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Livestock Grazing Impacts on Herbage and Shrub Dynamics in a Mediterranean Natural Park

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Abstract

Shrub encroachment can be explained by the abandonment of extensive livestock farming and changes to land use, and it is a common problem in the Mediterranean mountain pastures of Europe, with direct effects on biodiversity and landscape quality. In this paper, the effects of livestock exclusion vs. grazing on the dynamics of shrub and herbaceous vegetation were analyzed in a Spanish natural park located in a dry Mediterranean mountain area over a 5-yr period. Twelve 10×10 m exclosures were set up in six representative pasture areas of the park (with two replicates per location). Each year, the shrub number, volume, and biomass were measured in April, and the herbage height, biomass, and quality were measured in April and December (which represent the start and end of the vegetative growth season). A sustained increase of the shrub population and individual biomass was observed throughout the study, which was reflected in total shrub biomass per ha. Growth was greater in nongrazed exclosures (2 563 kg dry matter $[\mathrm{DM}] \cdot \mathrm{ha}^{-1} \cdot \mathrm{yr}^{-1}$), but it also happened in the grazed control areas (1 173 kg $\mathrm{DM} \cdot \mathrm{ha}^{-1} \cdot \mathrm{yr}^{-1}$), with different patterns depending on the location and shrub species. Herbage biomass did not change when grazing was maintained, but it did increase in places where grazing was excluded (291 kg $\mathrm{DM} \cdot \mathrm{ha}^{-1} \cdot \mathrm{yr}^{-1}$), mostly as a consequence of the accumulation of dead material, with a concomitant reduction in herbage quality. It was concluded that at the current stocking rates and management regimes, grazing alone is not enough to prevent the intense dynamics of shrub encroachment, and further reductions in grazing pressure should be avoided.

Key Words: agro-sylvo-pastoral systems, cultural landscape, grazing exclusion, herbage biomass and quality, shrub encroachment

INTRODUCTION

Most Mediterranean pastures are categorized by the European Environmental Agency (EEA) as High Nature Value (HNV) agro-ecosystems that are undergoing complex environmental, social, and economic processes (EEA 2004). Traditional agriculture and livestock production have molded the landscape in these areas and are currently considered vital for the maintenance of biodiversity (Henle et al. 2008) as well as for risk reduction of environmental hazards such as forest fires (Kramer et al. 2003), which are perceived as an environmental problem of paramount importance by policy makers and local communities. During the second half of the last century, the decreasing human populations in mountainous areas led to a continuous reduction in livestock farming (MacDonald et al. 2000), which resulted in deep changes in land use (Lasanta-Martínez et al. 2005). Additionally, new economic activities have also emerged in the last decades that are linked to the recreational values of these areas, which, in many cases, are substituting the primary agricultural activities with tourism (Martin-Yaseli and Lasanta-Martinez 2003; García-Martínez et al. 2009).

Like other Mediterranean mountain areas, the Sierra y Cañones de Guara Natural Park (SCGNP, NE Spain) has suffered a serious decline in farming activity in recent decades (Riedel et al. 2007) with increasing tourism and an associated service sector, which have resulted in technical changes in farm management and land use (Bernués et al. 2005). These changes have been particularly intense in farms that are owned by the younger and more innovative farmers, who have adopted intensive reproduction and feeding technologies leading to reduced grazing periods and livestock concentration on high-quality pastures, mostly in agricultural areas (Riedel et al. 2007). In contrast, the more traditional farms with extensive pasture use have the lowest chances of long-term maintenance, mostly as a consequence of the lack of intergenerational succession (Bernués et al. 2011). The disappearance of these farms would therefore increase the abandonment of large grazing areas in the future.

The abandonment of farmland is associated with a process of vegetative succession that is characterized by shrub encroachment. This has multiple implications for the ecologic values of these areas: the colonization of open spaces by a reduced number of competitive shrub species (Manier and Hobbs 2006), accumulation of low-quality herbaceous biomass (Hope et al. 1996), or landscape closure that represents a threat to several bird species whose conservation status is unfavorable in Europe (Fonderflick et al. 2010). Shrub encroachment has even been related to a shift in the fire regime that is common in

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Mediterranean ecosystems; Pausas and Fernández-Muñoz (2012) observed that rural depopulation and farm abandonment in the 1970s resulted in an increase in the amount of and continuity of fuel in the western Mediterranean basin leading to an increase in fire frequency and size that made vegetation more prone to fire under dry climatic conditions.

Grazing controls the dominance of certain plant species and favors less competitive ones, increasing biodiversity (Collins et al. 1998; Rosen and Bakker 2005) and enhancing structural heterogeneity by selective defoliation, trampling, nutrient cycling, and propagule dispersal (Rook and Tallowin 2003). Therefore, enhancing livestock grazing may be a useful management tool to prevent or reduce shrub encroachment and its associated environmental risks, which would help to maintain the open structure of Mediterranean woodland pastures (Casasús et al. 2007), which in turn may be of particular interest in HNV farmland (EEA 2004; Caballero 2007). In contrast, reduced grazing pressure or even the disappearance of grazing livestock in some areas is a consistent trend that has been described by Bernués et al. (2011) in Mediterranean countries, which may have detrimental effects on pastures. However, the consequences may depend on the vegetation type and its successional stage, grazing regime, and the socio-economic environment that is associated with livestock farming systems in the area of study.

Within this framework, the objectives of this study were to determine the effects of prevention from grazing on shrub and herbaceous vegetation in SCGNP pastures and to compare them with adjacent control areas that were grazed at the current moderate-to-low stocking rates.

