COMPARATIVE ASSESSMENT OF GOODS AND SERVICES PROVIDED BY GRAZING REGULATION AND REFORESTATION IN DEGRADED MEDITERRANEAN RANGELANDS

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ABSTRACT

Several management actions are applied to restore ecosystem services in degraded Mediterranean rangelands, which range from adjusting the grazing pressure to the removal of grazers and pine plantations. Four such actions were assessed in *Quercus coccifera* L. shrublands in northern Greece: (1) moderate grazing by goats and sheep, (2) no grazing, (3) no grazing plus pine (*Pinus pinaster* Aiton) plantation in forest gaps (gap reforestation), and (4) no grazing plus full reforestation of shrubland areas, also with *P. pinaster*. In addition, heavy grazing was also assessed to serve as a control action. We comparatively assessed the impact of these actions on key provisioning, regulating and supporting ecosystem services by using ground-based indicators. Depending on the ecosystem service considered, the management actions were ranked differently. However, the overall provision of services was particularly favored under moderate and no grazing management options, with moderate grazing outranking any other action in provisioning services, and the no grazing action presenting the most balanced provision of services. Pine reforestations largely contributed to water and soil conservation and C sequestration, but had a negative impact on plant diversity when implemented at the expense of removing natural vegetation in the area. Heavy grazing had the lowest provision of ecosystem services. It is concluded that degraded rangelands can be restored by moderating the grazing pressure rather than completely banning livestock grazing or converting them into pine plantations.

KEY WORDS: Carbon sequestration, ecosystem services, forage, grazing management, landscape functional analysis, plant diversity, restoration actions, soil functions

INTRODUCTION

Land degradation in the Mediterranean is a multifaceted process that results from the independent or combined effect of factors such as land cover conversion (Alphan, 2012), agriculture and rangeland mismanagement (Cerdà et al., 2009; García Orenes et al., 2010), and frequent or severe wildfires (Guénon et al. 2013; López-Poma et al. 2014), which are ultimately driven by social and economic changes (Abu Hammad & Tumeizi, 2012), and often lead to soil erosion (Cerdà et al. 2010) and long-term loss of natural vegetation (FAO, 2013). Specifically for rangelands, overgrazing is considered as the key degradation factor in many regions of the world, as it results in increased soil and water losses, decrease and degradation of the vegetation cover, and critical changes in regulating ecosystem services (Cerdà & Lavee, 1999; Mekuria & Aynekulu, 2011; Angassa, 2014). However, grazing can be also seeing as a sustainable use (Vetter & Bond, 2012; Alvarez Martínez et al., 2013; Palacio et al., 2014) if it is appropriately managed (Papanastasis 2009).

Rangelands of the Mediterranean region amount to 830,000 km^2 and are grazed by 270 million sheep-equivalents, corresponding to a stocking rate of about 2.2 sheep equivalents per hectare, which suggests overgrazing (Le Houérou, 1981). However, the grazing pressure is not evenly distributed all over the Mediterranean rangelands; it is higher in the southern than in the northern Mediterranean (Puigdefabregas & Mendizabal, 1998). In Mediterranean Europe, in particular, it is unevenly distributed with areas being highly overgrazed or undergrazed (Roeder et al., 2007; Papanastasis et al., 2009).

The restoration of degraded rangelands requires overcoming two main types of thresholds: biotic (e.g., species composition) and abiotic (e.g., soil degradation and erosion) limitations (Hobbs & Harris, 2001). The former type can be addressed by applying appropriate grazing management. This may include soft measures such as adjustment of livestock numbers to grazing capacity, selection of the right kind of animals, and adoption of the right grazing system (Papanastasis, 2009), and also hard measures such as access roads and watering points to improve animal distribution (Vallentine, 2001). The damaged abiotic conditions, on the contrary, cannot be repaired by the grazing process alone; active restoration measures such as soil rehabilitation and reforestation are common actions to address this problem (Vallejo et al., 2006; Chirino et al., 2009).

