Ecological Engineering 36 (2010) 1290–1298

Contents lists available at ScienceDirect

Ecological Engineering

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

Dominant plant species modulate responses to hydroseeding, irrigation and fertilization during the restoration of semiarid motorway slopes

Pablo García-Palacios [a,b,*], Santiago Soliveres [a,b], Fernando T. Maestre [a], Adrián Escudero [a], Andrea P. Castillo-Monroy [a], Fernando Valladares [a,b]

[a] Departamento de Biología y Geología, Área de Biodiversidad y Conservación, Escuela Superior de Ciencias Experimentales y Tecnología, Universidad Rey Juan Carlos, c/Tulipán s/n, 28933 Móstoles, Spain
[b] Instituto de Recursos Naturales, Centro de Ciencias Medioambientales, CSIC, C/Serrano 115-bis, 28006 Madrid, Spain

ARTICLE INFO

Article history:
Received 10 November 2009
Received in revised form 5 April 2010
Accepted 5 June 2010

Keywords:
Dominant species
Fertilization
Irrigation
Hydroseeding
Grasslands
Plant composition
Roadside slopes
Semiarid
Soil erosion

ABSTRACT

Restoring roadside slopes in semiarid regions of the Mediterranean Basin is often constrained by the difficulties arising when developing restoration projects (absence of nearby natural ecosystems serving as reference sites and slow natural colonization) and by the contradictions found between short-term (reduce soil erosion) and long-term (increase plant diversity) restoration goals. Restoration techniques developed in temperate climates are commonly applied in these regions without taking into account their specific characteristics; as a consequence, they often fail. We evaluated the effectiveness of three treatments widely used by practitioners (hydroseeding, fertilization and irrigation) to foster community composition changes that control soil erosion and increase species diversity (restoration goals) during the restoration of motorway embankments. The study was carried out during an 18-month period in five embankments from semiarid central Spain. The most outstanding result was that responses of the plant community to the treatments evaluated were site-specific. Several fast-growing dominant species, some hydroseeded and some already present in the study sites, were responsible for this idiosyncratic variation between sites. On embankments, where plant cover can easily reach values high enough to prevent erosion, the use of non-native herbs that can potentially dominate the community should be avoided. These fast-growing species, although effective as starters the first years following motorway building, can constrain vegetation dynamics in the long term. Our results indicate that these species should be controlled in the field, and their presence avoided in the commercial seed mixtures when the target is to enhance biodiversity and ecosystem stability and resilience.

1. Introduction

Roads are one of the human constructions causing a major environmental impact (Briggs and Giordano, 1992). The adjacent slopes created or transformed after road building are abundant habitats worldwide (Valladares et al., 2008). Despite the abundance and environmental impacts that these constructions promote, there is a lack of basic information on the ecology of the degraded lands resulting from motorway construction (Bradshaw and Huttl, 2001), as well as on the best strategies to restore them (Matesanz et al., 2006). However, designing projects to restore roadside slopes in semiarid areas of the Mediterranean Basin is not an easy task. First of all, these environments have been intensively transformed by humans for centuries (Naveh and Dan, 1973), making it difficult to select a nearby natural ecosystem as a reference site (Hobbs et al., 2006). Secondly, the slow colonization of these slopes by natural vegetation is seriously conditioned by water availability (García-Fayos et al., 2000), the lack of propagules (Tormo et al., 2007) and the adverse climatic and soil conditions (Bochet and García-Fayos, 2004) characterizing these environments. In addition, there are apparent contradictions between the achievement of short-term (e.g., controlling soil erosion, Andrés et al., 1996) and long-term (e.g., increasing plant diversity to enhance ecosystem resilience to future environmental conditions and disturbances, Hooper et al., 2005) restoration goals.

The existence of widespread landscape transformations like extensive agriculture, introduction of non-native species (Blondel and Aronson, 1995) or wildfire incidence (Pausas, 2004) can complicate the selection of the reference site that could serve as

* Corresponding author at: Departamento de Biología y Geología, Área de Biodiversidad y Conservación, Escuela Superior de Ciencias Experimentales y Tecnología, Universidad Rey Juan Carlos, c/Tulipán s/n, 28933 Móstoles, Spain.
Tel.: +34 914888517; fax: +34 916647490.
E-mail address: pablo.palacios@urjc.es (P. García-Palacios).

0925-8574/$ – see front matter © 2010 Elsevier B.V. All rights reserved.
doi:10.1016/j.ecoleng.2010.06.005

a model for planning a restoration project (SER, 2004). Moreover, ongoing environmental changes and the increasing prevalent anthropogenic disturbance may result in novel ecosystems whose composition and/or function differ from any historical system (Jackson and Hobbs, 2009). Thus, little is known about the factors controlling ecosystem functioning in this potentially new successional context, as well as about the suitability of widely used restoration treatments for the recovery of vegetation in these novel ecosystems (Matesanz et al., 2006). In these cases, key questions to be answered are which restoration goals must be set and what is the baseline for comparisons and reference. Recovering ecosystem services and promoting ecosystem functioning could be an appropriate restoration objective (Hobbs et al., 2006) and a critical aspect in disturbed Mediterranean environments (Méndez et al., 2008). There is consensus that diversity is essential for maintaining ecosystem functioning and the stability of ecosystem processes in human-dominated and fast-changing environments (Loreau et al., 2001). Therefore, promoting shifts in community composition that increase species diversity at short-time scales seems a reasonable strategy to restore degraded ecosystems when reference sites are not available.

The colonization of semiarid motorway slopes by natural vegetation is typically very slow (Bochet and García-Fayos, 2004). This process is restricted by low water availability levels (García-Fayos et al., 2000), lack of propagules due to the prevalence of inefficient dispersal based on mixospermy and even atelechory (Tormo et al., 2007), and by the adverse climatic and soil conditions found for most commercial seed mixtures in many restoration sites (Bochet and García-Fayos, 2004). The stabilization of slopes and the control of soil erosion through the establishment of a dense herbaceous cover is a priority in the restoration of recently built roadside slopes (Andrés et al., 1996). Hydroseeding is widely used for this purpose in temperate climates (Sheldon and Bradshaw, 1997) and has also become very popular in semiarid Mediterranean areas to restore roadside slopes (Andrés et al., 1996; Albaladejo et al., 2000; Matesanz et al., 2006; Tormo et al., 2007). The commercial hydroseeding mixtures are mainly composed by highly competitive forage grass and legume species non-native to these areas (Martínez-Ruiz et al., 2007; Tormo et al., 2007). However, in many cases, this technique renders poor results in terms of species richness and aboveground biomass (Matesanz et al., 2006), and therefore leads to poor protection from soil erosion (Andrés et al., 1996; Tormo et al., 2007).

Numerous studies have investigated the effects of resource availability over entire plant communities (Fransen, 1998; Cahill, 1999), but few of these studies have been conducted on motorway slopes (Holmes, 2001; Paschke et al., 2000). Furthermore, it has been shown that interactions between the availability of water and nutrients can largely determine the response of herbaceous assemblages (Maestre and Reynolds, 2007). Although fertilization and irrigation often increase herbaceous productivity in the short term (Hooper et al., 2005), they can also prevent long-term vegetation development because of competition with spontaneous colonizers (Holl, 2002). Atmospheric fertilization is more marked in novel ecosystems such as roadside slopes, where N deposition from vehicle emissions represents an important N input to the system (Cape et al., 2004). However, the contribution of hydroseeding, and its joint effects with the increase in soil resources (water and nutrients), to the trade-off between the first step of establishing a dense herbaceous cover that controls soil erosion (Petersen et al., 2004) and the second step of facilitating the establishment of late-successional species is still controversial (Martínez-Ruiz et al., 2007). Therefore, it is necessary to evaluate the potential of these costly techniques and their interactions for restoring semiarid motorway slopes.

In this article, we experimentally evaluated the effects of both hydroseeding and simultaneous changes in the availability of water (irrigation) and nutrients (fertilization) on the restoration of vegetation in degraded motorway slopes. We did this in different motorway embankments located in Central Spain. We considered plant cover and soil erosion as surrogates of slope stability (Norris et al., 2008) and plant diversity and community composition as surrogates of ecological restoration success (Pywell et al., 2002; Smith et al., 2003). As responses are likely to be context dependent in roadside slopes such as studied (Matesanz et al., 2006), we have selected several similar sites sharing a similar climate and construction characteristics. This multi-site approach allows the evaluation of the benefits and generality of the experimental treatments applied, adding further value to this study.

