available at www.sciencedirect.com

ScienceDirect

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

Vegetation succession in reclaimed coal-mining slopes in a Mediterranean-dry environment

M. Moreno-de las Heras, J.M. Nicolau, T. Espigares*

Departamento de Ecología, Edificio de Ciencias, Universidad de Alcalá, Campus externo de Alcalá, Ctra. Madrid-Barcelona km 33.600, Alcalá de Henares, E-28871 Madrid, Spain

ARTICLE INFO

Article history:
Received 27 February 2008
Received in revised form
16 July 2008
Accepted 16 July 2008

Keywords:
Drylands
Inhibition
Local disturbances
Restoration
Soil erosion
Teruel coalfield

ABSTRACT

Mining reclamation results obtained in the Teruel coalfield (Mediterranean-dry Spain) during the last 30 years have been quite limited. In order to improve restoration operations we conducted a study to analyse the trajectories of ecological succession and identify the main driving forces that control vegetation dynamic in reclaimed artificial slopes. A total of 87 slopes of different ages and restoration treatments were classified and characterized after recording different variables related to topography, restoration techniques, vegetation, local disturbances and soil erosion. Successional trends were inferred from gradient analysis as well as the factors, mechanisms and processes implied. We found a wide variety of plant communities and successional trajectories. Initial conditions (soil quality and revegetation treatments) as well as the environmental scenario (climatic conditions and vicinity of preserved propagule sources) were the main driving forces directing vegetation succession. Soil erosion triggered by external run-on coming from surrounding structures was also identified as a key factor determining the evolution of vegetation in these dry environments. Other local disturbances (grazing and fungal pests) can favour vegetation transition in communities dominated by highly competitive non-native sown species to more diverse shrub communities. Some practical considerations for future reclamation projects are suggested.

1. Introduction

Failures in reclamation have been common in spite of the significant development of mining reclamation techniques during the last decades (Haigh, 2000); consequently, there are still poorly understood aspects (Plass, 2000). A deeper knowledge of the driving forces of community succession in reclaimed areas would improve their evolution towards ecosystems that contribute to provide territorial stability and facilitate the regeneration of fundamental ecological processes (Sänger and Jetschke, 2004).

In temperate areas, favourable conditions for spontaneous succession bring high potential for using it in reclamation processes (Prach and Pysek, 2001; Pietrzykowski and Krzaklewski, 2007). In these areas the main factors that control ecological succession are the following: regional meso-climatic differences, landscape factors related to the presence of preserved nearby vegetation, and local factors related to nutrient cycling and physico-chemical soil characteristics (Wiegleb and Felinks, 2001; Novak and Konvicka, 2006; Prach et al., 2007). Often, severe soil deficiencies and toxicity are the most relevant constraints (Bradshaw, 1997).

In reclaimed areas under dry climates, other factors related to water shortages and soil erosion may also affect vegetation dynamics (Whisenant, 2002; Martinez-Ruiz et al., 2007). Overland flow can be a driving force in these cases, since it

* *Corresponding author*. Tel.: +34 918856407; fax: +34 918854929.
E-mail address: mariano.moreno@uah.es (M. Moreno-de las Heras).
0925-8574/$ – see front matter © 2008 Elsevier B.V. All rights reserved.
doi:10.1016/j.ecoleng.2008.07.017

redistributes soil particles (erosion and sedimentation) and water (runoff and soil moisture) at the slope scale (Lavee et al., 1998; Puigdefábregas et al., 1999). In man-made slopes, the incipient soil development favours overland flow run-off and limits rainfall infiltration (Ward et al., 1983; Guebert and Gardner, 2001). In some cases, extra overland flow runs (as run-on) into reclaimed slopes coming from the top and can promote soil erosion, with dramatic consequences for plant dynamics (Moreno-de las Heras et al., 2005).

Regardless of the particular environmental context, ecological succession occurring in restored environments is subjected to a wide range of contingencies, reducing the predictability of succession (Parker and Pickett, 1997). Some contingencies result particularly important in reclaimed mining ecosystems: mistakes in the execution of reclamation, changes in legislation, post-mining land uses, and the impact of local disturbances unconsidered during reclamation management (Nicolau and Moreno-de las Heras, 2005).

Surface mining in the Teruel coalfield (Mediterranean-dry Spain) started in 1976. During the last 30 years, 24 mines have been opened, affecting around 3000 hectares. During these three decades, reclamation practices have evolved as a consequence of legal requirements. Nevertheless, reclamation outcomes have been rather limited in most mines, and demand research in order to develop a successful restoration protocol (Mellado, 2006).

We studied the pattern and factors controlling vegetation succession in artificial slopes derived from opencast mining activities in the Teruel coalfield. Taking into account the presence of an environmental gradient related to climate continentality and landscape conservation, and the great heterogeneity of reclamation treatments in our study area, we expected a complex scenario expressed by different successional pathways in built slopes through time (about 20 years). We also expected to find that hydrologic dysfunction (mainly caused by erosion processes) and contingencies play a relevant role in vegetation dynamics, given the strong influence of macro-climate and the contingent nature of mining landscapes. The specific questions we wanted to answer are the following:

1. What is the relative importance of the environmental context and initial conditions on vegetation dynamics?
2. Can soil erosion be highlighted as a significant driving force for succession in these artificial slopes?
3. What is the role of contingences in the regional succesional pattern?

2. Materials and methods

2.1. *Study area*

This study was carried out in the Teruel coalfield (4900 km^2; Fig. 1), central-eastern Spain. The climate is Mediterranean. The regional moisture regime, classified as Mediterranean-dry (sensu Papadakis, 1966), is characterized by the concentration of rainy episodes in spring and autumn, and a period of summer drought.

A general environmental gradient related to meso-climatic differences and landscape conservation was defined according to geographic location within the study area:

1. In the eastern third, altitude is about 600–700 m.a.s.l. Climate continentality in this area is attenuated by maritime influence: mean annual temperature is 12.9–13.3 °C, and