METHODS

Study Area

The research was carried out in the Sierra y Cañones de Guara Natural Park, which is a protected Mediterranean area of over 80 000 ha (lat 47°17′N, long 0°13′W, Huesca, Spain). The park is located in a calcareous mountain chain that is rich in karstic formations, with altitudes ranging between 430 and 2 077 m. The Guara mountain range longitudinally divides the park into two faces: the northern face is characterized by 900–1 000 mm of annual rainfall, 10°C average annual temperature, and a summer water balance of –250 mm, while the southern face is characterized by values of 600–700 mm, 13°C, and –550 mm, respectively (Del Valle 1996). Concerning fire regime, a 24-yr study (Gobierno de Aragón 1998) indicated that the frequency of wildfires and burned areas in the park were on average 4.2 and 431 ha per year, respectively, which occurred mostly in the shrublands. Summer was the main fire season (52% of the fires were registered during July and August), with a second peak in the winter (22%). Forest fires that were caused by lightning made up 33% of fires (occurring more frequently in the summer), while anthropogenic causes were responsible for 25% of fire events (which were mostly associated with agricultural practices in the winter and negligence in the summer). A decreasing trend was observed over this period, and the fire risk is currently very low or low in most of the park's area (90.6%). Using fire for the improvement of pastures has not been permitted in the natural park since 1997, and it is only allowed for agricultural purposes under explicit permission during the winter.

The most abundant vegetation in the park are shrub pastures (49%), followed by dense (29%) or open (7%) pine (*Pinus sylvestris*) or holm oak (*Quercus ilex subsp. rotundifolia*) forest pastures, agricultural crops (7%), or mountain summer pastures (1%), with the remaining 6% in nonproductive land (Asensio and Casasús 2004). A livestock census was conducted in the region by the same authors, revealing a population of 32 651 meat sheep, 1 199 beef cows, 700 adult goats, and 259 mares. Only 53% of the total area of the park was grazed by domestic animals, with an average stocking density of 0.15 animal units (AUs) per ha (in 92% of the grazed area, the annual stocking density was lower than 0.25 $AU \cdot ha^{-1}$). Grazing took place in different pasture types according to their relative abundance, productivity, and accessibility: the main grazed plant types were shrub and dense forest pastures (54% and 25% of the total grazed area, respectively), whereas agricultural crops (9%), open forest pastures (7%), summer mountain pastures (1%), and nonproductive land (4%) were less important. More information on grazing and general herd management can be found in an article by Riedel et al. (2007).

Experimental Design

The study consisted of a randomized complete block, where shrub and herbage characteristics were compared between nongrazed and grazed areas (treatment vs. control) over the course of 5 yr (1 to 5) with six replications (locations 1 to 6) and two subsamples per location.

Six locations (1 to 6) were selected in pastures that are currently grazed by sheep or beef cattle and dominated by shrubs, according to pasture type, shrub cover, and geographical location within the park (Asensio and Casasús 2004), so that the most characteristic areas were represented. Farmer willingness to collaborate in the study was also considered. The locations were at an average altitude of 994 m (with values from 715 to 1 220 m; i.e., the medium altitude range of the park) and homogeneously distributed across the N and S sides; all of them were placed on public land belonging either to the municipality or the regional government. According to the information that was provided by farmers at the start of the experiment, grazing periods and stocking rates were similar to those observed during the period of study in all locations during the previous decade at the least, although they all agreed that in a more distant past, the stocking rates were higher.

The shrub cover in these locations was visually appraised at each site by two independent technicians at the start of the study and was on average 30% (ranging from 15% to 48%). Table 1 offers a detailed description of the locations and their physical characteristics, pasture types, and grazing management (livestock species, length of the grazing season, and stocking rate). All aspects of grazing management remained constant during the 5-yr study period in all locations.

Two neighboring plots from each location that were considered by the farmers to be representative of each pasture area and were less than 50 m apart were selected as replicates. Although we are aware that this may not have been sufficient to capture the existing variability in vegetation of the whole area,

Table 1. Characteristics of the pasture locations that were selected for study.

Location	Altitude, face	Vegetation: dominant shrub species, [shrub cover][1]	Livestock use: flock size and species, total area, no. grazing months [stocking rate][2]
1	715 m South	*Thymus* sp., *Genista scorpius, Lavandula angustifolia, Santolina chamaecyparissus* [48%]	700 ewes, 500 ha, 6 mo (April–September) [1.68 AUM · ha^{-1}]
2	1 062 m North	*Genista scorpius, Thymus* sp., *Juniperus communis* [35%]	1 000 ewes, 1 000 ha, 6 mo (May–October) [1.2 AUM · ha^{-1}]
3	1 048 m North	*Genista scorpius, Echinospartum horridum*, *Juniperus communis* [15%]	200 cows, 500 ha, 3 mo (July–September)[1.2 AUM · ha^{-1}]
4	1 076 m South	*Prunus spinosa, Genista scorpius*, *Thymus* sp. [29%]	1 600 ewes, 1 500 ha, 4 months (June–September) [0.85 AUM · ha^{-1}]
5	845 m South	*Genista scorpius*, *Lavandula angustifolia*, *Thymus* sp. [20%]	550 ewes, 1 000 ha, 6 mo (April–September) [0.66 AUM · ha^{-1}]
6	1 220 m North	*Echinospartum horridum*, *Crataegus monogyna, Genista scorpius* [33%]	100 cows, 1 200 ha, 6 mo (May–October)[0.5 AUM · ha^{-1}]

[1]Shrub cover: average shrub cover at the study plots appraised at the start of the study.
[2]To calculate stocking rate, animal unit months (AUMs) per ha were used, where one animal unit (AU) equals one mature 500-kg cow or five sheep.

we assumed that these characteristic plots will reliably reflect general trends in the shrubland pastures that they represent.