2. Methods

2.1. Study area

The study area is located in the R4 and AP36 motorways, between Pinto (Madrid; 40°14′N, 3°43′W) and Corral de Almaguer (Toledo; 39°45′N, 3°03′W), in the centre of the Iberian Peninsula (altitude *c.* 700 m a.s.l.). The climate is semiarid, with cold winters and a severe summer drought. The mean temperature and precipitation is 15 °C and 450 mm, respectively (Getafe Air Base climatic station 40°18′N, 3°44′W, 710 m.a.s.l., 1971–2000). A meteorological station (Onset, Pocasset, MA, USA) was located in the AP36 motorway to get a more detailed description of the local climatic conditions during the study.

In order to homogenize the slope selection, and to minimize main sources of variation when restoring semiarid roadside slopes (Matesanz et al., 2006), we selected five sites of similar slope type (embankments) and size (greater than 15 m long from top to bottom of the slope and 30 m wide), slope aspect (south) and inclination (between 20 and 30°). Three and two of these sites were located in the R4 (sites 1–3) and AP36 (sites 4 and 5) motorways, respectively. The five sites have poor and alkaline soils with low water holding capacity, but differ in their soil type and construction age (Table 1). The artificial soils of these embankments are constructed using local parent material, gravels and components from external sources that are stored for a while before motorway building (Forman et al., 2003; confirmed by information provided by the motorway builder). In fact, substrate differences between sites are not linked with soil type, but probably caused by the heterogeneity of materials used for embankment construction. Therefore, potential differences in restoration treatment effectiveness between the two motorways studied are not expected to be related with their soil types.

2.2. Experimental design

Three restoration treatments (hydroseeding [HS], fertilization [Fe] and irrigation [Ir]) were evaluated in this study. The full experiment included five different factors (HS, Fe, Ir, block and site). Six blocks containing 12 1 m × 1 m plots per block, with at least 1 m buffer between plots, were randomly established at each of the five sites (Fig. 1). Each block contained a full factorial design with the three treatments employed (HS, Fe and I), which were randomly assigned to the plots within each block and site.

We added three HS levels (control, seeding and seeding + mulch) during December 2006. The control and seeding addition levels consisted of no seeding addition and the application of a commercial seed mixture (Zulueta Corp., La Rioja, Spain; dose of 30 g/m^2; Table 2), respectively. The seed mixture and seed application rate

Table 1
Main characteristics and soil properties of the five sites studied. Numerical values are means ± SE ($n = 10$).

	Site 1	Site 2	Site 3	Site 4	Site 5
Coordinates	40°16′08″N 3°43′14″W	40°14′11″N 3°44′12″W	40°13′39″N 3°44′34″W	40°53′39″N 3°28′22″W	40°47′46″N 3°12′34″W
Motorway	R4	R4	R4	AP36	AP36
Construction age (years)	3	3	3	0	0
Soil type	Gypsum	Gypsum	Gypsum	Calcareous	Calcareous
Angle (°)	20	20	21	30	26
Initial plant cover (%)	37	58	35	12	52
Water holding capacity (ml water g soil^{-1})	0.63 ± 0.01	0.43 ± 0.03	0.54 ± 0.03	0.36 ± 0.03	0.49 ± 0.03
Total N (mg N g soil^{-1})	0.25 ± 0.01	0.34 ± 0.04	0.37 ± 0.02	0.14 ± 0.01	0.22 ± 0.03
Total P (mg P g soil^{-1})	0.58 ± 0.04	0.35 ± 0.01	0.44 ± 0.04	0.16 ± 0.01	0.15 ± 0.02
pH	8.21 ± 0.14	8.06 ± 0.15	7.88 ± 0.14	8.35 ± 0.14	8.37 ± 0.15

Table 2
Species composition of the commercial seed mixture used in the hydroseeding treatment at a dose of 30 g/m^2 (Zulueta Corporation, La Rioja, Spain).

	% of seeds
Agropyrum cristatum	10
Agropyrum desertorum	15
Colutea arborescens	1
Genista scorpius	1
Lavandula latifolia	1.5
Lolium multoflorum var. westertwoldicum	30
Medicago sativa	15
Melilotus officinalis	10
Moricandia arvensis	1.5
Piptatherum milliaceum	2.5
Retama sphaerocarpa	2.5
Vicia sativa	10

were copied from the protocols commonly used by restoration practitioners in the study area (Matesanz et al., 2006). Most of the species hydroseeded are non-native to Spain, which is the norm in restoration projects in the Mediterranean Basin (Méndez et al., 2008). The ingredients of the seeding + mulch level were stabilizer (Stable, dose of 10 g/m^2; Projar, Valencia, Spain), wood fiber mulch (Hortifibre, dose of 100 g/m^2, Projar, Valencia, Spain), water (dose of 3 l/m^2) and the seed mixture. To isolate the effects of mulch and stabilizer from that of the seed mixture, we included the seeding level. Fertilization was applied twice (December 2006 and January 2008) at two levels (control and fertilized). Fertilized plots received a 20 g/m^2 dose of a slow release N:P:K (16:11:11) inorganic fertilizer (Scott Corp.), while control plots were not fertilized. Irrigation was conducted from March to June in both 2007 and 2008, coinciding with the peak growing season of annual plant communities characterizing our study sites. This treatment was applied at two levels (0 and 50% of the monthly total precipitation median from the 1971 to 2000 period). The amounts of water added kept this 50% increase in relation with the current year values recorded in the on site meteorological station in 2008 (Table A.1).

2.3. Plant community responses

In May 2007 and 2008, the optimal phenological moment to measure the herbaceous communities studied, we visually determined the absolute cover of each plant species on the whole 1 m × 1 m plots. To avoid different bias in visual estimations among researchers, all the plots were surveyed by the same person. From these values, we estimated for each plot total plant cover, which can exceed 100% because of overlapping vegetation strata, species diversity (Shannon's Index; Shannon and Weaver, 1963) and community composition.

2.4. Soil measurements

We placed 16 ECH$_2$O humidity sensors (Decagon Devices Inc., Pullman, USA) in the soil at a depth of 5 cm at sites 2 and 4 to

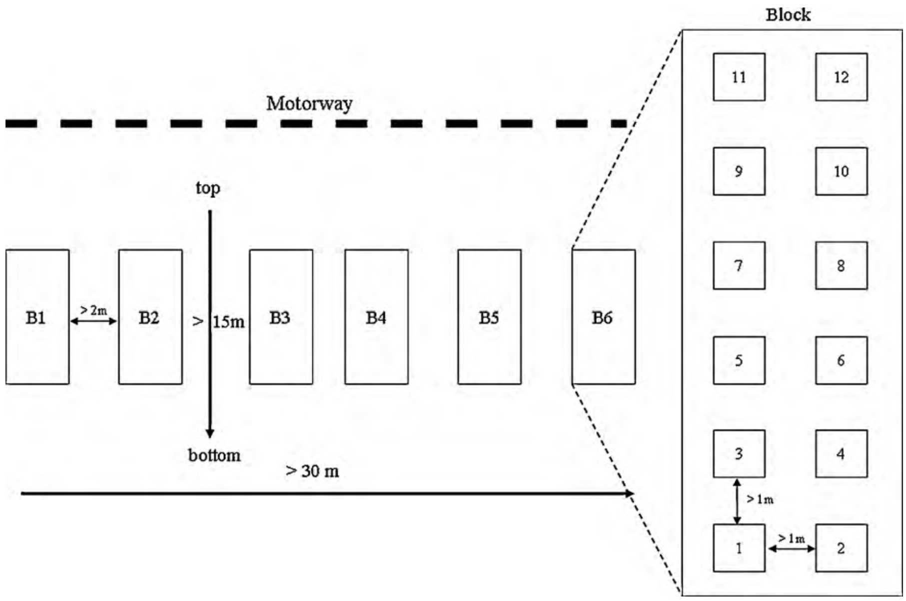

Fig. 1. Experimental design replicated in the five sites studied. The six blocks (B1–B6) were randomly established at each site. Each block contained a full factorial design with the three treatments employed (three hydroseeding levels × two fertilization levels × two irrigation levels = 12 plots). The treatment combinations were randomly assigned to the plots within each block. Double and single head arrows represent buffer zones between plots and blocks and motorway slope measures, respectively.

assess the effects of irrigation on soil moisture dynamics during the study period (2 sites × 2 irrigation levels × 4 replicates). These measurements were recorded every 90 min. Ten soil cores (10 cm × 4 cm × 4 cm) were randomly collected at each site at the beginning of the experiment for laboratory analysis. Soil water holding capacity was determined volumetrically (Parkin et al., 1996). Total N and P were obtained on a SKALAR San^{++} Analyzer (Skalar, Breda, The Netherlands) after digestion with sulphuric acid. The pH was measured with a pH meter, in a 1:2.5 mass:volume soil and water suspension.