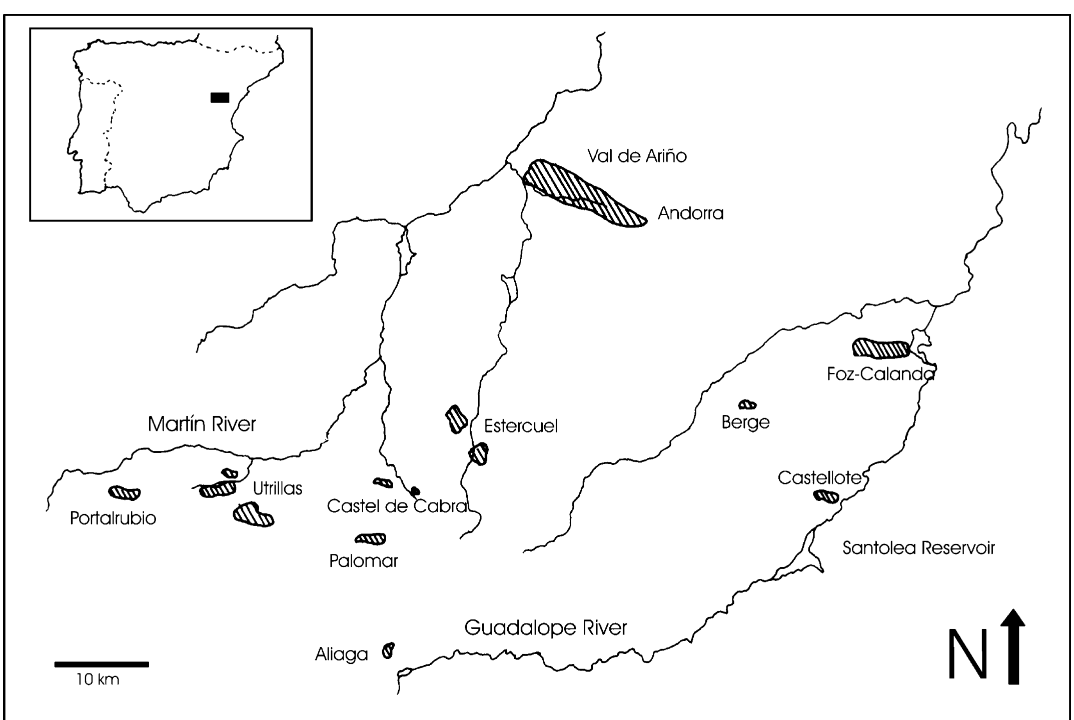

Fig. 1 – Geographic location of opencast coal mining in the Teruel coalfield. Stripes indicate areas affected by opencast coal mining.

annual thermal amplitude (difference between the annual absolute max. and min. temperature) is 43.2–45.8 °C. Mean annual precipitation and potential evapotranspiration are 413–478 and 727–740 mm, respectively. Vegetation surrounding the areas affected by opencast coal mining is better preserved. It is formed by a mosaic of preserved forest patches (mainly dominated by *Pinus halepensis* and *Quercus ilex*) and abandoned terraces vegetated by shrubs (*Rosmarinus officinalis*, *Genista scorpius*, *Pistacea lentiscus*, *Thymus vulgaris* and *Dorycnium pentaphyllum*).

2. Altitude in the rest of the study area is about 700–1000 m.a.s.l. Climate is more continental: mean annual temperature is 11.0–14.0 °C, and annual thermal amplitude is between 44.7–50.4 °C. Mean annual precipitation and potential evapotranspiration are 466–480 and 758–769 mm, respectively. Vegetation surrounding the mines is more degraded, mainly because of overgrazing. Patches of natural forests are very scarce. Most of the landscape is covered by sparse shrub communities (dominated by *G. scorpius*) on abandoned terraces and cereal crops.

Regional natural soils (unaffected by mining) range from *Typic* or *Lithic Xerorthent* to *Calcic Xerochrept* (sensu Soil Survey Staff, 1998), with a low content of organic matter (<3%) and basic pH (Arranz, 2004).

2.2. Reclamation operations

Reclamation practices in the Teruel coalfield have progressed considerably in the last decades. Landform design has evolved from the oldest restored landscapes based on platforms, banks and ditches to catchments structured by gentle slopes and watercourses (Nicolau, 2003). Soil management practices tend to favour the use of topsoil instead of overburden materials to cover the new forms. Nevertheless, in cases where revegetation is undertaken, a general mixture of non-native herbaceous grasses and legumes is sown, and only in a few cases this is followed by a further plantation of shrubs and/or tree species (see Appendix A for species introduced by revegetation operations).

2.3. Field sampling

Eighty-seven artificial slopes of different ages and reclamation treatments were sampled in two different field campaigns separated by a 15-year period (Table 1). During spring 1987 and 1988, information about 51 recently built slopes (0–6 years) was recorded. 36 older artificial slopes (10–20 years) were sampled in spring 2002 and 2003. Six slopes sampled in the second campaign coincide with slopes sampled in the first campaign (Table 1).

Common variables related to topography, such as steepness (°) and slope length (m) were measured. Likewise, the distance from natural seed sources – unaltered by mining – was recorded and transformed into a dummy variable (<200 and >800 m, as no slope between 200 and 800 m was sampled due to accessibility problems).

Each slope was divided into three 30 m cross-slope equidistant transects. Nine equidistant 0.25 m^2 plots were laid in each transect. For each plot, canopy cover (%) of each species was recorded. This sampling procedure for vegetation survey has been successfully tested in reclaimed mining slopes of Mediterranean-dry Spain, encompassing more than 90% of species (Martinez-Ruiz et al., 2007). Mean species composition per slope was calculated from data recorded at the plot scale and used for statistical analyses. The proportion of total vegetation cover (%), as well as two diversity indices (Shannon's index-H and Species richness-S), were calculated from mean slope species composition. In the same way, the relative abundance of species introduced by sowing (%) was calculated as the ratio between canopy cover of all sown species and total vegetation cover.

Three composite soil samples (each sample formed by six homogeneously mixed subsamples, randomly distributed in each parallel transect) were taken from the first 15 cm of the soil profile in each slope. Soil parameters were analysed following the standardized methods proposed by the Spanish Ministry of Agriculture (MAPA, 1994). Thus, stoniness (%) was determined as the content of soil particles >2 mm; textural particle size distributions were analysed following the Bouyoucos method and USDA classification; soil organic matter (%) and total nitrogen (%) were determined using the Walkley–Black and the Kjeldahl's methods, respectively; and soil pH was determined in a 1/2 (weight/volume) aqueous solution with a Crison® mod.2001 pH-meter. Similarly to vegetation data, mean soil properties for each slope were calculated from the three composite soil samples and used for statistical analyses.

Every rill section with depth >1 cm was counted and measured (width and depth) for each transect. Mean number of rills per transect in each slope was determined. Rill erosion rate was calculated from the rill network dimensions following the methodology described by Morgan (1995). Variations of rock fragment cover can reflect variations in past erosion

Table 1 – Location of sampled slopes

Localization	Sampled slopes		
	1987–1988 field campaign	2002–2003 field campaign	Coincident sampled slopes[a]
Utrillas	8	22	
Estercuel	12	6	2
Castellote	10	8	4
Palomar	10		
Val de Ariño	11		

[a] Slopes sampled during both field campaigns.

Table 2 – General description of the plant community groups derived from TWINSPAN analysis

Code	N	Restoration treatments	Environmental factors	Plant traits	Erosion traits	Characteristic species	p
CT1	18		↓pH	↓↓Vc ↓↓S ↓↓H	↑↑Re	*Polygonum aviculare* *Iberis* sp	0.007 0.019
CT2	11	Revegetated	Continental Inf ↓Age	↓Vc ↓S	↑Re	*Lolium perenne* *Onobrychis viciifolia*	0.025 0.042
CT3	10	Topsoiled	Maritime Inf ↓Age	↓Vc ↓S	↑Re	*Medicago minima* *Silene muscipula* *Raphanus raphanistrum* *Erodium cicutarium*	0.004 0.004 0.005 0.012
CT4	4	Topsoiled	Maritime Inf ↑Age	↑Vc ↑S ↑H		*Coronilla scorpioides* *Helicrysum stoechas* *Astragalus hamosus* *Artemisia campestris*	0.001 0.001 0.002 0.002
CT5	17	Revegetated	Continental Inf Run-on.	↓↓Vc ↓↓S ↓↓H ↑Sspp	↑↑Re ↑Se	*Medicago sativa*	0.010
CT6	14	Revegetated	Continental Inf ↑Age	↑Vc ↓S ↑Sspp		*Festuca arundinacea* *Sonchus asper* *Filago vulgaris* *Anacyclus clavatus*	0.001 0.006 0.010 0.012
CT7	6	Revegetated	Continental Inf Local disturbances ↑Age	↑Vc ↑S ↑H		*Thymus vulgaris* *Xeranthemum annuum* *Santolina chamaecyparissus* *Plantago lanceolata*	0.002 0.002 0.003 0.003
CT8	4	Topsoiled	Maritime Inf Closed to Nss ↑Age	↑Vc ↑S ↑H		*Rosmarinus officinalis* *Genista scorpius* *Dorycnium pentaphyllum* *Plantago albicans*	0.002 0.002 0.005 0.008

