projection of each plant species in the soil horizontal plane by means of a grid (5 x 5 cm unit). Species richness and Shannon's diversity index (H), were calculated on each plot.

Species were grouped in life forms (trees, shrubs, dwarf shrubs, and herbs: perennial grasses, perennial forbs and annuals). The differences between the vegetation characteristics in the hydrological plots and the landscape patches, surveys were analysed by using the Kruskal-Wallis test and the post-hoc Wilcoxon test. The statistical analyses on plant diversity and hydrological variables were carried out by doing a one-way ANOVA on log transformed data. The statistical comparison of the vegetation cover types in the hydrological plots was carried out by Principal Components Analysis—PCA (Statistix, version 3.0). Vegetation composition and cover of hydrological plots was analysed through means of Detrended Correspondences Analysis—DCA (Ter Braak and Prentice, 1988) using CANOCO version 4 (Ter Braak and Smilauer, 1998).

4. Results and discussion

4.1. Analysis of the physical characteristics and vegetation composition of hydrological plots

In order to prove the homogeneity of the characteristics of the different hydrological plots in the cover types analysed, we carried out a Principal Components Analysis (PCA) using several variables that characterised the plots and the accumulated values observed in runoff and soil loss treatments. A correlation matrix between the 12 variables (runoff, soil loss, slope, total LAI, total vegetation cover, trees stratum cover, shrub stratum cover, herbaceous stratum cover, moss stratum cover, bare soil surface, stones cover, and litter cover), ratified the existence of significant correlation (p > 0.05) between the variables runoff and soil loss with the total vegetation cover and the bare soil percentages. After this analysis we rejected those variables that introduced redundant information in the PCA analysis. The PCA results (Fig. 1), indicate eigenvalues (λ) of 2.93, 2.13 and 1.57 for factors 1, 2 and 3, respectively. The first factor explains 36.6% of the variance and is composed of vegetation cover, and trees and shrub stratum; factor 2 explains 26.7% and is made up of the LAI and the herbaceous stratum and factor 3 explains the 19.6% represented by the slope.

Grassland plots showed more variability in factors 1 and 2, regarding differences in vegetation cover (Fig. 1A). Shrub plots also showed some variability in relation to factor 3 (Fig. 1B), as consequence of slope, which ratifies the effect of this variable in the generation of greater runoff in S in comparison to G. These results showed that the hydrological response in the plots is a function of the vegetation structure (vegetation strata) and physical characteristics, mainly slope. The PCA ordination is consistent with the plot selection by structure and vegetation cover types.

In order to verify the homogeneity of the vegetation of hydrological plots, a DCA analysis on species composition and cover was performed using an input matrix of 12 samples (plots with vegetation cover) by 52 species with 224 occurrence samples. DCA was analysed with detrending.
by segments and downweighting of rare species. Ordination
eigenvalues were higher for the first axis ($\lambda_1=0.320$) than
for the second ($\lambda_2=0.139$). Cumulative variances of species
data improve greatly through addition of the second and
third axis ($\gamma_1=28.4$; $\gamma_2=40.3$; $\gamma_3=44.0$). The DCA results
(Fig. 2) showed that ordination of vegetation composition is
consistent with the selected plot design. Ordination revealed
a close relationship of species cover between hydrological
plots pertaining to each land cover type. Meanwhile
Brachypodium retusum (Pers.) Beauv. was the most abundant
species in all plots, and specially in G. plots under this
land cover type also presented Corynilla juncea L. and
Fumana ericoides (Cav.) Gand. Cistus albidus L. was
recorded in pine understory of $\tilde{A}S$ and $AG$ but other dwarf
shrubs were absent in forested plots (Globularia alypum L.,
Helichrysum stoechas (L.) Moench., Lithodora fruticosa
(L.)Griseb. and T. vulgaris L.). On the other hand, Q.
coccifera L. showed high cover values in $S$ and $\tilde{A}S$. Despite
the existence of a P. halepensis Miller layer, differences in
composition and cover of hydrological plots between S and
$\tilde{A}S$ were due mainly to the abundance of E. multiflora L. in
$S$ and the abundance of P. lentiscus L. and Rhhamnus
lycocton L. and the absence of dwarf shrubs in $\tilde{A}S$. The first
DCA axis is negatively correlated with LAI (Kendall’s
correlation coeff. $-0.879$, $p<0.000$, n=12), meanwhile the
second axis is negatively correlated with tree cover
(Kendall’s correlation coeff. $-0.638$, $p<0.006$, n=12), and
positively correlated with sediment yield (Kendall’s
correlation coeff. $-0.545$, $p<0.014$, n=12), and slope
(Kendall’s correlation coeff. $-0.450$, $p<0.045$, n=12).