In each of the 12 replicate plots, 10×10 m exclosures were built at the start of the study (spring yr 1), with 1.8-m-high fences to prevent large animal grazing in order to establish comparisons between the herbage and shrub characteristics among the nongrazed exclosures and the grazed control areas.

Herbaceous and shrub vegetation were characterized inside (nongrazed) and outside (grazed) the fenced areas before (April) and after (December) each vegetative growth season (spring to autumn) from yr 1 (2001) to yr 5 (2005), a period that consisted of five grazing seasons. Samples in the grazed areas were taken approximately 5 m outside of the exclosure to ensure a sufficient proximity that minimized spatial variation while livestock distribution was not disturbed by the presence of the exclosure (Casasús et al. 2007).

Shrub Vegetation

Shrubs were studied in two fixed transects of 1×10 m per replicate, with one inside (nongrazed) and one outside (grazed) each exclosure. The most abundant shrub species were *Genista scorpius* (a perennial shrub of the *Leguminosae* family that is 1–2 m high with dense branches and thorn-like leaves), *Thymus* sp. (an aromatic shrub of the *Labiatae* family with dense branches and tiny leaves), *Echinospartum horridum* (a thorny-cushioned legume that forms large, dense monospecific patches), *Juniperus communis* (a low spreading shrub of the family *Cupressaceae,* with needle-like leaves and medium canopy density), *Santolina chamaecyparissus* (a dwarf evergreen aromatic shrub of the *Asteraceae* family with dense leaves and flowers), and *Lavandula angustifolia* (a small flowering aromatic shrub with leaves that form a rosette near the soil, family *Lamiaceae*). All of these plants are native to the area and are characteristic of the *Buxo-Quercetum rotundifoliae* and *Echinospartum-Lavanduletum pyrenaicae* associations, which prevail in the area (S and N sides, respectively; Montserrat 1986). The dominant species at each location are described in Table 1.

All individuals with roots within the transect (total $n = 367$ in 24 transects at the start of the study) were counted, marked, identified by species and maximum height, and their longitudinal and transverse diameters were measured once each year in the early spring (April). The shrub biomass was estimated using allometric regression equations that were developed by Torrano (2001) and Riedel et al. (2005), which are species-specific (Table 2). To complete these equations, the height as well as the longitudinal and transverse diameters of 20 individuals per species were measured directly in the field with a 1.5-m-long ruler to the nearest 0.5 cm. Shrub theoretical volume was estimated by considering them to be cylinders with an ellipsoid base (volume $= (\pi/4) \times$ height $\times$ longitudinal diameter $\times$ transverse diameter). After taking these measurements, the shrubs were cut at soil level and oven-dried to a constant weight, which was considered to be the real aerial shrub phytomass. Equations were then obtained by linear regression of the shrub aerial phytomass for the plant's apparent volume. Data for the volume and biomass of individual shrubs were pooled per plot, and the plot means of individual volumes and biomass (total

Table 2. Prediction equations for shrub biomass for the most important shrub species.

Species	Equation	Reference
Genista scorpius (L.) DC.	$DM(g) = 1\,175.6 \cdot Vol.\ (m^3)$	Torrano (2001)
Buxus sempervirens L.	$DM(g) = 1\,267.9 \cdot Vol.\ (m^3)$	Torrano (2001)
Prunus spinosa L.	$DM(g) = 696.7 \cdot Vol.\ (m^3)$	Torrano (2001)
Thymus sp.	$DM(g) = 1\,801.2 \cdot Vol.\ (m^3)$	Torrano (2001)
Santolina chamaecyparissus L.	$DM(g) = 3\,551.1 \cdot Vol.\ (m^3)$	Riedel et al. (2005)
Echinospartum horridum (Vahl) Rothm.	$DM(g) = 7\,252.8 \cdot Vol.\ (m^3)$	Riedel et al. (2005)

and per shrub species) were retained for statistical analyses. Finally, while the species-specific equations were applied to 77% of the shrub individuals that were identified in all 10-m^2 transects, total shrub biomass per ha was inferred by multiplying the obtained values by a correction factor of 1.3 (=100/77).

Herbaceous Vegetation

The most abundant herbaceous species in the study pastures were perennial grasses, mostly *Brachypodium pinnatum* (L.), *Brachypodium retusum* (L.), *Festuca arundinacea* subsp. *arundinacea*, and *Bromus erectus*, with a lower presence of legumes (*Vicia cracca* L., *Lotus corniculatus* L.), other graminoids (*Carex* spp.), and forbs (*Aphyllanthes monspeliensis* L., *Helianthemum nummularium* L.); all of them were native to these pastures.

Sward height was measured twice each year, once before (April) and once after the vegetative growth period (December) with a Hill Farm Research Organization (HFRO; Barthram, 1986) sward-stick to the nearest 0.5 cm at 60 points that were chosen at random within the 10×10 m plots in the grazed and nongrazed areas. Biomass availability was then calculated according to an equation that was developed by Casasús et al. (2004) for these pastures:

$$\begin{aligned} &\mathit{Herbage\ biomass}(\mathit{kg\ dry\ matter}\ [\mathrm{DM}]\cdot \mathit{ha}^{-1}) \\ &\quad = 86.59 \times \mathit{sward\ height}(\mathit{cm}) + 531.12 \rightarrow (R^2 = 0.75) \end{aligned}$$

This regression equation was obtained at the start of the study by measuring herbage height with the HFRO sward-stick at 10 points inside ten 0.25-m^2 quadrats that were randomly located outside the exclosures for each of the 12 replicates. Herbage within the quadrats was then clipped with an electric mower to 2 cm above ground level, and then oven-dried to a constant weight. A general equation relating herbage biomass to sward height was derived for all plots.