In the Mediterranean rangelands, socio-economic reasons often prevent an appropriate regulation of grazing management, particularly in communally used areas. Shepherds usually resist to any changes of their traditional way of grazing state-owned rangelands, especially the reduction of the excess animals in the case of overgrazing (Le Houérou, 1981). To cope with this problem land managers and developmental agencies usually resort to banning of grazing in degraded rangelands for restoration purposes, or to implementing reforestation programs, normally with pine species, and subsequently forbidding livestock grazing (Anthopoulou et al., 2006; Vallejo et al., 2006). In northern Ethiopia, for example, establishing exclosures in degraded communal grazing lands has been suggested in order to restore ecosystem structural and functional properties such as plant species richness and biomass, carbon stocks, and nutrient availability (Mekuria & Aynekulu, 2011; Mekuria 2013). In the Mediterranean region, protection from grazing increases perennial plant biomass, often at the expense of annual plants (Aidou et al., 1998; Tsiouvaras et al., 1998), plant species diversity and evenness (Koutsidou & Margaris, 1998; Papanastasis et.al.2002), plant cover (Papanastasis et al. 2003), and soil conservation (Aidou et al., 1998). Less clear are the effects of pine reforestation accompanied by prohibition of grazing. This is because pine plantations have often yielded very poor results in Mediterranean drylands, hampering the development of spontaneous vegetation (Maestre & Cortina, 2004), decreasing woody species richness (Andrés & Ojeda, 2002), and often being ineffective in regulating water and sediment fluxes (Bautista et al., 2010; Nunes et al., 2011). Although these previous works have provided useful information on the effects of a variety of management and restoration actions in degraded drylands, there is a demand for integrated comparative assessments on the potential of different management approaches to restore the overall functioning and the provision of ecosystem services in drylands (Rojo et al., 2012). Such integrated assessments are lacking for Mediterranean rangelands.

The ecosystem service concept (i.e., the benefits people obtain from ecosystems; MA 2005) offers a harmonized framework for the integrated assessment of conservation and restoration efforts (Egoh et al., 2007; Rey Benayas et al., 2009; Bautista & Lamb, 2013). The assessment of ecosystem condition and services at a management scale should be based on easily measured indicators, requiring minimum training and a reasonable level of effort and cost. Several such indicators have been proposed over the last 10-15 years for the assessment of dryland and rangeland ecosystems (Pyke et al., 2002; Tongway & Hindley, 2004; Mayor et al., 2008). They are based on vegetation attributes and soil properties, which reflect the water, carbon, and nutrient fluxes that determine the functioning of these ecosystems, and therefore the services provided. General indicators of community structure, such as species richness, diversity, and evenness also provide useful information for evaluating the ecosystem response to restoration, as biodiversity has proven to be positively related to the ecological functions that support the provision of ecosystem services in restored areas (Rey Benayas et al. 2009).

In this paper, our aim was to assess the overall provision of ecosystem services and resulting trade-offs between services for standard restoration actions applied in Mediterranean rangelands, which involved regulation of grazing management and pine reforestation. Using ground-based biophysical indicators, we compare the impact of a variety of these practices on key provisioning services (forage production), regulating and supporting ecosystem services (soil and water conservation, carbon sequestration), and biodiversity.

METHODS

Study area

The study was carried out in Lagadas county of central Macedonia, in North Greece. Climate is semi-arid to sub-humid Mediterranean. Soils are shallow, sandy loam and acid mainly derived from metamorphic rocks. The study area amounts to 206 ha and is dominated by rangelands, mainly consisting of *Quercus coccifera* L. shrublands, yet other woody species such as *Quercus pubescens* Wild., *Pyrus amygdaliformis* Vill., *Ligustrum vulgare* L., and *Cistus incanus* L. are also present. Grazing animals are goats and sheep while cattle are less important. Rangelands are public lands but local people who own livestock have the right of free grazing. As a result, they are managed as commons, often resulting in overgrazing and land degradation, particularly in the areas around villages and animal sheds (Roeder et al., 2007).