4.2. Effects of afforestation on runoff and erosion

Many authors have studied the relationships between
vegetation cover, runoff and sediment yield (Elwell and
Stocking, 1976; Morgan, 1986; Trimble, 1988; Francis and
Thomes, 1990b; Teixeira and Mistry, 1997; Albadalejo et al.,
1998). This relationship is a function of several factors:
rainfall characteristics (volume, intensity, duration and
frequency); soil parameters (physics and hydrophysics);
and also vegetation structure. All the research carried out in
the study zone (Drouin et al., 1996; Bellot et al., 1998;
2001; Chirino et al., 2001) has indicated that the protective
effect of the vegetation cover controls runoff and soil
erosion. Vegetation cover diminishes the quantity of water
that reaches the soil through the effect of interception
(Bellot and Escarré, 1998), and also increases the hetero-
geneity of the spatial distribution of the rainwater toward the
soil, through the stemflow and throughfall (López, 1989;
Bellot and Escarré, 1998).

According to the structure of vegetation cover, some
differences were expected in runoff and soil erosion between
$AG$, $\tilde{A}S$, $S$ and G. Through the analysis of the entire research
period we observe the lower values of the annual runoff
coefficients in the vegetated plots ($\tilde{A}S$, $AG$, $G$ and $S$), which
do not exceed 1.0% of precipitation (Table 1), and approx-
imately increased 8-fold in open land plots ($B$). In conse-
quence the calculated annual soil loss rates (through runoff)
are also low (Table 1). The average values of the annual soil
loss rates were calculated in all vegetated plots (less than 0.05
Mg ha$^{-1}$ year$^{-1}$). The open lands ($B$) obtained higher values
(1.90 Mg ha$^{-1}$ year$^{-1}$). Table 1 also expresses the sediment
yield as percentage of bare soil annual values. The result
appears to indicate that afforested communities ($AG$ and $\tilde{A}S$)
show small mean values with respect to the non-afforested
ones (the half of the $G$ and $S$, respectively), despite the high
interannual variability of each cover type.

The statistical analysis of the runoff values and soil
losses (Table 2), detected significant differences only
between the accumulated runoff ($F_{COV}$ type $8.105$, $p<0.004$, $n=15$), and soil loss ($F_{COV}$ type $16.554$, $p<0.000$, $n=15$), of the vegetated plots ($\tilde{A}S$, $AG$, $S$ and
$G$) in comparison to the open lands ($B$). These results
indicate that in the aforementioned conditions, a 30-year-old
afforestation with Aleppo pine ($\tilde{A}S$ and $AG$) does not
significantly reduce the runoff and erosion in comparison
to natural vegetation ($S$ and $G$), which is very important for
management. In summary, these results are entirely consist-
tent with early expectations and we are relieved that there is
no significant difference between cover types.

