Nova Acta Científica Compostelana (Bioloxía), **19**: 69-81 (2010) - ISSN 1130-9717

Burning in the management of heathlands of *Erica ciliaris* and *Erica tetralix*: effects on structure and diversity

A. Muñoz [1,3], R. Álvarez [1] , X.M. Pesqueira [1], J. García-Duro [1], O. Reyes [1,2] & M. Casal [1]

[1] *Departamento de Biología Celular y Ecología, Facultad de Biología*
Universidad de Santiago de Compostela
Campus Vida. 15782 Santiago de Compostela
[2] *Departamento de Bioloxía Celular e Ecoloxía. E.P.S. Universidade de Santiago de Compostela*
Campus de Lugo. 27002 Lugo
[3] *Corresponding author. E-mail: anamunozespasandin@gmail.com. Phone: +34881813318*

(Recibido, julio de 2011. Aceptado, noviembre de 2011)

Abstract

Muñoz, A., Álvarez, R., Pesqueira, X.M., García-Duro, J., Reyes, O. & Casal, M. (2010). Burning in the management of heathlands of *Erica ciliaris* and *Erica tetralix*: effects on structure and diversity. *Nova Acta Científica Compostelana (Bioloxía)*, **19**: 69-81

Can controlled burning be used as a management tool of *Erica ciliaris* and *Erica tetralix* wet heathlands? Two *E. ciliaris* and *E. tetralix* communities were selected and two 5 x 5 m plots were established in each. These were then characterised on the basis of frequency and cover values and plant species composition. They were subjected to experimental burning, after which the plots were sampled twice a year during the following four and a half years. The results show that the cover of woody species very quickly attained the values of the Control Plots. Diversity and species composition did not suffer notable changes during this period, however, temporal heterogeneity indicates that the main changes occur in the first 18 months of secondary succession. The multivariate analysis showed that the samples registered during this time were grouped as a function of the cover values of the species, which shows that stages exist in the vegetation recovery of these communities. The damage produced by fire in the community is minor, as a rapid recovery of the vegetation was observed, so controlled burning is a useful tool in the management of these heathlands.

Keywords: Wet heathland, traditional uses, community dynamics, multivariate analysis.

Resumen

Muñoz, A., Álvarez, R., Pesqueira, X.M., García-Duro, J., Reyes, O. & Casal, M. (2010). La quema en la gestión de matorrales de *Erica ciliaris* y *Erica tetralix*: efectos sobre la estructura y diversidad. *Nova Acta Científica Compostelana (Bioloxía)*, **19**: 69-81

¿Podría utilizarse la quema controlada como herramienta en la gestión de brezales de *Erica ciliaris* y *Erica tetralix*? Se seleccionaron dos comunidades de *E. ciliaris* y *E. tetralix,* fijando en cada una dos parcelas de 5x5m que se caracterizaron en base a sus valores de frecuencia, cobertura y composición específica de la vegetación. Posteriormente se sometieron a una quema experimental, tras la cual las parcelas se muestrearon dos veces al año durante los cuatro años y medio siguientes. Los resultados indican que la cobertura de especies leñosas alcanza en poco tiempo los valores de la parcela Control. La diversidad y la composición específica no sufren cambios notables durante estos años, sin embargo la heterogeneidad temporal indica que los principales cambios se producen en los primeros 18 meses de sucesión secundaria. Los análisis multivariantes indican que los muestreos realizados durante estos años se agrupan en función de los valores de cobertura de las especies, lo que nos indica que existen etapas en la recuperación de la vegetación de estas comunidades. El daño que produce el fuego en la comunidad es leve, observándose una rápida recuperación de la vegetación, por lo que se concluye que las quemas controladas constituyen una buena herramienta en la gestión de estos brezales.

Palabras clave: Humedales, usos tradicionales, dinámica de la comunidad, análisis multivariantes.

INTRODUCTION

Fire has been used as a management tool for improving and extending grazing areas and is considered to be a very important factor in both the formation and maintenance of many European heathlands, principally for two reasons (GIMINGHAM *et al.*, 1979; HOBBS & GIMINGHAM, 1984; WEBB, 1986; COULSON *et al.*, 1992): 1) It modifies the vegetation structure, preparing the regeneration of woodland, which is the climax habitat of most of the heathlands (IZCO *et al.*, 1999; SYMES & DAY, 2003), favouring recovery of the shrubland through resprouting and/or germination (CASAL, 1987; CASAL *et al.*, 1984; ÁLVAREZ *et al.*, 2005, 2007). 2) It assures a low concentration of nutrients over the long term (WEBB, 1986), which is a normal characteristic of the shrubland communities. One of the effects of fire is that the nutrients that are retained in an inaccessible form in the plants are converted into a more readily assimilated form through the increase in fertility of the soil (COULSON *et al.*, 1992). However, at the same time, a loss in the cation exchange capacity is produced that facilitates the emigration of certain nutrients from the soil, reducing the initial fertility. On the other hand, some nutrients are transformed during combustion into gasses or minute particles and abandon the area of the fire in the smoke and in the convections caused by the heat. Others are retained in the ashes and pass to the soil, becoming available to the plants once more (COULSON *et al.*, 1992).

The plant regeneration after fire can follow two reproductive strategies, resprouting and germination, that have been studied by numerous authors (CALVO *et al.*, 1998; PEREIRAS & CASAL, 2002; VALBUENA & VERA, 2002; ÁLVAREZ *et al.*, 2005, 2007; CLARKE *et al.*, 2005; RIVAS *et al.*, 2006). The resprouting capacity of shrubland species are influenced by diverse factors, such as the age of the plant when it was burnt, or the intensity, frequency and period of the fire (GILL & GROVES, 1981; GIMINGHAM, 1981; CASAL, 1987; REYES *et al.*, 2000; PEREIRAS & CASAL, 2002; PEREIRAS *et al.*, 1990). The germinative capacity as a response to fire depends on the species, life cycle of the plants, type of fire and local post-fire conditions (REYES *et al.*, 1997; ÁLVAREZ *et al.*, 2005, 2007). In most of the species present in Galicia belonging to the Cistaceae, Ericaceae and Leguminosae families, heat shocks stimulate germination (REYES & CASAL, 2008; REYES *et al.*, 2000). These two reproductive strategies are not exclusive, as some species reproduce both by resprouting and by germination.

