POLISH JOURNAL OF ECOLOGY (Pol. J. Ecol.)	61	3	505–518	2013

Regular research paper

Andrea CATORCI[1*], Sabrina CESARETTI[2], Renata GATTI[1]

[1] School of Environmental Sciences, University of Camerino – via Pontoni 5, 62032 Camerino (MC), Italy, *e-mail: andrea.catorci@unicam.it (*corresponding author*)
[2] School of Advanced Studies, PhD Course in Environmental Sciences and Public Health, University of Camerino, 62032 Camerino (MC), Italy

EFFECT OF LONG-TERM ABANDONMENT AND SPRING GRAZING ON FLORISTIC AND FUNCTIONAL COMPOSITION OF DRY GRASSLANDS IN A CENTRAL APENNINE FARMLAND

ABSTRACT: Semi-natural dry grasslands host some of the most valuable habitats in Europe, due to their biodiversity heritage. Nevertheless, a strong decline in their extension, due to the cessation of traditional management, has been observed in the last decades. The aim of the study was to assess plant community changes due to abandonment and the effect of spring grazing in sub-Mediterranean dry grasslands, focusing on the plant functional traits involved in this turnover.

The study area is located in the central Apennines (Italy), where grasslands were grazed by sheep in late winter and spring until 1980 and are nowadays abandoned. Relevés sampled (using the Braun-Blanquet method) in different years, namely in 1976–1980 (grazed pasture) and again in 2010 (abandoned pasture) were compared. Results indicated that abandonment leads to the increase of species richness. Traits and strategies indicator sets were: therophyte for the grazed pasture; geophytes, flower palatability, early flowering strategy, clonal ability and presence of storage organs for the abandoned ones. Traits related to low levels of stress (tolerance strategies) are heavily reduced in grazed systems, and thus the functional composition of plant community is mostly characterised by traits promoting avoidance strategies. In abandoned conditions a higher number of species can co-exist thanks to the micro-scale variation of soil features and niche diversification. The research findings also revealed grazing timing as a key factor for understanding changes of plant functional trait patterns and spring grazing as a threat for orchid species.

KEY WORDS: sub-Mediterranean climate, grassland management, grazing, plant functional traits, orchid conservation

1. INTRODUCTION

Semi-natural calcareous grassland are priority habitats for European biodiversity (92/43/EEC Directive), worthy of conservation for the valuable biodiversity heritage they host (e.g. Myklestad and Sætersdal 2004, Klimek *et al.* 2006). However, throughout Europe, there is a strong decline in their extension because of the cessation of traditional management (Sebastià *et al.* 2008). The same trend has been observed in the pastoral landscape of central Italy (Bracchetti *et al.* 2012).

The impact on species composition owing to grassland abandonment or management change is well documented worldwide (e.g. Hadar *et al.* 1999, Lavorel *et al.* 1999a, b, Sternberg *et al.* 2000, Noy-Meir and Oron 2001, Adler *et al.* 2001, Bullock *et al.* 2001, de Bello *et al.* 2006, Catorci *et al.* 2011b, c, Peco *et al.* 2012). Nevertheless, very few data are available concerning the sub-Mediterranean pasture ecosystems where winter cold stress alternate with summer drought stress and cause a more complex response than that observed in temperate regions and in arid context (Catorci *et al.* 2012d). Moreover, in sub-Mediterranean mountains, altitude, slope aspect and angle, besides soil micro-scale features, are key factors in determining the spatial distribution and the species composition of plant communities, because they strongly influence the environmental constraints of sites (Catorci *et al.* 2007, Catorci and Gatti 2010).

Considerable evidence supports the hypothesis that co-existence of plants is conditioned by stress and disturbance intensities (Grime 2001). These factors act in a complex way, affecting species competitive ability and consequently plant community diversity (Lauenroth and Aguilera 1998). Changes in species richness and diversity should be regarded as an outcome of ecological assembly processes, not a causal driver of ecosystem function (Mayfield *et al.* 2010). Instead, in investigating plant community assemblage and shifts due to management modification, functional plant traits assessment proved to be a key tool (e.g. Diaz and Cabido 2001, Hunt *et al.* 2004, de Bello *et al.* 2005, Catorci *et al.* 2012a). Indeed, plant competitive ability is a function of the activity and the distribution in space and time of the plant surface through which resources are intercepted (Grime 2001).

Grazing implies changes in the ability of plants to acquire resources and hence influences competitive interactions (Louda *et al.* 1990). On the other hand, plants respond to herbivory by modifying resources allocation or both compensating and minimizing the loss of tissues by means of resistance strategies (Grubb 1992, Grime 2001). The theory about plant response to grazing identifies two strategies that enable plants to survive and grow in grazed systems: avoidance and tolerance. Grazing avoidance involves mechanisms that reduce the probability and severity of grazing (mechanical and chemical defence or escape strategy), while grazing tolerance consists of mechanisms that promote regrowth after defoliation (Briske 1996). It has been hypothesised that the relative importance of these plant strategies depends on the level of primary production that, in turn, is related to stress intensity (Silvertown *et al.* 1994). Bullock *et al.* (2001) asserted that this dichotomy may be simplistic, while Watkinson and Ormerod (2001) suggested that variations in the density or seasonality of grazing are key factors in understanding how grazing influences plant community composition.

Diaz *et al.* (2007) argued that plant functional type and response rules need to be defined for each different climatic context and grazing history. Nevertheless, Diaz *et al.* (2006) demonstrated that worldwide grazing favoured annual over perennial species, short plants over tall ones, prostrate over erect plants, stoloniferous and rosette architecture over tussock architecture. Besides, in semimesophilous sub-Mediterranean pastures, late flowering and clonal ability have been linked to grazing cessation and spring grazing respectively (Catorci *et al.* 2011a, Vitasović Kosić *et al.* 2011, Catorci *et al.* 2012c).

The main goal of this study was to assess plant community response to long-term abandonment in spring grazed sub-Mediterranean dry grasslands. We hypothesised that the abandonment of sub-Mediterranean dry grasslands affects the plant community functional composition leading to a shift of trait modes especially from those related to avoidance strategies to that ascribed to tolerance strategies.

Specific research goals were to: i/ compare floristic turnover between grazed and abandoned pastures; ii/ highlight the functional traits set involved in this turnover, and iii/ understand its ecological meaning. Our interest was also focused on the functional response of the systems to spring grazing, because it was demonstrated that grazing timing may have a strong influence on functional and species composition of pastures (Catorci *et al.* 2012c).