Herbage samples were taken at the same dates in grazed and nongrazed areas by clipping all plant material that was 2 cm above ground level with an electric mower in six 0.25×0.25 m quadrats that were randomly located in the replicate plots. Green and dead herbage fractions were hand-separated and oven-dried at 60°C for 48 h in order to calculate their relative contribution to the total herbage biomass. After drying, the green and dead samples obtained from each quadrat were pooled, milled through a 0.75-mm screen, and stored for further analysis. Crude protein (CP) was analyzed using the Dumas method with an Elemental NA2100 Protein Analyzer. Neutral detergent fiber (NDF), acid detergent fiber (ADF), and acid detergent lignin (ADL) were analyzed according to standard procedures (Goering and Van Soest 1970).

Statistical Analysis

The normal distribution of data was tested with the Shapiro-Wilk test using SAS statistical software (version 9.1, SAS Institute, Inc, Cary, NC). The normality of the variable number of individuals per 10-m^2 transect from yr 1 to 5 could not be confirmed and was therefore analyzed using the Kruskal-Wallis nonparametric one-way analysis of variance (ANOVA) test, using the NPAR1WAY procedure from SAS. The effects that were tested in each one-way analysis were the management (nongrazed treatment vs. grazed control), year (1 to 5), location (1 to 6), and shrub species (using the most abundant ones as described in Table 2), with the plot as the statistical unit. The nonparametric two-sample Wilcoxon rank-sum test was used to contrast paired samples (nongrazed vs. grazed within the year, yr 1 vs. yr 5 within nongrazed or grazed plots). The data are reported as the means and their associated standard errors (SEs).

The rest of the variables were normally distributed and were therefore analyzed using a mixed model ANOVA for repeated measures with the PROC MIXED procedure of SAS. The dependent variables under consideration were individual shrub volume (cm^3) and biomass (g), total shrub biomass (kg $DM \cdot ha^{-1}$), herbage height (cm), total herbage biomass (kg $DM \cdot ha^{-1}$), percentage of green and dead fraction (%), green fraction biomass (kg $DM \cdot ha^{-1}$), dead fraction biomass (kg $DM \cdot ha^{-1}$), and quality attributes (% NDF, ADF, ADL, and CP). The plot was considered to be the statistical unit.

Management (nongrazed vs. grazed control), year (1 to 5), season (spring vs. autumn, not considered within the analyses of total shrub biomass per ha because it was only measured in spring), location (1 to 6), and their interactions were initially tested as fixed effects, but nonsignificant interactions were eliminated from the analysis. The replicate plot (1 to 12) was included as a random effect in the repeated measurement analysis with an unstructured covariance matrix within the plot that provided the closest-to-zero Akaike information criterion. For analyses of individual shrub biomass per species, the mean values per plot and species were considered, and in addition to those mentioned above, the species was also included as a fixed effect in the mixed model ANOVA.

The results are presented as least squares (LS) means and residual standard deviation (RSD), with differences between LS means tested with a *t*-test. The rates of annual accumulation of shrub and herbage biomass per ha were estimated using the REG procedure in SAS.

Abiotic factors (altitude and face) and grazing management elements (livestock species and stocking rate) were initially considered in the models but were eventually left out because they did not reach significance. In all cases, only the effects with P values < 0.05 were declared significant, while P values < 0.10 are discussed as trends.

RESULTS

Shrub Vegetation

Grazing exclusion over a 5-yr period resulted in increased individual shrub volume and biomass as well as an increase in the total number of shrubs per transect from the start to the end of the study (Table 3). In the adjacent control areas that were grazed at moderate-to-low stocking rates, only the total number of shrubs tended to increase during the same period.

There were no significant differences between nongrazed exclosures and control grazed areas in shrub dimensions or above-ground biomass at the start of the experiment (yr 1). However, at the end of the study (yr 5), the shrub volume was higher ($P < 0.001$), and biomass tended to be higher ($P = 0.09$) in the nongrazed areas. Individual shrub volume and biomass almost doubled in nongrazed areas from the start to the end of

Table 3. Individual shrub volume and biomass (plot means) and number of shrubs per 10-m^2 transect in nongrazed and grazed areas during yr 1 and yr 5 of the study.

	Yr 1	Yr 5	Yr 1 vs. 5[1]	RSD
Individual volume,[2] cm^3 $\times$ 10^3				
Nongrazed	272	550	**	—
Grazed	126	182	NS[5]	189.0
Nongrazed vs. grazed[3]	NS	***	—	—
Individual DM[2], g[2]				
Nongrazed	262	539	$P = 0.06$	—
Grazed	139	251	NS	317.4
Nongrazed vs. grazed	NS	$P = 0.09$	—	—
No. individuals/transect[4]				
Nongrazed	17.8 ± 4.99	45.2 ± 5.00	*	—
Grazed	18.4 ± 4.99	41.4 ± 5.00	$P = 0.07$	—
Nongrazed vs. grazed	NS	NS	—	—

[1]Significance of year effect (yr 1 vs. yr 5) within nongrazed or grazed plots.
[2]Normally distributed: differences explained with a mixed model analysis of variance and data presented as LS means.
[3]Significance of differences between nongrazed vs. grazed areas within yr 1 or yr 5.
[4]Nonnormally distributed: differences explained with nonparametric Kruskal–Wallis test and data presented as means ± standard error.
[5]NS indicates not significant; RSD, residual standard deviation.

the study ($P < 0.01$ and $P = 0.06$, respectively), while both traits remained close to their initial levels in the control grazed areas.