To measure the volume of soil mobilized in each plot, we buried ten 20 cm long microprofiles 10 cm under the soil surface in August 2007 in sites 2–5. We calculated the initial length (L_0) of the microprofiles standing out of the soil at this date and again in February 2008 (L_1). We used the mean differences between L_1 and L_0 as our surrogate of soil erosion (Andrés and Jorba, 2000). Positive values indicate the exportation of soil downward and negative values indicate the deposition of soil. Therefore, the final result represents the balance between erosion and sedimentation processes. To avoid intrinsic slope effects of each block on the surrogate of soil erosion measured, we standardized the values in each block using the index RII (Armas et al., 2004). In each plot, RII was calculated as $(SE_t - SE_c)/(SE_t + SE_c)$, where SE_t and SE_c are the absolute mean differences between L_1 and L_0 in a given combination of treatments and control plots, respectively. RII ranges from −1 to +1: a value of zero indicates equal soil erosion on both plots (no treatment effect). Increasing positive values indicate increasing soil erosion (the volume of soil mobilized is higher in that treatment than in the control).

2.5. Statistical analyses

A model including all the sources of variation considered would be extremely complex and difficult to interpret. For instance, it is well known that extreme fluctuations between years in the structure and composition of the herbaceous communities characterizing the studied motorway slopes is the norm, especially under semiarid climates (Wali, 1999). In addition, the plant community responses between the two motorways may vary because of uncontrolled constraints regarding motorway construction or traffic intensity (Cape et al., 2004). Thus, we built independent models for each motorway (R4 and AP36) and monitoring year (2007 and 2008). A total of 216 1 m × 1 m plots in the R4 sites and 144 plots in the AP36 sites were measured every year. We evaluated the treatment effects on plant diversity and cover and soil erosion using a five-way nested ANOVA model. We used site and block as between plot factors, and HS, Fe and Ir as within plot factors; all the factors except block (random) were fixed. Relationships among response variables and the cover of the most dominant species (mean cover ≥ 20%) were examined using the Spearman correlation coefficient because of lack of normality in the data. The most dominant species used in the analysis were chosen independently of their different origins: hydroseeding, seed bank or colonization from nearby forage crops. In sites 1 (2008), 4 (2007) and 5 (2008), the cover of the two most dominant species was summed to reach this 20%. Where appropriate, Tukey's *B*-test was used for *post hoc* comparisons.

Differences in community composition between treatments were evaluated using the semi-parametric PERMANOVA approach (Anderson, 2001). It allows including experimental designs in the analysis of multivariate datasets on the basis of any distance measured using permutation methods. For these analyses, we used Bray–Curtis distance and 9999 permutations (permutation of raw data). The *p*-values used in the analyses were obtained from a random Monte Carlo sample from the asymptotic distribution of the pseudo *F*-statistic under permutation (appropriate for limited unique permutations because of the low number of permutable units dictated by the denominator used to construct the *F*-statistic; Anderson and Willis, 2003). Additionally, square root transformations were used to increase the influence of rare species (Lepš and Šmilauer, 2003). To visualize patterns in multivariate data, we used a principal coordinate analysis (hereafter PCO) with the same distance measure (see Anderson, 2003 for a detailed description of the method). To identify which species were responsible for the patterns found, the values on the first two axes of each plot were correlated with the cover of each species using the Spearman correlation coefficient; species with a correlation coefficient greater than 0.7 were represented by vectors. The length and direction of each vector indicate the strength and sign, respectively, of the relationship between a given species and the PCO axes. Each vector begins at the centre of the circle and ends at the coordinates consisting of the correlations between that species and each of the PCO axes.

Nested ANOVAs and correlation analyses were carried out with SPSS version 14.0 (SPSS Inc., Chicago, IL, USA). PERMANOVA and PCO analyses were carried out with PERMANOVA+ for PRIMER (PRIMER-E Ltd., Plymouth Marine Laboratory, UK).

3. Results

3.1. Plant cover and soil erosion

In the R4 sites in 2007, HS increased the total plant cover in 10%. This effect was mainly due to the effect of the seeding level (Table 3; Table A.3; *post hoc* results not shown), but disappeared when the plots were also fertilized (HS × Fe; $F_{2,30} = 3.63$, $p = 0.039$). In 2008, significant site ($F_{1,15} = 9.59$, $p < 0.001$) and Ir effects ($F_{1,15} = 34.67$, $p < 0.001$) were found on total cover. This variable was about 10% higher in irrigated plots. In the AP36 sites, there were significant differences between the levels of HS in 2008 ($F_{2,20} = 5.43$, $p = 0.013$) when analyzing plant cover. The seeding and seeding + mulch levels increased the plant cover by 16% respect to the control values (Table 3; Table A.3; *post hoc* results not shown).

Our surrogate of soil erosion (mean $L_1 - L_0$) was negative in both motorways (R4 = −3.8 and AP36 = −0.85; $n = 72$), indicating that no overall erosion occurred in our study slopes. No significant effects or interactions among treatments were found on this variable (as measured with the RII index) in the R4 sites (data not shown), but a significant site × Ir interaction was found in the AP36 sites ($F_{1,10} = 6.07$, $p = 0.033$; Table A.4). The Ir treatment decreased soil erosion at site 4 (Fig. 2A), but had no consistent effects at site 5 (Fig. 2B).

3.2. Plant diversity and community composition

In the R4 sites in 2007, HS increased the species diversity by 10%, mainly because of the effects of the seeding level (Table 4; Table A.3; *post hoc* results not shown). However, Fe reduced that positive effect by 5% (HS × Fe; $F_{2,30} = 6.37$, $p = 0.005$). A significant site effect was also found for this variable $F_{1,15} = 17.00$, $p < 0.001$). The differences between sites in the Shannon's Index (site 1 = 0.70, site 2 = 0.57 and site 3 = 0.31) were the consequence of the dominance of *Carduus tenuiflorus* in site 3 (67% of total plant cover). In 2008, a significant HS × Fe interaction ($F_{2,30} = 4.11$, $p = 0.027$) was found on species diversity, but this effect was very weak. In the AP36 sites, we found significant site × HS × Ir ($F_{2,20} = 5.63$, $p = 0.012$) and site × Fe ($F_{1,10} = 21.70$, $p < 0.001$) interactions in 2007 and 2008, respectively. The later result was due to the 67% increase of species diversity found under fertilization at site 4 (Table 4; Table A.3).

Table 3
Total plant cover in the R4 (1–3) and AP36 sites (4 and 5) in both 2007 and 2008. Values are means ± 1 SE, $n = 6$. The control levels correspond to no treatment addition. The seeding level consisted of the application of a commercial seed mixture (dose of 30 g/m^2; Table 2). The seeding + mulch level consisted of stabilizer, wood fiber mulch, water and the seed mixture. The fertilization and irrigation levels consisted of 20 g/m^2 of a slow release N:P:K (16:11:11) inorganic fertilizer and 50% of the monthly total precipitation median from 1971 to 2000 period (Table A.1), respectively. See Table A.3 for the statistical analysis of these data.