In order to restore degraded rangelands, national and regional agencies initiated a number of actions in the 1980's, with funds mainly coming from the European Union. The actions included building access roads and watering points so that animal flocks are evenly distributed and moderate grazing is encouraged; banning livestock grazing from degraded rangelands; and combining pine plantations with banning of livestock grazing, so that rangelands are converted to forests and further grazing-driven degradation is halted. For this last type of action, two approaches were followed: reforestations of forest gaps were implemented in areas with moderate previous grazing pressure and enough presence of spontaneous forest tree species; in this case, pine seedlings were planted in digged out holes in the tree openings. Most extensively, full (covering the whole target area) pine reforestations were implemented in heavily grazed areas with no trees; in this case, the soil was mechanically prepared to create troughs and banks along contours to hold the water, with pine seedlings planted in the troughs.

In the Lofiscos village community of the Lagadas county, the whole variety of actions described above were implemented in a rather homogeneous area of degraded rangelands. We selected the following specific actions for assessment (Figure 1) : (a) moderate grazing applied in an area dominated by *Q. coccifera* with a stocking rate of about one sheep equivalent (sheep or goat)/ha/year; (b) no grazing applied for the last 20 years in an area covered by *Q. coccifera* and other deciduous species (e.g., *P. amygdaliformis*); (c) gap reforestation (about 30 years old) with *Pinus pinaster* Aiton, resulting in mixed stands of pines and spontaneous oak species (e.g., *Q.coccifera*, *Q. pubescens*); and (d) full reforestation (about 20 years old), again with *P. pinaster*, that resulted in pure pine stands. In addition, an area dominated by *Q. coccifera* but heavily grazed with a stocking rate of more than three sheep equivalents/ha/year was taken as (d) control action. Each type of action was implemented on a minimum area of about 10 ha.

Assessment of ecosystem services

A variety of ground-based indicators were measured in order to assess the provision of a well-balanced basket of ecosystem services. The assessment indicators were selected according to their potential for describing critical ecosystem services in drylands, and included: (1) landscape pattern metrics and soil functional indicators (following the LFA methodology; Tongway & Hindley, 2004), all of which have proven to relate very well with water and soil conservation functions in dryland ecosystems (Tongway & Hindley, 2000; Maestre & Puche, 2009; Mayor & Bautista, 2012); (2) plant above-ground biomass, which provides information on forage provision (herbaceous biomass) and on carbon sequestration (total aboveground biomass; Brown, 2002); soil organic carbon, which also provides information on carbon stocks and is commonly used as indicator for the carbon sequestration service (Lal, 2009); and plant species richness, Shannon-Wiener diversity and evenness indices, which were used to assess action impact on biodiversity. The field work started in early June and lasted till mid September of 2011. Three slopes were assessed per action type. On each slope, we established three 50-m long transects along and around which the various indicators were assessed. To facilitate comparisons among actions, all slopes had similar angle (20-25%) and aspect (east-facing).

Landscape functional analysis (LFA)

Landscape pattern metrics and soil functional condition were assessed by following the LFA methodology, proposed by Tongway & Hindley (2004). First the various patches and inter-patches on the landscape were identified. A patch is defined as a long-lived feature which obstructs or diverts water flow and collects or filters out material from runoff, while an inter-patch is characterized as a zone where resources such as water, soil materials and litter are freely transported (Tongway & Hindley, 2004). In this study, patches included various species of trees, shrubs, and herbaceous plants as well as troughs and banks while inter-patches included bare soil and litter.