Individual shrub size and biomass differed among the six locations over the length of the study, with higher individual volume and biomass in location 4 (601×10^3 cm^3 and 541 g) than the rest (ranging from 122 to 347×10^3 cm^3 and 148 to 254 g, $P < 0.001$), although the effects of exclusion vs. grazing were similar at all sites. These parameters were also dependent on shrub species ($P < 0.001$), with a range in average volume from 10.1 ± 1.7 SE$\times 10^3$ cm^3 in *Thymus* spp. to $1\,492 \pm 454$ SE$\times 10^3$ cm^3 in *Rosa canina*, and in average biomass from 18 ± 3.1 SE g in *Thymus* spp. to 946 ± 343 SE in *Echinospartum horridum* ($P < 0.001$). Some species are characteristically smaller and lighter in biomass than others (e.g., *Thymus* spp. and *Santolina chamaecyparissus*, although their canopy is dense as shown in Table 2); some were of average size and different densities (e.g., *Lavandula angustifolia, Dorycnium pentaphyllum*, or *Echinospartum horridum*, the second being particularly thin, while the latter was the thickest in the sample), and others were larger (e.g., *Prunus spinosa, Genista scorpius, Buxus sempervirens, Juniperus spp*., and *Rosa canina)* but of a medium to low canopy density. The biomass of the most abundant shrub species increased from yr 1 to yr 5 (such as *Genista scorpius* and *Echinospartum horridum,* which made up 31% and 11% of the total observations, respectively), but only the response of *Thymus* spp. (16% of individuals) differed significantly between the exclosures and the control areas. In this species, the size and biomass doubled throughout the study in nongrazed areas while they remained at their initial levels in the grazed control areas and were therefore significantly different during yr 5 (30 ± 6.6 SE vs. 15 ± 2.6 SE g in nongrazed and grazed areas, respectively, $P < 0.05$).

The total number of shrubs per transect did not differ between nongrazed and grazed areas either at the start or at the end of the study (Table 3). A large increase of the shrub population density was observed both in nongrazed and grazed pastures between yr 1 and 5, and although it was only significant in nongrazed exclosures, a trend ($P = 0.07$) was also observed in grazed control areas. Population trends differed slightly among locations ($P < 0.05$), with an increase in shrub numbers from yr 1 to 5 at locations 1, 5, and 6 (+38.9 individuals per 10-m^2 transect), and no significant temporal changes in the rest of the locations (+11.9 individuals). However, all locations all responded similarly to exclusion vs. grazing. Although the population numbers of shrubs increased throughout the study for most species (particularly *Echinospartum horridum*, *Prunus spinosa*, and *Crataegus monogyna*), the increase only reached significance in *Thymus* spp. (from 7.5 ± 1.8 SE shrubs per 10-m^2 transect in yr 1 to 44.2 ± 9.3 SE, $P < 0.05$) in yr 5, which was observed both in the exclosures and the control areas.

The changes in total shrub biomass per ha during the study from nongrazed exclosures and control grazed areas are presented in Figure 1. There were no significant differences between them at the start of the experiment (3 610 vs. 2 145 kg DM $\cdot$ ha^{-1} in nongrazed and grazed areas in yr 1, respectively, NS), but thereafter, a trend was noted toward higher shrub

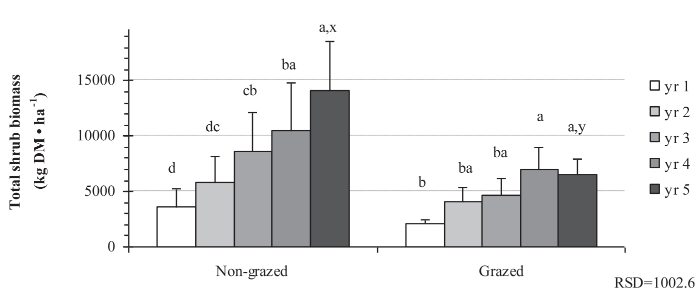

Figure 1. Effect of time after the start of the experiment on aboveground total shrub biomass (kg DM $\cdot$ ha^{-1}) in nongrazed and grazed areas. Inferred from the values obtained in the species with existing prediction equations (77% of the individuals). a, b, c, d: means within nongrazed or grazed areas lacking a common superscript letter differ among years ($P < 0.05$). x, y: means within year lacking a common superscript letter differ among nongrazed and grazed areas ($P < 0.05$). Vertical bars indicate the standard error of the mean.