General abbreviations: *N*: number of representative slopes in each community group; *p*: Monte Carlo test significance for characteristic species assigned by the indicator species analysis.
Abbreviations for environmental factors: Inf: influence; Nss: natural seed sources (unaltered by mining).
Abbreviations for plant traits: Vc: total vegetation cover; S: species richness; H: Shannon's index; Sspp: relative sown species abundance.
Abbreviations for erosion traits: Re: rill erosion; Se: accumulated sheet erosion. ↑: high; ↑↑: very high; ↓: low; ↓↓: very low (based on the quantitative differences summarized in Appendix B).

rates, due to the accumulated loss of fine soil particles from the soil surface, as a consequence of sheet erosion (Poesen et al., 1998). In the same way, variations in rock fragment cover can also reflect spatial variations of rock fragment contents in the bulk soil. An accumulated sheet erosion index (ASEI), proportional to accumulated sheet erosion (soil particle loss due to sheet erosion since slope construction) and independent from the rock fragment content in soil was used. It was calculated as the quotient between mean rock fragment cover of the soil surface (calculated from rock fragment cover measured in 0.25 m^2 plots) and mean soil stoniness.

Information about restoration treatments (slope age, type of substrate, revegetation operations, and the presence of functional up-slope protective structures from run-on) was recorded as dummy variables by means of direct observation and the inspection of mining operation registers.

The occurrence of contingent local disturbances (grazing and fungal diseases) was also recorded as a dummy variable related to presence of sheep grazing signs (grazed vegetation, sheep wool and dung pellets), and fungal pests (fungal disease stains on leafs and micelle presence in stems and roots) on plot vegetation. Specifically, we found one species (*Medicago sativa*) highly affected by fungal attacks in some particular slopes.

2.4. *Data analysis*

To define the plant communities, slopes were grouped based on classification analysis and the identification of characteristic species (Velazquez and Gómez-Sal, 2007). TWINSPAN analysis (Hill, 1979) was performed to classify plant communities using a matrix with the slope plant composition. Three bare slopes from a total of 87 were excluded for analyses. Species present in less than 5% of slopes were eliminated to reduce erratic results (Sokal and Rohlf, 1995). Setup parameters of the classification analysis were: 3 as maximum number of indicators per division, 4 as maximum level of division and 10 as minimum group size for division. Indicator Species Analysis (ISA, Dufrene and Legendre, 1997) was used to establish the characteristic species of each group.

In order to characterize plant communities and check whether significant differences existed, non-parametric Kruskall–Wallis tests and post hoc Mann–Whitney *U* tests were performed. To control the rate of spurious significance of related tests sequential Bonferroni corrections (Rice, 1989) were used ($\alpha = 0.01$ for main tests and $\alpha = 0.05$ for post hoc tests).

The classified and characterised plant communities were reflected by a Detrented Correspondence Analysis (DCA, Hill and Gauch, 1980), performed with the same matrix, to define the underlying ecological structure (van Groenewoud, 1992). Relevant passive variables (environmental factors as well as plant and erosion traits) were projected in DCA configuration as vectors to interpret the gradients expressed by DCA axes. To avoid redundancy in gradient interpretation, passive variables fitted to DCA configuration were previously selected within significant variables for community characterization by cross-correlation (Vogiatzakis et al., 2003).

PC-ORD Version 4 (McCune and Mefford, 1999) was used for TWINSPAN and ISA. The Vegan package from the R system (Oksanen et al., 2007) was used for DCA and vector fitting. STATISTICA 6.0 (Statsoft, 2001) was used for the complementary statistical analyses.

3. Results

3.1. Plant community classification and characterization

Considering floristic composition (83 species appeared in more than 5% plots of the 84 vegetated slopes; Appendix A), and after the grouping of slopes according to the TWINSPAN analysis and their characteristic species (established using Indicator Species Analysis), eight different plant communities were identified (Table 2). These plant communities were related to both the general restoration treatments applied and environmental factors (meso-climate and landscape context influence), and were very different in terms of plant and erosion traits (Table 2; Appendix B).

A very sparse (almost bare) and poor community (CT1; Table 2) was identified on unreclaimed slopes, characterised by the presence of acid overburden substrates.

On the eastern mining sites (specifically the Castellote mining site), where climate is less continental and surrounding vegetation is better preserved, three different plant communities (CT3, CT4 and CT8; Table 2) were identified on unrevegetated slopes covered with topsoil from abandoned agricultural terraces. These three communities differed basically in vegetation cover, diversity and age; the oldest ones (CT4, CT8) were better covered and more diverse, and in the case of the CT8 community (slopes $\leq$200 m from natural seed sources), its composition was characterised by the presence of different shrub species (*G. scorpius*, *R. officinalis* and *D. pentaphyllum*).

Four different plant communities (CT2, CT5, CT6 and CT7; Table 2) were identified on slopes sown with herbaceous seed blends on the western and central mining sites, where climate is more continental and surrounding areas are degraded by overgrazing. With respect to substrate, these groups were fairly heterogeneous as both overburden and topsoil were used to cover the slopes where these communities appeared. These four groups differed on vegetation complexity and degradation through erosion. Soil erosion processes in these slopes were mainly triggered by the presence of active external sources of overland flow ($F(1; 84) = 40.41$; $p < 0.000$). In fact, a highly eroded community type (CT5) was found on slopes of different ages where up-slope protective structures from run-on (channels and berms which should be designed to isolate slopes from mining tracks and banks) were broken or absent. This community was characterised by a very low cover, vegetated by almost only one species, *M. sativa*, a sown perennial legume. Two less eroded herbaceous communities (CT2, CT6) were found, differing in age and vegetation cover. Soil surface was less covered by vegetation in the youngest group (CT2); the oldest one (CT6), better covered, was characterized by the massive presence of sown species (up to 70% of total cover), especially the legume *M. sativa*. Finally, a more diverse shrub community was identified (CT7) on old slopes with sings of sheep grazing on vegetation and/or fungal affection on *M. sativa* populations.

3.2. Gradient analysis: the underlying ecological processes

The identified plant communities were represented using DCA (Fig. 2), performed with the same data matrix as the classification analysis. The first correspondence axis (27.2% of explained variation) was positively linked with rill erosion and inversely with age and soil pH (Fig. 2). Likewise, the second axis (13.1% of explained variation) was positively linked with diversity and occurrence of local disturbances, and inversely linked with the distance from seed sources and relative sown species abundance (Fig. 2). Other factors (climate and conservation of surrounding landscape) may play a role in the gradient expressed by axis 2. Thus, CT3, CT4 and CT8 community type groups, which are formed by topsoiled and unsown slopes from the Castellote mining site in the eastern third of the studied area (lowest climatic continentality and surrounding vegetation better preserved), were distributed in DCA biplot along the second half of axis 2 (Fig. 2).