We measured patch length and width as well as inter-patch length along 50-m long transects, starting on the upper edge of each sampling slope and following the maximum slope angle. From these measurements were derived patch cover (percentage of total transect length that was covered by patch zones), patch width, and average inter-patch length for each action. The values obtained for these metrics were then standardized and averaged to derive a single landscape pattern indicator. Soil surface condition for patch and inter-patch areas was assessed on quadrats (50x50 cm in size, divided into 10x10 cm grids) randomly distributed among the main patch types and inter-patch areas of each transect, totaling five replicates for each type of area and action. On these quadrats and following a rating scale (Tongway & Hindley, 2004), we estimated perennial plant cover, litter input (cover, origin and degree of decomposition), biological crust (biocrust) cover, microtopography (surface roughness), compaction (surface resistance to penetration), slake test (stability of natural soil fragments to rapid wetting), and soil texture. For the latter variable, soil samples were taken from a depth of 0-5cm, combined into a single soil sample per transect, and transferred to the soil laboratory for analysis. The same samples were used to measure soil organic carbon. The soil surface condition variables were combined to calculate an integrated value per transect for the three functional indices proposed by Tongway & Hindley (2004): (a) stability index, by incorporating litter cover, biocrust cover, compaction and slake test; (b) infiltration index, by incorporating perennial plant cover, litter input (cover, origin and degree of decomposition), microtopography, compaction, slake test, and soil texture; and (c) nutrient cycling index, by incorporating perennial plant cover, litter input and microtopography. Data from patch and inter-patch quadrats were averaged taking into account the percentage of each type of area for each transect (Tongway & Hindley, 2004). The values for the three indices were averaged to yield a single soil functional index.

Plant cover and diversity

Plant cover and diversity were assessed along the same three transects used for the LFA assessment. Specifically, we applied the line-point sampling method by taking records of plant species, litter (dead material), rock or bare ground (uncovered by vegetation) every 50 cm along each transect, resulting in a total of 100 contacts per transect. Species overlapping in each contact were also recorded. Additionally, 10 equally distributed quadrats (50 x 50 cm each) were systematically placed along each transect and all species present in each one were recorded. Three diversity indices (Shannon-Wiener, evenness and species richness) were calculated for each transect (Magurran, 2004). The Shannon-Wiener and evenness indices were based on the cover data collected from the line-point method. Species richness was estimated as the mean number of species per transect.

Plant biomass

Herbaceous and shrubby biomass was also measured along the 50m transects used for the LFA assessment. Specifically, shrub biomass was estimated for 10 (1 x 1 m) plots that were systematically (every 5m) located along each transect. Within each plot, a 50x50cm subplot was randomly placed for estimating herbaceous biomass. The double sampling technique of visual weight estimation calibrated by harvesting was applied for measuring the biomass (Tadmor et al., 1975). For tree biomass, individual tree biomass was assessed on two 30x30m plots for each of the three actions that had trees, namely no grazing, gap and full reforestation actions. In each plot, the diameter at the breast height (DBH) of all the trees (wider than 5cm DBH) was measured. Subsequently, the above ground tree biomass (Y) was estimated in kg dry weight based on the following equations (Jenkins et al., 2003):

For the oak trees: Y=Exp (-2.0127+2.4342*ln DBH)

For the pine trees: Y = Exp (-2.5356+2.4349*ln DBH)

Total aboveground biomass per replicate was estimated as the addition of the average herbaceous and shrub biomass per transect and the average tree biomass of the action. Herbaceous biomass was used as indicator of forage production, while total biomass was used as indicator of carbon sequestration service.

Data analysis

Both landscape pattern metrics and soil functional indices were calculated using a Microsoft Excel template developed by Tongway (Ecosystem Function Analysis, 2008). The various indicators assessed were compared among actions using One-way ANOVA, followed by a Tukey test for between-action comparisons. To integrate the information provided by the various indicators assessed into an overall assessment of ecosystem services provision, the indicator values per sampling slope were standardized (by subtracting the general mean and dividing by the general standard deviation from all the sampling transects and actions) and then used to yield a composite value per slope for biodiversity (averaging species richness and Shannon-Wiener diversity and evenness indices) and for each type of ecosystem service: forage provision (herbaceous biomass); water and soil conservation (averaging the landscape pattern and soil functional indices); and carbon sequestration (averaging soil organic carbon and total biomass). A global value for ecosystem services provision was estimated for each action as the average of the integrated standardized values for each type of service, including plant diversity.