During many centuries, the Galician shrubland communities have been subject to human activities such as burning, cutting and grazing. For this reason they are considered to be a part of anthropised landscapes that require a certain amount of perturbation as a form of sustainable management. The abandonment of farming in the last few years has caused a decrease in these types of ecosystems, not only in Galicia, but in a large part of Europe (GIMINGHAM *et al.*, 1979; ROSE *et al.*, 2000; IZCO *et al.*, 2006). There are many studies that highlight the ecological, landscaping, educative and cultural importance of *Erica ciliaris* Loefl. ex L .and *Erica tetralix* L. communities (SMIDT, 1995; BRAGG & TALLIS, 2001). This importance is recognised in the Habitats Directive (92/43/EEC) in which they are considered to be Priority Habitats Types. The recovery of the traditional customs, such as controlled burning, would help in the conservation of this type of heathland that depend, to a great measure, on their inclusion in natural conservation programmes (GIMINGHAM, 1981).

The general objective of this study is to understand how the composition, structural and ecological characteristics of *E. ciliaris* and *E. tetralix* communities respond to cultural burning. The hypothesis posed is that: 1) Fire causes a change in the structure, diversity and species composition *E. ciliaris* and *E. tetralix* communities, recoverable in a short time. 2) These communities have a great resilience after cultural burning.

METHODS

Field methods

Two heathland communities were located and selected in Galicia (NW Spain), sampling sites of Baio and Masgalán, both of one hectare and ten years since their last perturbation. The altitude of the areas studied is of 100 and 800 m, respectively. The annual rainfalls is around 1650 mm for both, whilst average temperatures is between 10º C for Masgalán and 14º C for Baio, with minimums of 5º C and maximums of 17º C (NINYEROLA *et al.*, 2005).

In each community, two 5 x 5m Control Plots (CP) were established in order to characterise the vegetation on the basis of cover, frequency and species composition, being the final data the average of them both. Afterwards, these plots were subjected to a controlled burning (CB), with the intention of studying the community dynamics after such a perturbation. The Baio station was burnt in October 2001 and the Masgalán station in March 2002. After they were burnt, these same plots were sampled twice a year, in spring and autumn, during the following four and a half years, following a diachronic approach. The first sampled was performed a few months after the controlled burning. To record the Species Composition, an inventory was made of all the plants, woody and herbs species, that were present in each of the samplings carried out. The Frequency values were calculated from the presence - absence data for each of the woody and herbs species obtained using a 30 x 30 cm quadrat placed randomly a total of 30 times in each plot. To determine the Vertical structure, the percentage occupied by the vegetation was estimated at six previously established vertical layers, proposed by GODRON *et al.* (1968) and later modified by BASANTA *et al.* (1988). Finally, the Linear Cover was measured using five 5m long parallel lines in each 5 x 5 m plot, separated by a distance of one metre. The centimetres occupied in each line by each woody species and by the Bare Ground (Herbs, Bryophytes, Necromass, Soil, Litter, Stones and Blocks) were noted. The cover of Herbs and Bryophytes was taken as a single variable, without differentiating the species. Only when there was no overlayering of woody vegetation was the ground considered to be Bare Ground.

Data treatment

The Life Forms of all species present in each samplings were identified according to RAUNKIAER (1934) modified by AIZPURU *et al.* (1999). For each life form, the relative abundance (%) was calcutale, using the presence-absence data of frequencies. The percentage of overlayering of the woody species was calculated from the linear cover data using the formula proposed by BASANTA *et al.* (1988) and modified later by PEREIRAS *et al.* (1990). Various diversity measurements were calculated for the woody and herbs species linear cover data, applying the Shannon Diversity Index (H´), the Simpson Dominance Index (λ) and the Pielou Equitability Index (J). The Margalef Index (MARGALEF, 1974) was also used to calculate Temporal Heterogeneity.

A PCA and a Cluster Analysis was applied to the cover data of the woody species that were present in more than 10% of the samplings, using Statgraphics Plus 5.1. The Cluster Analysis was carried out using the UPGMA Method. To establish significant differences between the samplings in both sites, the Wilconxon Test for related pairs (α=0.05) was applied, using SPSS 15.0.

RESULTS

Species composition and life forms

A total of 35 and 25 species were found in Masgalán and Baio, respectively, in the total of samples performed. The woody and herbs species present in the Control Plots of both sampling sites are shown in Table I. The average number of woody species present in each sampling in both stations is similar and practically constant after the controlled burning (≈ 6 species). However, there is a greater variability in the number of herbs species (Table II). In Baio, *Carex durieui* Steud. ex Kunze, endemic in the NW of the Iberian Peninsula, dominates among the herbs species present, followed by *Potentilla erecta* (L.) Raeusch. and *Molinia caerulea* (L.) Moench (Fig. 1). These three species show a slow increase in abundance over a period of time. However, in Masgalán, *M. caerulea* stands out, followed by *Pseudoarrenatherum longifolium* (Thore) Rouy (whose abundance notably increases in the final years) and *P. erecta*.

TABLE I. List of species in Control Plots (CP) of Baio (B) and Masgalán (M)

Species	Sampling Site
Woody species	
Calluna vulgaris (L.) Hull	M
Daboecia cantabrica (Huds.) K. Koch	M
Erica ciliaris Loefl. ex L.	B,M
Erica cinerea L.	M
Erica erigena R. Ross	B
Erica tetralix L.	B,M
Genista berberidea Lange	B
Salix atrocinerea Brot.	B
Ulex gallii Planch.	B
Ulex minor Roth	M
Herbs species	
Carex durieui Steud. ex Kunze	B
Cuscuta epithymum (L.) L.	M
Danthonia decumbens (L.) DC.	B
Eriophorum angustifolium Honck.	M
Globularia nudicaulis L.	M
Hypericum elodes L.	M
Molinia caerulea (L.) Moench	B,M
Narthecium ossifragum (L.) Huds.	M
Parnassia palustris L.	M
Potentilla erecta (L.) Raeusch.	B
Pseudarrhenatherum longifolium (Thore) Rouy	M
Schoenus nigricans L.	B

With respect to life forms (Fig. 2), the temporal pattern observed in both sampling sites was characterised by a dominance of the Chamaephytes over the rest, followed by the group of Hemicriptophytes, despite the fact that the number of Hemicriptophytes is higher in the samplings from both stations, representing from 40-60% of total species. In the Control Plots of both sites, Chamaephytes are dominant (>75%), Hemicriptophytes form the second most important group with an abundance of 20%, whilst Phanerophytes are scarcely represented by *Salix atrocinerea* Brot. (<1%) in Baio. After burning, Chamaephytes dominate with percentages which varie from 55-70%. Hemicriptophytes are the second most abundant species in all the samplings (25-40%), increasing their abundance with respect to the Control. The presence of Geophytes is notable in Baio after 18 months (close to 2%). Their importance increases during the following 18 months and reaches its maximum in the samplings performed at 37 months (>8%), whilst they vary during the four and a half years of samplings in Masgalán.