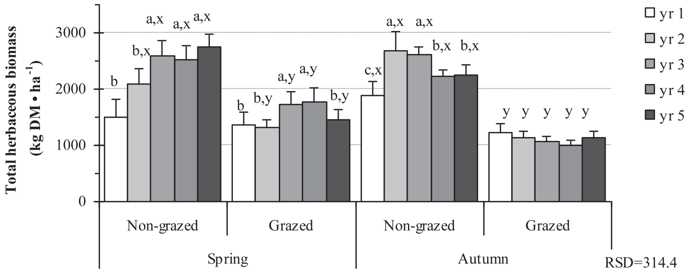

Figure 2. Effects of time after the start of the experiment on aboveground total herbaceous biomass (kg DM $\cdot$ ha^{-1}) in nongrazed and grazed areas during the spring and autumn. a, b, c: means within nongrazed or grazed areas and season lacking a common superscript letter differ among years ($P < 0.05$). x, y: means within year and season lacking a common superscript letter differ among nongrazed and grazed areas ($P < 0.05$). Vertical bars indicate the standard error of the mean.

biomass in the nongrazed areas ($P = 0.09$ in yr 3), which resulted in significance at the end of the study (yr 5, 14 092 vs. 6 536 kg DM $\cdot$ ha^{-1} in nongrazed and grazed areas, respectively, $P < 0.01$). Shrub biomass per ha was different among the six locations throughout the study, with lower values in location 2 (711 kg DM $\cdot$ ha^{-1}), intermediate in locations 3, 5, and 6 (6 008, 5 962, and 5 588 kg DM $\cdot$ ha^{-1}, respectively), and higher values in locations 1 and 4 (11 113 and 10 910 kg DM $\cdot$ ha^{-1}, respectively, RSD = 1 002.6, $P < 0.001$), but there was no interaction between the location and the effect of exclusion vs. grazing. According to the model, the rate of annual accretion for shrub biomass was 2 563 $\pm$ 1 030.2 SE kg DM $\cdot$ ha^{-1} $\cdot$ yr^{-1} in nongrazed exclosures ($P < 0.05$), although a net increment was also shown for grazed areas (1 173 $\pm$ 424.3 SE kg DM $\cdot$ ha^{-1} $\cdot$ yr^{-1}, $P < 0.05$).

Herbaceous Vegetation

Sward height was similar in nongrazed and grazed areas in the spring of yr 1 (11.3 vs. 9.5 cm, respectively, RSD = 3.63, NS), but the influence of grazing management was already evident in the autumn of yr 1 after the first grazing season (15.7 vs. 8.1 cm, respectively, RSD = 3.63, $P < 0.001$), and the values in the exclosures and grazed control areas were consistently different thereafter. Accordingly, the estimated herbaceous biomass per ha was significantly different in nongrazed and grazed areas from the autumn of yr 1 onwards (Fig. 2). Herbage biomass per ha was different among the sites, with lower values in locations 1, 3, 5, and 6 (1 396, 1 572, 1 525, and 1 567 kg DM $\cdot$ ha^{-1}, respectively), intermediate in location 2 (2 064 kg DM $\cdot$ ha^{-1}) and at a maximum in location 4 (2 783 kg DM $\cdot$ ha^{-1}, RSD = 314.4, $P < 0.001$) throughout the study, but there was no significant interaction between location and exclusion vs. grazing. The rate of annual accumulation for herbage biomass, which was measured in the spring, was 291 $\pm$ 85.3 SE kg DM $\cdot$ ha^{-1} $\cdot$ yr^{-1} in nongrazed areas ($P < 0.001$), while no increment was observed in the grazed areas (NS).

The effect of exclusion vs. grazing on the proportion of green herbage and total green and dead herbage biomass per ha is shown in Table 4. While there were no differences in the relative proportions of green and dead herbage in the spring at the start of the study (yr 1), a lower proportion of green herbage was observed for the nongrazed exclosures by the autumn of yr 1, which was constant throughout the rest of the study. As a consequence of both herbage biomass and dead fraction increments, total green herbage biomass and especially total dead herbage biomass were significantly higher in nongrazed areas at the end of the study (yr 5), where the latter almost doubled the initial values that were observed in the spring.

The effects of time after the start of the study and exclusion vs. grazing on herbage quality are described in Table 5. There were no differences in herbage quality at the start of the experiment (spring yr 1), but the herbage inside the exclosures had lower protein during the autumn after the first grazing season as well as a higher fiber content (NDF and ADF), which corresponded with the observed proportions of green and dead herbage. Although the differences were slight and not consistent throughout the study, some relevant effects on

Table 4. Characteristics of herbage biomass in nongrazed and grazed areas in yr 1 and yr 5 of the study.[1]

	Spring			Autumn			
	Yr 1	Yr 5	Yr 1 vs. 5[2]	Yr 1	Yr 5	Yr 1 vs. 5	RSD
% Green herbage							
Nongrazed	60%	58%	NS[4]	36%	27%	*	0.1%
Grazed	60%	79%	***	47%	54%	$P = 0.07$	
Nongrazed vs. grazed[3]	NS	***		**	***		—
Green biomass, kg DM $\cdot$ ha^{-1}							
Nongrazed	864	1 575	***	619	605	NS	223.8
Grazed	764	1 171	***	536	593	NS	
Nongrazed vs. grazed	NS	***	—	NS	NS	—	—
Dead biomass, kg DM $\cdot$ ha^{-1}							
Nongrazed	645	1 165	***	1 268	1 654	**	292.70
Grazed	588	293	*	695	551	NS	
Nongrazed vs. grazed	NS	***	—	***	***	—	—

[1]Normally distributed: differences explained with a mixed model analysis of variance and data presented as LS means.
[2]Significance of year effect (yr 1 vs. yr 5) within nongrazed or grazed plots.
[3]Significance of differences between nongrazed vs. grazed areas within yr 1 or yr 5.
[4]NS indicates not significant; RSD, residual standard deviation.