4. Discussion

4.1. Successional pattern and driving forces

A wide variety of plant communities and trajectories were identified in the Teruel coalfield (Fig. 3). This is expected in post-mining ecosystems, where vegetation dynamics can be considered a type of primary succession with low probability of vegetation convergence (Wiegleb and Felinks, 2001; Novak and Prach, 2003).

Based on gradient analysis and differences between structural and environmental features, three main pathways can be inferred (Fig. 3):

(a) On acid soils, plant establishment and community development are highly limited, directing the system towards very simple and poor states (CT1).

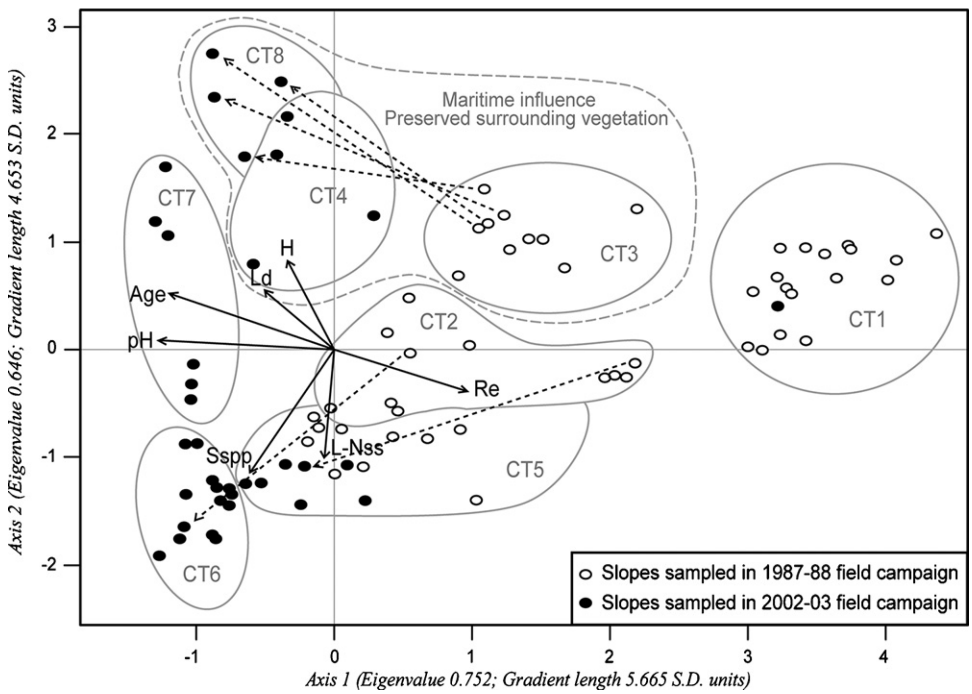

Fig. 2 – DCA biplot showing ordination of sampled slopes. Black solid vectors represent the strength and direction of significant correlations ($\alpha = 0.01$) between representative passive variables (environmental factors as well as plant and erosion traits) and ordination axes. Black dotted vectors represent the temporal transition of the six coincident sampled slopes. Solid gray borderlines group the community types derived from TWINSPAN analysis. The broken gray borderline groups slopes located in the Castellote mining site (eastern extreme of the environmental gradient), where climate is less continental and surrounding vegetation is better preserved. Abbreviations for passive variables: Re: rill erosion; H: Shannon's diversity index; Ld: local disturbances; Sspp: relative sown species abundance; L-Nss: length from seed sources.

(b) Where the environmental conditions are less restrictive (lower continentality, better soil quality and vicinity of preserved natural forests) the transition from the initial community (CT3) to more diverse ones (CT4 and CT8) is always observed. In this case, the main driving force is the proximity of propagule sources.

(c) Where the environmental conditions are more restrictive, other driving forces become more relevant. These forces are related to initial conditions (communities based on competitive non-native species) and the occurrence of some disturbances as rill erosion, mainly triggered by external overland flow, or grazing and fungal pests. In very rilled slopes only a sparse and simple community (CT5) can develop. Where extra overland flow does not run into slopes, initial sown communities (CT2) progress into a persistent herbaceous community (CT6) dominated by a non-native sown legume (*M. sativa*). In this case, grazing by sheep and/or fungal pests drives the system towards a more diverse community, dominated by shrub species (CT7).

4.2. *Environmental context and initial conditions*

In the Teruel coalfield area, the divergent successional pattern is primarily ruled by both the initial conditions of mining slopes and the environmental gradient associated to climate and conservation of surrounding vegetation. Similar results have been found in central European man-made habitats, where the initial conditions of altered areas (specifically soil pH) and meso-climatic differences drive successional patterns (Prach et al., 2007).

Regarding the chemical soil characteristics, the temporal successional progress was related to soil pH levels (Fig. 2). In the study area, plant immigration, vegetation establishment and species replacement on acid sites were highly constrained (Fig. 3). This is probably due to the absence of appropriate species in nearby propagule sources (which grow on basic soils) and occasional toxicity associated to extremely acid substrates. Analogous restrictions to natural recovery associated to soil acidity and toxicity have been frequently reported in mining sites (Bradshaw, 1997; Prach and Pysek, 2001). In the Teruel coalfield, this problem mainly affects derelict mines which operated before the regulation introduced by the first Spanish reclamation law in 1982 (Spanish Royal Decree, RD 2994/1982). Indeed, prior to the application of the cited regulation, no procedure of substrate selection to cover the altered surfaces was applied by local mining companies.

In the absence of serious restrictions caused by soil characteristics, climate and the presence of well-preserved surrounding vegetation were responsible for the different successional trends. In this way, all slopes located in the Castellote mining site (situated in the least continental area of the Teruel coalfield) were grouped towards the top of DCA axis 2. Such influences caused by meso-climatic differences