Vertical structure

Baio (Fig. 3) is characterised by the presence in the Control Plots of an almost total cover in the three intermediate layers (5 to 100 cm), which decreases to 60% in the next layer. After the controlled burning, a slow recovery of the lower layers was observed after the first 18 months (< 75% in any sampling or layer). Layer II have the greatest cover in succeeding samplings (maximum 90%), until the sampling at 55 months, in which Layer III (25-50 cm) was greater than the rest of the layers (80%). In Masgalán (Fig. 3), the Control Plots showed the greatest cover in the two intermediate layers (5-50 cm), together with stunted vegetation of not higher than one metre. However, after the controlled burning, the

TABLE II. Number of woody species (WS) and herbs species (HS) present in the sampling sites of Baio and Masgalán along time (months after burning) after controlled burning and in the Control Plots (CP)

TIME		**7**	**12**	**18**	**25**	**30**	**37**	**45**	**50**	**55**	**CP**
Baio	WS	6	6	6	7	6	6	6	7	6	6
	HS	4	6	12	7	9	10	9	7	10	5
		4	**14**	**16**	**23**	**26**	**33**	**40**	**46**	**50**	**CP**
Masgalán	WS	4	6	6	5	5	7	6	6	6	6
	HS	6	17	13	9	11	12	11	11	15	8

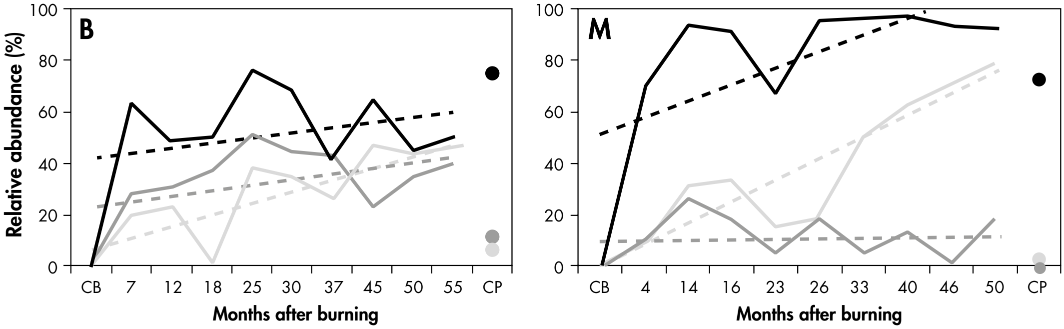

Fig.1. Relative abundance (frequency, %) of the dominant herbaceous species, continuous line, and the regression line of each one, dashed line, in Baio (B) and Masgalán (M) after controlled burning (CB) and in the Control Plots (CP). In Baio: —— *Carex durieui,* —— *Potentilla erecta* and —— *Molinia caerulea.* In Masgalán: —— *Molinia caerulea,* —— *Pseudarrhenatherum longifolium* and —— *Potentilla erecta.*

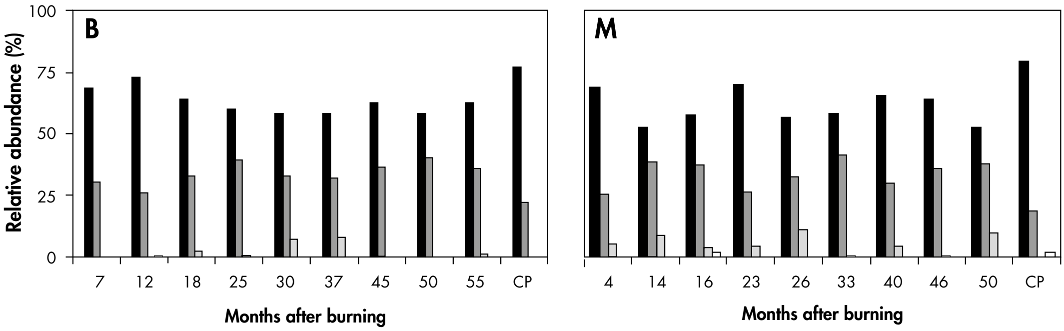

Fig.2. Relative abundance (frequency, %) of the life forms in after burning and in the Control Plots (CP) in the sampling sites of Baio (B) and Masgalán (M). Phanerophytes, Chamaephytes, Hemicriptophytes, Geophytes and Therophytes.

two lowest cover layers recovered very quickly, showing values of 90% in the first sampling. Layer III soon reached cover values of around 75% (26 months) and these three layers continued to present the largest cover values during the rest of the samplings performed.

Linear cover and overlayering

In the first sampling performed in both stations after burning, the cover of woody species did not reach 50% of the total cover and Bare Ground dominated, but in the following sampling performed the woody cover became dominant,

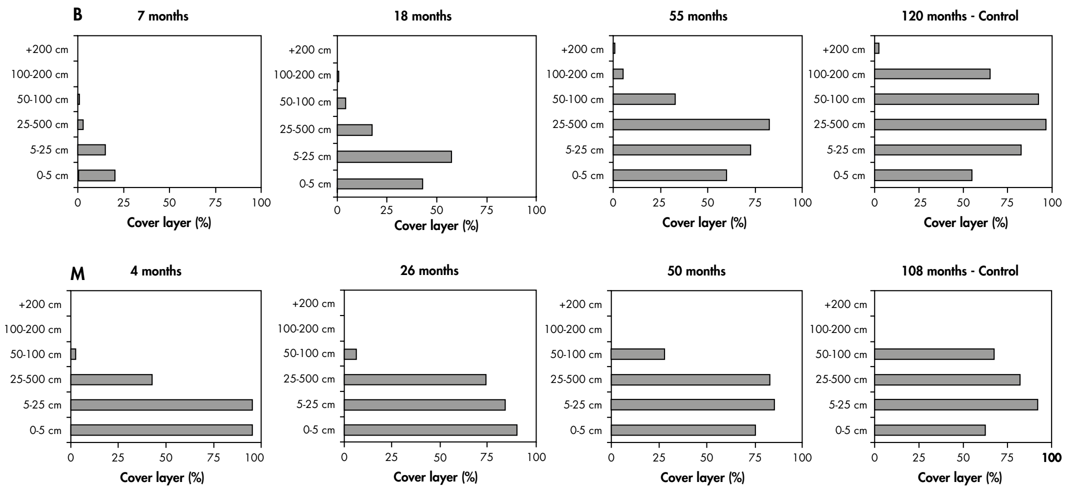

Fig. 3. Vertical structure of the vegetation in some samplings in Baio (B) and Masgalán (M)