These results are due to the high percentage of vegetation
cover in non-afforested plots. These values rise to 70–95% in
all (Table 1), which are enough to reduce the runoff and soil
loss in every type of vegetation cover analysed. If we take into account the estimated values of net precipitation
using the hydrological model VENTOS (Bellot et al., 2001),
31–40% of the rainfall is intercepted by vegetation cover in
plots, reducing both the runoff by 85%, and the soil loss by
95%, with respect to the open lands. Several authors
(Francis and Thomes, 1990a; Cerda, 1997; Andreu et al.,
1998) have reported similar figures. In our study, shrublands
and grasslands with more than 70% of vegetation cover
protect the soil in the same way as the afforested
communities, however, the AS plots reveal higher protection
(0.40% of runoff and 0.019 Mg ha$^{-1}$ year$^{-1}$ of

Fig. 2. DCA ordination plot of the plant species cover showing the position
of each hydrological plots. $S$, filled circle; $\tilde{A}S$, open circle; $G$, open square;
$AG$, filled square.
Table 1

<table>
<thead>
<tr>
<th>Characteristics</th>
<th>Slope and vegetation cover of hydrological plots</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>B</td>
<td>G</td>
<td>S</td>
</tr>
<tr>
<td>Slope (°)</td>
<td>22</td>
<td>26</td>
<td>23</td>
</tr>
<tr>
<td>Vegetation cover (%)</td>
<td>0</td>
<td>70</td>
<td>90.2</td>
</tr>
</tbody>
</table>

Annual runoff coefficient and soil loss in hydrological plots by vegetation type cover

<table>
<thead>
<tr>
<th>Years</th>
<th>Acum. rainfall (mm)</th>
<th>Annual runoff coefficient (%)</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>B</td>
<td>G</td>
<td>S</td>
<td>AG</td>
</tr>
<tr>
<td>1996</td>
<td>308.9</td>
<td>nd*</td>
<td>0.60</td>
<td>0.46</td>
</tr>
<tr>
<td>1997</td>
<td>483.3</td>
<td>5.36</td>
<td>0.62</td>
<td>0.63</td>
</tr>
<tr>
<td>1998</td>
<td>288.3</td>
<td>3.80</td>
<td>0.45</td>
<td>0.52</td>
</tr>
<tr>
<td>1999</td>
<td>241.1</td>
<td>4.11</td>
<td>0.52</td>
<td>0.77</td>
</tr>
<tr>
<td>Mean</td>
<td>330.4</td>
<td>4.42</td>
<td>0.55</td>
<td>0.60</td>
</tr>
<tr>
<td>2010-97b</td>
<td>134.6</td>
<td>12.54</td>
<td>0.59</td>
<td>0.86</td>
</tr>
<tr>
<td>2011</td>
<td>483.3</td>
<td>2.07</td>
<td>0.100</td>
<td>0.055</td>
</tr>
<tr>
<td>1998</td>
<td>288.3</td>
<td>0.873</td>
<td>0.003</td>
<td>0.004</td>
</tr>
<tr>
<td>1999</td>
<td>241.1</td>
<td>2.761</td>
<td>0.064</td>
<td>0.067</td>
</tr>
<tr>
<td>Mean</td>
<td>337.6</td>
<td>1.901</td>
<td>0.049</td>
<td>0.042</td>
</tr>
<tr>
<td>2010-97</td>
<td>134.6</td>
<td>1.70</td>
<td>0.07</td>
<td>0.05</td>
</tr>
<tr>
<td>2011</td>
<td>483.3</td>
<td>2.07</td>
<td>4.83</td>
<td>2.65</td>
</tr>
<tr>
<td>2012</td>
<td>288.3</td>
<td>0.873</td>
<td>0.34</td>
<td>0.46</td>
</tr>
<tr>
<td>2013</td>
<td>241.1</td>
<td>2.761</td>
<td>0.14</td>
<td>0.24</td>
</tr>
<tr>
<td>Mean</td>
<td>337.6</td>
<td>1.901</td>
<td>0.257</td>
<td>2.21</td>
</tr>
<tr>
<td>2010-97</td>
<td>134.6</td>
<td>1.70</td>
<td>4.11</td>
<td>2.94</td>
</tr>
</tbody>
</table>

* Treatment not available.
* Extraordinary event.