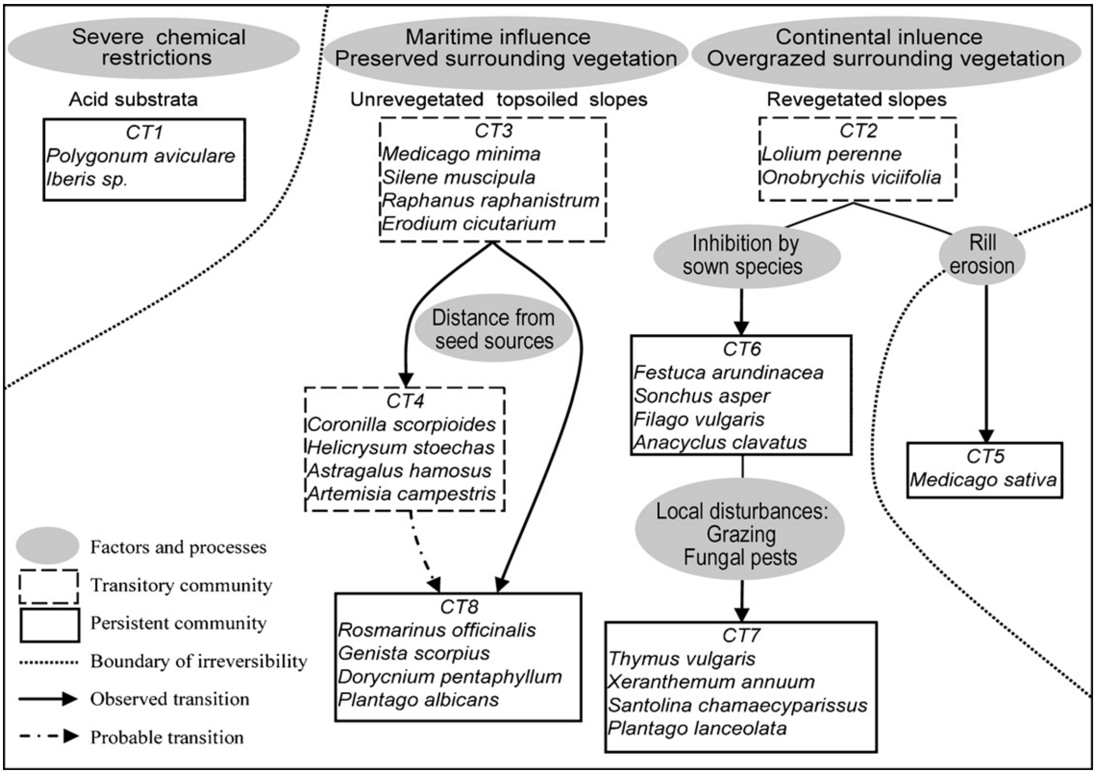

Fig. 3 – General scheme of succession in reclaimed artificial slopes in the Teruel coalfield. Displayed plant communities and characteristic species has been identified using TWINSPAN and species indicator analysis. Community transitions as well as the relevant factors and processes involved have been inferred from community characterization and gradient analysis.

have been generally attributed to variations in the species pool (Otto et al., 2006). Nevertheless, other factors could also play a role in successional differentiation and subsequent community transitions identified in this mining site. In fact, these slopes were reclaimed with good quality topsoil and were surrounded by preserved forest patches of *P. halepensis* and diverse shrub communities (*Rosmarino-Ericion multiflorae*; sensu Rivas-Matinez, 1987). This aspect is especially important for reclaimed mining sites, where community structure is usually limited by long distances from seed sources and reduced dispersal ability of species (Parrotta and Knowles, 2001; Wiegleb and Felinks, 2001; Novak and Konvicka, 2006). Indeed, the establishment of some shrub species characterised by limited barochorous seed dispersal (as *G. scorpius* and *D. pentaphyllum*) was only possible in slopes nearest to preserved patches, resulting in a more diverse community (CT8, Fig. 3).

In more continental areas, a strong effect caused by the use of mixtures of non-native fast growing herbaceous species appeared. This is supported by DCA axis 2, which confronts the relative abundance of sown species with species diversity. These herbaceous mixtures were generally used to simply “paint slopes green” and to comply with legal requirements for soil erosion and water quality control. In consequence, succession was arrested in most of these revegetated slopes, leading to a persistent herbaceous community (CT6) mainly dominated by *M. sativa*. Similarly, it has been generally mentioned that revegetation with fast growing herbaceous species can prevent long-term vegetation development due to competition with spontaneous colonizers (Holl, 2002). This result emphasizes the need to modify the composition of revegetation seed mixtures using native species; these mixtures should include some successful colonizing shrub species which may be restricted by the availability and distance of the natural propagule sources.

4.3. *The role of soil erosion*

Soil erosion must be considered as a significant process for restored vegetation dynamics in the studied area. This is shown by the opposite relationship with successional advance (Fig. 2). Accordingly, soil erosion, and particularly rill erosion, has been highlighted as a key limiting factor for vegetation establishment and succession in both natural and reclaimed slopes (Nicolau and Asensio, 2000; Wang et al., 2007). In the studied slopes, rill erosion is frequently triggered because of the presence of overland flow contributing areas at the top of reclaimed slopes. Failures in the design, as the absence or collapse of up-slope diverting structures (channels and berms), allow overland flow run-on the slopes promoting soil erosion.

In the present study, severe rill erosion processes are associated with the development of a very sparse and simple community essentially composed of a few plants of *M. sativa* (Fig. 3). Similar communities have been documented in other severely eroded Mediterranean-dry environments, as a consequence of the increase of mechanical disturbance and water stress supported by vegetation (Guardia, 1995;

Guerrero-Campo and Montserrat-Martí, 2004). Accordingly, other studies carried out in reclaimed slopes of the Teruel coalfield have pointed at the negative influence of rill erosion on plant colonization and establishment as a consequence of a drastic reduction in water availability (Moreno-de las Heras et al., 2005). The main mechanisms involved are the reduction of water infiltration by surface crust formation and soil surface roughness reduction, and the efficient evacuation of overland flow from the slopes by rill networks (Nicolau, 2002; Moreno-de las Heras et al., 2007). The resprouting ability and deep rooting system of *M. sativa* could explain plant survival in these harsh conditions, lightening the impact of mechanical disturbance and water stress caused by soil erosion (Bell et al., 2007).

4.4. *Implications of local disturbances*

As expected, some contingent factors were involved in community changes. A positive connection between diversity increases and the occurrence of local disturbances (sheep grazing and fungal diseases) on dominant herbaceous populations is reflected in DCA axis 2 (Fig. 2). In fact, local disturbances has been stressed as important sources of successional change, controlling inhibition mechanisms (Pickett et al., 1987). In the present study, the transition from persistent herbaceous communities (CT6) to a more diverse shrub community (CT7) was related to the occurrence of both occasional fungal diseases in the population of *M. sativa*, and sheep grazing. The primary effect of both factors is the creation of gaps free of competition which enhance the chances of seed germination and plant establishment for new species, especially anemochorous shrubs (as *Santolina chamaecyparissus* and *T. vulgaris*). In addition to site control, grazing can also act as a vector of seed dispersal, since large quantities of seeds can be transported through seed retention in fur and the alimentary tract (Mitlacher et al., 2002). These results highlight the potential function of sheep grazing as a restoration tool in the studied area, bearing in mind the involved benefits from seed dispersal and the control of dominant species introduced by sowing.

5. Conclusions

The evolution of vegetation communities in reclaimed slopes in the Mediterranean-dry area of the Teruel coalfield showed a complex pattern in which three main trends can be identified: (i) very poor communities or even bare slopes on acid soils, (ii) spontaneous communities in which diversity increases with time in less continental areas surrounded by preserved vegetation, and (iii) revegetated communities in more continental areas surrounded by overgrazed vegetation, in which case, transitions are highly dependent on the occurrence of disturbances (rill erosion, grazing or fungal pests). Therefore, initial conditions (soil characteristics and revegetation treatments) and the environmental scenario (climate continentality and presence of surrounding preserved vegetation) appear as the main driving forces directing vegetation succession in reclaimed slopes.