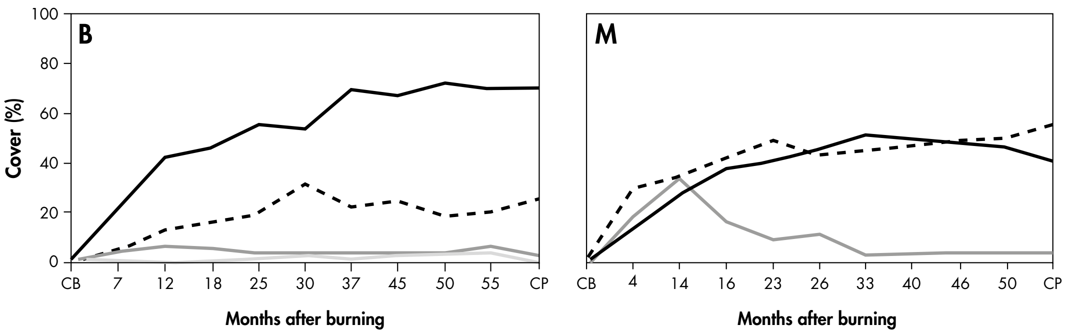

Fig. 4. Cover (%) of the different ecological groups (– – Ericaceae, — Leguminosae, — Other woody species and — Herbs) after control burning (CB) and in the Control Plots (CP), in Baio (B) and Masgalán (M).

increasing rapidly until they reached cover values (90-95%) very similar to the Control Plots in both Baio and Masgalán. *E. tetralix* and *E. ciliaris* are dominant in Baio and recovered their cover values very quickly after the burning, reaching a combined cover of 60% of total cover. On the other hand, *Ulex minor* Roth followed by *E. ciliaris* dominate in Masgalán with a combined cover reaching 70% of total cover. Bare Ground decrease their cover as time passes. There is a rapid recovery of these species, probably due to the resprouting capacity of most of these species The overlayering values increase following the pattern of cover values of the woody species, reaching the Control Plots values (95%) in both sites in a few months.

On analysing the cover values as a function of the established groups, Leguminosae, Ericaceae, Other Woody Species and Herbs (Fig. 4), similar behaviour was observed in them all, characterised by the dominance of the woody shrub species. The Ericaceae clearly dominated in Baio and the Leguminosae is the most abundant in most of the samplings performed in Masgalán, where an important occupation of Herbs was observed during the first 14 months after the burning.

Diversity measurements

Few temporal changes happen in the values of diversity, dominance and evenness (Fig. 5). In Baio and Masgalán, the diversity and evenness present the highest values in the Control Plots.

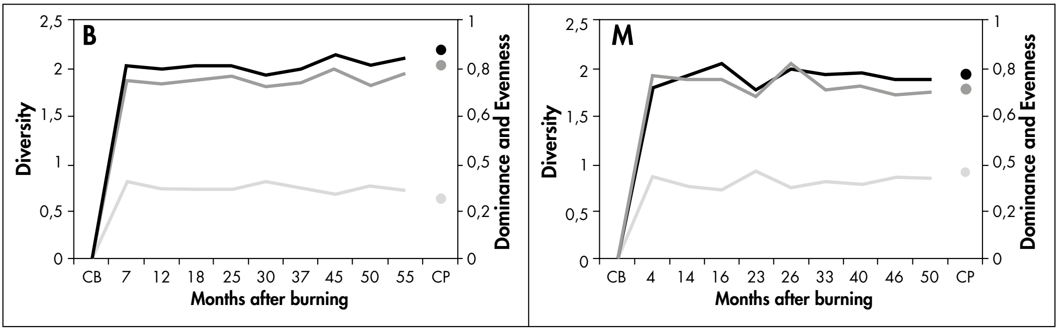

Fig. 5. Temporal evolution of diversity (—), dominance (—) and evenness (—) in the Control Plots (CP) in Baio (B) and Masgalán (M) after controlled burning (CB).

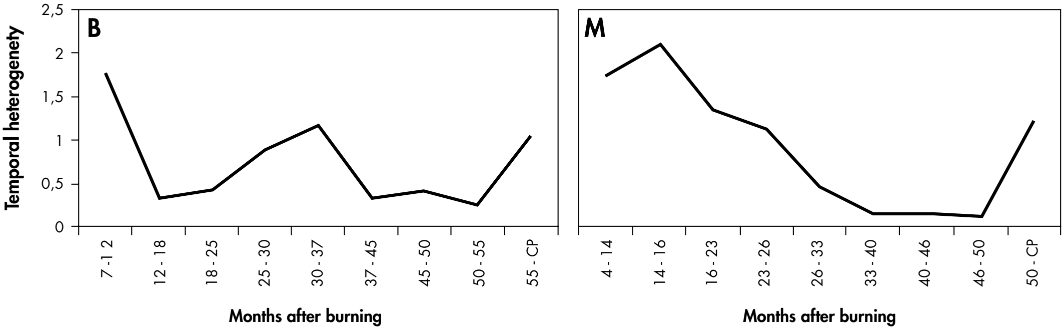

Fig. 6. Temporal heterogeneity according to the time passed since the controlled burning and the Control Plots (CP) in Baio (B) and Masgalán (M).

