

COLONIZATION PATTERNS OF WOODY SPECIES ON LANDS MINED FOR COAL IN SPAIN: PRELIMINARY INSIGHTS FOR FOREST EXPANSION

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ABSTRACT

In this study, we combined the analysis of a coenocline with Huisman–Olf–Fresco (HOF) modelling of species behaviour to describe colonization patterns of woody species on reclaimed coal wastes in northern Spain. We hypothesize that fine-scale variation in abiotic factors along the forest–grassland gradient that appears after mining restoration affects colonization pattern of woody species. To understand these factors, we analysed changes in floristic composition and abundance of the main woody colonizers according to the spatial distance from forest limit, as well as the spatial variability of some edaphic and microclimatic parameters. Our results showed how the primary coenocline extracted from DCAI reflected a gradient in environmental conditions influencing the species composition of the woody plant communities, from the more shade, moist and dense forest on acid and rich soils to the open, dry and oligotrophic grassland. The colonization pattern of woody species is affected by fine-scale variations in abiotic factors, such as the increase in pH, P, light, and the decrease in K, N, C/N, organic matter and soil moisture, from the forest to the restored mine. Most species HOF modelled showed monotone responses with decreasing trends along the environmental gradient from forest to mine, being *Quercus petraea* the species clearly dominant in the forest. On the contrary, *Cytisus scoparius* and *Genista florida* colonize intensely in the mine area, reaching the maximum abundance around 15 and 10 m distance from the forest limit, respectively. We conclude that the composition of woody species in the forest is not a good predictor of the colonizing intensity, not even of the species composition in the mine area. Copyright © 2011 John Wiley & Sons, Ltd.

KEY WORDS: coenocline; forest–grassland boundaries; Mediterranean deciduous forests; species responses; surface mining; Spain; oak

INTRODUCTION

In the northern of Palencia, Spain opencast coal mining plays an important role in the dynamics of landscape fragmentation (Bradshaw and Chadwick, 1980; Hardt and Forman, 1989; Prach and Pyšek, 1994). As a result, a spatial variation in the physical and biological structure of plant communities is produced, since the restored coal mines become scars healing in a forest matrix. Therefore, the structure of the new plant community making up on the reclaimed mine area both temporal and spatially varies (Matlack, 1994; Saunders *et al.*, 1999; Newmak, 2001). The process is strongly affected by interactions between the forest matrix and the initial grassland patch (Bramble and Ashley, 1955; López-Barrera, 2004; López-Barrera *et al.*, 2005), but also by the new environmental gradient that emerged (Williams-Linera, 1990; Kapos *et al.*, 1997; Oosterhoorn and Kapelle, 2000) from the forest to the open mined area.

The forest matrix, as a seed source, partly determines the species composition of woody colonizers (Hardt and Forman, 1989; Milder *et al.*, 2008) while there is often a lack of seed bank in coal reclamation areas (González-Alday *et al.*, 2009). Animals living in forest edges also play an important role as seed dispersers or consumers (Parmenter *et al.*, 1985; Gómez *et al.*, 2003; López-Barrera *et al.*, 2005). In addition, grazing can directly limit recruitment of woody species (Gómez *et al.*, 2003; López-Barrera *et al.*, 2005), and indirectly by specifically affecting the few seeds dispersed to the high-quality microsites (Callaway, 1992; Rousset and Lepart, 2000).

Species may respond to fine-scale variations in light (Chen *et al.*, 1995), pH (Anderson *et al.*, 1995), soil temperature and moisture (Chen *et al.*, 1995) or nutrients (Johnson and Wedin, 1997) along the forest–mine environmental gradient, while climatic factors may be important on broader scales (Westman, 1991). Therefore, species performance along the forest–mine changing environment also determines their success of establishment once they managed to arrive.

From a biological point of view, the essence of understanding environmental gradients is the extent to

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which they, at a fixed spatial scale, are accompanied by changes in species composition which are not captured by measured environmental variables (Økland, 1990; Lawesson and Oksanen, 2002). A promising statistical approach to describe species responses that may result from environmental conditions was suggested by Huisman *et al.* (1993), in which a set of hierarchical models which include a skewed response and a symmetric response, are fitted by a maximum likelihood estimation [the so-called Huisman–Olff–Fresco (HOF) models].