Distance from seed sources is the key factor that directs community development in areas where environmental conditions are less restrictive. Erroneous revegetation practices and design mistakes (presence of overland flow contributing areas at the top of reclaimed slopes) have an important weight in the explanation of vegetation dynamics under restrictive environmental conditions. The use of non-native fast growing herbs can seriously constrain vegetation dynamics. In these cases, local disturbances (such as the occurrence of fungal diseases and, especially, sheep grazing on dominant herbaceous populations) can promote transitions to more diverse shrub communities. Soil erosion triggered by run-on coming from the top of reclaimed slopes constitutes a significant driving force for vegetation succession in reclaimed slopes in Mediterranean-dry environments. On intensively rilled slopes plant establishment was severely restricted, probably because of an increase in water stress and physical disturbance caused by accelerated soil erosion.

This work is not the result of an experimental design aimed at the study of all the possible combinations of factors involved on vegetation dynamics (i.e.: environmental scenario and initial conditions). Thus, this research would greatly benefit from experimental confirmation in the future. Nevertheless, we can draw the following practical considerations from our results: special attention must be placed in the selection of a substrate free of important physico-chemical restrictive factors (extreme pH and toxicity), up-slope protective structures (channels and berms) must be preserved to control run-on fluxes from the top of reclaimed slopes, and revegetation with fast growing allochtonous species must be avoided in order to prevent the inhibition of spontaneous colonization.

Acknowledgments

The authors would like to thank Carlos Ontiveros, Luis Jimenez and Maria José de la Torre for field assistance and Dr. Carmen Bartolomé for technical assistance. We are also very grateful to the Minas y Ferrocarril de Utrillas SA and ENDESA mining companies as well as the Utrillas Council for their active collaboration. An earlier version of this manuscript benefited from the helpful comments of Professor Karel Prach and four anonymous referees; we also thank Lucia Gálvez and Jesús Romero for language corrections. This research has been supported by the project UAH PI2004/024 from the Universidad de Alcalá and the CICYT project CGL2004-00355/BOS from the Spanish government. Mariano Moreno-de las Heras is supported by a research contract from the Regional Authority of Madrid.

Appendix A

See Table A.1 .

Appendix B

See Table B.1 .

Table A.1 – List of species identified

Boraginaceae	*Geraniaceae*	*Medicago minima*
Echium vulgare	*Erodium ciconium*	*Medicago polymorpha*
Myosotis arvensis	*Erodium cicutarium*	*Medicago sativa*[b]
Neatostema apulum	*Erodium malacoides*[a]	*Melilotus indica*
Caryophyllaceae	*Graminae*	*Melilotus officinalis*[b]
Silene muscipula	*Aegylops geniculata*	*Onobrychis viciifolia*[b]
Silene nocturna	*Arrhenatherum elatius*[a]	*Ononis spinosa*[a]
Vaccaria hispanica	*Avena sterilis*	*Psoralea bituminosa*
Chenopodiaceae	*Avenula bromoides*[a]	*Scorpiurus muricatus*
Atriplex halimus[a,b]	*Brachypodium retusum*	*Trifolium pratense*[a]
Atriplex sp	*Bromus willdenowii*[a,b]	*Vicia villosa*
Salsola kali	*Bromus diandrus*[a]	*Linaceae*
Salsola vermiculata	*Bromus erectus*	*Linum narbonense*[a]
Cistaceae	*Bromus hordeaceus*	*Linum suffruticosum*[a]
Cistus clusii	*Bromus rubens*	*Malvaceae*
Helianthemum apenninum	*Bromus tectorum*	*Althaea hirsuta*
Helianthemum nummularium	*Cynodon dactylon*	*Malva hispanica*[a]
Compositae	*Dactylis glomerata*	*Papaveraceae*
Anacyclus clavatus	*Desmazeria rigida*	*Fumaria parviflora*[a]
Artemisia campestris	*Elymus hispidus*	*Papaver rhoeas*
Bellis perennis[a]	*Festuca arundinacea*[b]	*Pinaceae*
Carduus pycnocephalus	*Holcus lanatus*	*Pinus halepensis*[a,b]
Centaurea aspera[a]	*Hordeum murinum*	*Plantaginaceae*
Filago vulgaris	*Lolium perenne*[b]	*Plantago albicans*
Helichrysum stoechas	*Phleum pratense*	*Plantago lanceolata*
Hieracium pilosella[a]	*Sorghum bicolor*[a]	*Plantago sempervirens*
Mantisalca salmantica[a]	*Stipa iberica*[a]	*Polygonaceae*
Pallenis spinosa	*Stipa pennata*[a]	*Polygonum aviculare*
Santolina chamaecyparissus	*Triticum aestivum*[a]	*Primulaceae*
Senecio vulgaris[a]	*Wangenheimia lima*[a]	*Anagallis arvensis*
Sonchus asper	*Labiatae*	*Androsace maxima*
Xeranthemum annuum	*Lavandula angustifolia*[a]	*Coris monspeliensis*[a]
Convolvulaceae	*Marrubium supinum*	*Ranunculaceae*
Convolvulus arvensis	*Rosmarinus officinalis*	*Aquilegia vulgaris*[a]
Crassulaceae	*Satureja* sp[a]	*Rosaceae*
Sedum album[a]	*Thymus vulgaris*	*Potentilla crantzii*
Cruciferae	*Leguminosae*	*Rosa canina*[a]
Alyssum alyssoides	*Anthyllis cytisoides*	*Sanguisorba minor*
Diplotaxis erucoides	*Argyrolobium zanonii*	*Rubiaceae*
Eruca vesicaria	*Astragalus hamosus*	*Galium verum*
Iberis sp	*Coronilla scorpioides*	*Sanatalaceae*
Matthiola fruticulosa[a]	*Dorycnium pentaphyllum*	*Thesium humifusum*
Raphanus raphanistrum	*Genista scorpius*	*Umbelliferae*
Sisymbrium orientale	*Hippocrepis glauca*	*Bupleurum baldense*
Euphoriaceae	*Hippocrepis comosa*	*Daucus carota*[a]
Euphorbia serrata[a]	*Medicago lupulina*	*Eryngium campestre*

Nomenclature follows Tutin et al. (1964–1980).
[a] Species present in less than 5% of the slopes.
[b] Species introduced by revegetation operations.

Table B.1 – Comparison of plant, environmental and erosion traits associated to the community groups derived from TWINSPAN analysis (mean ± S.D.)

	Plant community							
	CT1	CT2	CT3	CT4	CT5	CT6	CT7	CT8
Plant traits								
Veget cover (%)	5.6 ± 5.1a	21.3 ± 7.0b	21.1 ± 12.3b	34.6 ± 2.7bc	13.5 ± 8.9ab	38.2 ± 18.6bc	51.2 ± 10.8c	29.5 ± 9.3bc
Species richness: S	2.6 ± 2.0a	8.2 ± 3.4b	12.9 ± 6.1bc	27.3 ± 4.5cd	5.7 ± 5.0ab	16.6 ± 3.8c	30.7 ± 9.4d	25.3 ± 7.3c
Shannon's index: H	0.6 ± 0.6a	1.5 ± 0.4b	2.0 ± 0.5b	1.67 ± 0.4b	0.8 ± 0.7ab	1.5 ± 0.5b	2.3 ± 0.6b	2.0 ± 0.3b
Relative Sspp (%)	0.0 ± 0.0a	47.2 ± 17.1c	0.0 ± 0.0a	0.0 ± 0.0a	75.9 ± 28.9c	71.4 ± 17.5c	26.9 ± 19.3bc	0.0 ± 0.0a
Environmental variables								
Age	4.0 ± 1.1a	2.7 ± 0.5a	3.1 ± 1.4a	19.0 ± 0.0c	10.4 ± 6.9a	12.4 ± 1.6b	14.8 ± 0.4b	19.0 ± 0.0c