After controlled burning, both diversity and evenness experiment a little increase with time, but none of the samplings reaches the values of the Control Plots. Dominance decreases with time in Baio after controlled burning, while in Masgalán it remains practically constant. The number of woody species can be considered to be practically constant, as this increase is due to the appearance of species with very scarce cover (< 1%), *U. europaeus* L. in Baio and *Lithodora diffusa* (Lag.) I.M. Johnston and *Erica cinerea* L. in Masgalán, for which reason the increase of diversity is due to changes in the abundance of the species present.

The temporal heterogeneity values (Fig. 6), calculated between consecutive samplings, indicate that the maximum inter-sample variability is registered in the first stages of secondary succession. Althought temporal heterogeneity shows little differences between both sampling sites, it is clear that the most notable changes in both communities occur during the first 18 months after the controlled burning.

Multivariate analysis

The initial data matrix consists of 10 cases in both Baio and Masgalán, corresponding to the total of samplings and the Control Plots, and seven and six variables in Baio and Masgalán, respectively (Fig. 7 and Fig. 8). The variant absorption percentage for the first two axes for each of the PCA is higher than 89%. In Baio and Masgalán, the samplings performed during the four and a half years are distributed along the Component I from left to right as a function of

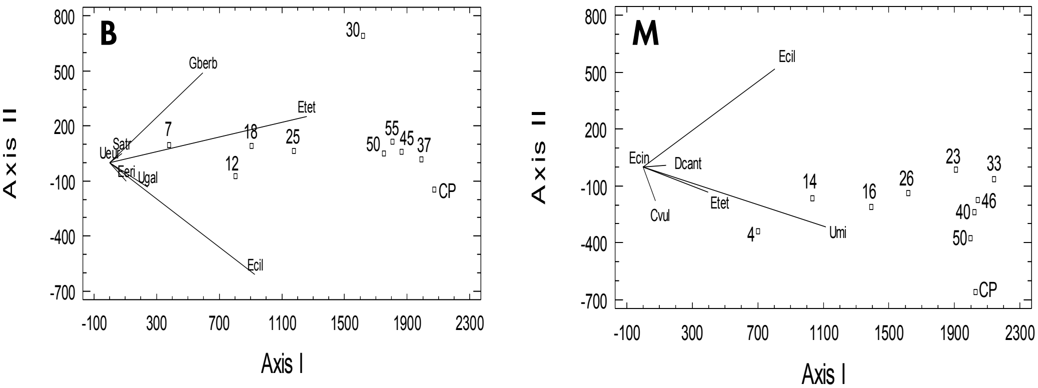

Fig. 7. Situation of samplings and species in the plane defined by the first two axes of PCA for Baio (B) and Masgalán (M). The number by each sample indicates the recovery time (in months) after burning. *Calluna vulgaris* (Cvul); *Daboecia cantabrica* (Dcant); *Erica ciliaris* (Ecil); *Erica cinerea* (Ecin); *Eerica erigena* (Eeri); *Erica tetralix* (Etet); *Genista berberidea* (Gberb); *Salix atrocinerea* (Satr); *Ulex europaeus* (Ueur); *Ulex gallii* (Ugal); *Ulex minor* (Umi).

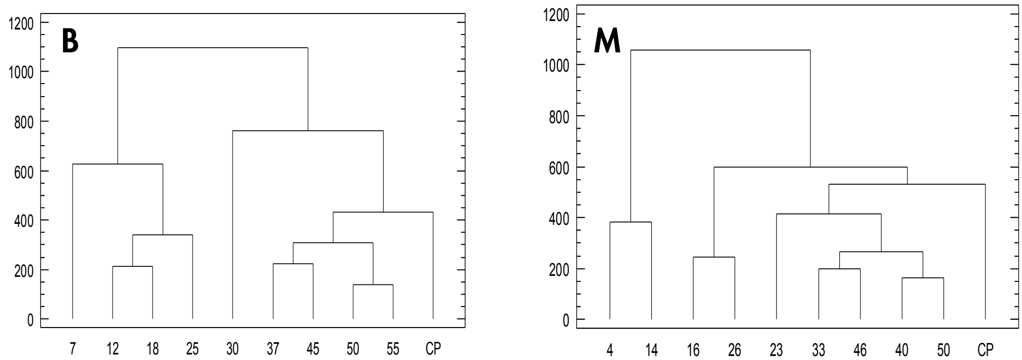

Fig. 8. Dendrogram resulting from the UPGMA grouping of all the samples performed and the Control Plot (CP) in Baio (B) and Masgalán (M). The number by each sample indicates the recovery time (in months) after burning.

the time passed since the burning. The biplot (Fig. 7) shows that the most influential species in the ordination of samplings in Baio are *E. tetralix* and *E. ciliaris*, with an abundance that increases with time after the perturbation. Whilst in Masgalán, the most influential species in the ordination of samplings are *U. minor* and *E. ciliaris*, whose cover increases with the age of the community.

In both stations, the Cluster Analysis groups the samplings (Fig. 8). The Wilconxon test suggests that significant differences exist ($p<0.05$) between the samplings. As a function of the significant differences detected and the Cluster Analysis, these samplings can be grouped in stages. The stages established for each station are significantly different ($p<0.05$) from each other. In Baio, the samplings would be grouped in three stages in the following manner: Stage 1, formed by the first sampling performed after the burning (B7); Stage 2, formed by the samplings performed between 12 and 25 months after the burning (B12, B18 and B25); Stage 3, formed by the rest of the samplings together with the Control Plots (B30, B37, B45, B50, B55 and

B120). In Masgalán, the three stages would be formed by the following samplings: Stage 1, formed by the first two samplings after the burning (M4 and M14); Stage 2, formed by the samplings performed between 16 and 26 months (M16, M23 and M26); Stage 3, formed by the rest of the samplings together with the Control Plots (M33, M40, M46, M50 and M 108).

DISCUSSION

The recovery pattern of vegetation after controlled burning is similar in both stations, although slightly faster in Masgalán. This could be influenced by various factors, like the burning season, Masgalán was burnt in spring, whilst Baio was burnt in autumn. Existing studies demonstrate that vegetation subjected to burning in spring recovers faster than when burnt in autumn (Gill & Groves, 1981; Casal, 1987). On the other hand, the species composition of the two stations is quite different, which shows that species variability exists between *E. ciliaris* and *E. tetralix* communities, an aspect included in the studies of Rodríguez Oubiña (1982, 1987). This variability is observed at the family level of woody species, as, in Baio, the Ericaceae dominates over the Leguminosae, whilst the contrary occurs in Masgalán. The fact that a different family dominates each community influences the trend in the recovery of the vegetation, as not all of the species have the same resprouting intensity. Pereiras *et al.* (1990) found that, in different shrubland ecosystems of Galicia, the Leguminosae species regain their covers after fire before the Ericaceae species.