RESULTS

Plant cover

Plant cover was more than 90% in all actions, except for heavy grazing, which was about 57% (Figure 2). However, its components largely varied among the actions. Specifically, tree cover was absent from the grazing areas, either with moderate or heavy grazing. Shrub cover was particularly high in the heavily grazed and in the gap reforestation areas. Herbaceous cover (grasses and forbs) was dominant in the moderate grazing areas and very low in the reforested and heavily grazed areas.

Landscape Functional Analysis

Two of the landscape pattern metrics assessed, patch cover and width, significantly varied among the five actions assessed (F=19.2, P<0.001, and F=32.7; P<0.001, for patch cover and width respectively; One-way ANOVA). Both patch cover and patch width were higher in moderate grazing, no grazing and full reforestation areas than in heavy grazing and gap reforestation areas (Figure 3). Inter-patch length did not significantly differ between actions (F=2.4; P=0.117). The soil functional index (average of the three LFA indices) significantly varied among the actions (F=116.8; P<0.001), being significantly lower (P<0.005; Tukey test) for the grazing actions, significantly higher for the two reforestation actions, and showing intermediate values for the no grazing action. These differences were consistent for the three original LFA indices assessed (Figure 4).

Plant biomass and soil carbon

Moderate grazing far exceeded any other action in herbaceous biomass, while heavy grazing and gap reforestation far exceeded the other three actions in shrub biomass (Table 1). Tree biomass was the dominant biomass fraction in both types of reforestation actions but also in no grazing areas due to the contribution of oak trees. Tree size was small, with diameter at breast height ranging from 0.10-0.11 m for oaks and 0.14-0.18 for pine trees. Total aboveground biomass significantly varied among the actions (F=319.3; P<0.001), with the highest biomass in gap reforestation, followed by full reforestation and no grazing areas, and with the lowest total biomass in the grazed areas, both heavy and moderate grazing areas (Figure 5).

Soil organic carbon was different among the five actions (*F*=2.9; *P*=0.047), but differences were small (Figure 5) and no particular action showed significantly higher or lower values (p>0.05; Tukey test).

Plant diversity

The five actions assessed showed clear differences in plant diversity (Table 2). Specifically, the highest species richness and Shannon diversity index were found in the no grazing and moderate grazing actions, followed by the gap reforestation action, while much lower values were found in the heavy grazing and full reforestation actions. . The evenness index was not much different among all actions except for the full reforestation one, which had the lowest value (Table 2).

Integrated provision of ecosystem services

Figure 6 shows a synthetic picture of the variation among actions in the provision of four main categories of ecosystem services and biodiversity. Water and soil conservation was similar among all the actions assessed except for heavy grazing, which showed a much lower value; carbon sequestration was mostly favored in no grazing and gap reforestation; forage provision was clearly favored in moderate grazing areas; and diversity was favored in no grazing and moderate grazing areas. When all services were pooled together, the overall provision of ecosystem services showed a decreasing trend from moderate grazing and no grazing areas, with the highest values, to the full reforestation and heavy grazing areas, with the lowest values (Figure 7).

DISCUSSION

The set of indicators assessed for the five management actions selected informed on the impact of these actions on the provision of a well-balanced basket of ecosystem services, including provisioning, regulating and supporting services, and biodiversity (MA, 2005). Depending on the type of ecosystem service addressed, the comparison of the actions yielded different rankings, highlighting the existing trade-offs among services (Carpenter et al., 2009).

For forage provisioning, probably the most relevant provisioning service in rangelands, only herbaceous forage was considered in this work, as it is suitable for all kind of livestock in the area. According to this indicator, moderate grazing far exceeds any other type of management in provisioning of ecosystem goods. The fact that the moderate grazing action had the highest herbage production makes it particularly suitable for grazers such as sheep and cattle, yet browsers such as goats also feed on this type of forage (Yiakoulaki & Nastis, 1994). For goats, however, the heavy grazing and the gap reforestation actions, which had the highest shrub production, were more suitable than the fully reforested areas, which had no shrubs at all. A higher shrub cover in heavily grazed than in moderately grazed areas should be attributed to the fact that *Q. coccifera*, the dominant shrub in the area, is not grazed by sheep, not all of its annual production is acceptable to goats, since some parts of the twigs are hard, and accessibility to new twigs and leaves is often reduced by the overgrowth and high density of these shrubs in many areas (Papanastasis & Liacos, 1983). This means that heavy grazing, even by goats, could not control the expansion of *Q. coccifera* at the expense of the herbaceous species. Wood production, which was only relevant in the reforestation actions and the no grazing areas, could also be considered as a provisioning service. However, due to the small size and slow growth rate of trees, the wood produced could be used only as firewood.