Table 5. Herbage quality in nongrazed and grazed areas in yr 1 and yr 5 of the study.[1]

	Spring			Autumn			
	Yr 1	Yr 5	Yr 1 vs. 5[2]	Yr 1	Yr 5	Yr 1 vs. 5	RSD
CP,[3] $g \cdot kg^{-1}$ DM							
Nongrazed	72.4	80.3	NS	59.8	87.6	***	14.8
Grazed	81.9	84.6	NS	84.7	99.1	*	
Nongrazed vs. grazed[4]	NS	NS	—	***	0.06	—	—
NDF, $g \cdot kg^{-1}$ DM							
Nongrazed	654.6	707.1	***	695.0	717.3	NS	36.2
Grazed	659.7	702.6	*	638.3	701.5	***	
Nongrazed vs. grazed	NS	NS	—	**	NS	—	—
ADF, $g \cdot kg^{-1}$ DM							
Nongrazed	351.8	372.7	**	374.0	385.6	NS	17.2
Grazed	351.6	358.3	NS	336.7	355.1	*	
Nongrazed vs. grazed	NS	*	—	***	***	—	—
ADL, $g \cdot kg^{-1}$ DM							
Nongrazed	45.4	58.5	***	53.1	63.4	***	7.00
Grazed	46.4	50.3	NS	52.5	60.6	*	
Nongrazed vs. grazed	NS	**	—	NS	NS	—	—

[1]Normally distributed: differences explained with a mixed model analysis of variance and data presented as LS means.
[2]Significance of year effect (yr 1 vs. yr 5) within nongrazed or grazed plots.
[3]ADF, acid detergent fiber; ADL, acid detergent lignin; CP, crude protein; NDF, neutral detergent fiber; NS, not significant; RSD, residual standard deviation.
[4]Significance of differences between nongrazed vs. grazed areas within yr 1 or yr 5.

forage quality, mainly in relation to fiber content, appeared in the autumn and/or spring of yr 5 that were associated with grazing management. All fiber fractions (NDF, ADF, and ADL) increased during the experiment in the nongrazed exclosures for the spring samples, although it is remarkable that increases were also observed in some traits and samples from the grazed control areas.

DISCUSSION

Grazing Exclusion and Shrub Population and Biomass

Overall, we found increased recruitment of shrubs in this natural park, typical of Mediterranean mid-mountain areas, regardless of the presence or absence of livestock. However, the exclusion from grazing by domestic animals had an impact on shrub dimensions and biomass; both parameters were higher in nongrazed exclosures than in the adjacent grazed areas after 5 yr. Because the exclosures and grazed control areas were only 5 m apart within the location and replicate and did not differ at the start of the experiment, final differences can largely be attributed to grazing management.

The accretion of total shrub biomass per ha was the consequence of an increase in both shrub populations and individual phytomass in nongrazed areas, as Casasús et al. (2007) had observed in Mediterranean pine forests, and only for shrub numbers in grazed areas. This finding may be explained by the fact that most of the native species that were identified here produce many seedlings, even under grazing conditions. However, their architecture can be affected by the presence of livestock; i.e., both consumption and physical damage such as animal trampling can influence their growth and reproduction patterns (Magda et al. 2009), mostly directly but also indirectly, by modifications of the hydrology and soil conditions.

The large increment of total shrub phytomass that was observed in nongrazed areas is in accordance with the results of Prieto et al. (2009), who found that shrub biomass in Mediterranean shrubland doubled in undisturbed, nongrazed areas within a similar period. Similar responses to reduced pasture use or even abandonment have been described in other Mediterranean mountain areas during different periods of study (Bartolomé et al. 2000; Roura-Pascual et al. 2005).

The fact that total shrub biomass per ha also increased in grazed areas, although to a lesser extent than in ungrazed ones, indicates that the stocking rates and grazing regimes presented here were able to modulate but not stop the succession of shrub vegetation toward the climax vegetation types. According to Montserrat (1987), these populations would consist of dense holm oak forests on the southern side (in association with *Quercetum rotundifoliae* subas. *rhamnetosum infectoriae*) and of oaks on the northern side (*Buxo-Quercetum pubescentis* subas. *quercetosum subpyrenicae*). Although Casasús et al. (2007) described livestock grazing at moderate (but higher than in our study) stocking rates that could halt shrub encroachment in Mediterranean forests, similar results to those obtained here have been shown in other studies that were also conducted in Mediterranean areas (Carmel and Kadmon 1999; Bartolomé et al. 2000; Henkin et al. 2005).

This increased woody plant density responds to the natural succession process because most Mediterranean grasslands are derived from original forests that were altered through centuries of interaction with grazing livestock (Zarovali et al. 2007). When grazing pressure is reduced, as has occurred in this natural park over recent decades (Riedel et al. 2007), grasslands might gradually return to the previous situation. However, secondary succession is not a continuous process but rather a series of discrete alternative states (Suding et al. 2004), which seem relatively permanent by human-scale time periods, between which transitions may occur as a result of spontaneous or human-caused disturbances (Westoby et al. 1989). Because most of the species that are described here can be viewed as colonizers following a disturbance (Gallego Fernández et al. 2004), this finding would indicate that the transition toward closed shrubland was still incipient but already established in these pastures.

The results that were obtained here may be generalized to larger areas, as stocking rates and vegetation types in the current study are common to many extensive ruminant production systems in Mediterranean mid-mountain rangelands (Barrantes et al. 2009; Jouven et al. 2010), and so are the problems related to the abandonment of marginal land and the decline of traditional farming practices (Bernués et al. 2011). In a similar vein, Evlagon et al. (2012) indicated that the potential carrying capacity of Mediterranean forests in fire-prone environments was considerably greater than the number of livestock that were available for grazing the combustible shrubs; they therefore suggested an increase of the grazing pressure to effectively reduce the fire hazard.