2. MATERIALS AND METHODS

2.1. Study area

The study area, covering about 200 hectares, is located in the central Apennines (Italy) between 700 and 850 m a.s.l (centroid coordinates 43°06'24"N, 13°00'49"E). Rock substrata consist of limestone covered by shallow soils (5–15 cm), with south eastern-facing aspect and slopes with angle ranging from 10 to 25 degree. From a bioclimatic point of view, this area lies within the Temperate region, near the border of the Mediterranean region (Rivas-Martínez and Rivas-Saenz 1996–2009), and is characterised by alternation of winter cold stress and summer drought stress, occurring between late June and mid September (Orsomando and Catorci 2000). The growing season lasts from April to mid July. A phase of plant tissue re-growth, after the summer leaves yellowing, occurs from mid September to the end of October. Nevertheless, in this period very few species are in bloom (Catorci *et al.* 2012a).

The annual average temperature is 12°C, and annual rainfall amounts to about 900 mm (Biondi and Baldoni 1995). As stated by Catorci *et al.* (2007), pasture plant community belongs to the *Festuco-Brometea* class (Biondi *et al.* 2005). Grassland productivity ranges between 80 and 100 g m^{-2} year of dry forage (Gatti *et al.* 2007b). Until the 1980s, the study area was grazed by sheep in late winter and spring. After this period the site was abandoned and nowadays is partly covered by shrub communities or by woods (*Quercus pubescens* Willd).

2.2. Data collection

We compared two groups of relevés sampled using the phytosociological method (Braun-Blanquet 1964) in 1976–1980 (grazed pasture) and again in 2010 (abandoned pasture). The first group was recorded by Hruška Dell'Uomo (1976), and by Ballelli S. (unpublished data of 1980), while the second was re-sampled on the same sites (relevés of 1976 and 1980 were mapped).

Jandt *et al.* (2011) argued that, in default of permanent plots, vegetation databases represent very valuable resources for the analysis of temporal changes in species occurrence. In addition, they stated that any evaluation of databases that includes preferentially sampled data requires a confinement to particular vegetation types with a strong environmental stratification of samples. (Catorci *et al.* 2011a). According to these considerations, we selected a pool of samples representing a single vegetation typology (dry communities dominated by *Bromus erectus*). To increase the samples homogeneity, we did not consider the plots currently covered by shrubs or woody species (even when their cover value was very low), those of uncertain placement or with heterogeneous morphology (e.g. presence of outcropping rocks or mosaic landforms) and the ones sampled after mid June (to avoid error in presence/absence data of the early flowering species). Each considered relevé covered the same surface area (50 m^2); a total of 60 relevés (30 for each management condition) were used to build up the data set used for the statistical elaborations.

Floristic nomenclature refers to Conti *et al.* (2005).

To assess the functional response of the plant community to abandonment and to the spring grazing management, the following plant traits and strategies were considered.

- Life forms (chamaephyte, geophyte, hemicryptophyte, and therophyte), and Liira and Zobel (2000) guilds (sedges, grasses, rosette forbs and upright forbs). The combinations of these architectural traits give information on plant community responses to disturbance.
- Plant palatability (we divide the species set into non palatable and palatable plants). This trait mode gives information on the plant-herbivorous interplay indicating the presence of resistance strategies against herbivory.
- Palatability of flowers, as it may affect the sexual reproduction and the species establishment and persistence inside the plant community (Pelliza *et al.* 1993).
- Flowering strategies. Plants were divided into early flowering species (from April to May) and late flowering species (from June to July).
- Presence/absence of clonal ability, as it was demonstrated that clonal strategies are more common both where harsh environmental conditions limit seed set (Eckert 2002) and when plant community undergoes a heavy grazing disturbance (Catorci *et al.* 2012c).

- Presence/absence of storage organs, as this trait mode gives information about the plant strategy of resources storage and exploitation (Grime 2001, Catorci *et al.* 2012b).
- Dominant strategy (*sensu* Grime 2001). We evaluated the presence of this strategy because it was argued that dominant plants tend to spread throughout pastures after grazing cessation (Grime 2001). As we ascertained that in the whole data set only *Brachypodium rupestre* may be considered as a potentially dominant species, we only considered the cover values of this species.

Data on functional traits and strategies were gathered from Klotz *et al.* (2002), Klimešová and Klimes (2006) and Pignatti (1982), or inferred by field observations (e.g. flower palatability); data on flowering strategy were obtained from Gatti *et al.* (2007a); data on palatability were gathered from Roggero *et al.* (2002).

Following some authors (e.g. Ertsen *et al.* 1998, Catorci *et al.* 2012e), in order to detect changes in environmental condition and shifts in ecological features of the plant community, five bioindication values (Ellenberg *et al.* 1991) were attributed to all species set. They were: L – light radiation; T – temperature; U – soil moisture or water availability; R – soil chemical reaction; N – nutrients. Bioindication values were gathered from Pignatti (2005), following the scale of values proposed by the author.

2.3. Data elaboration

Braun-Blanquet scale values were transformed in percent values using the average cover value of each class to asses the species abundance (Podani 2007). Mean cover values were then calculated for each species, in each of the two management conditions, to delineate differences in species abundance.

Principal Coordinate Analysis (PCoA) was performed on a species × relevés matrix, based on the Bray-Curtis measure of dissimilarity.

Total species richness and average number of species per relevé were calculated for each treatment. To assess the floristic diversity and community similarity between the grazed and abandoned conditions, the Shannon-Wiener (H'), Evenness (E_H-Sannon's equitability), Simpson diversity (D) and Sørensen (CC) indices (Magurran 1988) were computed using the vegetation relevé data sets.

To identify the indicator species set between grazed and abandoned pasture, Indicator Species Analysis (ISA) was performed on two matrices [relevés × species abundances and relevés × management type (grazed or abandoned)]. Only indicator values greater than 50 (P <0.05) were considered of interest. ISA is a non-parametric method for identifying those species that show significantly preferential distribution (frequency and abundance) with respect to an *a priori* treatment group. An indicator value is calculated by multiplying the relative abundance of each species in a given group and the relative frequency of the species occurrence in the sample of the same group (Dufrêne and Legendre 1997, McCune and Grace 2002). The statistical significance of the observed maximum indicator values for species is evaluated through the Monte Carlo test, based on 4,999 permutations, where samples are reassigned and recalculated. The number of randomised indicator values higher than the observed ones are used to calculate the probability value (McCune and Grace 2002).

