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RESEARCH ARTICLE

Performance of Vegetation in Reclaimed Slopes Affected by Soil Erosion

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Abstract

Soil erosion in reclaimed mines may affect plant colonization and performance, and may compromise restoration success; however, the magnitude of this effect has seldom been quantified. We monitored the dynamics of vegetation (seed bank density, seedling emergence, plant mortality, and seed production) during a growing season (2003–2004) in three constructed slopes with differing past erosion rates. The slopes are located in the Utrillas coalfield in Spain, which experiences a Mediterranean-continental climate. In the most eroded slope, soil water availability was lower—especially in the interrill areas—and seedling emergence rate, plant survival, and seed production were also significantly lower than on the less eroded slopes. We found that vegetation recovery is dramatically constrained when rill erosion rate is 17 t ha^{-1} yr^{-1} and plant cover is 30%, but this effect disappears when plant cover is higher than 60%. Soil erosion in constructed slopes appears to inhibit natural plant colonization processes by increasing runoff water loss over the long-term. Thus, when rill erosion networks develop, human intervention would be needed to minimize the loss of water and facilitate vegetation colonization.

Key words: **constructed slopes, Mediterranean environment, mining reclamation, overland flow, rill erosion, water availability.**

Introduction

Soil erosion is one of the most significant barriers to the success of restoration practices in constructed slopes derived from mining activities (Whisenant 2005). Many failures in successful restoration of mining areas are the result of soil erosion processes triggered by an excess of overland flow on structurally poor soils with generally low infiltration rates (Nicolau & Asensio 2000).

Traditionally, the role of soil erosion on ecosystems has been interpreted as an abiotic exploitation agent, responsible for the loss of nutrients (Lü et al. 2007) and propagules (Cerdà & García-Fayos 1997; Chambers 2000). Soil erosion may reduce water availability by reducing soil depth and contributing to the formation of surface soil crusts (Pimentel et al. 1995; Sarah 2004). If soil erosion promotes the development of rill networks, water availability may be further reduced, since rills increase runoff connectivity on the slopes and provide efficient pathways to drive water out of the system (Favis-Mortlock et al. 2000; Bracken & Croke 2007). In this way, erosion can negatively affect plant colonization and performance by reducing the availability of seeds, nutrients, and water in soil.

On the other hand, it is well known that vegetation cover reduces soil erosion (Elwell & Stocking 1976; Francis & Thornes 1990; Bochet & García-Fayos 2004) by decreasing soil erodibility, effective precipitation, and kinetic energy of runoff and raindrops (Brandt 1989; Domingo et al. 1998; Martínez-Mena et al. 2000). Thus, one of the primary targets in the reclamation of artificial slopes is to reach optimal vegetative cover for the prevention of soil erosion (Redente & Depuit 1988; Andrés & Jorba 2000).

The relationships between erosion and vegetation are complex since the soil–vegetation system has feedback mechanisms which regulate soil formation, plant development, and erosion–sedimentation processes (Kirkby et al. 1998; Puigdefábregas et al. 1999). However, more attention has been paid to the effects of vegetation on erosion than the reverse (García-Fayos & Cerdà 1997). During the last decade, many studies have tried to identify the limiting factors controlling plant colonization in highly eroded areas of Mediterranean environments subjected to seasonal drought. García-Fayos et al. (1995) found that the lack of vegetation in badlands was not caused by seed removal as few seeds were lost by erosion (<13%), and seed rain was always greater than seed outputs. It is also known that the hardness of soil crusts adversely affects plant establishment by interfering with seedling emergence and reducing the infiltration rate, causing loss of water through runoff (Awadhwal & Thierstein 1985). Various studies suggest that the primary factor controlling plant colonization

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in these areas is the effect of water availability on germination and plant establishment (García-Fayos et al. 2000; Bochet et al. 2007).

Water availability is certainly one of the primary limiting factors affecting performance of vegetation in reclaimed slopes, especially in areas subjected to water stress such as Mediterranean environments (Martínez-Ruiz & Marrs 2007). Several authors have documented that high mortality rates of seedlings are related to water stress in artificial slopes created by mining activity (Bell & Ungar 1981). Also, studies on ecological succession in mine-reclaimed slopes have identified water availability as a major driver of successional trajectories (Wiegleb & Felinks 2001).

In reclaimed mining areas of Teruel province (Spain), Nicolau (2002) evaluated some mechanisms by which soil erosion can reduce water availability in eroded artificial slopes, namely: reduction of total soil depth and soil surface roughness, crust formation, and soil structure degradation. In this study, the hydrological response of reclaimed slopes was monitored 3 and 7 years after their construction, and substantial differences were observed: slopes in which rills formed in the early phase of slope development lost their functionality as ecological succession proceeded, and slopes in which the density of rills increased experienced high losses of water by surface runoff and lower rates of plant colonization. It can therefore be concluded that rill development in artificial slopes may limit water storage in soils and negatively affect plant colonization and the establishment of protective cover. Indeed, the loss of water resources has been associated with the initiation of feedback loops in both natural and reclaimed water-restricted environments, which causes sharp changes in ecosystem structure and functionality (Moreno-de las Heras et al. 2008; Turnbull et al. 2008).