These heathlands present a large number of vegetal species, as much herbs as woody, with a markedly hydrophilic character, some of which are exclusive to these communities (Basanta *et al.*, 1988; 1989). Many of the species present in Baio and Masgalán are species adapted to waterlogged, a condition they suffer during a large part of year. Examples would be *M. caerulea, Genista berberidea* Lange or *E. tetralix* (Basanta *et al.*, 1988). On the other hand, there are species with a wider ecological range, such as: *Calluna vulgaris* (L.) Hull*, E. cinerea, U. europaeus* and *Erica erigena* R. Ross.

These heathlands have a very characteristic qualitative and quantitative composition, which differentiates them from each other and from the rest (Rodríguez Oubiña, 1982, 1987; Basanta *et al.*, 1989) and are, together with the *Erica umbellata* Loefl. ex L. shrublands, those that have the largest number of species within the shrublands studied by Basanta *et al.* (1989). However, important differences at the level of life forms do not exist, as in both stations there is a clear dominance of Chamaephytes, with the Hemicriptophytes forming the second most abundant group. Gimingham *et al.* (1979) analysed the spectrum of life forms in two examples of heath communities in Western Europe and the two most abundant types with regard to the number of species were Chamaephytes and Hemicriptophytes, with a dominance of the former. The scarce representation of the Therophytes coincides with the results obtained by other authors in humid shrublands (Basanta *et al.*, 1988, 1989).

E.ciliaris and *E. tetralix* heathlands are characterised by the high cover and stratification values of the shrub species. Bare Ground is scarce and is generally occupied by herbs species (Basanta *et al.*, 1988). However, these values change after the controlled burning. During the first year of secondary succession, the cover value of Bare Ground is higher than the sum of the cover values of the woody species in both stations, a situation found in studies of other types of shrublands, such as those of *U. europaeus* (Casal, 1987; Pereiras & Casal, 2002), those of *Erica australis* L. (Calvo *et al.*, 1998, 2002b) or those of *Cistus* spp. (Tárrega *et al.*, 1998). Three years after the controlled burning, the linear cover values of the woody species are very similar to the Control Plots. Pereiras & Casal (2002) found in their study that, in the stations subject to controlled burning, the cover developed very quickly, producing the greatest changes in the first three years, trends that were also found by other authors (Casal *et al.*, 1984; Calvo *et al.*, 2002a). Pereiras *et al.* (1990) describe the same pattern in different communities of *U.*

europaeus shrublands, but underscore a more rapid recovery in shrublands with deep and humid soils. Moreover, the growth of the vegetation is influenced by the climate of each zone and also by the meteorological conditions after the controlled burning, the fire intensity, the plants age, the duration of waterlogged period, the nutrients level in soil, the pre-existing vegetation, etc. (GILL & GROVES, 1981; GIMINGHAM, 1981; BERENDSE, 1994; SANSEN & KOEDAM, 1996; ROEM *et al.*, 2002; CLARKE *et al.*, 2005). The overlayering values obtained in both stations are similar to those obtained by PEREIRAS *et al.* (1990). This means that after a moderately intense controlled burning, the shrublands very quickly recover their cover values of woody species, independently, in this case, of the type of community.

The recovery pattern of species after fire is strongly influenced by the life history and morphological attributes of the species of the burnt communities (NORTON & DE LANGE, 2003) and by the life form to which they belong, or the regeneration strategies they may have. Attention should be drawn to the fact that all the species present in Baio are resprouters: *Daboecia cantabrica* (Huds.) K. Koch, *E. cinerea, E. ciliaris, E. erigena, E. tetralix*, *G. berberidea*, *Ulex gallii* Planch., *U. europaeus* and *U. minor* (MALLIK & GIMINGHAM, 1985; STOKES *et al.*, 2003, 2004; REYES & CASAL, 2008; REYES *et al.*, 2000). Resprouting was not observed in the specie *C. vulgaris*, present in Masgalán, germination being its method of recruiting new individuals. This specie germinated with a scarce representation 18 months after the controlled burning, slightly increasing its cover at a later date. There are authors who affirm that *C. vulgaris* is seeder (ANDRÉS & OJEDA, 2002; NILSEN *et al.*, 2005), but many others defend the resprouting capacity of this species, although this aspect can be frustrated as a function of the age of the plant or the frequency of fire (GIMINGHAM, 1981; GIMINGHAM *et al.*, 1979; MALLIK & GIMINGHAM, 1985; WEBB, 1986). In other studies, the germination of *C. vulgaris* after burning is produced in the second year (CALVO *et al.*, 2002a) or even in the third (SEDLÁKOVA & CHYTRÝ, 1999). *L. diffussa* is an obligate seeder (ANDRÉS & OJEDA, 2002). According to a classification proposed by REYES & CASAL (2008), most of the species present in these two stations are characterised by a high resprouting intensity (*E. erigena, E. tetralix, G. berberidea* and *U. europaeus*), a medium resprouting intensity (*D. cantabrica, E. ciliaris, U. gallii* and *U. minor*) and a low resprouting intensity (*C. vulgaris*), which explains their high recuperation intensity.

The Diversity and Evenness values are fairly high, a characteristic found by BASANTA *et al.* (1988, 1989) within the shrublands that they studied. The diversity shows the same behaviour found in other types of shrublands in Galicia (PEREIRAS & CASAL, 2002). The diversity increases due to a progressively more equal sharing between the species, a pattern that is repeated in the study made by PEREIRAS & CASAL (2002), which is practically unaffected by the change in the number of species. The temporal heterogeneity is high in the first samplings and decreases as the succession advances, which indicates that the differences are attenuated over time. The same pattern was found in the studies made by CALVO *et al.* (2002b).