2007		Control					Fertilization				
		Site 1	Site 2	Site 3	Site 4	Site 5	Site 1	Site 2	Site 3	Site 4	Site 5
Control	Control	47 ± 4.3	54 ± 6.7	60 ± 4.9	43 ± 12.8	61 ± 7.4	47 ± 3.8	62 ± 8.2	67 ± 1.5	54 ± 8.8	64 ± 6.0
	Seeding	55 ± 4.6	65 ± 5.8	63 ± 4.5	49 ± 10.2	68 ± 3.4	55 ± 9.1	54 ± 6.7	65 ± 1.1	49 ± 12.2	72 ± 2.5
	Seeding + mulch	49 ± 6.6	57 ± 8.2	58 ± 8.2	40 ± 8.4	66 ± 4.1	59 ± 6.2	66 ± 8.0	67 ± 8.0	34 ± 12.2	66 ± 8.1
Irrigation	Control	50 ± 2.8	63 ± 3.7	63 ± 1.5	35 ± 12.2	74 ± 6.2	65 ± 9.3	64 ± 6.8	68 ± 1.5	48 ± 16.6	66 ± 3.8
	Seeding	61 ± 7.9	58 ± 6.7	72 ± 5.3	54 ± 10.7	89 ± 4.9	49 ± 4.8	63 ± 8.0	68 ± 2.8	43 ± 6.9	75 ± 6.7
	Seeding + mulch	53 ± 5.0	63 ± 7.4	64 ± 7.4	43 ± 13.3	67 ± 6.4	60 ± 8.4	59 ± 10.2	60 ± 10.2	45 ± 7.7	67 ± 4.1
2008											
Control	Control	52 ± 9.9	59 ± 8.15	60 ± 4.1	48 ± 15.2	42 ± 8.8	51 ± 2.9	56 ± 9.7	72 ± 5.9	59 ± 12.8	44 ± 2.6
	Seeding	62 ± 7.5	55 ± 3.1	71 ± 3.4	59 ± 10.5	61 ± 7.5	49 ± 7.1	53 ± 4.9	72 ± 4.3	45 ± 10.5	66 ± 5.5
	Seeding + mulch	61 ± 8.2	58 ± 8.7	63 ± 3.8	56 ± 7.8	67 ± 3.1	67 ± 5.3	64 ± 4.3	75 ± 4.1	61 ± 11.1	69 ± 6.9
Irrigation	Control	70 ± 7.5	67 ± 4.0	74 ± 3.7	36 ± 11.2	56 ± 10.3	64 ± 8.4	59 ± 3.8	84 ± 2.8	42 ± 15.9	42 ± 8.9
	Seeding	55 ± 2.7	54 ± 9.1	72 ± 3.4	63 ± 11.0	55 ± 6.0	76 ± 8.3	60 ± 8.6	79 ± 5.1	50 ± 8.3	72 ± 8.6
	Seeding + mulch	61 ± 6.1	64 ± 6.4	81 ± 3.1	41 ± 9.3	62 ± 5.7	60 ± 5.3	65 ± 4.3	73 ± 3.8	62 ± 6.6	66 ± 5.6

Table 4
Species diversity, measured with the Shannon's Index, in the R4 (1–3) and AP36 sites (4 and 5) in both 2007 and 2008. Values are means ± 1 SE, $n = 6$. Rest of legend as in Table 3.

2007		Control					Fertilization				
		Site 1	Site 2	Site 3	Site 4	Site 5	Site 1	Site 2	Site 3	Site 4	Site 5
Control	Control	0.70 ± 0.04	0.48 ± 0.06	0.29 ± 0.12	0.39 ± 0.09	0.29 ± 0.02	0.75 ± 0.03	0.52 ± 0.04	0.31 ± 0.11	0.46 ± 0.02	0.34 ± 0.04
	Seeding	0.73 ± 0.03	0.64 ± 0.05	0.29 ± 0.08	0.44 ± 0.08	0.37 ± 0.03	0.64 ± 0.05	0.54 ± 0.06	0.36 ± 0.07	0.43 ± 0.09	0.38 ± 0.02
	Seeding + mulch	0.66 ± 0.03	0.57 ± 0.06	0.23 ± 0.12	0.38 ± 0.06	0.35 ± 0.02	0.75 ± 0.04	0.63 ± 0.06	0.32 ± 0.09	0.49 ± 0.06	0.32 ± 0.07
Irrigation	Control	0.68 ± 0.04	0.56 ± 0.01	0.31 ± 0.08	0.30 ± 0.11	0.34 ± 0.04	0.72 ± 0.04	0.51 ± 0.06	0.30 ± 0.11	0.37 ± 0.08	0.35 ± 0.03
	Seeding	0.69 ± 0.02	0.63 ± 0.07	0.45 ± 0.11	0.46 ± 0.07	0.46 ± 0.05	0.75 ± 0.06	0.58 ± 0.05	0.25 ± 0.10	0.39 ± 0.07	0.36 ± 0.03
	Seeding + mulch	0.61 ± 0.04	0.52 ± 0.06	0.35 ± 0.11	0.55 ± 0.07	0.35 ± 0.01	0.73 ± 0.04	0.60 ± 0.02	0.28 ± 0.10	0.50 ± 0.04	0.32 ± 0.03
2008											
Control	Control	1.60 ± 0.14	1.53 ± 0.07	1.43 ± 0.16	0.60 ± 0.22	0.94 ± 0.21	1.47 ± 0.12	1.38 ± 0.18	1.41 ± 0.17	1.05 ± 0.09	0.86 ± 0.08
	Seeding	1.64 ± 0.09	1.51 ± 0.14	1.58 ± 0.14	0.49 ± 0.13	1.159 ± 0.18	1.43 ± 0.08	1.32 ± 0.22	1.50 ± 0.21	1.22 ± 0.23	1.27 ± 0.13
	Seeding + mulch	1.44 ± 0.16	1.23 ± 0.15	1.54 ± 0.08	0.70 ± 0.20	0.74 ± 0.18	1.53 ± 0.13	1.46 ± 0.10	1.71 ± 0.10	1.03 ± 0.11	0.91 ± 0.22
Irrigation	Control	1.69 ± 0.09	1.41 ± 0.16	1.73 ± 0.04	0.54 ± 0.23	0.93 ± 0.17	1.50 ± 0.13	1.42 ± 0.17	1.40 ± 0.12	0.80 ± 0.20	0.79 ± 0.12
	Seeding	1.51 ± 0.07	1.41 ± 0.20	1.54 ± 0.06	0.64 ± 0.09	1.10 ± 0.13	1.53 ± 0.13	1.52 ± 0.21	1.57 ± 0.21	1.04 ± 0.12	1.13 ± 0.19
	Seeding + mulch	1.30 ± 0.18	1.47 ± 0.11	1.38 ± 0.08	0.77 ± 0.16	1.08 ± 0.15	1.45 ± 0.16	1.53 ± 0.12	1.51 ± 0.11	1.15 ± 0.14	1.09 ± 0.21

When analyzing community composition, a significant site effect was found in 2007 ($F_{2,15} = 24.93$, $p < 0.001$) and 2008 ($F_{2,15} = 22.56$, $p < 0.001$) in the R4 sites, where we also found a weak site × HS ($F_{4,30} = 1.63$, $p = 0.041$) interaction in 2008 (Fig. 3A and B; Table A.5). In the AP36 sites, we found a significant site ($F_{1,10} = 29.45$, $p < 0.001$) and HS effects ($F_{2,20} = 6.37$, $p < 0.001$), as well as a site × HS interaction ($F_{2,20} = 2.40$, $p = 0.035$), in 2007 (Fig. 4A; Table A.5). The seeding + mulch level produced a shift in the community in sites 4 and 5 (*post hoc* results not shown). In 2008, several significant effects and interactions were found (Fig. 4; Table A.5): site ($F_{1,10} = 7.30$, $p < 0.001$), HS ($F_{2,20} = 4.13$, $p < 0.001$), Fe ($F_{1,10} = 11.87$, $p < 0.001$), site × HS ($F_{2,20} = 3.98$, $p < 0.001$), HS × Fe ($F_{2,20} = 1.99$, $p = 0.029$), site × HS × Fe ($F_{2,20} = 2.83$, $p = 0.002$) and site × HS × Ir ($F_{2,20} = 2.30$, $p = 0.020$). Separate ANOVAs were conducted in each site to help in the interpretation of these interactions. HS significantly affected the community composition of sites 4 ($F_{2,10} = 5.56$, $p = 0.001$) and 5 ($F_{2,10} = 8.28$, $p = 0.001$). The seeding + mulch level was responsible for this variation at site 4, as it promoted an increase in the cover of *Melilotus officinalis*, *Morincadia arvensis* and *Onobrychis viciifolia*. No level was found significant in site 5 (*post hoc* results not shown). The addition of nutrients changed the composition by increasing the community diversity, mainly at site 4. The species more correlated with the variation explained by the PCO axes were responsible for differences among sites.