In this paper, the combined analysis of a coenocline with HOF modelling of species behaviour is used for describing colonization patterns of woody species on reclaimed coal wastes in northern Spain. We hypothesize that fine-scale variation in abiotic factors along the forest–mine gradient affects colonization pattern of woody species. To understand these factors, we analysed changes in the floristic composition and in the abundance of the main woody colonizers according to the distance from the forest–mine border, as well as the spatial variability of the main edaphic and microclimatic parameters.

MATERIALS AND METHODS

Study Area

This study examines three close sites, all surface-mined for coal, in the province of Palencia (northern Spain), near the village of Guardo (1110 a.m.s.l.; 42°47'N, 4°50'W), in the 'Montaña Palentina' area on predominantly limestone of Paleozoic age, with a presence of clay and to a lesser degree of sand (Milder *et al.*, 2008). The climate is sub-humid Mediterranean (MAPA, 1991); with a rainy season in autumn and spring and dry season in July through August with only 8 per cent of the annual rainfall that is 973 mm (average 1973–2007; data provided by the Spanish Meteorological National Agency from the Meteorological Station at Guardo). The mean annual temperature for this period is 9.2°C, the mean minimum in the coldest month (January) is –2.7°C, and the mean maximum in the warmest month (August) is 25.9°C. Most of the area surrounding the sites is forested with relatively diverse vegetation associated to the sessile oak (*Quercus petraea*) forests. The most abundant tree species associated to this type of forest in this area are: *Prunus avium*, *Ilex aquifolium*, *Malus sylvestris* and *Corylus avellana*, with an understory composed of *Cornus sanguinea*, *Euonymus europaeus*, *Crataegus monogyna*, *Lonicera xylosteum*, *L. periclymenum*, *Genista florida*, *Cytisus scoparius* and other shrub species.

Study sites were reclaimed around 15 years before our autumn 2005 sampling. All sites were regraded to approximately the original contour, and the surface was covered with 30 cm of topsoil amended with cattle manure

(30 t ha⁻¹) and fertilizer (8N:15P:15K; 150 kg ha⁻¹). All sites were revegetated, in autumn 1990, with a commercial seed mix (210 kg ha⁻¹) containing grasses (80 per cent in weight) and legume herbs (20 per cent in weight). By 2005 the three sampling sites were colonized by native herbaceous and shrub species from the surrounding areas, regardless of the other species originally sown.

The three reclaimed mine sites are described as follows: Site I: 325 m × 185 m, eight transects sampled from northeast to east edges, slope varying from 30 degrees to flat; Site II: 100 m × 100 m, six transects sampled from northeast to east edges, slopes 18–22 degrees; Site III: 100 m × 370 m, nine transects sampled from northwest to southwest edges, slope ca. 18 degrees.

Vegetation Sampling

The sampling design followed that proposed by Hardt and Forman (1989). Transects 74 m long, extending 63 m onto the mine site and 11 m into the forest, were laid out perpendicular to the forest–mine boundary. Nineteen 2 m × 2 m plots, centered at 4 m intervals along each transect, were sampled (3 plots into the forest and 16 into the mine). Transects traversing grossly disturbed areas such as roads or extensive gullies were omitted.

An initial transect point was established at the southeasternmost corner of each mine site, and subsequent transects were established around the forested boundary in a clockwise fashion at 30 m intervals at larger sites, and at 15 m intervals at the smaller site. The initial transect point was omitted from sampling at each site to minimize possible bias in starting point selection. The number of transects varied with the size of the mine, ranging from 6 to 9 transects per site; a total of 23 transects were sampled through the study.

Within each 2 m × 2 m sample plot, the number of individuals of all woody plants was censused; only 19 species were recorded through the study. Separate stems at ground level were recorded as individuals. Species nomenclature follows Tutin *et al.* (1964–1980).