In the present study we analyze the performance of vegetation throughout a growing season in three reclaimed slopes of the opencast coal mining area of Teruel that have been subjected to different soil erosion intensities since their construction. The slopes were reclaimed in 1988, with the same substrate, orientation, and general restoration treatments, but they differ in the density of rills as a result of failures in their geomorphological design. We expected that soil erosion processes would affect plant performance negatively in terms of the rate of seedling emergence, plant mortality, and seed production. We also expected that plant performance would be less successful in the slopes with higher density of rills, corresponding with lower soil water availability. In a previous study in the area (Moreno-de las Heras et al. 2008), we identified rill erosion as a driving force in plant succession on constructed slopes, leading to plant communities with low diversity and cover. The present study intends to identify the mechanisms through which soil erosion affects plant communities in reclaimed slopes. Halle and Fattorini (2004) emphasized the need to analyze the processes of germination, growth, and reproduction of plants in restored sites to understand the underlying processes involved. Our study follows this path, and can contribute to the understanding of the ecological effects of soil erosion in reclaimed ecosystems.

Methods

Study Site

The study was carried out in the "Utrillas fieldsite," located in the mine *El Moral*, in the Utrillas coalfield, Central-Eastern Spain (lat 40°47′24″ N, long 0°49′28″ W).

This area is situated in the *Iberian* chain with a mean altitude of 1,100 m a.s.l. Mean annual air temperature is 11°C (mean monthly temperature 6.8°C in December and 23.5°C in July), and the air frost period runs from October to April. The climate is Mediterranean-continental and the local moisture regime is Mediterranean-dry, according to Papadakis (1966). The rainy period primarily occurs in spring and autumn, with a mean annual precipitation of 466 mm. Potential evapotranspiration is 758 mm, with a hydrological deficit of 292 mm between June and October. The mean of the annual rainfall events is approximately 50 mm, which includes some convective rainstorms of high intensity, particularly in summer (Peña et al. 2002).

Mining activities in the area were completed in 1988–1989, at which time the company *Minas y Ferrocarril de Utrillas S.A.* undertook restoration practices. We selected three adjacent artificial north-facing slopes, with a rectilinear shape and 20° slope angle. All slopes received the same reclamation treatments, consisting of a 1-m superficial layer of overburden substrate revegetated by seeding a mixture of perennial grasses and leguminous herbs (*Festuca rubra, F. arundinacea, Poa pratensis, Lolium perenne, Medicago sativa*, and *Onobrychis viciifolia;* nomenclature follows Tutin et al. 1964–1980), with a sowing density of 350–400 kg seeds/ha. The overburden substratum came from the *Escucha* Cretacic formation of *Albian* age; this is a non-sodic and clay loam textured substratum with a basic pH (Table 1). In spite of having very similar initial features, monitoring of the slopes 2 years after their construction showed differences in the intensity of soil erosion influenced by the accumulation of overland flow in water-contributing areas upslope, as a result of design flaws (Nicolau 1996, 2002). No replicated slopes for a given erosion level were available. Thus, we considered sub-plots within each slope as replicates.

Data Sampling for Characterization of Vegetation, Soil, and Hydrology of Slopes

In order to characterize soil traits, five composite soil samples were collected from the top 10 cm in each experimental slope (each formed by six subsamples distributed at random, which were homogeneously mixed in an attempt to minimize spatial variability). Soil samples were air dried and sieved (2-mm sieve). General physicochemical characteristics were determined using the standardized methods proposed by the Spanish Ministry of Agriculture (MAPA 1994). Soil water retention at both field capacity and the willing point (−0.03 and −1.50 MPa, respectively) was determined following the pressure chamber procedure (Klute 1986). Water-holding capacity was subsequently calculated as the difference between these values (Descheemaeker et al. 2006). Finally, soil bulk density

Table 1. Basic characteristics (soil, surface traits, and hydrological features) of the three experimental slopes.