Relationships between the cover of the most dominant species and species diversity showed contrasting results depending on the motorway considered (Table 5). In the R4 sites, the dominant

Table 5
Relationships between diversity (Shannon's Index; H) and the absolute cover of the most dominant species (mean cover ≥ 20%) in both R4 (1–3) and AP36 (4–5) sites in 2007 and 2008. In sites 1 (2008), 4 (2007) and 5 (2008), the sum of the two most dominant species was used to reach the 20% of the cover. Spearman's correlation (ρ) coefficient and p-values are shown ($n = 72$).

Site	Year	Dominant species	H	
			ρ	p
1	2007	*Bromus rubens*	−0.186	0.116
1	2008	*Bromus rubens* + *Medicago sativa*	−0.252	0.033
2	2007	*Anacyclus clavatus*	0.163	0.172
2	2008	*Carduus tenuiflorus*	−0.258	0.029
3	2007	*Carduus tenuiflorus*	−0.858	< 0.001
3	2008	*Carduus tenuiflorus*	−0.174	0.144
4	2007	*Diplotaxis erucoides* + *Lolium multiflorum*	0.548	< 0.001
4	2008	*Melilotus officinalis*	0.06	0.616
5	2007	*Hirschfeldia incana*	−0.574	< 0.001
5	2008	*Melilotus officinalis* + *Lolium rigidum*	0.307	0.009

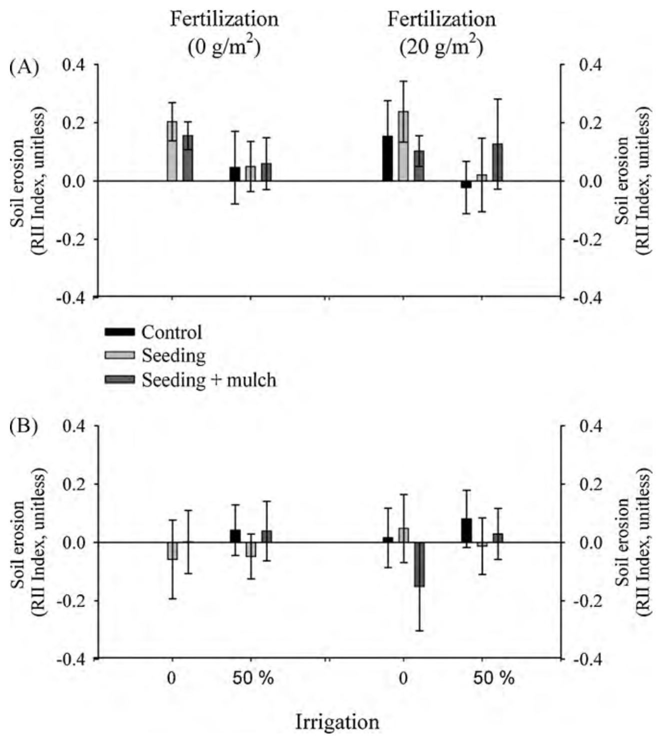

Fig. 2. Volume of soil mobilized, our surrogate of soil erosion, as measured with the RII index in the AP36 sites (A, site 4; B, site 5). Values are means ± SE ($n = 6$). Irrigation levels (0 and 50%) correspond to the control and 50% of the monthly precipitation median, respectively, and fertilization levels (0 and 20 g/m^2) to the control and the dose applied, respectively. The control and seeding levels correspond to no seeding addition and the application of a commercial seed mixture (dose of 30 g/m^2; Table 2). The seeding + mulch level consisted of stabilizer, wood fiber mulch, water and the seed mixture.

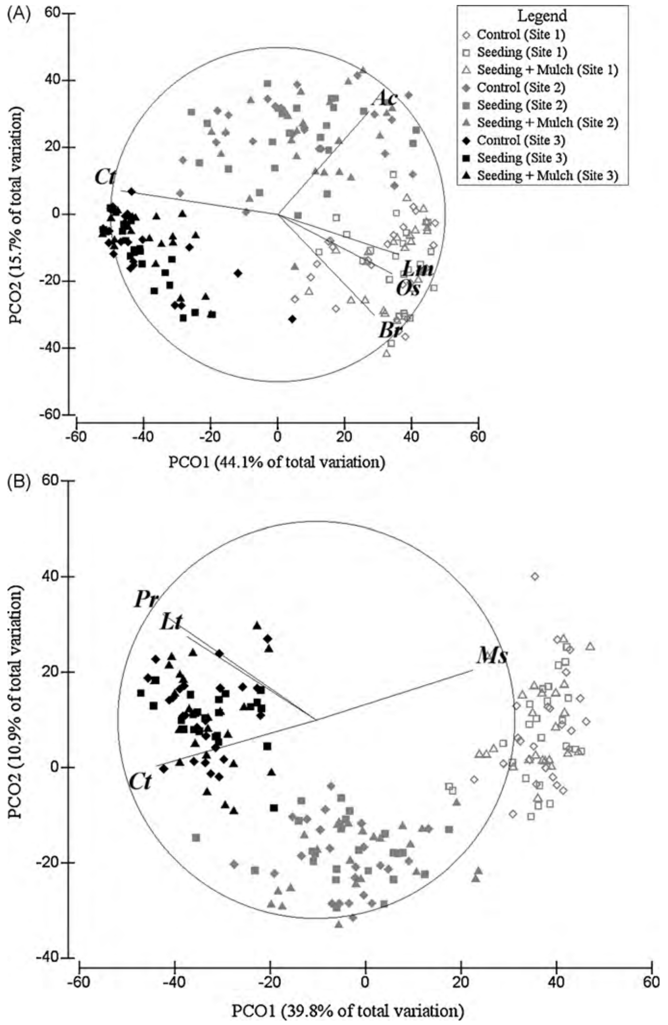

Fig. 3. Principal coordinate analysis of the species characterizing the sampling plots in the R4 sites, showing the effects of hydroseeding and site on community composition in 2007 (A) and 2008 (B). The effects of fertilization and irrigation were not significant, and we pooled these treatments to simplify the visualization of the results. The control and seeding levels correspond to no seeding addition and the application of a commercial seed mixture (dose of 30 g/m^2; Table 2). The seeding + mulch level consisted of stabilizer, wood fiber mulch, water and the seed mixture. Vectors represent those species with Spearman correlations coefficients greater than 0.7. Ac = *Anacyclus clavatus*, Br = *Bromus rubens*, Ct = *Carduus tenuiflorus*, Lm = *Lolium multiflorum*, Lt = *Leontodon taraxacoides*, Ms = *Medicago sativa*, Os = *Ononis spinosa* L. and Pr = *Papaver rhoeas*. Percentage of variance explained by each axis is shown in brackets. See Table A.4 for the statistical analysis of these data.

species were negatively correlated with plant diversity in five of the six situations evaluated (3 sites × 2 years). However, in the AP36 sites we found the opposite pattern, with positive correlations in three of the four situations studied (2 sites × 2 years). The identity of the dominant species changed between sites. The sign of the relationship between the cover of the dominant species and species diversity was independent from the mean cover of the former, because all of them represented at least 20% of total site cover.

4. Discussion

Plant community responses to the treatments evaluated were site-specific. We found several significant site effects on plant cover, diversity and community composition. Our results indicate that the development of plant communities after the construction of motorways depends more on biotic variables, such as the identity of neighbors in early life stages (Luzuriaga and Escudero, 2008), than on the resources added by restoration treatments. Several fast-growing dominant species, some hydroseeded and some already present in the study sites, were responsible for this idiosyncratic plant community variation between sites (Figs. 3 and 4). The species that modulated community composition shifts were also dominant in each site in terms of total cover (Table A.2). Treatments promoting an increase in resource availability (nutrients and water) had little effects when any fast-growing species already present in the field can potentially dominate the community (Baer et al., 2004).