Soil Sampling and Analyses

Once the vegetation sampling was finished, soil was sampled to a depth of 15 cm with a soil auger (diameter 3 cm) at one random point near each of the established 2 m × 2 m plots. The 23 soil samples from the same position along transects were pooled and thoroughly mixed to obtain a uniform sample of the soil characteristics of each position; hence there were 19 composited soil samples to evaluate potential differences in soil characteristics along the forest–mine gradient (see Table I).

Soil samples were air-dried, sieved (≤2 mm) and analysed for soil pH using a conductivity meter in a 1:2.5 soil/deionized water slurry (Allen, 1974). Soil organic matter

Table I. Spatial variability of some soil characteristics, and Ellenberg indicator values for light (L) and soil moisture (F), along the forest–mine gradient

Plot number along a transect	pH	Organic matter (per cent)	Available P (ppm)	Exchangeable K (ppm)	Total N (ppm)	Ratio C/N	L (1–9) ^a	F (1–12) ^a
1 ^b	5.21	11.95	5.1	289.20	4900	18.63	6.63	4.64
2 ^b	5.46	15.58	<4.4	193.46	4400	24.67	6.62	4.65
3 ^b	5.46	9.78	4.5	219.29	4200	15.74	6.66	4.60
4	5.46	7.86	<4.4	139.07	2800	17.55	6.86	4.55
5	5.56	5.20	11.7	92.81	2900	12.89	6.45	3.71
6	6.10	6.61	37.9	87.44	4100	11.22	6.64	3.85
7	5.95	3.83	10.9	117.01	2400	14.68	5.87	3.56
8	5.95	4.44	14.0	66.90	2200	15.67	6.06	3.26
9	6.29	4.55	23.3	136.90	2800	13.43	5.07	2.72
10	6.18	5.53	22.9	81.29	4100	11.26	5.09	7.72
11	5.69	4.20	6.9	107.46	2700	13.35	4.86	2.69
12	5.44	3.97	7.7	90.10	3000	11.24	4.35	2.41
13	5.58	3.01	<4.4	82.09	2500	12.63	4.65	2.62
14	5.65	5.21	4.6	96.11	2500	14.77	5.24	3.07
15	6.04	4.67	<4.4	77.98	2500	15.44	4.98	2.78
16	4.55	4.96	7.5	124.37	2500	15.58	3.74	1.98
17	5.60	4.62	4.6	87.29	2900	13.29	4.39	2.37
18	5.83	3.25	9.7	89.56	2500	10.09	4.48	2.30
19	6.07	9.24	23.7	85.39	4900	13.31	3.37	1.80

^aCategories for light and moisture scales according with the ECOFACT database; an increase in the number of category means an increase in requirements for light or moisture of species present within each plot along the transect (the forest–mine gradient).

^bPlots into the forest; the rest belonging to the mine.

was determined using the K-dichromate oxidation method (Walkley, 1947). Soil total nitrogen was measured using the Kjeldahl-method (Bremner and Mulvaney, 1982). Available P was measured using the Olsen method (Olsen and Sommers, 1982), and exchangeable K⁺ was extracted with 1 N ammonium acetate at pH 7 by shaking for 30 min and determined using atomic absorption spectroscopy (Allen, 1974).

Data Analyses

Detrended correspondence analysis (DCA) was used to extract the primary ordination axes or coenocline (Lawesson and Oksanen, 2002). DCA constructs ordination axes from eigenvectors, and the positions of the sample plots along these axes relative to each other represent their floristic similarity, expressed in standard deviation (SD) units that can be interpreted in terms of diversity or compositional turnover (Økland, 1990; Lawesson and Oksanen, 2002). The extracted ordination axes may conveniently be considered as standardized, abstract complex environmental gradients or coenoclines (Lawesson and Oksanen, 2002). For simplicity, only the first ordination axis (DCA1) is used in this paper. Besides indirect interpretation of this ordination axis, sample ordination scores were tested for a significant correlation with the explanatory variables by means of Kendall's τ . The explanatory variables considered in this study were soil parameters (pH, organic matter, N, P, K and

C/N), and the unweighted Ellenberg indicator values (Ellenberg *et al.*, 1991) for light (L) and soil moisture (F).