	N	*Slope 1*	*Slope 2*	*Slope 3*
Topography				
Slope length (m)		50	75	60
Slope gradient (°)		20	20	20
Length of water-contributing area (m)		8.0	6.5	na
Soil traits				
Stoniness (%)	15	27.4 ± 3.5^a	26.2 ± 4.1^a	24.5 ± 3.3^a
Sand (%)	15	33.5 ± 3.7^a	33.8 ± 3.0^a	36.3 ± 2.7^a
Silt (%)	15	33.8 ± 1.6^a	30.8 ± 1.8^{ab}	26.6 ± 4.5^b
Clay (%)	15	32.8 ± 2.9^a	35.4 ± 2.1^a	37.1 ± 2.9^a
Texture	15	Clay loam	Clay loam	Clay loam
WHC (%, w/w)	15	11.9 ± 0.2^a	10.7 ± 0.5^a	10.6 ± 1.5^a
pH H_2O; w/v: 1/2-	15	8.0 ± 0.1^a	8.0 ± 0.1^a	7.9 ± 0.1^a
EC w/v: 1/2- (dS/m)	15	0.26 ± 0.14^a	0.20 ± 0.10^a	0.23 ± 0.03^a
CEC (cmol^c/kg)	15	23.3 ± 3.3^a	28.3 ± 1.0^a	22.1 ± 3.5^a
ESP (%)	15	0.27 ± 0.11^a	0.13 ± 0.01^a	0.18 ± 0.03^a
Bulk density (g/cm^3)	45	1.49 ± 0.12^a	1.39 ± 0.17^a	1.23 ± 0.17^b
Organic matter (%)	15	0.56 ± 0.23^a	1.27 ± 0.35^a	2.00 ± 0.74^b
Surface traits				
Vegetation cover (%)	42	9.9 ± 12.2^a	26.2 ± 23.8^a	59.4 ± 17.6^b
Aboveground biomass (g/m^2)	42	32.5 ± 43.7^a	56.9 ± 54.7^a	205.4 ± 96.4^b
Hydrological features				
Runoff coefficient (%)		21.0	15.9	4.5
Rill density (m/m^2)		0.78	0.58	0.00
Rill erosion rate (t ha^{-1} year^{-1})		45.0	16.9	0.0

Mean ± standard deviation values are shown. Different letters within rows indicate differences at $\alpha = 0.05$ (Kruskal–Wallis test and post hoc Mann–Whitney).
N, number of samples; WHC, water-holding capacity; w/w, ratio of water (weight)/soil (weight); EC, electrical conductivity; w/v, ratio of soil (weight)/water (volume); CEC, cation exchange capacity; ESP, exchangeable sodium percentage; na, not applicable.

was determined from 15 unaltered soil sample cores (3-cm height by 5-cm diameter) collected at random in each slope.

Vegetation sampling took place in June 2004. In each slope, vegetation cover was quantified in fourteen 50 × 50–cm quadrats, distributed at random. Aboveground biomass was collected in these quadrats, and subsequently oven-dried (60°C; 72 h) and weighed.

To assess soil erosion processes, rill density was measured in each slope (m of linear rill/m^2 of area). Rill sections (width and depth) were directly determined in all rills intercepted by three equidistant cross-slope transects of 30-m length. Historical averaged rill erosion rate (t ha^{-1} yr^{-1}) in each slope was quantified using the former rill network dimensions as well as the bulk soil density and slope age from reclamation (Morgan 1995).

Cumulative runoff rate in the analyzed experimental slopes is provided for a later hydrological year (from October 2005 to October 2006). This data were obtained in 3 × 15–m runoff plots (Moreno-de las Heras et al. 2007).

Experimental Design

We established ten 5 × 5–m plots randomly distributed in two positions on each slope: five upslope (upper half of the slope) and five downslope. In slopes 1 and 2, with rills already developed, we compared the dynamics of vegetation between rill and interrill sites. It must be stressed that rill microsites have very little vegetative cover due to the mechanical perturbations provoked by water flow. Slope 3 was considered an interrill area as it had no rills.

Soil Moisture

Volumetric soil moisture content (%) in the upper 15 cm of soil was measured with TDR technology (Topp et al. 1980; Topp & Davis 1985) by using a TRIME FM (Imko®) instrument. Soil moisture was sampled during vegetation data collection beside each permanent plot established for the monitoring of plant mortality.

Soil Seed Bank Composition and Seedling Emergence

In September 2003, before the arrival of autumn rains, we collected soil samples (60 cm^2 × 4–cm depth) to analyze the composition of the soil seed banks. In each plot we collected eight soil samples located at random, four in rills and four in interrills. Each of these samples was subdivided into four subsamples that were placed in 250 ml plastic containers over a 5-cm vermiculite layer. The floristic composition of the soil seed banks was determined by means of germination under optimal conditions in a greenhouse.

In November 2003, after the conclusion of autumn germination in the field, we measured the number of emerged seedlings in 20 × 20–cm quadrats randomly distributed in each plot, three in rills and three in interrills. Relative emergence rate for each plot was calculated by dividing the mean number of emergences recorded in the field by the density of germinable seeds in soil obtained in the greenhouse experiment.