The short-term restoration goal of increasing total plant cover, and hence reducing soil erosion, was achieved in the studied motorway slopes, although the restoration treatments employed did not contribute to this result. Hydroseeding was able to increase total plant cover by 10 and 16% in the R4 and AP36 sites, respectively, but did not affect our surrogates of soil erosion (Andrés et al., 1996). The mean plant cover during the period when erosion was measured was higher than 50% in all the sites, which has been established by Andrés and Jorba (2000) as a minimum to prevent erosion. Therefore, no treatments are needed to effectively control soil erosion when plant cover is higher than 50% in semiarid Mediterranean embankments. Although no overall erosion was found in our study slopes, irrigation contributed to reduce it in one of the AP36 sites (Fig. 2A). Since no significant effect was found on plant cover in this site, irrigation could be promoting soil erosion control through an indirect effect on soil structure. Previous studies demonstrated that an increase in water availability enhance aggregate stability (Lavee et al., 1996) and promote root biomass (Maestre and

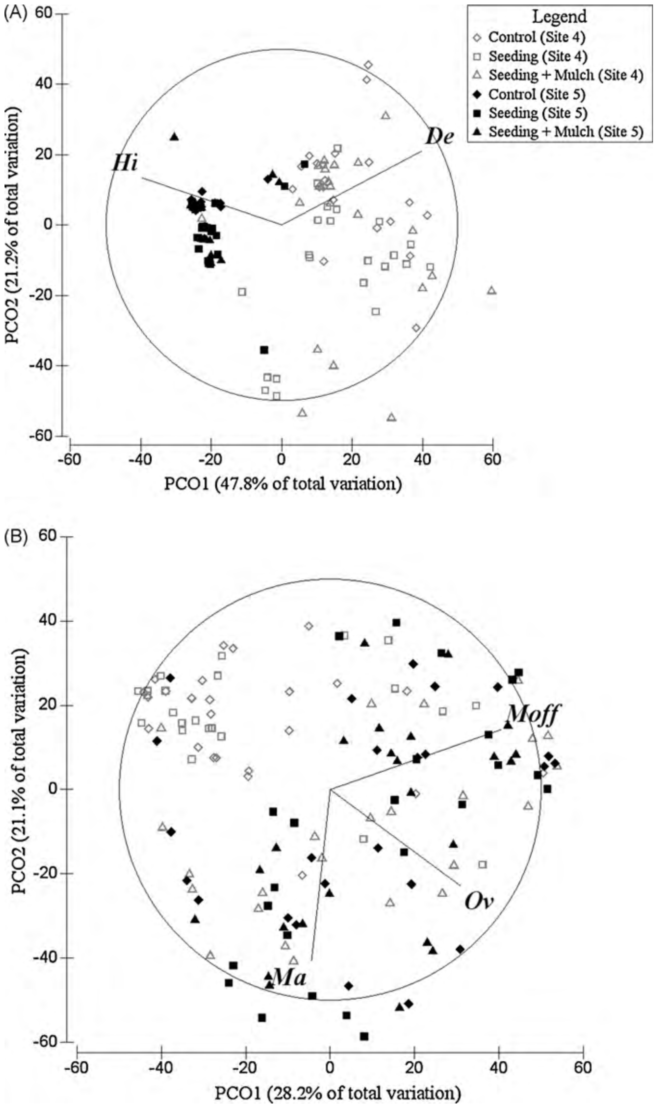

Fig. 4. Principal coordinate analysis of the species characterizing the sampling plots in the AP36 sites, showing the effects of hydroseeding and site on community composition in 2007 (A) and 2008 (B). The effects of fertilization and irrigation, both significant, were pooled to simplify the visualization of the results. The control and seeding levels correspond to no seeding addition and the application of a commercial seed mixture (dose of 30 g/m^2; Table 2). The seeding + mulch level consisted of stabilizer, wood fiber mulch, water and the seed mixture. Vectors represent the species with Spearman correlations coefficients greater than 0.7. De = *Diplotaxis erucoides*, Hi = *Hirschfeldia incana*, Ma = *Moricandia arvensis*, Moff = *Melilotus officinalis* and Ov = *Onobrychis viciifolia*. Percentage of variance explained by each axis is shown in brackets. See Table A.4 for the statistical analysis of these data.

Reynolds, 2007), therefore improving soil structure via biotic and abiotic mechanisms.

The effectiveness of the treatments evaluated to promote community composition shifts differed between the two motorways studied. In the R4 sites, hydroseeding did not produce any compositional change (as also found by Tormo et al., 2007), but was able to promote a shift in the AP36 sites in both 2007 and 2008, but not towards a more diverse plant community (Table 4; Martínez-Ruiz et al., 2007). This effect seems to be driven by an increase in the cover of hydroseeded species, most of them exotics, such as *Lolium multiflorum* in 2007 and *M. officinalis* and *M. arvensis* in 2008 ($F_{2,20} = 18.70$, $p = 0.001$; $F_{2,20} = 18.40$, $p = 0.001$ and $F_{2,20} = 7.75$, $p = 0.019$, respectively). On the other hand, fertilization also affected diversity, but to a lesser extent. This treatment directly increased the community diversity (Elmarsdottir et al., 2003) at one of the AP36 sites (site 4), but partially decreased this variable in all the R4 sites (Rajaniemi, 2002). The unexpected absence of any irrigation effect (excepting 10% increase in plant cover in the R4 sites found in 2008) could be due to the high rainfall found during the irrigation period (200 mm from March to June 2008). Such rainfall maintained soil moisture content at levels high enough for plant requirements in both irrigated and non-irrigated plots (Fig. A.1).

The patterns found when evaluating the relationships between the most dominant species and community diversity were the opposite for the R4 and AP36 sites (Table 5), highlighting the existence of local effects that determine vegetation dynamics between the two motorways (Moreno-de las Heras et al., 2008). Because of the nature of the studied slopes (Table 1), we cannot statistically separate construction age from soil type effects. However, we believe that the different ages of each motorway may sustain in a large extent these differences in the relations between the most dominant species and community diversity, and the former in the vegetation responses to the treatments evaluated. Several features of the studied slopes suggest that ecosystem function and structure may not be as different as the differences in their parental soil type may indicate. First of all, as commented before in the text, the embankment soils are artificially constructed using substrates from multiple sources. Therefore, the stressful physical and chemical properties that gypsum materials impose for plants (Palacio et al., 2007) may not be so relevant in our study sites. As recently shown (Romao and Escudero, 2005), soil physical surface hardness is the most relevant constraint of these soils under semiarid conditions, and its importance on steep slopes on our embankments is irrelevant (Escudero, pers. comm.). Moreover, N, P and water holding capacity (indicators of soil organic matter; Hudson, 1994) are higher in the gypsum sites (Table 1). Both substrates held a plant community dominated by annual herbs with similar plant cover before the addition of treatments. In addition, gypsum and calcareous sites shared 65% of the species along the 2 years of the study (Table A.2). The positive effect of dominant species on community diversity found in the recently built AP36 slopes is likely due to higher niche availability at these sites, which may prevent competition between species (Tilman, 2004). In this situation, if dominant species, as *Melilotus officinalis* (included in the hydroseeding), perform as starters, they could enhance the retention of resources like water, nutrients or seeds through an increase in plant cover, preparing the way for successive groups of species (Wali, 1999). This effect should be more pronounced in poor soils like those found in motorway slopes. Here, fast-growing species could provide high quality litter (i.e., low C:N) that stimulates bacterial-dominated food webs linked with fast nutrient cycling (Bardgett, 2005), promoting community diversity. However, in the 3-year-old R4 slopes, plant cover increases and the effect of the dominant species over diversity becomes negative, with a likely intense competition occurring for resources like space and light (Martínez-Alonso and Valladares, 2002). In this case, the pre-existing species *Anacyclus clavatus*, *Bromus rubens* and *Carduus tenuiflorus* are responsible of this negative relationship with diversity.