Grazing exclusion affected sites in a similar manner, but there were noteworthy differences in the magnitude of response among the sites. This difference may be associated with the different disturbance histories of each pasture, agro-climatic condition, vegetation type, and species pool (Lunt et al. 2007; Sebastiá et al. 2008), although no effects from any single factor or simple combination proved significant within the current study. We found no significant effect on vegetation dynamics in relation to the species of livestock that was present at each site. Small ruminants, particularly goats, are known to include a high proportion of shrub components in their diets (Bartolomé et al. 1998), but cattle in SCGNP also incorporate a relatively high proportion of shrubs into their diets when herbage resources are scarce (Casasús et al. 2009). The relative abundance of the different shrub species may also account for site differences, as they may respond differently to grazing according to their acceptability by livestock and to their preferences for light intensity or soil conditions (Pykälä 2005). Some of them can even be enhanced by endozoochorous seed dispersal, while in other species, such as *Thymus* sp., the population has increased irrespective of the presence or absence of livestock, but individual shrub biomass is lower in grazed areas than in exclosures, which suggests that they are adapted to herbivory (Navarro et al. 2006). In summary, the differences that were observed between the sites suggest that the succession direction process is consistent in the area of study but that the intensity is not, although the response to a reduction in grazing pressure is unequivocal.

The shrubs in these ecosystems play important roles in terms of landscape heterogeneity and protection of soils because they prevent sediment yield and surface runoff, particularly on steep slopes (Molinillo et al. 1997). In terms of livestock production, they can constitute a strategic resource for grazing flocks, particularly during periods of scarcity for other forages (Papachristou et al. 2005). However, the threshold shrub densities that have been quantified above indicate the negative consequences that may be observed on landscape connectivity and biodiversity (McEvoy et al. 2006; Fonderflick et al. 2010), as well as the risks and difficulties related to controlling forest fires (Kramer et al. 2003). Bartolomé et al. (2000) set this threshold shrub cover at approximately 70% in similar Mediterranean areas, which is far from the values that were observed in all study sites, but it is possible that this shrub cover may be reached in the medium-long term, particularly if grazing pressure is allowed to decrease further, as seems to be the case (Riedel et al. 2007).

Grazing Exclusion and Herbaceous Biomass and Quality

Grazing exclusion caused a significant increase in herbage height and biomass in comparison to the initial conditions, while both parameters were constant under grazing conditions.

The effect of reduced grazing pressure on herbage biomass has been reported in other studies of natural pastures (Hope et al. 1996), which were sometimes concomitant with a higher shrub cover (Zarovali et al. 2007). Other authors also describe lower forage accumulation in grazed areas, together with reductions in vegetation cover (Cole et al. 2004; McEvoy et al. 2006), but the effects on herbage quality seem to be less evident. The variability among locations, lower than that of shrub biomass, could be related to pasture type and climatic conditions because herbage biomass tended to be higher at sites with greater altitude and precipitation, which are determinants of grassland productivity (Smit et al. 2008).

In the current study, the herbage quality was reduced and the proportion of dead material was increased in nongrazed exclosures. Other authors have also noted lower quality in the ungrazed areas or highly encroached pastures (Molinillo et al. 1997; Zarovali et al. 2007) or a positive response in herbage quality after grazing re-introduction (Bokdam and Wallis de Vries 1992). This pasture response may be explained by the compensatory growth of grazed plants (McEvoy et al. 2006) or the higher proportion of young, greener plants as mediated by the dispersion of seeds and provision of fertility by grazers (Rook and Tallowin 2003). However, Kesting et al. (2009) found a positive relationship between shrub occurrence and herbage quality, mostly in relation to the different proportions of grasses, legumes, and forbs. The fact that some herbage quality traits were slightly reduced in grazed areas at some time points in our study would indicate that livestock could control the accumulation of forage biomass, but the low stocking rates would allow for the selection of preferred species and would in turn increase the frequency of avoided, low-quality species (Mayer et al. 2009).

The consequences of an accumulation of biomass and mostly dead material in nongrazed areas can be detrimental to the preservation of the traditional landscape and can contribute to the risk of forest fires, which are associated with high fuel loads and a particular vertical structure of the vegetation (Kramer et al. 2003), while the lower forage quality would reduce the carrying capacity of these areas. The current herbage quality is relatively low when compared to intensively managed pastures, and as in other areas, it precludes sustained year-round grazing (Bokdam and Wallis de Vries 1992). In fact, only livestock with moderate nutritional requirements had adequate performance on these pastures (Riedel et al. 2004), and other complementary foraging areas or stall-fed diets were needed to complete the annual production cycles.

IMPLICATIONS

The shrub vegetation in this representative Mediterranean mountain area is undergoing a significant process of succession toward encroachment, which would be particularly dramatic in the event of further reductions in grazing pressure or livestock disappearance. Livestock grazing can reduce the intensity of this process and the accumulation of low quality herbage, but current stocking rates and grazing management regimes may not be enough to stop the succession. These changes can affect biodiversity and landscape quality; thus, further abandonment of farming and/or intensification should be prevented in these areas. This prevention may be achieved by specific supporting schemes (agri-environmental policies) that promote livestock as a vegetation management tool for environmental services that are increasingly recognized by society.

Further research is needed to determine optimal stocking rates and grazing periods for managing these diverse rangelands and the mechanisms underlying the direction and extent of vegetation response. To ensure the survival of extensively managed livestock that can be used to address conservation goals, it seems crucial that a compromise is reached between optimal livestock productivity and environmental performance. To determine this balance, it is important to objectively value the various ecosystem services provided by pasture-based livestock farming systems.

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