To verify whether changes in grassland management are related to functional changes, the above mentioned plant traits were attributed to the indicator species (coming from the ISA). We assumed that the indicator species set can be considered a key tool to understand the plant community changes, as the management modification most affects their frequency and abundance.

The percentage value of occurrence of each trait or strategy inside the indicator species set was calculated as well. Then ISA on two matrices [indicator species (cover %) × traits occurrence and indicator species × management type (grazed or abandoned pasture)] was performed. For the functional assessment of the indicator species set of each management regime, binary data (trait presence/absence) were transformed into quan-

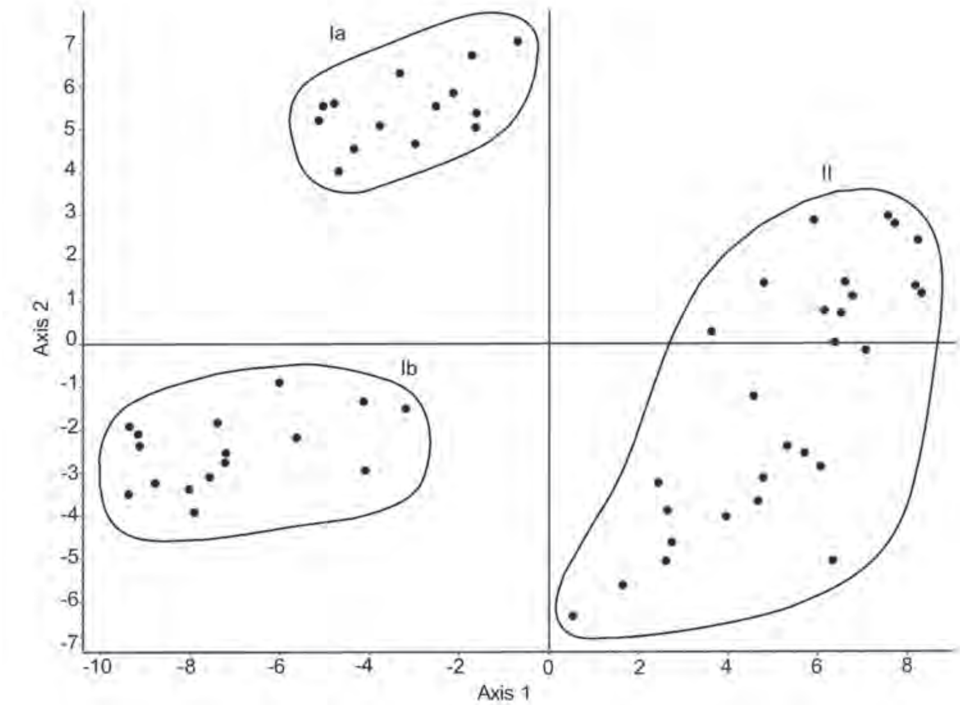

Fig. 1. Ordination diagram representing the first two axis of a Principal Coordinates Analysis (PCoA). Circles (•) represent the relevés carried out in two different management types (Ia and Ib: abandoned pasture; II: grazed pasture).

Table 1. Values of floristic richness, diversity and similarity indices calculated for each disturbance type. Floristic richness (S), average number species per relevè (mean S) Shannon-Wiener (H'), Evenness (E_H) Simpson diversity (D) and Sørensen (CC).

Treatment	S	mean S	H'	E_H	D	CC
Grazed	42	39	2.92	0.78	0.90	0.70
Abandoned	49	48	2.60	0.67	0.84	

Table 2. Indicator species identified by ISA (*P* <0.05) in grazed (cluster I of Fig. 1) and abandoned (clusters IIa and IIb of Fig. 1) condition.

Species identified by ISA for grazed treatment	Species identified by ISA for abandoned condition
Anagallis arvensis L.; *Bupleurum baldense* Turra; *Carthamus lanatus* L. subsp. *lanatus; Convolvulus cantabrica* L.; *Crepis pulchra* L. subsp. *pulchra; Euphorbia exigua* L.; *Fumana procumbens* (Dunal) Godr. & Gren.; *Globularia bisnagarica* L.; *Gnaphalium uliginosum* L. subsp. *uliginosum; Helichrysum italicum* (Roth) Don subsp. *italicum*; *Hieracium pilosella* L.; *Linum tenuifolium* L.; *Linum trigynum* L.; *Lotus corniculatus* L. subsp. *corniculatus*; *Ononis pusilla* L.; *Ononis reclinata* L.; *Seseli tommasinii* Rchb. f.; *Sideritis italica* (Mill.) Greuter & Burdet; *Trifolium campestre* Schreb.; *Trifolium stellatum* L.; *Trifolium tomentosum* L.; *Trigonella gladiata* Steven; *Trigonella monspeliaca* L.; *Xeranthemum inapertum* (L.) Mill.	*Anacamptis pyramidalis* (L.) Rich.; *Asperula purpurea* (L.) Ehrend.; *Asterolinon linum-stellatum* (L.) Duby; *Avenula praetutiana* (Parl.) Pignatti; *Bellis sylvestris* Cyr.; *Bromus erectus* Huds. subsp. *erectus*; *Bunium bulbocastanum* L.; *Cerastium arvense* L. subsp. *suffruticosum* (L.) Ces.; *Dactylorhiza sambucina* (L.) Soó; *Helianthemum oelandicum* (L.) DC. subsp. *incanum* (Willk.) G. López; *Helianthemum salicifolium* (L.) Mill.; *Hippocrepis comosa* L. subsp. *comosa*; *Hypericum perforatum* L.; *Inula montana* L.; *Knautia purpurea* (Vill.) Borbás; *Koeleria lobata* (M. Bieb) Roem. & Schult; *Linum bienne* Mill.; *Ophrys fusca* Link subsp. *fusca*; *Orchis morio* L.; *Orchis pauciflora* Ten.; *Orchis tridentata* Scop.; *Phleum hirsutum* Honck. subsp. *ambiguum* (Ten.) Tzvelev; *Plantago argentea* Chaix subsp. *argentea*; *Poa bulbosa* L.; *Sedum rupestre* L. subsp. *rupestre*; *Sedum sexangulare* L.; *Sherardia arvensis* L.; *Thymus sp. pl.*; *Trinia glauca* (L.) Dumort. subsp. *carniolica* (A. Kern. ex Janch.) H. Wolff; *Veronica arvensis* L.

titative data. For this purpose, the matrix of relevés × species (cover %) data was multiplied by the matrix of species × trait (presence/absence) to provide a matrix of trait × relevés, which formed the basis for the following analyses (Pakeman *et al.* 2009, Catorci *et al.* 2012e). Only indicator values greater than 20 (P <0.05) were considered of interest.