The negative effect of dominant species, hydroseeded or not, over community diversity may be enhanced during the course of succession because some species become an important part of the seed bank (González-Alday et al., 2009), which may cause long-term problems. This situation gets worse when distance to natural communities prevents from colonization (Ash et al., 1994), as is the case of the motorway slopes studied (García-Palacios, pers. observ.). Nevertheless, further studies are needed to assess the long-term dynamics of fast-growing herbaceous species and native recruited species and their effects on the ecosystem functioning and resilience of the restored roadside slopes.

5. Conclusions

Grasslands dominated by annuals are a common stage in human-disturbed environments in which a few species perform as invasives and dominate over potentially perennial communities (Seabloom et al., 2003). These herbaceous communities, typical of ecosystems such as motorway slopes, are extraordinarily dynamic, with rapid structural and compositional changes taking place from one year to another (Wali, 1999). Our results indicate that, in semiarid Mediterranean environments, this variation is driven by a pool of dominant species that turns plant community responses to restoration treatments site and context dependent. Evident differences can be found between slope ages, but also between sites closely located with a common management history. Our results differ from those from Rentch et al. (2005) in roadside plant communities of USA. They concluded that standard techniques could be implemented in different sites. However, the main management implication of our study is that site differences mediated by dominant species can mask the effectiveness of these restoration treatments. Hydroseeding did not have a strong impact on vegetation cover and erosion rates and some of the species included can even dominate the community after the restoration works. Therefore, hydroseeding can be considered of little value to restore semiarid roadside slopes. In order to be successful in semiarid Mediterranean environments, different approaches should be taken into account depending on the characteristics of the target site. On embankments, where plant cover can easily reach values high enough to prevent erosion, the use of non-native herbs that potentially can dominate the community should be avoided. These fast-growing species, although effective as starters, can constrain vegetation dynamics (Moreno-de las Heras et al., 2008). In these situations, autochthonous seed mixtures can be an alternative to enhance natural colonization (Prach and Pyšec, 2001).

Acknowledgements

Dr. Matthew Bowker provided useful comments and English corrections in previous versions of this manuscript We thank Patricia Izquierdo, Cristina Alcalá, Enrique Pigem, Marta Carpio and María D. Puche for their help in the laboratory and field work, two anonymous reviewers for comments and suggestions on a previous version of the manuscript, and Jesús Álvarez and all the CINTRA personnel for the facilities and support given during all the stages of this research. PGP and SS were supported by PhD fellowships from Proyecto Expertal, funded by Fundación Biodiversidad and CINTRA. APC was supported by a PhD fellowship from the INTERCAMBIO (BIOCON06/105) project, funded by Fundación BBVA. FTM was supported by a Ramón y Cajal contract from the Spanish Ministerio de Ciencia e Innovación (co-funded by the European Social Fund), and by the British Ecological Society (ECPG 231/607 and Studentship 231/1975). This research was supported by the EXPERTAL and REMEDINAL (S0505/AMB/0335) projects, funded by Fundación Biodiversidad-Cintra S.A. and the Comunidad de Madrid, respectively.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecoleng.2010.06.005.

References

Albaladejo, J., Álvarez, J., Querejeta, J., Díaz, E., Castillo, V., 2000. Three hydro-seeding revegetation techniques for soil erosion control on anthropic steep slopes. Land Degrad. Dev. 11, 315–325.

Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. Aust. Ecol. 26, 32–46.

Anderson, M.J., 2003. PCO: a FORTRAN computer program for principal coordinate analysis. Department of Statistics, University of Auckland, New Zealand.

Anderson, M.J., Willis, T.J., 2003. Canonical analysis of principal coordinates: a useful method of constrained ordination for ecology. Ecology 84, 511–525.

Andrés, P., Jorba, M., 2000. Mitigation strategies in some motorway embankments (Catalonia, Spain). Rest. Ecol. 8, 268–275.

Andrés, P., Zapater, V., Pamplona, M., 1996. Stabilization of motorway slopes with herbaceous cover, Catalonia, Spain. Rest. Ecol. 4, 51–60.

Armas, C., Pugnaire, F.I., Ordiales, R., 2004. Measuring plant interactions: a new comparative index. Ecology 85, 2682–2686.

Ash, H., Gemmell, R.P., Bradshaw, A.D., 1994. The introduction of native plant species on industrial waste heaps a test of immigration and other factors affecting primary succession. J. Appl. Ecol. 31, 74–84.

Baer, S.G., Blair, J.M., Collins, S.L., Knapp, A.K., 2004. Plant community responses to resource availability and heterogeneity during restoration. Oecologia 139, 617–629.

Bardgett, R., 2005. The Biology of Soils: A Community and Ecosystem Approach. Oxford University Press.

Blondel, J., Aronson, J., 1995. Biodiversity and ecosystem function in the Mediterranean Basin: human and non-human determinants. In: in: Davis, W., Richardson, D.M. (Eds.), Mediterranean-type Ecosystems: The Function of Biodiversity. Ecological Studies, vol. 109. Springer-Verlag, Berlin, pp. 43–119.

Bochet, E., García-Fayos, P., 2004. Factors controlling vegetation establishment and water erosion on motorway slopes in Valencia, Spain. Rest. Ecol. 12, 166–174.

Bradshaw, A.D., Huttl, R.F., 2001. Future minesite restoration involves a broader approach. Ecol. Eng. 17, 87–90.

Briggs, D., Giordano, A., 1992. CORINE Soil Erosion Risk and Important Land Resources in the Southern Regions of the European Community. Commission of the European Communities Publication EUR 13233 EN.

Cahill, J.F., 1999. Fertilization effects on interactions between above- and belowground competition in an old field. Ecology 80, 466–480.

Cape, J.N., Tang, Y.S., van Dijk, N., Love, L., Sutton, M.A., Palmer, S.C.F., 2004. Concentrations of ammonia and nitrogen dioxide at roadside verges, and their contribution to nitrogen deposition. Environ. Pollut. 132, 469–478.

Elmarsdottir, A., Aradottir, A.L., Trlica, M.J., 2003. Microsite availability and establishment of native species on degraded and reclaimed sites. J. Appl. Ecol. 40, 815–823.

Forman, R.T.T., Sperling, D., Bissonette, J.A., Clevenger, A.P., Cutshall, C.D., Dale, V.H., Fahrig, L., France, R., Goldman, C.R., Heanuem, K., Jones, J.A., Swanson, F.J., Turrentine, T., Winter, T.C., 2003. Road Ecology: Science and Solutions. Island Press, Washington (DC).

Fransen, B., de Kroon, H., Berendse, F., 1998. Root morphological plasticity and nutrient acquisition of perennial grass species from habitats of different nutrient availability. Oecologia 115, 351–358.

García-Fayos, P., García-Ventoso, P., Cerda, A., 2000. Limitations to plant establishment on eroded slopes in southeastern Spain. J. Veg. Sci. 11, 77–86.

González-Alday, J., Marrs, R.H., Martínez-Ruiz, M., 2009. Soil seed bank formation during early revegetation after hydroseeding in reclaimed coal wastes. Ecol. Eng. 35, 1062–1069.

Hobbs, R.J., Arico, S., Aronson, J., Baron, J.S., Bridgewater, P., Cramer, V.A., et al., 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global Ecol. Biogeogr. 15, 1–7.

Holl, K.D., 2002. Long-term vegetation recovery on reclaimed coal surface mines in the eastern USA. J. Appl. Ecol. 39, 960–970.

Holmes, P.M., 2001. Shrubland restoration following woody alien invasion and mining: effects of topsoil depth, seed source, and fertilizer addition. Rest. Ecol. 9, 71–84.

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Setälä, H., Symstad, A.J., Vandermeer, J., Wardle, D.A., 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecol. Monogr. 75, 3–35.

Hudson, B.D., 1994. Soil organic matter and available water capacity. J. Soil Water Conserv. 49 (2), 189–194.

Jackson, S.T., Hobbs, R.J., 2009. Ecological restoration in the light of ecological history. Science 325, 567–569.

Lavee, H., Sarah, P., Imeson, A., 1996. Aggregate stability dynamics is affected by soil temperature and moisture regimes. Geografiska Annaler 78 (1), 73–82.

Lepš, J., Šmilauer, P., 2003. Multivariate Analysis of Ecological Data Using CANOCO. Cambridge University Press, Cambridge.