Plant Mortality

In October 2003 we established eight permanent 20×20–cm quadrats randomly distributed in each plot, four on rill and four on interrill areas to monitor seedling survival. Quadrats on bare soil were discarded, which may have importance for the interpretation of the results in rills where plant cover is very scarce and individuals concentrate in few safe sites. We recorded the performance of each seedling emerging from October 2003 to June 2004 twice a month in autumn and spring, and once a month in winter. During each sampling day we recorded every seedling failure and the cause of mortality. We used orthogonal photographs of each quadrat 1.30 m above ground (Andres & Jorba 2000) to complement field observations. This permitted us to have more observational data of seedlings in order to better assign the possible cause of death. We differentiated four causes of mortality: (1) frost (seedlings were dessicated, usually after becoming reddish, during winter months), (2) soil erosion (including seedlings that were removed with sediments and seedlings that were buried under sediments and did not re-emerge), (3) sheep-trampling, and (4) drought (seedlings dessicated in late spring, coinciding with the period of minimum water availability). Other factors may have influenced seedling mortality, although field observations suggested that these were not major causes of death. We calculated total and plot-specific mortality rate by dividing the number of dead seedlings by the total number of emerged seedlings in each quadrat. We also estimated the relative percentage of seeds in soil that became adult plants by multiplying the emergence rate of seedlings by their survival rate in each plot.

Seed Production

We analyzed the influence of soil erosion on seed production and seed weight. For this purpose we chose the species *Aegilops geniculata*, as it is one of the most conspicuous in the studied slopes (Table 2). In summer 2004, after the seeds had matured, we collected all *A. geniculata* caryopses in 50×50–cm quadrats, 12 in the upper part and 12 in the lower part of each slope (and within each part, 6 in the rills and 6 in the interrills, also randomly distributed). We separated all the seeds and weighed them after air-drying. To explore the differences in the weight of produced seeds between slopes, we differentiated between light (<0.005 g) and heavy seeds (>0.005g), as it is well known that *Aegilops* spp. produce two types of seeds within the same plant progeny that differ in mass and germinability (Marañón 1989).

Data Analysis

We used nonparametric statistical tests (Kruskall–Wallis and Mann–Whitney U tests) to explore the differences between the three slopes in the soil properties, the density of soil seed banks, seedling emergence rate, relative percentage of seeds in soil that became adult plants, and *A. geniculata* seed production and average seed weight. Nonparametric tests were selected because data could not be transformed. The use of nonparametric analyses also follows the recommendation of Ruxton and Beauchamp (2008) to use the Kruskal–Wallis test with environmental data, if "the sample distributions are similar and symmetric" and if "the variances of [these] ranks are similar for all the groups." A nonmetric multidimensional scaling (NMDS) analysis was used for data on floristic composition of the soil seed bank. The graphic representation of the samples in the ordination space enabled us to measure the heterogeneity in floristic composition of the different slopes. Soil moisture and relative percentage of causes of mortality were transformed (arcsine (square root(x))) to achieve normality. Soil moisture data from the different sampling dates were analyzed with repeated measures analysis of variance (ANOVAs; with time as within subjects factor and slope as between subjects factor). A multi-analysis of variance (MANOVA) test was used to analyze the differences between slopes in the relative percentage of causes of mortality, using the plot as experimental unit. Means were back-transformed to report the data. Multivariate analysis was performed with PC-ord package (McCune & Mefford 1997); the STATISTICA package was used for the remaining statistical analyses (Statsoft Inc. 1996).

Results

Soil Properties

Although general chemical properties (pH, conductivity, cation exchangeable capacity, and exchangeable sodium percentage), stoniness, texture, and the water-holding capacity of the soils were rather homogeneous between the three analyzed slopes (Table 1), some important differences regarding soil bulk density (Kruskall–Wallis $H = 14.74$, $N = 45$, $df = 2$, $p < 0.001$) and organic matter content (Kruskall–Wallis $H = 9.92$, $N = 15$, $df = 2$, $p = 0.007$) were detected. Soil organic matter in slopes 1 and 2 was significantly lower than in slope 3 (Table 1). Additionally, soil bulk density in slopes 1 and 2 was significantly higher than in slope 3 (Table 1), where a high density of herbaceous roots was observed.

Soil Moisture

More water was available in the rills than in the interrills in slopes 1 and 2 (repeated measures ANOVA, $F_{1,60} = 270.49$, $p < 0.001$, Fig. 1a). Because of the differences in hydrology between rills and interrills, we analyzed the water content of the soils from both sites separately. Results of the repeated measures ANOVA with the data on volumetric water content in interrills revealed significant differences between slopes; slope 1 had the lowest values whereas the uneroded slope had the highest ($F_{2,42} = 107.64$, $p < 0.001$) (Fig. 1b). Rills in slope 2 had more water than in slope 1 ($F_{1,28} = 107.50$, $p < 0.001$). Slope position had no effect in the water content.

Table 2. Mean (±SE) density of seeds in soil (seeds/m^2) of each of the identified species in each slope and erosion area (R, rills; IR, interrills).