Bioindication values were applied to all the species occurring in the vegetation relevés, and means were calculated for each relevé and treatment. The distributions were tested for normality and homogeneity of variance using the Kolmogorov-Smirnov and Levene tests. Since the data did not meet the assumptions required for parametric tests, the Mann-Whitney U-test was used to detect statistically significant differences.

As in abandoned condition PcoA highlighted two subclusters, ISA was performed on two matrices [relevés x species abundances and relevés x subcluster (Ia and Ib) in order to assess the indicator species set of the two subclusters. In addition, differences in bioindication values (of all the species data set) were assessed using the Mann-Whitney U-test.

Differences in *Brachypodium rupestre* cover value between abandoned and grazed conditions (cluster I and II) and between subclusters Ia and Ib of abandoned pasture were tested.

PCoA was run using the Syn-tax 2000 package (Podani 2001), ISA elaboration using PCOrd 5.0 software (McCune and Mefford 2006), and Kolmogorov-Smirnov, Levene and Mann-Whitney tests using SPSS 13.0 software (2005).

3. RESULTS

Principal Coordinate Analysis (PCoA) highlights two main groups of relevés (Fig. 1). Cluster I consists of relevés carried out in abandoned pasture, and is divided into two subclusters (Ia and Ib). Cluster II groups the relevés of grazed pasture.

Floristic richness and the average number of species per relevés are higher in the abandoned condition. The results of the Sørensen index of community similarity (CC) between the two plant communities indicate that the two treatments have a quite high floristic similarity (70%). The outcomes of the Shannon-Wiener (H'), Evenness (E_H) and Simpson (D) indices pointed out that the abandoned pasture has lower floristic diversity, lower equitability of species distribution and greater dominance (Table 1).

Indicator Species Analysis (ISA) identified 24 indicator species for the grazed pasture and 30 for the abandoned conditions (Table 2).

The indicator plant traits highlighted by ISA are shown in Table 3. The assessment of mean cover values of indicator traits (grazed vs. abandoned grassland) highlights a higher value for therophyte in grazed conditions and a higher value for geophyte, flower palatability, early flowering strategy, clonal ability and storage organs in the abandoned conditions (Table 4). On the contrary, chamaephyte, hemicryptophyte, sedges, grasses, rosette forbs, upright forbs, non palatable and palatable plants, and late flowering strategies do not show significant indicator values following the results of the Monte Carlo test.

As far ecological needs is concerning (Table 5), the abandoned condition is marked by lower light radiation (L) and temperature (T) bioindication values, and by a higher value of soil moisture (U). Differences were significant (P <0.001). Indeed, differences in soil chemical reaction (R) and nutrients request (N) are not significant.

As regards the two subclusters highlighted by PCoA for abandoned condition, ISA indicated 6 indicator species for the subcluster Ia and 18 for the subcluster Ib (Table 6).

Outcomes of the Mann-Whitney U-test indicates significant differences (P <0.05) between the two subclusters in ecological needs for temperature (T) and soil moisture (U) bioindicators. Particularly, a higher value for temperature and a lower value for moisture were observed in subcluster Ia (Table 7).

About the *Brachypodium rupestre* cover value, the difference between abandoned (mean cover value 1.60%) and grazed conditions (mean cover value 0.50%) was not significant. Conversely, a significant difference (P <0.05) has been highlighted between subclusters Ia and Ib (mean cover values 0.5 and 3.1% respectively).

Table 3. List of indicator traits as determined by indicator species analysis (ISA) for each management types (G: grazed; A: abandoned); only traits with P <0.05 and indicator value > 20 are reported.

Trait	Treatment	Observed indicator value	*P*
Therophyte	G	41.1	0.0030
Geophyte	A	26.7	0.0148
Flower palatability	A	26.7	0.0168
Early flowering	A	23.3	0.0306
Clonal ability	A	59.7	0.0046
Storage organs	A	63.6	0.0472

Table 4. Trend of mean cover values and standard deviation of indicator traits in grazed and abandoned treatments.

Trait	Grazed	Abandoned
Therophyte	7.79 ± 0.75	1.13 ± 0.12
Geophyte	0.00	4.78 ± 0.44
Flower palatability	0.00	14.74 ± 1.97
Early flowering	0.00	4.32 ± 0.48
Clonal ability	9.73 ± 1.35	70.87 ± 8.03
Storage organs	16.19 ± 1.45	78.54 ± 8.17

Table 5. Mean bioindication values (see Pignatti 2005) with standard deviation (SD) for each treatment (G: grazed; A: abandoned). * Differences statistically significant at P <0.001.

Bioindication values	Treatment	Mean values ± SD
Light radiation*	G	8.90 ± 0.15
	A	8.39 ± 0.26
Temperature*	G	6.92 ± 0.20
	A	6.61 ± 0.26
Soil moisture or water availability*	G	3.10 ± 0.11
	A	3.18 ± 0.14
Soil chemical reaction	G	6.52 ± 0.17
	A	6.55 ± 0.13
Nutrients	G	2.45 ± 0.14
	A	2.48 ± 0.20

4. DISCUSSION

In accordance with the findings of many authors (e.g. Myklestad and Sætersdal 2004, Klimek *et al.* 2006, de Bello *et al.* 2006, Peco *et al.* 2006) abandonment leads to plant community floristic change. A slight decrease of evenness and of floristic diversity were demonstrated in abandoned conditions, probably owing to the higher cover value of grass species and due to the closure of gaps in the herbaceous sward after abandonment. An increasing trend of species richness was highlighted as well. This could be explained by the fact that the study area is located near the border of Mediterranean areas. In fact it was demonstrated that where drought limits plant growth, grazing does not increase species richness (e.g. Austrheim and Eriksson 2001, Catorci *et al.* 2011a).