336 erosion), but the difference was not statistically significant from the other vegetated plots. Vicca et al. (2000) estimate a runoff coefficient of 0.65-1.54%, and erosion rates between 0.03 and 0.05 Mg ha⁻¹ in plots of 20 m² covered by herbaceous plants and shrubs, while in Eucalyptus sp. plots (15 years old and 25% vegetation cover) the annual erosion rates were 2.01% and 0.19 Mg ha⁻¹, respectively. The registered rates of soil erosion in our vegetated plots are lower than those reported in other research works carried out in semi-arid environments of Spain. Romero-Diaz et al. (1988) calculated annual soil losses of 0.08-2.55 Mg ha⁻¹ year⁻¹ in a catchment with 35% of vegetation cover with bare soil as substrate. In a microcatchment with 60% of vegetation cover, Albadaledo and Stocking (1989) determined rates between 0.5 and 1.2 Mg ha⁻¹ year⁻¹, and Lopez Bermudez et al. (1994) reported annual losses of 0.1 Mg ha⁻¹ year⁻¹ in plots with 8% shrub cover. Areas with reduced vegetation cover (lower than 50%) caused by human interference or affected by wildfires can increase soil loss in the first years after disturbance (Sánchez, 1997; Soto and Díaz-Fierros, 1997; Bautista, 1999). In our experimental area, the bare soil (B) registered higher values of runoff coefficient and soil loss (4.42% and 1.90 Mg ha⁻¹ year⁻¹, respectively). Cerda (1998) obtained similar results when comparing discontinuous Stipa tenacissima steppes, herb cover and dwarf shrub communities by means of rainfall simulation experiments.

4.3. Effects of precipitation characteristics on runoff and erosion

Besides the vegetation cover (%), rainfall characteristics determine the runoff and erosion magnitude (Francis and Thornes, 1990a; Cerda, 1998). Runoff and loss of soil caused by rainfall events reflect small values in both variables, as responses to the vegetation cover and to the characteristics of the rain. 75% of events in vegetated plots and 53% in bare soils showed runoff values of ≤ 0.1 L m⁻².

Table 2

<table>
<thead>
<tr>
<th>Vegetation type cover</th>
<th>B</th>
<th>G</th>
<th>S</th>
<th>AG</th>
<th>AS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Runoff (L m⁻²)</td>
<td>20.8 ± 15.46 a</td>
<td>2.56 ± 0.36 b</td>
<td>3.36 ± 0.75 b</td>
<td>4.01 ± 3.12 b</td>
<td>2.20 ± 1.15 b</td>
</tr>
<tr>
<td>Soil loss (Mg ha⁻¹)</td>
<td>3.64 ± 2.23 a</td>
<td>0.05 ± 0.00 b</td>
<td>0.07 ± 0.01 b</td>
<td>0.02 ± 0.01 b</td>
<td>0.02 ± 0.00 b</td>
</tr>
</tbody>
</table>

Values of runoff and soil loss followed by same letter are not significantly different at p ≤ 0.05.
During the whole study period we registered 94 rainfall events, with an annual mean of 330.4 mm. 90% of these events present volumes lower than 10 mm (Fig. 3), with an average intensity lower than 5 mm h⁻¹ and short duration (less than 2 h). These characteristics of precipitation and the vegetation cover (higher of 70%) may justify the lower measured runoff and erosion rates. The temporal distribution of rainfall events is also relevant in this region. These are concentrated in a few days of very heavy rain. The rainfall events with more than 10.1 mm (10% of the events) represent 57% of accumulated annual precipitation volume, with a mean maximum intensity ($I_{15}$) of 18.9 mm h⁻¹ and a mean duration of 3 h. Also storm events are frequent in the semi-arid area, called "flood", "cold drop" or "cut-off low", similar to that which occurred on 30-09-97, with 134.6 mm in 7 h 30 min, which represented 24% of the annual rainfall.

Considering that 90% of the precipitation events do not exceed 10 mm, and that these events generated between 81% and 89% of the runoff, and between 92% and 96% of soil loss in the different vegetation cover type, we carried out a second statistical analysis considering two rainfall sets (events until 10 mm and rainfall over 10 mm). The results (not shown) reinforce the previous interpretations detecting significant statistical differences between the means of the accumulated runoff and soil loss of the vegetated plots (IS, AG, S and G) and open land (B), but not between vegetation communities (Table 2). The effects of the extraordinary event of 30-09-97 only just increased the values of runoff and soil loss (Table 1), but not the relationships between different vegetation cover types.