For regulating and supporting services, the services assessed were soil and water conservation, and carbon sequestration. Several indicators were jointly considered for water and soil conservation, ranging from structural features of the vegetation cover to soil functional attributes, which consistently pointed to heavy grazing as the management action with the least potential for conserving resources and maintaining soil functioning in the area. Plant cover plays an important role in water regulation as well as in soil retention. The U.S.A. National Research Council (1994) considers 40% plant cover as a "threshold for rangeland health", while Papanastasis et al. (2003) and Gutierrez & Hernandez (1996) have set up the critical level for land degradation higher (45-70%). This means that in the moderate grazing, no grazing, and reforestation actions no relevant soil erosion should occur since plant cover exceeded 90%. In the heavy grazing action, however, where plant cover was less than 60%, thus letting almost 40% of the ground bare, soil erosion could be an active process, particularly because the soils of the study area are sandy loam, very susceptible to soil erosion. Besides plant cover, plant pattern is also important for soil and water conservation (Ludwig et al. 2000; Bautista et al. 2007). It refers to the spatial arrangement of run-on, i.e. patches, and run-off zones, i.e. inter-patches, which together form functional units (Tongway & Hindley, 2000; Urgeghe & Bautista, 2014). Larger patch cover and width, and therefore larger areas with resource sink capacity (Ludwig & Tongway, 2000), were found in moderate grazing, no grazing and full reforestation, in the latter case probably due to the contribution of the banks and troughs created along the contours. Together with heavy grazing, gap reforestation showed the least functional landscape in structural terms, probably due to the lack of a well-developed understory that contributed to ground (patch) cover and the absence of banks and troughs. However, this fact was counterbalanced by very high values in soil functioning. The soil functional indices, namely stability, infiltration and nutrient cycling, provide information on how resistant is the soil to erosion and how well water infiltration and nutrient cycling function (Ludwig & Tongway, 2000; Mayor & Bautista, 2012). These indices had highest values for the no grazing actions, suggesting that avoidance of grazing together with the reforestation of tree openings, significantly improved the soil surface condition and, consequently, the soil, water and nutrient conservation. In a critical review of the literature, Gilford & Hawkins (1978) found that even moderate grazing may cause soil erosion compared to ungrazed conditions. The fact that the soil functional indices did not greatly differ between moderate grazing and heavy grazing in this study points the high sensitivity of the soil surface to even moderate pressure of livestock grazing.

Carbon sequestration was addressed by two indicators: soil carbon and total aboveground plant biomass. Our results suggest that moderate grazing did not have a negative effect on soil carbon sequestration as compared with the no grazing actions. However, organic carbon was measured on the surface soil, where herbaceous plants, the dominant species in the moderate grazing action, mainly lay their roots. The no grazing actions may promote higher soil organic carbon content in deeper soil layers, due to a higher below-ground biomass associated to the higher aboveground biomass of these actions. As expected, a much higher amount of carbon was stored as above-ground biomass in the no grazing areas, particularly in the reforested ones. As a matter of fact, reforestation is considered to be the most effective way of absorbing CO_2 and increasing carbon sinks in terrestrial ecosystems (Huang et al., 2012).