In grazed conditions, the only indicator trait was the therophyte life form. This trait affirmation can be linked to the presence of gaps in the herbaceous sward caused by livestock activities (Grime 2001, Kohler *et al.* 2006). Annual species are also known as indicator of high disturbance intensity worldwide (Grime 2001), thus their decrease trend may be linked to grazing cessation.

A part of the set of indicator traits of the abandoned condition (geophyte life form, flower palatability, early flowering strategy and presence of storage organs) may be explained considering that the studied pasture was grazed during the springtime, and thus it is possible to state that species with palatable flowers that bloom in such period are favoured when the pasture is not grazed in spring (as in abandoned condition). In fact, many of the indicator species of abandoned conditions are early flowering geophytes such as the orchid species: *Anacamptis pyramidalis* (L.) Rich., *Dactylorhiza sambucina* (L.) Soó, *Ophrys fusca* Link subsp. *fusca*, *Orchis morio* L., *Orchis pauciflora* Ten. and *Orchis tridentata* Scop.

The affirmation of graminoids (*Bromus erectus* Hudson subsp. *erectus*, *Phleum hirsutum* Honck. subsp. *ambiguum* (Ten.) Tzvelev, *Koeleria* sp. pl., *Avenula praetutiana* (Parl.) Pign.) with tolerance strategies and erect leaves in the abandoned condition (all of which had a higher cover value) is consistent with the findings of some authors (e.g. Skarpe 2001, Pykälä 2004) who argued that exclusion of grazers causes the decrease of prostrate species, while plants with erect leaves increase. It should be also considered that perennial grasses, which are sensitive to continuous grazing disturbance, are among the most preferred foods of large herbivores (O'Connor and Everson 1998). Furthermore, it was demonstrated that affirmation of tolerance strategies is favoured by grazing only in highly productive systems (Maschinsky and Whitham 1989) or when the system undergoes to a very low level of disturbance (Crofts and Jefferson 1999). Actually, the sub-Mediterranean grassland landscape could be considered a productive system only in springtime, whilst it is a very poorly productive system in summer, when a quite strong drought stress influences the plant communities (Catorci *et al.* 2012d). Therefore, it is possible to state that in the sub-Mediterranean climate, species with tolerance strategies are strongly affected by the intensity and timing of grazing, because tissue regrowth after the herbivory is not possible in summer. Thus it can be argued that, in the sub-Mediterranean most stressed habitats (such as the south-facing slopes), spread of grasses is promoted by the absence of herbivory (Peco *et al.* 2012).

This hypothesis is supported by the bioindication value assessment, showing a higher value of soil moisture and a lower value of temperature indices of the diagnostic species group related to the abandoned conditions. These findings indicate that species with the lower drought stress tolerance ability are enhanced in abandoned treatment. Nevertheless, our outcomes do not highlight differences between the considered treatments with respect to the palatability trait modes. Since Peltzer and Wilson (2001) argued that traits modes giving stress tolerance ability are often involved in the plants avoidance strategies (hence to plant palatability), it is possible to state that the drought stress acting in the system remains a key factor in plant community assemblage even in abandoned condition.

The lower value of light biondicator in the ungrazed condition is probably due to the higher density of the herbaceous canopy (Hurst and John 1999), with a consequent increase in the average cover value of plants demanding intermediate light intensity, that are those living under the canopy of tall species (Catorci *et al.* 2011a).

The spread of species with clonal ability in abandoned condition might be related both to the lack of disturbance and to the grass en-

Table 6. Indicator species identified by ISA ($P < 0.05$) in abandoned condition, between subclusters Ia and Ib.

Species identified by ISA for subcluster Ia related to abandoned condition	Species identified by ISA for subcluster Ib related to abandoned condition
Artemisia alba Turra; *Helianthemum salicifolium* (L.) Miller; *Poa bulbosa* L.; *Sedum rupestre* L.; *Sedum sexangulare* L.; *Sideritis italica* (Mill.) Greuter & Burdet	*Anthyllis* gr.; *Asperula purpurea* (L.) Ehrend.; *Avenula praetutiana* (Parl.) Pign.; *Bellis perennis* L.; *Brachypodium rupestre* (Host) R. et S.; *Cerastium arvense* L. subsp. *suffruticosum* (L.) Ces.; *Cerastium pumilum* Curtis; *Globularia bisnagarica* L.; *Helianthemum nummularium* (L.) Miller subsp. *obscurum* (Celak.) Holub; *Helianthemum oelandicum* (L.) DC. subsp. *incanum* (Willk.) G. López; *Hippocrepis comosa* L.; *Knautia purpurea* (Vill.) Borbas; *Lotus corniculatus* L.; *Onobrychis viciifolia* Scop.; *Orchis pauciflora* Ten.; *Plantago holosteum* Scop.; *Sanguisorba minor* Scop.; *Thymus praecox* Opiz subsp. *polytrichus* (Borbás) Jalas

Table 7. Mean bioindication values with standard deviation (SD) for two subclusters (Ia and Ib) emerged by PCoA as regards abandoned condition. * Difference statistically significant at P <0.05).

Bioindication values	Subcluster	Mean values ± SD
Light radiation	Ia	8.39 ± 0.32
	Ib	8.39 ± 0.16
Temperature*	Ia	6.76 ± 0.20
	Ib	6.42 ± 0.21
Soil moisture or water availability*	Ia	3.12 ± 0.13
	Ib	3.25 ± 0.11
Soil chemical reaction	Ia	6.51 ± 0.12
	Ib	6.61 ± 0.13
Nutrients	Ia	2.49 ± 0.23
	Ib	2.46 ± 0.14

croachment. Clonal ability is a reproductive strategy strengthened both by low and high level of environmental resources and by low and high disturbance intensities; instead in the intermediate condition sexual reproduction seems to be the main reproductive strategy (Eckert 2002). Moreover, clonal ability may be useful in face of neighbour competition, when resources have a micro-scale patchy distribution (Catorci *et al.* 2011b, 2012d). Clonal ability, especially later spread, seems also to have some role in dynamic processes (Klimešová and Pyšek 2011), thus our findings may be explained in terms of dynamic process acting in the system after abandonment.