The unweighted Ellenberg indicator values were calculated for each plot by averaging the available indicator values for the species present in each plot (see Table I). The available indicator value for each species was extracted from the ECOFACT database (Ellenberg, 1999), and it is an addimensional value assigned to each species that encapsulate information about its realised ecological niche with respect to different ecological factors such as light or moisture. The categories for light and moisture scales ranged from 1 to 9 and from 1 to 12, respectively, and species are assigned into each category according with their preference of light or moisture; increasing the number of category as light or moisture species requirements increase.

Species responses along the DCA1 coenocline were modelled by HOF models (Huisman *et al.*, 1993). HOF models are a means of describing species responses, which may result from both environmental conditions and intra- and inter-specific interactions (Lawesson and Oksanen, 2002). These are a hierarchical set of five response models, ranked by their increasing complexity (Model I, no species trend; Model II, increasing or decreasing trend; Model III, increasing or decreasing trend below maximum attainable response; Model IV, symmetrical response curve; Model V, skewed response curve). The AIC statistic (Akaike, 1973) was used to select the most appropriate response model for each species (Burnham and Anderson, 2002; Johnson and

Omland, 2004); smaller values of AIC indicate better (more parsimonious) models. Only HOF response curves of species sufficiently frequent to be modelled (10 of 19 included in DCA) were computed to illustrate the broad trends of compositional change.

DCA was carried out using the CANOCO 4.5[®] (Ter Braak and Šmilauer, 2002), with standard options and no downweighting of rare species. HOF models were carried out using the GRAVY package (Oksanen and Minchin 2002; Oksanen, 2004) implemented in the R software[®] environment (version 2.4.1; R Development Core Team, 2006). Correlation analyses were carried out using STATISTICA software v. 5.1[®].

RESULTS

DCA and Coenocline Interpretation

The DCA performed on the complete data set produced an ordination with a first axis gradient length of 3.87 SD units. The eigenvalues (λ) for the first four DCA axes were 0.85, 0.40, 0.29 and 0.22, respectively. The Kendall's τ coefficient indicated a highly significant negative correlation ($p < 0.001$) between plot scores along DCA1 and their belonging to the forest or mine area ($\tau = -0.72$), and with their distance to the forest boundary ($\tau = -0.59$). This suggests a gradient strongly associated with environment type, which influences the species composition of woody plant communities (Figure 1b), although modified by the effect of the distance from a propagules source (the adjacent forest). The plots into the forest and those in the mine area but close to the forest boundary cluster together on the right of the diagram, whereas plots in the open mine area cluster on the left (Figure 1a). Species composition gradually changes from the forest to the mine; from those exclusive of the forest (*Erica arborea*, *Ligustrum vulgare*, *Prunus spinosa*, *M. sylvestris*, *Vaccinium myrtillus*, *Viburnum*

lantana) to the right end of DCA1, and those more abundant in the open mine area (*C. scoparius*, *G. florida*) to the left end (Figure 1b).

The Kendall's τ coefficient also indicated a highly significant positive correlation ($p < 0.001$) between plot scores along DCA1 and organic matter ($\tau = 0.56$), potassium ($\tau = 0.57$), nitrogen ($\tau = 0.41$), C/N ratio ($\tau = 0.59$), and unweighted average Ellenberg value of soil moisture ($\tau = 0.51$), while pH ($\tau = -0.64$), phosphorus ($\tau = -0.40$) and unweighted average Ellenberg value of light ($\tau = -0.80$) were negatively correlated ($p < 0.001$) with DCA1 (see Table I for edaphic and microclimatic parameters). Therefore, variations in the species composition and number of colonizing stems produced along DCA1 (primary coenocline) can be caused by changes in soil conditions and light. Thus, colonization is not only determined by distance itself, but it is also a result of the existing gradient of these parameters (soil conditions and light) that runs perpendicular to the boundary, from the forest to the mine.

Although it is obvious that there is a gradient of environmental conditions from the forest to the mine, many differences can be found among plots into the mine area in relation with some edaphic parameters; particularly between plot 19 and the rest for N, P and organic matter percentage, showing a great heterogeneity in the own composition of coal wastes piled up (see Table I).