The structural differences detected with the Multivariate Analysis permits the differentiation of three stages in each of the Sampling Stations. This coincides with the results obtained by PEREIRAS & CASAL (2002) in the *U. europaeus* shrubland, in which they found a clear and significantly different structural divergence between the first year and those following in the succession. In the case of the two *E. ciliaris* and *E. tetralix* communities studied, the results obtained indicate that most of the differences in the cover community are detected in the first year after the perturbation, as, in both stations, the samplings performed in the first year show significant differences to the rest. This appears to be logical, because the community manifests the most notable changes during the first year. The perturbation causes an enormous decrease in the aereal biomass that changes a series of resources such as light, water or nutrients, which are used by the opportunist species, most of which are herbs according to the data obtained. During this first year, the woody species begin a rapid process of recovery because all the woody species, excepting *Calluna vulgaris* (L.) Hull and *L. diffusa*, are resprouters. On the other hand, it should be noted that the community looses structure, due to the burning. A dominance of the Bare Ground group is observed in the cover values during the first stage, which is characterised by a functional and structural instability that decreases as the community matures and the structure and dominant species become more defined.

CONCLUSIONS

E. ciliaris and *E. tetralix* communities were characterised by a large number of species with a clear dominance of Chamaephytes, principally represented by shrub species, followed by the Hemicriptophytes. The intermediate vertical layers (5-50 cm) are those that present the most cover, whilst the woody species dominate in horizontal cover and Bare Ground is very scarce. Both of the communities studied show high diversity and evenness values with low levels of dominance. After burning, the principle changes are produced in the following year, both at the level of vertical and liner cover. The controlled burning produced a loss of vertical structure that was quickly recovered. On the other hand, the Bare Ground cover dominated during this first year, but the cover of woody and herbs species recovered rapidly and, as a consequence, increased the overlayering values. The number of herbs species varied along the experiment reaching a maximum at 12-18 months after the burning. With regard to diversity, very few changes were observed, although, in general, the values are lower than in the Control Plots. Structurally, the samplings are grouped in three significantly different stages, from which it can be deduced that the communities take about 24 months to recover their structure. The damage produced by fire to the community is minor, as a rapid recuperation of the vegetation is observed, with little loss of diversity, number of species or layers. Therefore, controlled burning could be used as a sustainable management tool in this type of shrubland community.

ACKNOWLEDGEMENTS

We would like to thank Doctor Elvira Díaz-Vizcaíno for the methodological help given during the realisation of this study. We would also like to thank the personnel of the Forestry Districts of Vimianzo and Lalín for their help with the controlled burning in the communities studied. This study has been carried out with the support of the Xunta de Galicia with two projects (PGIDT00 MAM20001PR and PGIDIT02RFO20001PR) and a Pre-Doctoral Grant for a thesis.

REFERENCES

Aizpuru, I., Aseginolaza, P.M., Uribe-Echebarría, P.M., Urrutia, P. & Zorrakin, I. (1999). *Flora del País Vasco y territorios limítrofes*. Servicio Central de Publicaciones del Gobierno Vasco: Vitoria-Gasteiz.

Álvarez, R., Valbuena, L. & Calvo, L. (2005). The influence of tree age on seed germination response to environmental factors and inhibitory substances in Pinus pinaster. *Int. J. Wildland Fire*, **14**: 277-284.

Álvarez, R., Valbuena, L. & Calvo, L. (2007). Effect of high temperatures on seed germination and seedling survival in three pine species (*Pinus pinaster, P. sylvestris* and *P. nigra*). *Int. J. Wildland Fire*, **16**: 63-70.

Andrés, C. & Ojeda, F. (2002). Effects of afforestation with pines on woody plant diversity of Mediterranean heathlands in southern Spain. *Biodivers. Conserv.*, **11**: 1511-1520.

Basanta, M., Díaz-Vizcaíno, E. & Casal, M. (1988). Structure of shrubland communities in Galicia (NW SPAIN). *In*: During, H.J., Werger, M.J.A. & Willems, J.H. (Eds.), *Diversity and patterns in plant communities*: 25-36. SPB Academic Publishing, The Hague, The Netherlands.

Basanta, M., Díaz-Vizcaíno, E., Casal, M. & Morey, M. (1989). Diversity measurements in shrubland communities of Galicia (NW Spain). *Vegetatio*, **82**: 105-112.

Berendse, F. (1994). Litter decomposability: a neglected component of plant fitness. *J. Ecol.*, **82**: 187-190.

Bragg, O.M. & Tallis, J.H. (2001). The sensitivity of peat-covered upland landscapes. *Catena*, **42**: 345-360.

Calvo, L., Tárrega, R. & Luis, E. (1998). Space-time distribution patterns of *Erica australis* L. subsp. *aragonensis* (Willk) after experimental burning, cutting and ploughing. *Plant Ecol.,* **137**: 1-12.

Calvo, L., Tárrega, R. & Luis, E. (2002a). Regeneration patterns in a *Calluna vulgaris* heathland in the Cantabrian mountains (NW Spain): effects of burning, cutting and ploughing. *Acta Oecol.*, **23**: 81-90.

Calvo, L., Tárrega, R. & Luis, E. (2002b). The dynamics of Mediterranean shrub species over 12 years following perturbations. *Plant Ecol.*, **160**: 25-42.

Casal, M. (1987). Post-fire dynamics of shrublands dominated by Papilionaceae plants. *Ecol. Mediterranea*, **XIII 4**: 87-98.

Casal, M., Basanta, M. & García-Novo, F. (1984). *La regeneración de los montes incendiados en Galicia*. Monografía Nº 99. Universidad de Santiago de Compostela, Santiago de Compostela.

Clarke, P.J., Knox, K.J.E., Wills, K.E. & Campbell, M. (2005). Landscape patterns of woody plant response to crown fire: disturbance and productivity influence sprouting ability. *J. Ecol.*, **93**: 544-555.

Coulson, J.C., Fielding, C.A. & Goodyer, S.A. (1992). *The management of moorland areas to enhance their nature conservation interest*. JNCC Report Nº 134. Joint Nature Conservation Committee, Peterborough.

Gill, A.M. & Groves, R.H. (1981). Fire regimes in heathlands and their plant-ecological effects. *In*: Specht, R.L. (Ed.), *Heathlands and Related Shrublands. Analytical Studies*: 61-84. Elsevier Scientific Publishing Company, Amsterdam.

Gimingham, C.H. (1981). Conservation: European Heathlands. *In*: Specht, R.L. (Ed.), *Heathlands and Related Shrublands. Analytical Studies*: 249-260. Elsevier Scientific Publishing Company, Amsterdam.