The bioindication assessment showed an increasing trend of soil moisture need between subcluster Ia and Ib, probably due to the patchiness of local topographic conditions, characterised by variations in soil depth. A small difference in environmental constraints between the more productive and the less productive patches seems to lead to differences in long-term floristic composition change following grassland abandonment. Indeed, as emerged by ISA, the indicator species of the subcluster Ia (*Artemisia alba* Turra; *Helianthemum salicifolium* (L.) Miller; *Sedum rupestre* L.; *Sedum sexangulare* L.; *Sideritis italica* (Mill.) Greuter et Burdet) are mainly species of xeric habitat with shallow soils and a very strong and long summer drought stress, while the indicator species set of the subcluster Ib is mostly composed by semimesophilous plants (Biondi *et al.* 2005, Catorci *et al.* 2007).

Actually, soil moisture has been demonstrated to be one of the most important environmental features affecting plant species diversity (Alard and Poudevigne 2000, Ejrnæs and Bruun 2000, Galvánek and Lepš 2009). Furthermore, the patchiness of soil moisture reflects on the *Brachypodium rupestre* ability to spread, in fact its cover value is higher in the group of relevés with the higher value of moisture need.

5. CONCLUSIONS

The research findings indicate that in abandoned treatment, species with drought stress/avoidance strategies, plant with tolerance strategy and quite high soil resources need can co-exist probably owing to the variation of soil features at micro-scale level. Thus the lower species richness of grazed treatment could be caused by the suppression of the effect of spatial niche diversification at micro-scale level. On the other hand, the poor spreading of dominant grasses after the abandonment, owing to the high level of drought stress (Grime 2001), fosters a more straightforward biodiversity conservation. In fact, it was demonstrated that dominant species invasion can lead to the decrease of species richness (Bonanomi *et al.* 2009). These findings agree with Bennie *et al.* (2006) and Vitasović *et al.* (2011), who argued that in the case of cessation of management practices, dry grasslands are less vulnerable to the loss or change of species than the more productive ones.

Assessment of grazing timing emerged as a key factor for understanding changes of plant functional trait, allowing for the design of more suitable plans for biodiversity conservation. For example, the studied sub-Mediterranean dry grasslands are referred to the *6 210 habitat [(semi-natural dry grasslands on calcareous substrates (*Festuco-Brometalia*) (* important orchid sites) Biondi *et al.* 2009] which is of priority importance only when orchids are present. Landi *et al.* (2009) argued that the most important predictors of high abundance of orchids on calcareous grasslands are xerophilous and basophilous perennial plants and microenvironments with low herbaceous cover. Accordingly, it is possible to state that grazing can positively influence the fitness of orchids by limiting dominant species. Nevertheless, the high number of orchids observed among the indicator species of the abandoned communities, indicates that to conserve orchid populations, spring grazing must be forbidden.

6. REFERENCES

Adler P., Raff R., Lauenroth W.K. 2001 – The effect of grazing on the spatial heterogeneity of vegetation – Oecologia, 128: 465–479.

Alard D., Poudevigne I. 2000 – Diversity patterns in grasslands along a landscape gradient in northwestern France – J. Veg. Sci. 11: 287–294.

Austrheim G., Eriksson O. 2001– Plant species diversity and grazing in the Scandinavian mountains – patterns and processes at different spatial scales – Ecography, 24: 683–695.

Bennie J., Hill M.O., Baxter R., Huntley B. 2006 – Influence of slope and aspect on long-term vegetation change in British chalk grasslands – J. Ecol. 94: 355–368.

Biondi E., Baldoni M.A. 1995 – The climate and vegetation of peninsular Italy – Colloq. Phytosociol. 23: 675–721.

Biondi E., Allegrezza M., Zuccarello V. 2005 – Syntaxonomic revision of the Apennine grassland belonging to *Brometalia erecti* and an analysis of their relationship with the xerophilous vegetation of *Rosmarinetea officinalis* – Phytocoenol. 35: 129–163.

Biondi E., Blasi C., Burrascano S., Casavecchia S., Copiz R., Del Vico E., Galdenzi D., Gigante D., Lasen C., Spampinato G., Venanzoni R., Zivkovic L. 2009 – Italian interpretation manual of the 92/43/EEC Directive Habitats. http://vnr.unipg.it/habitat/index.jsp

Bonanomi G., Caporaso S., Allegrezza M. 2009 – Effects of nitrogen enrichment, plant litter removal and cutting on a species-rich Mediterranean calcareous grassland – Plant Biosyst. 143: 443–455.

Bracchetti L., Carotenuto L., Catorci A. 2012 – Land-cover changes in a remote area of central Apennines (Italy) and management directions – Landsc. Urban Plan. 104: 157–170.

Braun-Blanquet J. 1964 – Pflanzensoziologie – Springer, Wien, New York, 865 pp.

Briske D.D. 1996 – Strategies of plant survival in grazed systems: a functional interpretation (In: The Ecology and Management of Grazing Systems, Eds: J. Hodgson, A.W. Illius) – CAB International, Wallingford, pp. 37– 68.

Bullock J.M., Franklin J., Stevenson M.J., Silvertown J., Coulson S.J., Gregory S.J., Tofts R. 2001 – A plant trait analysis of responses to grazing in a long-term experiment – J. Appl. Ecol. 38: 253–267.

Catorci A., Gatti R. 2010 – Floristic composition and spatial distribution assessment of montane mesophilous grasslands in central Apennines, Italy: a multi-scale and diachronic approach – Plant Biosyst. 144: 793–804.

Catorci A., Gatti R., Ballelli S. 2007 – Studio fitosociologico della vegetazione delle praterie montane dell'Appennino maceratese (Italia centrale) – Braun-Blanquetia, 42: 101–143.

Catorci A., Ottaviani G., Cesaretti S 2011a – Functional and coenological changes under different long-term management conditions in Apennine meadows (central Italy) –Phytocoenol. 41: 45–58.

Catorci A., Cesaretti S., Gatti R., Ottaviani G. 2011b – Abiotic and biotic changes due to spread of Brachypodium genuense (DC.) Roem. & Schult. in sub-Mediterranean meadows – Community Ecol. 12: 117–125.

Catorci A., Ottaviani G., Ballelli S., Cesaretti S. 2011c – Functional differentiation of central Apennine grasslands under mowing and grazing disturbance regimes – Pol. J. Ecol. 59: 115–128.

Catorci A., Carotenuto L., Gatti R. 2012a – Flowering patterns in sub-Mediterranean grasslands: a functional approach – Plant Ecol. Evol. 145: 165–175.

Catorci A., Cesaretti S., Gatti R., Tardella F.M. 2012b – Trait-related flowering patterns in submediterranean mountain meadows – Plant Ecol. DOI: 10.1007/s11258-012-0090-9.