HOF Response Models

Most of the species HOF modelled (*Q. petraea*, *Rosa canina*, *Rubus ulmifolius*, *C. monogyna*, *E. europaeus*, *I. aquifolium* and *Sorbus aria*) showed monotone responses with decreasing trends (HOF Model II; Figure 2) along the environmental gradient from the forest to the open mined area. Species of this group had higher abundances in the forest, but as distance to forest boundary increased the number of colonizing stems decreased, at a higher rate in the

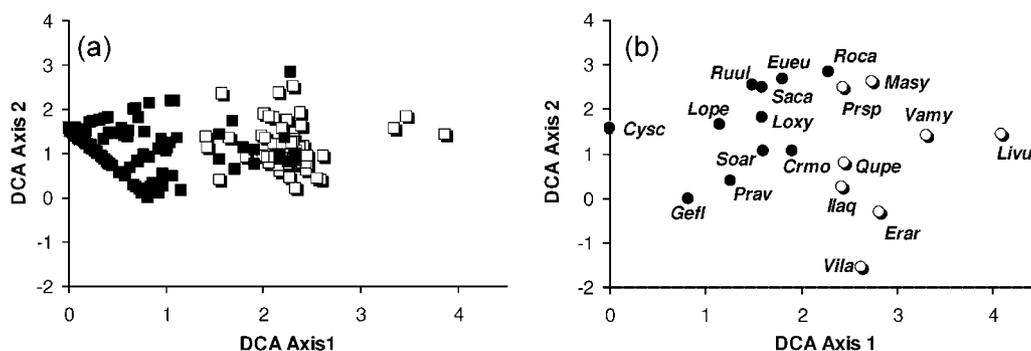


Figure 1. DCA ordination of (a) 437 plots (19 plots for each of 23 transects), and (b) 19 woody species, listed by first two letters of genus and species name; full and empty symbols refer to plots belonging to mine area or forest edge, respectively. Species codes: *Crataegus monogyna* (Crmo), *Cytisus scoparius* (Cysc), *Erica arborea* (Erar), *Euonymus europaeus* (Eueu), *Genista florida* (Geff), *Ilex aquifolium* (Ilaq), *Lonicera periclymenum* (Lope), *L. xylosteum* (Loxy), *Ligustrum vulgare* (Livu), *Malus sylvestris* (Masy), *Prunus spinosa* (Prsp), *Prunus avium* (Prav), *Quercus petraea* (Qupe), *Rosa canina* (Roca), *Rubus ulmifolius* (Ruul), *Salix caprea* (Saca), *Sorbus aria* (Soar), *Vaccinium myrtillus* (Vamy) and *Viburnum lantana* (Vila).

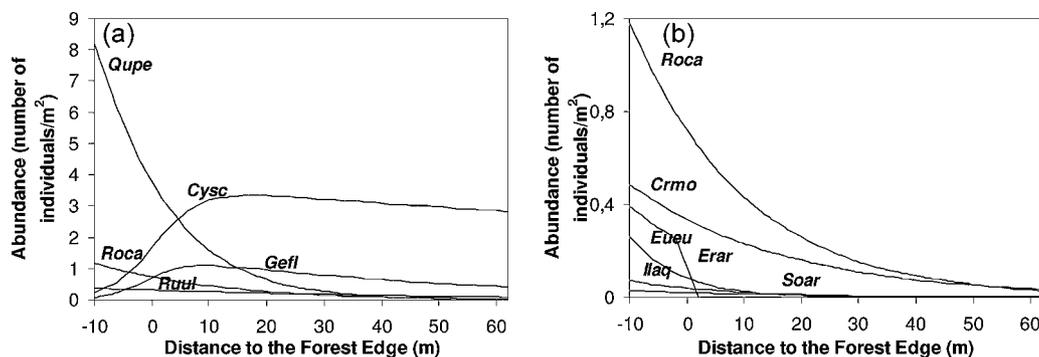


Figure 2. HOF-derived response curves showing the response of the most common species, relative to the environmental gradient from the forest (negative axis 1) to the mine area (positive axis 1). Graphs separated for clarity of scale including in both the trend of *Rosa canina* as a reference. For species codes see Figure 1.

first meters of distance to forest boundary. An aspect which should be emphasized is that *Q. petraea*, which is included in this group, is the dominant species in the forest. The rest of species included in this group, which are not highly abundant in the forest, do not colonize intensely the mine area.