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J.P., Hector, A., Hooper, D.U., Huston, M.U., Raffaelli, D., Schmid, B., Tilman, D., Wardle, D.W., 2001. Biodiversity and ecosystem functioning: current knowledge and future challenges. Science 294, 804–808.

Luzuriaga, A.L., Escudero, A., 2008. What determines emergence and net recruitment in an early succession plant community? Disentangling biotic and abiotic effects. J. Veg. Sci. 19, 445–456.

Maestre, F.T., Reynolds, J.F., 2007. Amount or pattern? Grassland responses to the heterogeneity and availability of two key resources. Ecology 88, 501–511.

Martínez-Alonso, C., Valladares, F., 2002. La pendiente y el tipo de talud alteran la relación entre la riqueza de especies y la cobertura de las comunidades herbáceas. Ecología 16, 59–71.

Martínez-Ruiz, C., Fernández-Santos, B., Putwain, P., Fernández-Gómez, J., 2007. Natural and man-induced revegetation on mining wastes: changes in the floristic composition during early succession. Ecol. Eng. 30, 286–294.
Matesanz, S., Valladares, F., Tena, D., Costa-Tenorio, M., Bote, D., 2006. Early dynamics of plant communities on revegetated motorway slopes from southern Spain: Is hydroseeding always needed? Rest. Ecol. 14, 297–307.
Méndez, M., García, D., Maestre, F.T., Escudero, A., 2008. More ecology is needed to restore mediterranean ecosystem. A reply to Valladares and Gianoli. Rest. Ecol. 16, 210–216.
Moreno-de las Heras, M., Nicolau, J.M., Espigares, T., 2008. Vegetation succession in reclaimed coal-mining slopes in a Mediterranean-dry environment. Ecol. Eng. 34, 168–178.
Naveh, Z., Dan, J., 1973. The Human Degradation of Mediterranean Landscapes in Israel. In: di Castri, F., Mooney, H.A. (Eds.), Mediterranean-type Ecosystems: Origin and Structure. Springer, Berlin, pp. 373–390.
Norris, J.E., Stokes, A., Mickovski, S.B., Cammeraat, E., van Beek, R., Nicoll, B.C., Achim, A., 2008. Slope Stability and Erosion Control: Ecotechnological Solutions. Springer, Dordrecht, The Netherlands.
Paschke, F.M.W., deLeo, C., Redente, E.F., 2000. Revegetation roadcut slopes in Mesa Verde National Park, USA. Rest. Ecol. 8, 276–282.
Parkin, T.B., Doran, J.W., Franco-Vizcaíno, E., 1996. Field and laboratory tests of soil respiration. In: Doran, J.W., Jones, J.W. (Eds.), Methods for Assessing Soil Quality. Soil Sci. Soc. Am. Spec. Publ., Madison, USA, pp. 231–246.
Palacio, S., Escudero, A., Montserrat-Martí, G., Maestro, M., Milla, R., Albert, M.J., 2007. Plants living on gypsum: beyond the specialist model. Ann. Bot. 99, 333–343.
Pausas, J.G., 2004. Changes in fire and climate in the eastern Iberian Peninsula (Mediterranean Basin). Climatic Change 63, 337–350.
Petersen, S.L., Roundy, B.A., Bryant, R.M., 2004. Revegetation methods for high-elevation roadsides at Bryce Canyon National Park, Utah. Rest. Ecol. 12, 248–257.
Prach, K., Pyšec, P., 2001. Using spontaneous succession for restoration of human-disturbed habitats. Experience from Central Europe. Ecol. Eng. 17, 55–62.
Pywell, R.F., Bullock, J.M., Hopkins, A., Walker, K.J., Sparks, T.H., Burke, M.J.W., Peel, S., 2002. Restoration of species-rich grassland on arable land: assessing the limiting processes using a multi-site experiment. J. Appl. Ecol. 39, 294–309.
Rajaniemi, T.J., 2002. Why does fertilization reduce plant species diversity? Testing three competition-based hypotheses. J. Ecol. 90, 316–324.
Rentch, J.S., Fortney, R.H., Stephenson, S.L., Adams, H.S., Grafton, W.N., Anderson, J.T., 2005. Vegetation-site relationships of roadside plant communities in West Virginia, USA. J. Appl. Ecol. 42, 129–138.
Romao, R., Escudero, A., 2005. Gypsum physical soil surface crusts and the existence of gypsophytes in semi-arid central Spain. Plant Ecol. 181, 1–11.
Seabloom, E., Harpole, W.S., Reichman, O.J., Tilman, D., 2003. Invasion, competitive dominance, and resource use by exotic and native California grassland species. Proc. Natl. Acad. Sci. 100, 13384–13389.
SER, 2004. BT The SER international primer on ecological restoration. www.ser.org & Tucson: Society for Ecological Restoration International. http://www.ser.org/pdf/primer3.pdf.
Shannon, C.E, Weaver, W., 1963. The Mathematical Theory of Communication. University of Illinois Press, Urbana, Illinois.
Sheldon, J.C., Bradshaw, A.D., 1997. The development of a hydraulic seeding technique for unstable sand slopes. I. Effects of fertilizers, mulches and stabilizers. J. Appl. Ecol. 14, 905–918.
Smith, R.S., Shiel, R.S., Bardgett, R.D., Millward, D., Corkhill, P., Rolph, G., Hobbs, P.J., Peacock, S., 2003. Soil microbial community, fertility, vegetation and diversity as targets in the restoration management of a meadow grassland. J. Appl. Ecol. 40, 51–64.
Tilman, D., 2004. Niche tradeoffs, neutrality, and community structure: a stochastic theory of resource competition, invasion, and community assembly. Proc. Natl. Acad. Sci. 101, 10854–10861.
Tormo, J., Bochet, E., García-Fayos, P., 2007. Roadfill revegetation in semi-arid Mediterranean environments. Part II: topsoiling, species selection, and hydroseeding. Rest. Ecol 15, 97–102.
Valladares, F., Tena, D., Matesanz, S., Bochet, E., Balaguer, L., Costa-Tenorio, M., Tormo, J., García-Fayos, P., 2008. Functional traits and phylogeny: what is the main ecological process determining species assemblage in roadside plant communities? J. Veg. Sci. 19, 381–392.
Wali, M.K., 1999. Ecological succession and the rehabilitation of disturbed terrestrial ecosystems. Plant Soil 213, 195–220.

Pablo García-Palacios is an ecosystem ecologist interested in the links between composition and structure of vegetation and soil biota and their effects on the ecosystem functioning. He is actually working in the restoration of the plant communities characterizing roadside slopes, with special focus on the mechanisms that controls secondary succession.

Santiago Soliveres is a plant ecologist interested on how plant–plant interactions changes along environmental abiotic and biotic stress gradients. He is also interested in quarry and motorway slopes restoration where he has tested the effects of different levels of soil resources in the plant community composition and structure, the relative abundance of introduced species and the equilibria between woody and herbaceous plants.

Fernando T. Maestre is an ecosystem ecologist with a strong emphasis on quantitative statistical methods and on applied research. Albeit not narrowed to a particular topic, his work is mostly devoted to understanding how semiarid ecosystem works, and how they are responding to the ongoing global change. His research uses a wide variety of tools (field observations and experiments, laboratory work and modeling) and biotic communities (vascular plants, biological soil crusts and soil microorganisms) and is being carried out at multiple scales, from single-site studies in semiarid ecosystems of Spain to large-scale field studies with sites located all over the world.

Adrian Escudero is a plant ecologist interested in the plants living on extreme environments, especially in the high Mediterranean mountain and in arid and semiarid systems. He was worked in several aspects related with conservation biology and restoration ecology in these stressful ecosystems.

Andrea Castillo-Monroy is a Ph D student, who works with processes as soil respiration, nitrogen availability, nutrient cycling and litter decomposition, and how these processes modulate the functioning of semiarid Mediterranean ecosystems, using the biological soil crusts as an experimental model.

Fernando Valladares is a plant ecologist working in the integration of ecophysiological research with plant demography and fitness under global change scenarios. He is especially interested in the evaluation of the ecological and evolutionary significance of phenotypic plasticity, and in the application of ecological theory into the restoration of degraded ecosystems.