	Slope 1		*Slope 2*		*Slope 3*
Species	*R*	*IR*	*R*	*IR*	*IR*
Aegilops geniculata	86.3 ± 35.4	16.4 ± 9.1	12.3 ± 12.3	49.3 ± 14.8	542.8 ± 126.5
Androsace maxima	0	0	0	0	4.1 ± 4.1
Anthemis tuberculata	0	0	4.1 ± 4.1	0	4.1 ± 4.1
Brachypodium retusum	0	0	16.4 ± 16.4	0	333.1 ± 234.2
Bromus hordeaceus	0	0	0	0	24.7 ± 24.7
Bromus rubens	78.1 ± 53.6	4.1 ± 4.1	49.3 ± 49.3	20.6 ± 9.2	53.5 ± 29.4
Bromus tectorum	0	0	0	0	4.1 ± 4.1
Desmazeria rigida	4.1 ± 4.1	0	20.6 ± 14.1	0	271.4 ± 104.6
Echinaria sp.	4.1 ± 4.1	4.1 ± 4.1	4.1 ± 4.1	0	0
Echium sp.	4.1 ± 4.1	0	0	0	0
Filago vulgaris	0	0	61.7 ± 37.9	24.7 ± 9.1	24.7 ± 16.4
Hieracium pilosella	0	0	0	0	4.1 ± 4.1
Hordeum murinum	20.6 ± 11.1	4.1 ± 4.1	4.1 ± 4.1	0	0
Juncus sp.	0	0	0	8.2 ± 8.2	0
Lamium amplexicaule	0		0	4.1 ± 4.1	4.1 ± 4.1
Lolium sp.	0	0	4.1 ± 4.1	0	4.1 ± 4.1
Medicago sativa	41.1 ± 33.0	4.1 ± 4.1	4.1 ± 4.1	4.1 ± 4.1	0
Papaver rhoeas	0	0	0	4.1 ± 4.1	4.1 ± 4.1
Plantago lanceolata	0	0	8.2 ± 8.2	4.1 ± 4.1	12.3 ± 8.8
Sanguisorba minor	0	0	4.1 ± 4.1	0	4.1 ± 4.1
Santolina chamecyparissus	0	0	78.1 ± 36.0	16.4 ± 9.1	4.1 ± 4.1
Senecio vulgaris	0		0	4.1 ± 4.1	4.1 ± 4.1
Sisymbrium orientale	24.7 ± 12.6	8.2 ± 8.2	24.7 ± 9.1	32.9 ± 12.0	4.1 ± 4.1
Thymus vulgaris	0	0	0	0	16.4 ± 12.6
Xeranthemum annum	0	0	41.1 ± 41.1	4.1 ± 4.1	123.4 ± 88.2
Undetermined graminoids	45.2 ± 15.6	24.7 ± 11.0	180.9 ± 109.0	37.0 ± 17.8	468.8 ± 176.2
Undetermined forbs	41.1 ± 13.7	12.3 ± 6.3	82.2 ± 39.2	32.9 ± 10.2	106.9 ± 26.1
Total seeds in soil	349.5 ± 102.1^a	78.1 ± 21.6^a	600.3 ± 170.3^b	246.7 ± 31.8^b	2023.0 ± 265.3^c
Emerged seedling density	34.3 ± 14.3^a	14.4 ± 4.9^a	170.8 ± 29.8^b	149.2 ± 27.3^b	1616.2 ± 146.3^c
Seedling emergence rate	16.9 ± 11.2^a	13.0 ± 4.9^a	75.7 ± 37.7^b	67.1 ± 13.7^b	90.0 ± 11.3^c
Total plant mortality	23.1 ± 5.7^a	76.6 ± 6.8^b	22.0 ± 4.8^a	70.0 ± 5.9^b	6.01 ± 1.7^c
% Seeds became adult plants	12.7 ± 8.3^a	2.7 ± 1.0^a	58.5 ± 28.7^b	22.3 ± 6.4^b	84.5 ± 10.6^c

Means (±SE) of total seed density in soil (seeds/m^2), emerged seedling density (seedlings/m^2), seedling emergence rate in the field (%), total plant mortality (%), and relative percentage of seeds in soil that became adult plants (%) are also indicated. Different letters indicate significant differences between slopes (Mann–Whitney U test, $p < 0.01$). Nomenclature follows Tutin et al. (1964–1980).