Another aspect evaluated in this work has been the contribution of rainfall intensity on runoff and soil losses. Using multiple regressions to estimate runoff, through rainfall amount, maximum intensity at 15 min, mean weighted intensity, and rainfall duration, as independent variables, we obtain significant correlation coefficients in all vegetation cover types (Table 3). However, maximum intensity seems to have made the major contribution, together with the rainfall amount to explain the daily runoff and soil loss variability.

4.4. Afforestation effects on vegetation structure, plant life forms, species richness and diversity

Despite the observed changes in species composition and cover (Fig. 2), the statistical analysis of the vegetation cover structure of the hydrology plots (Table 4) indicates that afforestation with Aleppo pine only modifies the previous structure introducing the tree strata which was not present in natural vegetation communities (G and S). The surface occupied by the shrub stratum in the AS plots does not significantly differ from the surface in the S plots, being significantly smaller in AG. Nor does the surface occupied by herbaceous stratum in AG differ from the plots with natural vegetation (S and G), but the differences were significant between these and AS. On the other hand, we did not find differences in mosses and bare soil between the different hydrology plots; but the difference exists with regard to the surface covered by litter, being similar in AS, AG and S, and different from G (Table 4).

The conflict between pine plantations and conservation of pre-existing thorn shrublands is a relevant topic in the

---

**Table 2**

<table>
<thead>
<tr>
<th>Rainfall</th>
<th>$I_{15}$</th>
<th>mw_int</th>
<th>D</th>
</tr>
</thead>
<tbody>
<tr>
<td>IS</td>
<td>268,783</td>
<td>5.40 (0.03 ***)</td>
<td>-6.05 (0.03 ***)</td>
</tr>
<tr>
<td>AS</td>
<td>185,250</td>
<td>1.01 (0.02 ***)</td>
<td>-1.01 (0.02 ***)</td>
</tr>
<tr>
<td>S</td>
<td>353,265</td>
<td>8.28 (0.01 ***)</td>
<td>-8.91 (0.02 ***)</td>
</tr>
<tr>
<td>G</td>
<td>220,442</td>
<td>4.22 (0.03 ***)</td>
<td>-1.14 (0.02 ***)</td>
</tr>
<tr>
<td>B</td>
<td>395,198</td>
<td>0.14 (0.01 ***)</td>
<td>-0.11 (0.01 ***)</td>
</tr>
</tbody>
</table>

---

**Table 3**

<table>
<thead>
<tr>
<th>$R^2$</th>
<th>F</th>
<th>Coefficient of the independent variable</th>
</tr>
</thead>
<tbody>
<tr>
<td>t3.5</td>
<td>0.926</td>
<td>rainfall, $b_1$ slope, $b_2$ maximum intensity 15 min ($I_{15}$), $b_3$ mean weigh intensity (mw_int), and $b_4$ rainfall duration (D): gl 86.</td>
</tr>
<tr>
<td>t3.6</td>
<td>0.896</td>
<td>268,783 ***</td>
</tr>
<tr>
<td>t3.7</td>
<td>0.942</td>
<td>185,250 ***</td>
</tr>
<tr>
<td>t3.8</td>
<td>0.884</td>
<td>353,265 ***</td>
</tr>
<tr>
<td>t3.9</td>
<td>0.968</td>
<td>220,442 ***</td>
</tr>
</tbody>
</table>

---

**Table 4**

<table>
<thead>
<tr>
<th>Species composition</th>
<th>Plant life forms</th>
<th>Species richness</th>
<th>Diversity</th>
</tr>
</thead>
<tbody>
<tr>
<td>IS</td>
<td>AS</td>
<td>S</td>
<td>G</td>
</tr>
<tr>
<td>Semi-arid area</td>
<td>Aleppo pine</td>
<td>Thicket</td>
<td>Tree</td>
</tr>
<tr>
<td>Descending</td>
<td>Ascending</td>
<td>Descending</td>
<td>Ascending</td>
</tr>
</tbody>
</table>