Gimingham, C.H., Chapman, S.B. & Webb, N.R. (1979). European Heathlands. *In*: Specht, R.L. (Ed.), *Heathlands and Related Shrublands. Descriptive Studies*: 131-142. Elsevier Scientific Publishing Company, Amsterdam.

Godron, M. *et al.* (1968). *Code pour le relevé méthodique de la végétation et du milieu*. CNRS, Paris.

Hobbs, R.J. & Gimingham, C.H. (1984). Studies on fire in Scottish heathland communities. I. Fire characteristics. *J. Ecol.*, **72**: 223-240.

Izco, J., Amigo, J., García San León, D. (1999). Análisis y clasificación de la vegetación leñosa de Galicia (España). *Lazaroa*, **20**:29-47.

Izco, J., Amigo, J., Ramil-Rego, P., Díaz, R. & Sánchez, J.M. (2006). Brezales: biodiversidad, usos y conservación. *Recursos Naturais*, **2**: 5-24.

Mallik, A.U. & Gimingham, C.H. (1985). Ecological effects of heater burning II. Effects on seed germination and vegetative regeneration. *J. Ecol.*, **73**: 633-644.

Margalef, R. (1974). *Ecología*. Omega, Barcelona.

Nilsen, L., Johansen, L. & Velle, L. (2005). Early stages of *Calluna vulgaris* regeneration after burning of coastal heath in central Norway. *Appl. Veg. Sci.*, **8**: 57-64.

Ninyerola, M., Pons, X. & Roure, J.M. (2005). *Atlas climático digital de la Península Ibérica. Metodología y aplicaciones en bioclimatología y geobotánica*. Universidad Autónoma de Barcelona, Bellaterra, Spain.

Norton, D.A. & De Lange, P.J. (2003). Fire and Vegetation in a Temperate Peat Bog: Implications for the Management of Threatened Species. *Conserv. Biol.*, **17**: 138-148.

Pereiras, J. & Casal, M. (2002). Dynamics of an *Ulex* shrubland community subjected to prescribed burning. *In*: Trabaud, L. & Prodon, R. (Eds.), *Fire and Biological Processes*: 43-56. Backhuys Publishers Leiden, The Netherlands.

PEREIRAS, J., CASAL, M., MONTERO, R. & PUENTES, A. (1990). Post-fire secondary succession under different ecological conditions in shrub vegetation of Galicia (NW SPAIN). *In*: Universidade de Coimbra (Ed.), *Proceedings of International Conf. Forest Fire Research*: 1-15. Coimbra.

RAUNKIAER, C. (1934). *The life form of plants and statistical plant geography*. Clarendon Press, Oxford.

REYES, O. & CASAL, M. (2008). Regeneration Functional Models and Plant Functional-Regenerative Types, related to the intensity of fire in Atlantic shrubland and woodland species. *J. Veg. Sci.*, **19**: 575-583.

REYES, O., BASANTA, M., CASAL, M. & DÍAZ-VIZCAÍNO, E. (2000). Functionating and dynamics of woody plant ecosystems in Galicia (NW Spain). *In*: Trabaud, L. (Ed.), *Life and Environment in the Mediterranean*: 1-41. WIT Press Southampton, Boston.

REYES, O., CASAL, M. & TRABAUD, L. (1997). The influence of population, fire and time of dissemination on the germination of *Betula pendula* seeds. *Plant Ecol.*, **133**: 201-208.

RIVAS, M., REYES, O. & CASAL, M. (2006). Influence of heat and smoke treatments on the germination of six leguminous shrubby species. *Int. J. Wildland Fire*, **15**: 73-80.

RODRÍGUEZ OUBIÑA, J. (1982). *Brañas en Galicia meridional: ecología, flora y vegetación*. Tesis de Licenciatura (Inéd). Universidad de Santiago de Compostela, Spain.

RODRÍGUEZ OUBIÑA, J. (1987). *Estudio fitosociológico de las brañas de la provincia de A Coruña*. Tesis Doctoral (Inéd.). Universidad de Santiago de Compostela, Spain.

ROEM, W.J., KLEES, H. & BERENDSE, F. (2002). Effects of nutrient addition and acidification on plant species diversity and seed germination in heathland. *J. Appl. Ecol.*, **39**: 937-948.

ROSE, R.J., WEBB, N.R., CLARKE, R.T. & TRAYNOR, C.H. (2000). Changes on the heathlands in Dorset, England, between 1987 and 1996. *Biol. Conserv.*, **93**: 117-125.

SANSEN, U. & KOEDAM, N. (1996). Use of sod cutting for restoration of wet heathlands: re-vegetation and establishment of typical species in relation to soil conditions. *J. Veg. Sci.*, **7**: 483-486.

SEDLÁKOVA, I. & CHYTRÝ, M. (1999). Regeneration patterns in a Central European dry heathland: effects of burning, sod-cutting and cutting. *Plant Ecol.*, **143**: 77-89.

SMIDT, J.T. (1995). The imminent destruction of Northwest European heaths due to atmospheric Nitrogen Deposition. *In*: Thompson, D.B.A., Hester, A.J. & Usher, M.B. (Eds.), *Heaths and Moorland: Cultural Landscapes*: 206-217. HMSO, Edinburgh.

STOKES, K.E., ALLCHIN, A.E., BULLOCK, J.M. & WATKINSON, A.R. (2004). Population responses of *Ulex* shrubs to fire in a lowland heath community. *J. Veg. Sci.*, **15**: 505-514.

STOKES, K.E., BULLOCK, J.M. & WATKINSON, A.R. (2003). *Ulex gallii* Planch and *Ulex minor* Roth. *J. Ecol.*, **91**: 1106-1124.

SYMES, N.C. & DAY, J. (2003). *A practical guide to the restoration and management of lowland heathland*. The RSPB, Sandy.

TÁRREGA, R., LUIS-CALABUIG, E. & VALBUENA, L. (1998). A comparative study of recovery in two *Cistus* communities subjected to experimental burning in Leon province (Spain). *In*: Trabaud, L. (Ed.), *Fire Management and Landscape Ecology*: 147-154. Fairfield, Washington.

VALBUENA, L. & VERA, M.L. (2002). The effects of thermal scarification and seed storage on germination of four heathland species. *Plant Ecol.*, **161**: 137- 144.

WEBB, N.R. (1986). *The New Naturalist Heathlands*. Collins, Grafton Street, London.