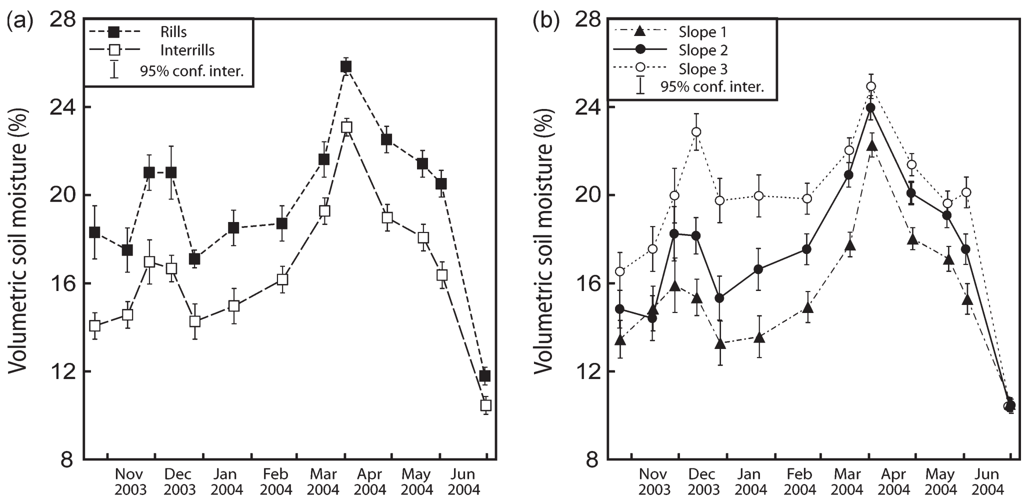

Figure 1. Mean volumetric soil water content and 95% confidence interval in the different sampling dates along the growing season in (a) rills and interrills and (b) interrills of the three slopes.

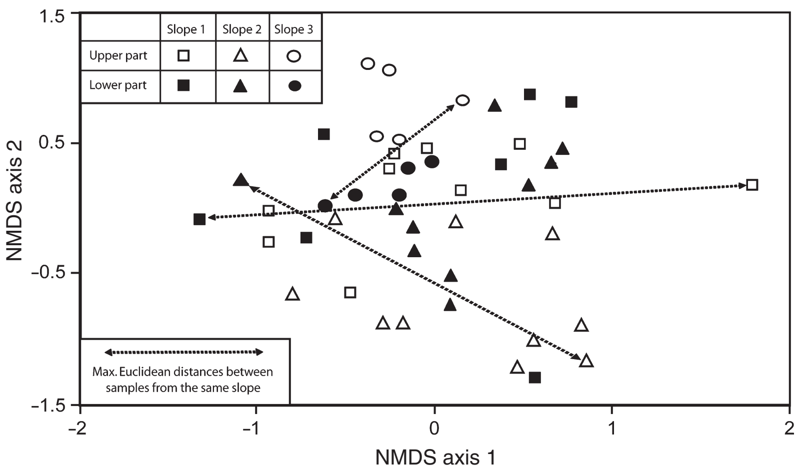

Figure 2. NMDS ordination graph with the floristic composition of the soil seed bank data.

Soil Seed Banks and Seedling Emergence

Twenty-five different species were identified in the soil seed banks of the three slopes (Table 2). The NMDS with the floristic composition data (final stress 0.24) showed a greater heterogeneity between samples in slopes 1 and 2 on the basis of Euclidean distances (Fig. 2) compared with samples in the uneroded slope (slope 3). We detected an effect of slope position on floristic composition in slope 3, with significant differences occurring between the axis 2 coordinates of the samples from the upper and lower part of this slope (Mann–Whitney U test, $Z = -2.61$, $p = 0.009$).

There were significant differences in the total density of seeds between the three slopes (Kruskall–Wallis $H = 26.73$, $N = 50$, $df = 2$, $p < 0.001$), with more seeds in slope 3 than in the eroded slopes (Table 2). Slope position did not affect seed density. Results of the evaluation of erosion zone data indicated that in slope 1 there were more seeds in the rills than in the interrills (Mann–Whitney U test, $Z = 2.60$, $p = 0.008$). Differences in seed density between rills and interrills in slope 2 followed the same pattern but with low statistical significance ($p = 0.057$).

Seedling emergence rate in the field was lower in the more eroded slope and reached the highest value in slope 3 (Kruskall–Wallis $H = 22.66$, $N = 50$, $df = 2$, $p < 0.001$) (Table 2). No significant effect of slope position or erosion zone (rills and interrills) on seedling emergence rate was found.

Plant Mortality

Total plant mortality during the whole growing season was higher in the slopes subjected to higher soil erosion ($F_{2,74} = 10.56$, $p < 0.001$, Table 2). No significant effect of slope position was detected. In slopes 1 and 2 total plant mortality was greater in interrills than in rills ($F_{1,56} = 67.6$, $p < 0.001$, Table 2). The MANOVA test with the relative percentages of mortality causes showed significant differences between slopes ($F_{8,148} = 3.29$, $p = 0.001$, Fig. 3). Drought was the greatest