---

**Footnotes**

1. $r_{1}$ rainfall, $b_{1}$ slope, $x_{1}$ maximum intensity 15 min ($I_{15}$), $x_{2}$ mean weigh intensity (mw_int), and $x_{4}$ rainfall duration (D): gl 86. **p < 0.001, **p < 0.01.
management of the Mediterranean semi-arid ecosystem (Esteve et al., 1990). From the medium to long-term, 432
Aleppo pine afforestation produces a positive effect increasing the overall vegetation cover with respect to non 433
afforested cover, which implies similar results regarding LAI (Table 4). On the other hand, a negative effect of the 434
afforestation is located in species richness values, being higher in natural vegetation communities where the estab-
435
ishment of dwarf shrubs is facilitated, in contrast to the afforested plots where the tree stratum reduces the presence 436
of other species in the understory. This effect is contrary to one of the priorities of the EU Environmental Policy, which 437
aims at the conservation of biodiversity of Mediterranean ecosystems (grassland and shrubland), through their 438
high rates of richness and species diversity (Peco et al., 1983).

439
To evaluate the representativeness of these results on structure, richness and diversity of species obtained in the 440
hydrology plots, we carried out another complementary study in landscape patches on the same community types. 441
Table 5 shows the statistical results of the comparative analysis between the natural vegetation communities with 442
their corresponding pairs of Aleppo pine afforestation (G – AG and S – AS). Overall richness and diversity present lower 443
values in afforested than in natural communities. Some shrubs like R. officinalis were present in S, but not in AS, and 444
the dominant Q. cocifera showed lower cover values in afforested plots (Table 6). These effects were especially 445

446

447

448

449

450

451

452

453

454

455

456

457

458

459
evident in G, where mean richness values decreased near 50% in afforested plots AG (Table 5). Moreover, afforestation with pines increased plant cover values in AS, increasing practically all the total available soil surface whereas there was not such an effect on AG. However, plant cover alone is not considered to be so important. There is a threshold cover value above which there is no significant impact on run-off and erosion.

The results indicate that afforested communities had a different species composition when compared with non-managed communities. Differences are clearer if we take into account the plant life forms as crude functional groups (Table 5). We did not find variation in forbs and annuals richness nor diversity in any of the cases that are compared. Grasses demonstrate lower richness and diversity values with pine afforestation in both cases, AG and AS. However, the perennial grass *B. retusum* seems to be more constant than other grasses (Table 6). Moreover, dwarf shrubs presented lower richness and diversity values with afforestation only in G. Shrub richness was lower in afforested communities than in non-afforested ones, whereas we obtained statistically significant differences in shrub diversity only between S and AS due to the low shrub abundance in G (Table 4). Most shrub species had fewer cover values in afforested plots, except *P. lentiscus* (a bird-dispersed shrub), which could have been facilitated through the presence of *P. halepensis*. Verdú and Garcia-Fayos (1996) described several examples of facilitation pathways in colonisation dynamics of this species by means of trees that can act as perches for frugivorous birds disseminating seeds.

In summary, the grassland communities presented greater shrub, dwarf shrub and grass species richness than in grassland afforestation. On the other hand, shrubland communities presented greater shrub and grass species richness and overall species diversity, with no differences in relation to dwarf shrubs. Several authors indicate similar effects associated with afforestation management.
(2002) and Watson et al. (2000) detected the decrease of
plant diversity of pre-existing native species after the
monospecific afforestations with pines in non-degraded
lands. Sometimes, current management techniques like
thinning or pruning, can affect the composition of forest
species (Freeman et al., 1996; Wagner et al., 1998). In spite
of this, it is recognised that plantations can be of benefit to
landscape composition by increasing the ecotone density
and landscape diversity (Estades and Temple, 1999; Bonet
et al., 2001).