Catorci A., Gatti R., Cesaretti S. 2012c – Effect of sheep and horse grazing on species and functional composition of sub-Me-

diterranean grasslands – Appl. Veg. Sci. 15: 459–469.
Catorci A., Ottaviani G., Vitasović Kosić I., Cesaretti S. 2012d – Effect of spatial and temporal patterns of stress and disturbance intensities in a sub-Mediterranean grassland – Plant Biosyst. 146: 352–367.
Catorci A., Vitanzi A., Tardella F.M., Hršak V. 2012e – Trait variations along a regenerative chronosequence in the herb layer of submediterranean forests – Acta Oecol. 43: 29–41.
Conti F., Abbate G., Alessandrini A., Blasi C. 2005 – An annotated checklist of the Italian vascular flora – Palombi Editori, Roma, 420 pp.
Crofts A., Jefferson R.G. 1999 – The lowland grassland management handbook. English Nature/The Wildlife Trusts, London.
de Bello F., Lepš J., Sebastià M.T. 2005 – Predictive value of plant traits to grazing along a climatic gradient in the Mediterranean – J. Appl. Ecol. 42: 824–833.
de Bello F., Lepš J., Sebastia M.T. 2006 – Variations in species and functional plant diversity along climatic and grazing gradients – Ecography, 29: 801–810.
Diaz S., Cabido M. 2001 – Vive la difference: plant functional diversity matters to ecosystem processes – Trends Ecol. Evol. 16: 646–655.
Díaz S., Lavorel S., Chapin F.S., Tecco P.A., Gurvich D.E., Grigulis K. 2007 – Functional diversity – at the crossroads between ecosystem functioning and environmental filters – (In: Terrestrial Ecosystem in a changing World, Eds: J.G. Canadell, D. Pataki, L. Pitelka) –Springer, Verlag, Berlin, Heidelberg, pp. 81–91.
Díaz S., Lavorel S., McIntyre S., Falczuk V., Casanoves F., Milchunas D.G., Skarpe C., Rusch G., Sternberg M., Noy-Meir I., Landsberg J., Zhang W., Clark H., Campbel B.D. 2006 – Plant trait responses to grazing – a global synthesis – Glob. Chang. Biol. 12: 1–29.
Dufrêne M., Legendre P. 1997 – Species assemblages and indicator species: the need for a flexible asymmetrical approach – Ecol. Monogr. 67: 345–366.
Eckert C.G. 2002 – The loss of sex in clonal plants – Evol. Ecol. 15: 501–520.
Ellenberg H., Weber H.E., Duell R., Wirth V., Werner W., Paulissen D. 1991 – Zeigerwerte von Pflanzen in Mitteleuropa – Scripta Geobot. 18: 1–248.
Ejrnæs R., Bruun H.H. 2000 – Gradient analysis of dry grassland vegetation in Denmark – J. Veg. Sci. 11: 573–584.
Ertsen A.C.D., Alkemade J.R.M., Wassen M.J. 1998 – Calibrating Ellenberg indicator values for moisture, acidity, nutrient availability and salinity in the Netherlands – Plant Ecol. 135: 113–124.
Galvanek D., Lepš J. 2009 – How do management and restoration needs of mountain grasslands depend on moisture regime? Experimental study from north-western Slovakia (Western Carpathians) – Appl. Veg. Sci. 12: 273–282.
Gatti R., Carotenuto L., Catorci A. 2007a – Sinfenologia di alcuni *syntaxa* prativi dell'Appennino Umbro-Marchigiano (Italia centrale) – Braun-Blanquetia 42: 179–202.
Gatti R., Vitanzi A., Cesaretti S., Catorci A. 2007b – Contributo alla quantificazione della fitomassa epigea di alcuni pascoli dell'Appennino umbro-marchigiano (Italia centrale) – Braun-Blanquetia 42: 255–266.
Grime J.P. 2001 – Plant Strategies, Vegetation Processes and Ecosystem Properties (Second Edition) – John Wiley and Sons, Chichester, 417 pp.
Grubb P.J. 1992 – A positive distrust in simplicity – lessons from plant defences and from competition among plants and among animals – J. Ecol. 80: 585–610.
Hadar L., Noy-Meir I., Perevolotsky A. 1999 – The effect of shrub clearing and grazing on the composition of a Mediterranean plant community: functional groups versus species – Veg. Sci. 10: 673–683.
Hruška Dell'Uomo K., 1976 – Contributo alla conoscenza dei pascoli aridi dell'Appennino marchigiano – Notiziario fitosociologico 12: 19–30.
Hunt R., Hodgson J.G., Thompson K., Bungener P., Dunnett N.P., Askew A.P. 2004 – A new practical tool for deriving a functional signature for herbaceous vegetation – Appl. Veg. Sci. 7: 163–170.
Hurst A., John E. 1999 – The biotic and abiotic changes associated with Brachypodium pinnatum dominance in chalk grassland in south-east England – Biol. Conserv. 88: 75–84.
Jandt U., von Wehrden H., Bruelheide H. 2011 – Exploring large vegetation databases to detect temporal trends in species occurrences – J. Veg. Sci. 22: 957–972.
Klimek S., Richter gen. Kemmermann A., Hofmann, M., Isselstein J. 2006 – Plant species richness and composition in managed grasslands: The relative importance of field management and environmental factors – Biol. Conserv. 134: 559–570.
Klimešová J., Klimes L. 2006 – Clo-Pla3: A database of clonal growth architecture of