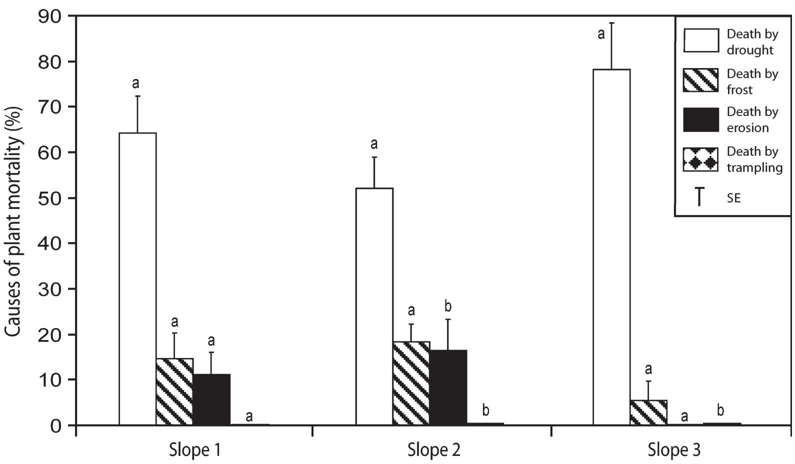

Figure 3. Relative percentage of the different causes of plant mortality in the three slopes. Different letters indicate significant differences between slopes (LSD test, $p < 0.05$).

Table 3. Mean number (±SE) of *Aegilops geniculata* seeds produced (seeds/m^2) and mean weight (±SE) of the two types of *A. geniculata* seeds in each slope.

	Slope 1	*Slope 2*	*Slope 3*	*Kruskal–Wallis (H)*	*N*	*p <*
Seeds produced (seeds/m^2)	56.6 ± 14.7^a	184.7 ± 47.6^{ab}	887.6 ± 382.4^b	13.35	62	0.001
Light seed weight (mg)	3.3 ± 0.08^a	3.7 ± 0.11^b	3.6 ± 0.04^b	14.58	892	0.001
Heavy seed weight (mg)	8.6 ± 0.18^a	10.9 ± 0.26^b	10.3 ± 0.15^b	48.83	2855	0.001

Different letters indicate significant differences between slopes (Mann–Whitney U test; $p < 0.05$).

cause of mortality in all slopes ($42.1 \pm 4.7\%$, mean ± SE) followed by erosion ($15.5 \pm 3.6\%$, mean ± SE) and frost ($14.6 \pm 2.9\%$, mean ± SE). Erosion affected only slopes 1 and 2. Plant mortality from trampling by sheep was almost negligible ($0.1 \pm 0.0\%$, mean ± SE) and only appeared in slopes 2 and 3 (Fig. 3). The relative percentage of seeds in soil that became adult plants was significantly higher in slope 3 (Kruskall–Wallis $H = 27.17$, $N = 47$, $df = 2$, $p < 0.001$, Table 2), and no effect of erosion zone (rill and interrill) or slope position was detected.

Seed Production and Seed Weight

We found a significantly higher density of *A. geniculata* seeds in slope 3 (Table 3); no effect of slope position or erosion zone was detected. Both heavy and light seeds from the more eroded slope had a significantly lower weight than the seeds produced in slopes 2 and 3 (Table 3). No effect of slope position or erosion zone (rill and interrill) was detected on seed weight.

Discussion

Monitoring of vegetation dynamics in mine-reclaimed slopes subjected to different soil erosion intensity during 16 years reveals that reduced plant performance is associated with increasing soil erosion rates. All development stages measured, seedling emergence rate, plant survival, seed production, and seed weight, were negatively affected as rill erosion increased.

Soil seed density was highly variable between slopes (ranging from 78 to 2,023 seeds/m^2), with the lowest values found in the most eroded slope. These values are comparable to the low seed densities found in natural badlands of Mediterranean Spain (García-Fayos et al. 1995, 2000). The differences in soil seed densities clearly reflect the variation in vegetation cover between slopes. In contrast, density of emerged seedlings varied much more between slopes than seed densities, ranging from 14 to 1,616 seedlings/m^2. Although seed densities were similar to other degraded areas, the rate of seedling emergence in the uneroded slope was higher; the maximum seedling emergence density found by García-Fayos et al. (2000) in Spanish badlands was 273 seedlings/m^2; and Elmarsdottir et al. (2003) found values around 300 seedlings/m^2 in reclaimed slopes of Iceland. Therefore, higher soil erosion rates imply a reduction in seedling emergence.

Soil erosion also affected the spatial heterogeneity of soil seed banks, because more seeds were found in the rills than in the interrills, possibly because rills trap seeds in the eroded slopes (Tsuyuzaki et al. 1997; Chambers 2000). We observed higher mortality in the eroded slopes, primarily because of drought. Hence, plant establishment is impeded in reclaimed slopes affected by soil erosion. Indeed, although an average of 84% of the seeds in soil become adult plants in slope 3, only 9% do so in slope 1. García-Fayos et al. (2000) found that plant mortality in Spanish badlands reached values up to 98%, and was mostly related to drought. In our reclaimed slopes, plant mortality ranged from 77% in interrill areas of the most eroded slope to 6% in the uneroded slope. The lower plant mortality found in rills is probably because of its higher soil moisture compared with interrills; however, this is not representative of the slope overall, because only a few safe sites in rills, such as small depressions protected from water flow by rock fragments, contain plants. Mortality directly related to erosion or frost was less common, each affecting about 5% of the plants. García-Fayos et al. (2000) found similar values for erosion-related mortality of plants in badlands of SE Spain.