E. arborea, which was only found in the forest edge, showed a decreasing trend below maximum attainable response (HOF Model III) (Figure 2b).

Finally, *C. scoparius* and *G. florida* showed skewed response curves (HOF Model V; Figure 2a). Both species were less abundant in the forest, but as distance to the forest boundary increased, the number of colonizing stems also increased, reaching the maximum abundance around 15 m and 10 m of distance to the forest boundary, respectively.

These results suggest that the species clearly dominant in the forest (*Q. petraea*) is one of the less abundant species in the mine area. On the contrary, species not especially abundant in the forest such as *C. scoparius* and *G. florida* colonize more intensely the mine area.

DISCUSSION

Our results showed how the primary coenocline extracted from DCA1 reflected a gradient in environmental conditions influencing the species composition of woody plant communities, from the more shade, moist and dense forest on acid and rich soils to the open, dry and oligotrophic grassland. Particularly, the colonization pattern of woody species might be affected in the study area by the fine-scale variation in abiotic factors, such as the increase in pH, P, light, and the decrease in K, N, C/N, organic matter and soil moisture, from the forest to the mine. It is known that habitat heterogeneity (Kollmann and Schill, 1996), as well as the distance from a propagules source (Müller *et al.*, 2007; Martínez-Ruiz and Marrs, 2007), affects the main processes of seed dispersal and establishment.

Difficulties in colonizing the mine area for species that are forest specialists, as *Q. petraea*, and other species also found in the study area such as *E. europaeus*, *I. aquifolium* and *S. aria*, could be explained by lower phenotypic plasticity to light (Chazdon *et al.*, 1996; Ribeiro *et al.*, 2005) and lower acclimation potential to increasing light (Fetcher *et al.*, 1983; Strauss-Debenedetti and Bazzaz, 1991) which is associated to late successional species (Rodríguez *et al.*, 2008). Compared to a forest, crops and grasslands allow more solar radiation (light) to reach the ground; consequently, temperatures tend to be higher near the ground. In contrast, the environment under the forest canopy is cooler, moister and more uniform (Murcia, 1995).

It is also known that *Q. petraea* prefers sites with deeper and more humid soils than other species of the same genus also present in the study area, such as *Q. pyrenaica* (Pardo *et al.*, 2004) with lower water and soil needs. Both the rapid growth and acclimation potential showed by *Q. petraea* in non-water-limited and productive sites, where among-species competition for resources is the driving force of regeneration (Keddy *et al.*, 1997), are likely to enhance its competitive ability. However, in Mediterranean ecosystems, as in our study area, recruitment relies to a greater extent on the capacity of seedlings to endure the combination of multiple stresses and disturbances, such as nutrient or water shortages, wildfires or herbivore damages (Rodríguez *et al.*, 2008). Probably, the modest presence of *Q. petraea* in Mediterranean ecosystems, where it only forms small stands in favourable areas at high altitudes or northern exposures, it is precisely due to this characteristic (Rodríguez *et al.*, 2008).

Similarly, responsiveness to resource availability is relatively low for some Mediterranean woody species, as a result of an adaptation to limiting stressful environments (Valladares *et al.*, 2002; Chambel *et al.*, 2005), so that one might expect a different response to light between seedlings of a sub-Mediterranean and a temperate species of similar successional status (Rodríguez *et al.*, 2008). Particularly,

C. scoparius and *G. florida* could be two Mediterranean species adapted to stressful environments, because they usually predominate in the forest degradation stages, just as it seems to happen in the study area where they are the most active colonizing species on lands mined for coal. Adaptation of species to extreme soil conditions is fundamental in the colonization of degraded areas (Bradshaw and Chadwick, 1980).