4.5. Relationships between vegetation cover, runoff and soil
loss

In order to contrast these results with the so called 30% rule (Francis and Thomes, 1990b), we carried out a
regression analysis between runoff and soil loss with
vegetation cover. The best equation obtained was the
negative exponential model (Fig. 4), as proposed by Elwell
and Stocking (1976), with \( R^2 = 0.9183 \). This function indicated
that in the presence of high vegetation cover percentages,
the responses in runoff and soil loss in the different
vegetation patches analysed tended to have little differen-
tiation. Obviously, small values of vegetation cover increase
soil losses, the most significant loss being for covers under
30% of surface. Further studies on Alpha grass steppe
(37.0% vegetation cover) reveal runoff and soil loss values
similar to the estimated ones using these equations (Fig. 4).

The response of the dry grassland is related to the
presence of \( B. \) retusum cover, that occupies 65.6% of the
surface of the plots (Table 6): being capable of intercepting
between 38% and 57% of the precipitation, according to
results of the rainfall simulations in laboratory (Derooiche et
al., 1996). Cerdá (1997) reported runoff coefficients of
0.03-0.06% in herbaceous micro-plots (with \( B. \) retusum as
dominant species) and soil loss of 0.014 Mg ha\(^{-1}\). Gutiérrez
and Hernández (1996) in experiments with rainfall simulator
(137 mm h\(^{-1}\)) and different herbaceous covers showed that
a cover of 50% of herbaceous (in growth phase) and 70%
in (dormancy phase) was sufficient to significantly reduce
the runoff.

Based on our results (Table 2), the runoff generated by
vegetation cover type should present the following order:
\( AS < AG < S < G < B \), but the measured annual runoff coeffi-
cients indicated a different order: \( AS < G < S < AG < B \).
Moreover it was found to be significant that \( G \) formation
generated less runoff than \( S \) and \( AG \). In this case the
difference between \( G \) and \( S \) may be because the mean slope
of the shrub plots (29.2°) exceeds the slope in the \( G \) plots
(26°), and despite the importance of soil surface roughness
(Lavee et al., 1995). The difference between \( G \) and \( AG \),
can be related to the characteristics of the upper soil layer on
the surface. Both vegetation cover types present soil upon
loamy limestone substrate, but the \( AG \) plots are found
located in the medium third of the hillside, the zone of
greater erosion, for which the surface presents loamy
limestone horizon, with a certain degree of crusting that
reduces the infiltration and favours the generation of runoff.

Similar explanations are reported by Cerdá (1999), which
obtained greater values of runoff and erosion in soil
supported by loamy limestone.

Most authors indicate that the soil loss tolerance rates
may differ according to several characteristics of the soils,
topography and vegetation cover. In our study area, the
values measured in all the vegetation cover types (afforested
and non-afforested vegetation communities) and bare soil
are lower than the 5.0 Mg ha\(^{-1}\) year\(^{-1}\) of annual rate of soil
loss indicated by Smith and Stoney (1965) and Mitchell and
Buhenazer (1980). Only bare soil (in years 1997 and 1999)
presented annual soil losses that exceeded the rate of 2.0 Mg
ha\(^{-1}\) year\(^{-1}\) established by Arnoldus (1977) for soils whose
roots reach 25 cm depth, as in our case.

5. Conclusions

Our study provides basic information on runoff, sediment
yield and plant diversity in semi-arid natural and managed
communities and can be used to design effective recovery
strategies for degraded semi-arid ecosystems. Although 4

Fig. 4. Exponential functions and regression coefficients for runoff and soil
loss as a function of vegetation type cover during the study period. Together
with hydrological plots studied in this paper, the data of Alfa grass plots
with 37% of soil cover is included to illustrate the relevance of the so called
"30% rule".


Rehabilitation in Mediterranean Environmental Conditions. CSIC, Murcia, Spain, pp. 87–115.
López, M.V., 1989. Estudio de flujos hídricos y de la contribución de la degradación seca y lavío a la aparición de nutrientes a un suelo forestal bajo encina (Q. ilex) y pino (P. sylvestris) Tesis Master de Sciences. CIEAM.