Finally, we have observed that soil erosion affects seed production negatively, as fewer and smaller seeds of the species *Aegilops geniculata* were found in the most eroded slope. Although density of seeds may be influenced by the differences in vegetation cover between slopes, the impact on seed weight must be an indirect consequence of soil erosion effects on soil water availability. The influence of limited resources, mainly water, on seed production, by means of a reduction in seed number or size is well known (Stephenson 1981; Pyke 1989). Baalbaki et al. (2006) observed that in several *Aegilops* species, seed number and weight were the attributes most affected by severe water stress in Mediterranean semiarid areas of Lebanon.

These negative effects of soil erosion on plant performance were associated with a reduction of the soil water content in the more eroded slopes. This would have constrained plants growing in the eroded slopes to withstand more intensive water stress. This might also explain the internal organization of the vegetation across the slope 3 (the differences in floristic composition between up- and downslope areas) and its absence on the eroded slopes, where the pattern of rills and interrills would be the primary factor influencing the structure of the plant community.

Our results suggest that long-term effects of soil erosion in constructed slopes inhibit vegetation development by limiting water availability. We found lower soil moisture in the slopes with more developed rill networks. In addition to the draining effect of rill networks, feedback mechanisms between vegetation and soil may be involved in these limitations. Indeed, 16 years of different soil erosion rates have clearly affected soil formation processes on the slopes in the study area and, consequently, water infiltration. This has resulted in limitations to the accumulation of organic matter and the impact of vegetation roots in the bulk density. Overall water availability limitations are also assessed by measured runoff coefficients along the 2005–2006 hydrological year (Moreno-de las Heras et al. 2007). Although all slopes received the same rainwater (615 mm), the high runoff rates of the eroded slopes may have limited their water availability. The runoff rate of slope 3 was 4.5%, which is similar to the values found in non-degraded natural slopes and successfully reclaimed mining slopes of the Mediterranean Spain (Puigdefábregas et al. 1999; Nicolau 2002; Martínez-Murillo & Ruiz-Sinoga 2007). Conversely, runoff rates in slopes 1 and 2 ranged from 15 to 20%, indicative of a noticeable loss of water by surface runoff. Other works carried out in Mediterranean Spain found a high similarity between the spontaneous flora in eroded zones and in uneroded areas naturally subjected to severe water stress, which agrees with the hypothesis that plants growing in eroded areas must face a higher water deficit related to soil erosion processes (Guerrero-Campo & Monserrat-Martí 2004).

The three constructed slopes represent three different points of a soil erosion intensity gradient. The analysis of vegetation dynamics in these slopes allows us to conclude that plant performance is more similar in slopes 1 and 2 than in slope 3. The model of competition for soil moisture between vegetation and soil erosion proposed by Thornes (1985) established instability regions in the erosion–vegetation system depending on critical values of soil erosion rate and vegetation cover. We can assume that slopes 1 and 2 fall into this instability situation, in which thresholds of erosion and vegetation cover have been surpassed and control of soil and water resources rely on erosional rather than vegetative processes. Therefore, threshold values of vegetation cover and erosion rates for our system may be close to those of slope 2: 30% and 17 t ha^{-1} yr^{-1}, respectively. Although these values should be evaluated in additional experimental studies, they agree with values proposed by other authors in Mediterranean Spain (Francis & Thornes 1990; Andres & Jorba 2000; Gimeno-García et al. 2007).

We must stress the importance of improving the design of constructed slopes to avoid the formation of water-contributing areas upslope that concentrate overland flow and may promote soil erosion processes. Hancock & Willgoose (2004) demonstrated that runoff control at the top of reclaimed slopes is crucial to the long-term stability of such structures. This is of particular importance for Mediterranean environments, where, when a threshold is passed, an extensive and costly human intervention is needed, and in some cases, a state of irreversibility is reached (Aronson et al. 1993).

We can thus conclude that soil erosion acting in the long term may impose serious limitations on plant performance that are associated with a reduction in water availability. In this sense, soil erosion could act as an ecological filter that may direct the community assembly process toward an irreversible situation in which natural plant colonization is heavily constrained.

Implications for Practice

- The early triggering of soil erosion processes in constructed slopes leads to situations in which performance of vegetation is seriously constrained by the loss of water resources.
- To avoid situations in which natural plant colonization is inhibited, vegetation cover of at least 30% and rill erosion rates below 17 t ha^{-1} yr^{-1} are required in Mediterranean-continental reclaimed environments.
- In highly eroded constructed slopes with developed rill networks, human intervention is needed to prevent the loss of water and facilitate vegetation recovery.

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