Table B.1 (*Continued.*)

	Plant community							
	CT1	CT2	CT3	CT4	CT5	CT6	CT7	CT8
Steepness (°)	27.3 ± 8.1ab	24.1 ± 4.9b	33.8 ± 3.0a	33.3 ± 0.6a	24.7 ± 7.8ab	17.5 ± 1.6b	16.7 ± 0.8b	32.8 ± 4.0a
Length (m)	21.6 ± 4.5a	32.1 ± 13.6ab	22.8 ± 4.2a	24.7 ± 8.1a	40.9 ± 17.7ab	43.9 ± 19.0ab	55.9 ± 11.6b	24.5 ± 7.8a
Stoniness (%)	21.7 ± 5.2b	32.5 ± 11.7ab	39.9 ± 14.1ab	41.0 ± 8.2a	23.6 ± 10.4b	21.7 ± 8.4b	34.8 ± 9.1ab	45.6 ± 7.4a
Sand (%)	34.8 ± 9.8a	39.4 ± 7.1a	37.9 ± 6.2a	44.0 ± 3.1a	36.9 ± 11.6a	41.7 ± 3.6a	43.7 ± 4.5a	39.1 ± 3.6a
Silt (%)	32.1 ± 7.1a	29.1 ± 3.8a	30.9 ± 3.1a	29.1 ± 2.9a	26.7 ± 7.1a	24.6 ± 3.3a	24.0 ± 2.5a	32.3 ± 1.7a
Clay (%)	33.1 ± 9.2a	31.5 ± 6.0a	31.6 ± 8.2a	26.8 ± 0.8a	36.4 ± 8.4a	33.5 ± 2.2a	32.5 ± 2.8a	28.6 ± 2.4a
pH	5.5 ± 1.6a	7.8 ± 0.6bc	7.4 ± 0.9bc	8.5 ± 0.1c	7.8 ± 0.4b	8.4 ± 0.1bc	8.3 ± 0.1bc	8.6 ± 0.1c
Organic matter (%)	2.1 ± 1.7a	1.3 ± 1.5a	1.0 ± 0.4a	1.4 ± 0.1a	1.3 ± 0.7a	1.6 ± 0.6a	2.3 ± 0.8a	1.0 ± 0.3a
Total nitrogen (%)	0.1 ± 0.1a	0.1 ± 0.1a	0.1 ± 0.1a	0.1 ± 0.1a	0.1 ± 0.1a	0.1 ± 0.1a	0.1 ± 0.1a	0.1 ± 0.1a
Erosion traits								
ASEI	0.6 ± 0.2b	0.6 ± 0.4b	0.6 ± 0.2b	1.1 ± 0.2a	1.7 ± 1.3a	0.8 ± 0.5a	0.8 ± 0.3a	1.3 ± 0.3a
Rills in transects	35.8 ± 10.3b	22.9 ± 15.0b	12.9 ± 12.5b	0.0 ± 0.0a	37.0 ± 12.9b	3.6 ± 5.2ab	3.8 ± 9.4ab	1.3 ± 1.9a
Re rate (t/(ha year))	33.2 ± 18.1c	11.1 ± 9.1b	19.8 ± 16.7b	0.0 ± 0.0a	28.8 ± 15.3c	3.1 ± 5.2ab	4.3 ± 10.6ab	2.1 ± 4.2a

Abbreviations for variables: Veget: vegetation; Sspp: sown species abundance; ASEI: accumulated sheet erosion index; Re: rill erosion Kruskal–Wallis and post hoc *U* tests were applied to detect significant differences between community types.
Different letters (a–c) within rows indicate differences at $\alpha = 0.05$ after the application of sequential Bonferroni corrections.

REFERENCES

Arranz, J.C., 2004. Propiedades, clasificación y evaluación de suelos mineros. Implicaciones sobre la ordenación y gestión de terrenos alterados por minería. Unpublished Ph.D. Thesis. Universidad Politécnica de Madrid, Madrid, Spain.

Bell, L.W., Williams, A.H., Ryan, M.H., Ewing, M.A., 2007. Water relations and adaptations to increasing water deficit in three perennial legumes, *Medicago sativa*, *Dorycnium hirsutum* and *Doricnyum rectum*. Plant Soil 290, 231–243.

Bradshaw, A., 1997. Restoration of mined lands using natural processes. Ecol. Eng. 8, 255–269.

Dufrene, M., Legendre, P., 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol. Monogr. 67, 345–366.

Guardia, R., 1995. La colonització vegetal de les áreas erosionades de la conca de la Baells (Alt Llobregat). Unpublished Ph.D. Thesis. Universitat de Barcelona, Barcelona, Spain.

Guebert, M.D., Gardner, T.W., 2001. Macropore flow on a reclaimed surface mine: infiltration and hillslope hydrology. Geomorphology 39, 151–169.

Guerrero-Campo, J., Montserrat-Martí, G., 2004. Comparison of floristic changes on vegetation affected by different levels of soil erosion in Miocene clays and Eocene marls from Northeast Spain. Plant Ecol. 173, 83–93.

Haigh, M.J., 2000. The aims of land reclamation. In: Haigh, M.J. (Ed.), Reclaimed Land. Erosion Control, Soils and Ecology. AA. Balkema, Rotterdam, The Netherlands, pp. 1–20.

Hill, M.O., 1979. TWINSPAN. A fortran program for arranging multivariate data in an ordered two-way table by classification of the individual attributes. In: Ecology and Systematics. Cornell University, Ithaca, NY, USA.

Hill, M.O., Gauch, H.G., 1980. Detrented correspondence analysis, an improved ordination technique. Vegetatio 42, 45–58.

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

Lavee, H., Imeson, A.C., Sarah, P., 1998. The impact of climate change on geomorphology and desertification along a Mediterranean-arid transect. Land Degrad. Dev. 9, 407–422.

MAPA, 1994. Métodos Oficiales de Análisis. Tomo III. Secretaría General de Alimentación. Dirección General de Política Alimentária. Ministerio de Agricultura, Pesca y Alimentación. Madrid, Spain.

Martinez-Ruiz, C., Fernandez-Santos, B., Putwain, P.D., Fernandez-Gomez, M.J., 2007. Natural and man-induced revegetation on mining wastes: changes in the floristic composition during early succession. Ecol. Eng. 30, 286–294.

McCune, B., Mefford, M.J., 1999. PC-ORD. Multivariate Analysis of Ecological Data, Version 4. MjM Software Design, Gleneden Beach, Oregon, USA.

Mellado, I., 2006. Evaluación ambiental de la minería del carbón a cielo abierto en Teruel. Unpublished Professional Thesis. Universidad de Alcalá, Alcalá de Henares, Spain.

Mitlacher, K., Poschlod, P., Rosen, E., Bakker, J.P., 2002. Restoration of wooded meadows: a comparative analysis along a chronosequence on Oeland, Sweden. Appl. Veg. Sci. 5, 63–73.