Furthermore, forest edges have always been associated with lower humidity and higher air temperatures (Kapos, 1989; Williams-Linera, 1990), higher soil temperatures and increased solar radiation (Brothers and Spingarn, 1992), and lower soil moisture (Camargo and Kapos, 1995; Jose *et al.*, 1996). These microclimatic conditions can reduce decomposition rates (Kapos *et al.*, 1993), which can favour a relatively lower soil organic carbon content near the edge relative to the interior (Jose *et al.*, 1996). These same microenvironmental factors, through their influence on decomposition rate, can control nutrient availability. Therefore, light is not the only factor determining the colonization pattern of woody species in the study area, edaphic factors have an important influence on forest regeneration, perhaps much more than microenvironmental factors (Jose *et al.*, 1996).

Hence, it is reasonable to assume that species that were found colonizing intensely the mine (*C. scoparius* and *G. florida*) may be adapted to conditions characterised by lower soil moisture and nutrients, and higher light availability, whereas other species with other preferences, such as *E. arborea*, are not capable of expanding out of the forest edge. Studies from temperate deciduous forest have shown that species composition differs markedly between forest edge and interior (Jose *et al.*, 1996). In the particular case of *E. arborea* the seeds do not disperse a great distance (Mesléard and Lepart, 1991) and most of them come from its persistent soil seed bank (Valbuena and Vera, 2002) that is not present in the coal mining wastes; thus its low dispersion is limited to the forest interior.

Similarly, *C. monogyna*, *R. canina* and *R. ulmifolius*, which are usually found forming the thorny border on this type of forests, decreases the number of colonizing individuals as the distance to the forest edge increases. Even more, most of the colonizing stems found at some distance to the forest edge were establish under the canopy of nurse shrubs, mainly *C. scoparius* and *G. florida*. Even though this species are usually found colonizing open areas in this forests, for their establishment is essential the existence of high-quality microsites with deep and cool soils.

Perspectives for Future Studies

Our results showed that natural woody colonization on lands mined for coal in Spain is scarce and concentrated in the

proximity of the forest, as occurs in other environments (Hardt and Forman, 1989); thereby colonization increases as distance to the forest decreases. Certain properties of ecotones (Jordana and Arbea, 1987) such as higher relative humidity than in nearby open areas, higher diversity and richness of edaphic fauna, higher biological activity that as a consequence generates a higher decomposition rate of organic matter, higher soil formation and stabilization rate, etc., which produces an increment in biomass and shrub density in the forest edge, can explain this trend. Furthermore, Hardt and Forman (1989) found that clonal species accounted for most of the immigration to the reclaimed mines.

Moreover, herbaceous vegetation can represent an important barrier for seed reproduction, retarding early growth of woody species from seeds (Plass, 1968; Vogel and Berg, 1973), whereas other woody colonizers could favour their establishment (Kitzberg *et al.*, 2000; Choler *et al.*, 2001). In fact, the only individual stems found over 5 m of distance to the edge, were capable to settle there thanks to the refuge that *C. scoparius* and *G. florida* offer, not only for *Q. petraea* but also for *C. monogyna*, *R. canina* and *R. ulmifolius*.

Therefore, topics such as the positive interactions among-species (facilitation), regenerative strategies of species, dispersal modes and influence of herbivory pressure should be approached in a deeper degree in future studies to provide more information on the mechanisms that can favour forest expansion in areas affected for coal mining all around the world.

CONCLUSIONS

In conclusion, the results suggest that the species clearly dominant in the forest (*Q. petraea*) is one of the less abundant species in the mine area, which can be due to the fact that forest woody species, in general, are incapable of dispersing their seeds long distances. On the other hand, the most active colonizers of the mine (*C. scoparius* and *G. florida*) are not specially abundant in the forest edge. The rest of species, with a not very high abundance in the forest, do not colonize intensely the mine area. Therefore, the composition of woody species in the forest is not a good predictor of the colonizing intensity, not even of the species composition in the mine area.

The knowledge of the mechanisms driving *Q. petraea* forest expansion has an especial value for the study area, since it is in the southern limit of the distribution of *Q. petraea* forests worldwide. These forests are among the most threatened of our country, and in the study area they have a very high environmental and landscape value. Therefore, they have to be managed in a rational way to

promote biodiversity conservation and forest expansion (Milder *et al.*, 2008).

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