Central-European plants – URL: http://clopla.butbn.cas.cz/.
Klimešová J., Pyšek P. 2011 – Current topics in clonal plants research: editorial – Preslia, 83: 275–279.
Klotz S., Kühn I., Durka W. 2002 – Biolflor: Eine Datenbank zu biologisch-okologischen Merkmalen der Gefäßpflanzen in Deutschland – Schriftenreihe für Vegetationskunde 38. Bonn, DE, Bundesamt für Naturschutz – URL: http://www.ufz.de/biolflor/index.jsp.
Kohler F., Gillet F., Gobat J. M., Buttler A. 2006 – Effect of cattle activities on gap colonization in mountain pastures – Folia Geobot. 41: 289–304.
Landi M., Frignani F., Lazzeri C., Angiolini C. 2009 – Abundance of orchids on calcareous grasslands in relation to community species, environmental, and vegetational conditions – Russ. J. Ecol. 40: 486–494.
Lauenroth W. K., Aguilera M.O. 1998 – Plant-plant interactions in grasses and grasslands (In: Population Biology of Grasses, Ed. G.P. Cheplick) – Cambridge University Press, Cambridge, 209–230.
Lavorel S., McIntyre S., Grigulis K. 1999a – Plant response to disturbance in a Mediterranean grassland: How many functional groups? – J. Veg. Sci. 10: 661–672.
Lavorel S., Rochette C., Lebreton J.D. 1999b – Functional groups for response to disturbance in Mediterranean old fields – Oikos, 84: 480–498.
Liira J., Zobel K. 2000 – Vertical structure of a species-rich grassland canopy, treated with additional illumination, fertilization and mowing – Plant Ecol. 146: 185–195.
Louda S.M., Keeler K.H., Holt R.D. 1990 – Herbivore influences on plant performance and competitive interactions (In: Perspectives in Plant Competition, Eds: J. Grace, D. Tilman) – Academic Press, San Diego, USA, pp. 414–444.
Magurran A.E. 1988 – Ecological diversity and its measurement – Chapman and Hall, London, 197 pp.
Maschinsky J., Whitham T.G. 1989 – The continuum of plant responses to herbivory: the influence of plant association, nutrient availability, and timing – Am. Nat. 134: 1–19.
Mayfield M.M., Bonser S.P., Morgan J.W., Aubin I., McNamara S., Vesk P.A. 2010 – What does species richness tell us about functional trait diversity? Predictions and evidence for responses of species and functional trait diversity to land-use change – Glob. Ecol. Biogeogr. 19: 423–431.
McCune B., Grace J.B. 2002 – Analysis of Ecological Communities – MjM Software Design. Gleneden Beach, Oregon, 300 pp.
McCune B., Mefford M.J. 2006 – PC-OR D. Multivariate Analysis of Ecological Data. Version 5 – MjM Software Design. Gleneden Beach, Oregon.
Myklestad Å., Sætersdal M. 2004 – The importance of traditional meadow management techniques for conservation of vascular plant species richness in Norway – Biol. Conserv. 118: 133–139.
Noy-Meir I., Oron T. 2001 – Effects of grazing on geophytes in Mediterranean vegetation. – J. Veg. Sci. 12: 749–760.
O'Connor T.G., Everson T.M., 1998 – Population dynamics of perennial grasses in African savanna and grassland (In: Population Biology of Grasses, Ed. G.P. Cheplick) – Cambridge University Press, Cambridge, 333–365.
Orsomando E., Catorci A. 2000 – The phytoclimate of Umbria – Parlatorea 6: 5–24.
Pakeman R.J., Lepš J., Kleyer M., Lavorel S., Garnier E. 2009 – Relative climatic, edaphic and management controls of plant functional trait signatures – J. Veg. Sci. 20: 148–159.
Peco B., Sánchez A. M., Azcárate F. M. 2006 – Abandonment in grazing systems: Consequences for vegetation and soil – Agric. Ecosyst. Environ. 113: 284–294.
Peco B., Carmona C.P., de Pablos I., Azcárate F.M. 2012 – Effects of grazing abandonment on functional and taxonomic diversity of Mediterranean grasslands – Agric. Ecos. Envir. 152: 27–32.
Pelliza A., Nakamatsu V., Feijoo S., Mansilla A., La Rosa F., Evans E., Vera J. 1993 – Composición botánica de la dieta de ovinos en el este de Río Negro y Chubut (Argentina). Proceedings of the XVI Reunión Argentina de Ecología, Puerto Madryn, Chubut, Argentina. Puerto Madryn: Centro Nacional Patagónico. 300 pp.
Peltzer D.A., Wilson S.D. 2001 – Competition and environmental stress in temperate grasslands. (In: Competition and succession in pastures, Eds: P.G. Tow, A. Lazenby) – CABI Publishing, Wallingford, 193–212.
Pignatti S. 1982 – Flora d'Italia, 1–3 – Edagricole, Bologna, 2302 pp.
Pignatti S. 2005 – Valori di bioindicazione delle piante vascolari della flora d'Italia – Braun-Blanquetia, 39: 1–97.
Podani J. 2001 – Syn-tax 2000 computer program for data analysis in ecology and systematics – Budapest.

Podani J. 2007 – Analisi ed esplorazione multivariata dei dati in ecologia e biologia – Liguori, Napoli, 515 pp.

Pykälä J. 2004 – Cattle grazing increases plant species richness of most species trait groups in mesic semi-natural grasslands – Plant Ecol. 175: 217–226.

Rivas-Martínez S., Rivas-Saenz S. 1996–2009 – Worldwide Bioclimatic Classification System, Phytosociological Research Center, Spain, http://www.globalbioclimatics.org

Roggero P.P., Bagella S., Farina R. 2002 – Un archivio dati di Indici specifici per la valutazione integrata del valore pastorale – Rivista di Agronomia, 36: 149–156.

Sebastià M.T., de Bello F., Puig L., Taull M. 2008 – Grazing as a factor structuring grasslands in the Pyrenees – Appl. Veg. Sci. 11: 215–222.

Silvertown J., Dodd M.E., McConway K., Potts J., Crawley M. 1994 – Rainfall, biomass variation, and community composition in the Park Grass Experiment – Ecology, 75: 430–437.

Skarpe C. 2001 – Effects of Large Herbivores on Competition and Succession in Natural Savannah Rangelands (In: Competition and succession in pastures, Eds: P.G. Tow, A. Lazenby) – CABI Publishing, Wallingford, pp. 175–192.

SPSS Inc. 2005 – SPSS for Windows – Version 13.0, Chicago, Illinois, NJ, USA.

Sternberg M., Gutman M., Perevolotsky A., Ungar E.D., Kigel J. 2000 – Vegetation response to grazing management in a Mediterranean community: a functional group approach – J. Appl. Ecol. 37: 224–237.

Vitasović Kosić I., Tardella F.M., Ruščić M., Catorci A. 2011– Assessment of floristic diversity, functional composition and management strategy of North Adriatic pastoral landscape (Croatia) – Pol. J. Ecol. 59: 765–776.

Watkinson A.R., Ormerod S.J. 2001 – Grasslands, grazing and biodiversity: editors' introduction – J. Appl. Ecol. 38: 233–237.

Received after revision October 2012