Moreno-de las Heras, M., Nicolau, J.M., Espigares, M.T., 2005. Interacción entre la erosión en regueros, contenido de humedad edáfica y colonización vegetal en laderas restauradas de la minería a cielo abierto del carbón en ambiente mediterráneo-continental (Teruel). In: Samper-Calvete, F.J., Paz-González, A. (Eds.), Estudios de la Zona No Saturada del Suelo, vol. VII. Universidade da Coruña, La Coruña, Spain, pp. 345–350.

Moreno-de las Heras, M., Nicolau, J.M., Espigares, M.T., 2007. Effects of soil erosion on ecohydrology of constructed slopes from opencast coal mining in a Mediterranean-Continental environment. EOS Transactions of the AGU, 88(52), Fall Meeting Supplement, Abstract H51H-0881.

Morgan, R.P.C., 1995. Soil Erosion and Conservation, 2nd ed. Longmann Group Lmtd., London, UK, pp.: 88.

Nicolau, J.M., 2002. Runoff generation and routing on artificial slopes in a Mediterranean-continental environment: the Teruel coalfield, Spain. Hydrol. Process. 16, 631–647.

Nicolau, J.M., 2003. Trends in relief design and construction in opencast mining reclamation. Land Degr. Dev. 14, 215–226.

Nicolau, J.M., Asensio, E., 2000. Rainfall erosion on opencast coal-mine lands: ecological perspective. In: Haig (Ed.), Land Reconstruction and Management. Reclaimed Land, Erosion Control, Soils and Ecology, vol. 1. A.A. Balkema, Rotterdam, The Netherlands, pp. 51–73.

Nicolau, J.M., Moreno-de las Heras, M., 2005. Opencast mining reclamation. In: Mansourian, S., Vallauri, D., Dudley, N. (Eds.),

Forest Restoration in Landscapes: Beyond Planting Trees. WWF/Springer, Berlin, Germany, pp. 370–376.
Novak, J., Konvicka, M., 2006. Proximity of valuable habitats affects succession patterns in abandoned quarries. Ecol. Eng. 26, 113–122.
Novak, J., Prach, K., 2003. Vegetation succession in basalt quarries: pattern on a landscape scale. Appl. Veg. Sci. 6, 111–116.
Oksanen, J., Kindt, R., Legendre, P., O'Hara, R.B., 2007. Vegan: Community Ecology Package. Version 1.8-5. http://cran.r-project.org/.
Otto, R., Krusi, B.O., Burga, C.A., Fernandez-Palacios, J.M., 2006. Old-field succession along a precipitation gradient in the semi-arid coastal region of Tenerife. J. Arid Environ. 65, 156–178.
Papadakis, J., 1966. Climates of the World and Their Agricultural Potentialities. Buenos Aires, Argentina.
Parker, V.T., Pickett, S.T.A., 1997. Restoration as an ecosystem process: implications of the modern ecological paradigm. In: Urbanska, K.M., Webb, N.R., Edwards, P.J. (Eds.), Restoration Ecology and Sustainable Development. Cambridge University Press, Cambridge, pp. 17–32.
Parrotta, J.A., Knowles, O.H., 2001. Restoring tropical forests on lands mined for bauxite: examples from the Brazilian Amazon. Ecol. Eng. 17, 219–239.
Pickett, S.T.A., Collins, S.L., Armesto, J.J., 1987. Models, mechanisms and pathways of succession. Bot. Rev. 53, 335–371.
Pietrzykowski, M., Krzaklewski, W., 2007. An assessment of energy efficiency in reclamation to forest. Ecol. Eng. 30, 341–348.
Plass, W.T., 2000. History of surface mining reclamation and associated legislation. In: Barnhisel, R.I., Darmody, R.G., Daniels, W.L. (Eds.), Reclamation of Drastically Disturbed Lands. American Society of Agronomy, Inc., Crop Science Society of America, Inc. and Soil Science Society of America Inc., Madison, WI, USA, pp. 1–20.
Poesen, J.W., van Wesemael, B., Bunte, K., Solé-Benet, A., 1998. Variation of rock fragment cover and size along semiarid hillsopes: a case-study from SE Spain. Geomorphology 23, 323–335.
Prach, K., Pysek, P., 2001. Using spontaneous succession for restoration of human-disturbed habitats. Experience from Central Europe. Ecol. Eng. 17, 55–62.
Prach, K., Pysek, P., Jarosik, V., 2007. Climate and pH as determinants of vegetation succession in Central-European man-made habitats. J. Veg. Sci. 18, 701–710.
Puigdefábregas, J., Solé, A., Gutiérrez, L., del Barrio, G., Boer, M., 1999. Scales and processes of water and sediment redistribution in drylands: results from the Rambla Honda field site in SE Spain. Earth Sci. Rev. 48, 39–70.
Rice, W.R., 1989. Analyzing tables of statistical test. Evolution 43, 223–225.
Rivas-Matinez, S., 1987. Memoria del Mapa de Series de Vegetación de España. Ministerio de Agricultura, Pesca y Alimentación, ICONA, Madrid, Spain, pp. 53–136.
Sänger, H., Jetschke, G., 2004. Are assembly rules apparent in the regeneration of a former uranium mining site? In: Temperton, V.M., Hoobs, R.J., Nuttle, T., Halle, S. (Eds.), Assembly Rules and Restoration Ecology: Bridging the Gap between Theory and Practice. SER, Island Press, Washington, USA, pp. 305–324.
Sokal, R.R., Rohlf, F.J., 1995. Biometry. W.H. Freeman, NY, USA.
Soil Survey Staff, 1998. Keys to Soil Taxonomy, 8th ed. USDA-Natural Resource Conservation Service, Washington, DC, USA.
Statsoft, 2001. STATISTICA Version 6. Data Analysis Software System. Statsoft, Tulsa, OK, USA.
Tutin, T.G., Heywood, V.H., Walters, S.M., Webb, D.A., 1964–1980. Flora Europaea, vols. 1–5. Cambridge University Press, Cambridge, UK.
van Groenewoud, H., 1992. The robustness of correspondence, detrented correspondence, and twinspan analysis. J. Veg. Sci. 3, 239–246.
Velazquez, E., Gómez-Sal., 2007. Environmental control of early succession on a large landslide in a Tropical dry ecosystem (Casita Volcano, Nicaragua). Biotropica 35, 601–609.
Vogiatzakis, I.N., Griffiths, G.H., Manion, A.M., 2003. Environmental factors and vegetation composition. Lefka Ori massif, Crete, S. Aegean. Global Ecol. Biogeogr. 12, 131–146.
Wang, F.X., Wang, Z.Y., Lee, J.H.W., 2007. Acceleration of vegetation succession on eroded land by reforestation in a subtropical zone. Ecol. Eng. 31, 232–241.
Ward, A.D., Wells, L.G., Phillips, R.E., 1983. Infiltration through reconstructed surface mined spoils and soils. Trans. ASAE 26, 821–832.
Whisenant, S.G., 2002. Manipulation of the physical environment. Terrestrial systems. In: Perrow, M.R., Davy, A.J. (Eds.), Handbook of Ecological Restoration. Principles of Restoration, vol. 1. Cambridge University Press, Cambridge, UK, pp. 83–105.
Wiegleb, G., Felinks, B., 2001. Primary succession in post-mining landscapes of Lower Lusatia-chance or necessity. Ecol. Eng. 17, 199–217.