Biodiversity, finally, was assessed as plant diversity, which was expressed through various indices. Moderate grazing and no grazing equally led to the highest plant diversity in the area. These results are in agreement with previous studies that showed maximum species richness and diversity at moderate levels of grazing and reduced values at higher intensities (e.g. Milchunas et al., 1988; Alados et al., 2004). A decrease in species richness with shrub invasion in grasslands was also found in a previous research in the same study area (Papadimitriou et al., 2004). The problem of diversity reduction is further aggravated if on top of previous degradation by heavy grazing, rangeland restoration is addressed by pine reforestation that imply heavy site preparation and removal of extant vegetation.

In this work, we did not address the cultural ecosystem services provided by the actions studied. However, livestock grazing plays an important role in vegetation dynamics, contributing to the creation of cultural landscapes, particularly traditional practices such as transhumance (Ispikoudis et al., 2004). Such landscapes are usually high nature value systems that can bring social cohesion in marginal areas, as they help limit depopulation rate, support off-farm economic activities such as tourism, and reduce fire risk (Caballero, 2007).

In sum, overall ecosystem services provision was particularly favored under moderate and no grazing management options, with moderate grazing outranking any other action in provisioning services, and the no grazing action presenting the most balanced provision of services. Pine reforestations largely contributed to water and soil conservation and C sequestration, which points out to the important role that reforestation actions can play in restoring ecosystem services (Bautista et al., 2010). However, when implemented at the expense of disturbing natural vegetation in the area, reforestation can have a negative impact on plant diversity, an important target in restoration of rangelands (Landsberg et al., 2004). The prioritization of the provision of specific services through management actions must therefore be based on the understanding of their trade-offs (Bautista & Lamb, 2013). Our integrated analysis of ecosystem services provision has been based on the assumption that all the four main types of services considered, as well as the various indicators used to measure them, are equally important. Obviously, the outcome of the integrated analysis could vary if the assessed services are considered to be of different importance by the target stakeholders and local communities (Derak & Cortina, 2014).

CONCLUSIONS

Banning of livestock grazing, with or without reforestation with pines, is not always an advantageous intervention for the restoration of degraded Mediterranean rangelands, particularly if reforestation is done after removing the extant vegetation. On the other hand, effective restoration can be achieved if grazing management is adjusted so that from heavy it becomes moderate, without depriving these areas from livestock use, which might create social unrest. Our results suggest that a relatively small suit of ground-based indicators can inform of a well-balanced basket of ecosystem services, and thus be effectively used for the assessment of management and restoration actions in degraded Mediterranean rangelands.

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Table 1. Above ground biomass (t/ha) per type of management action (mean values ± 1 SE).

			Trees	
Actions	Herbs	Shrubs	Pines	Oaks
Heavy grazing	0.48 ± 0.08	17.46 ± 5.12		
Moderate grazing	4.49 ± 0.22	4.09 ± 1.95		
No grazing	0.73 ± 0.07	5.04 ± 1.61		50.37 ± 4.99
Gap reforestation	0.14 ± 0.02	16.88 ± 3.88	51.65 ± 1.88	36.33 ± 3.12
Full reforestation	0.55 ± 0.13		71.43 ± 4.89	

Table 2. Species richness (S), Shannon-Wiener diversity index (H) and evenness index (J) per type of management action (mean values ± 1 SE); Results from One-way ANOVA for each variable (n=3). Different letters denote significant differences between actions (Tukey test)

Actions	Species richness (S)	Diversity index (H)	Evenness index (J)
Heavy grazing	11.67 ± 2.19^{C}	1.66 ± 0.13^{BC}	0.68 ± 0.01^{A}
Moderate grazing	26.67 ± 1.20^{AB}	2.29 ± 0.04^{A}	0.70 ± 0.01^{A}
No grazing	32.67 ± 1.45^{A}	2.42 ± 0.09^{A}	0.69 ± 0.02^{A}
Gap reforestation	20.33 ± 1.76^{B}	1.93 ± 0.11^{AB}	0.64 ± 0.02^{A}
Full reforestation	12.67 ± 0.67^{C}	1.08 ± 0.22^{C}	0.42 ± 0.08^{B}
ANOVA results: F(*P* values)	F = 34.2(<0.001)	F = 16.4(<0.001)	F = 9.9(0.002)

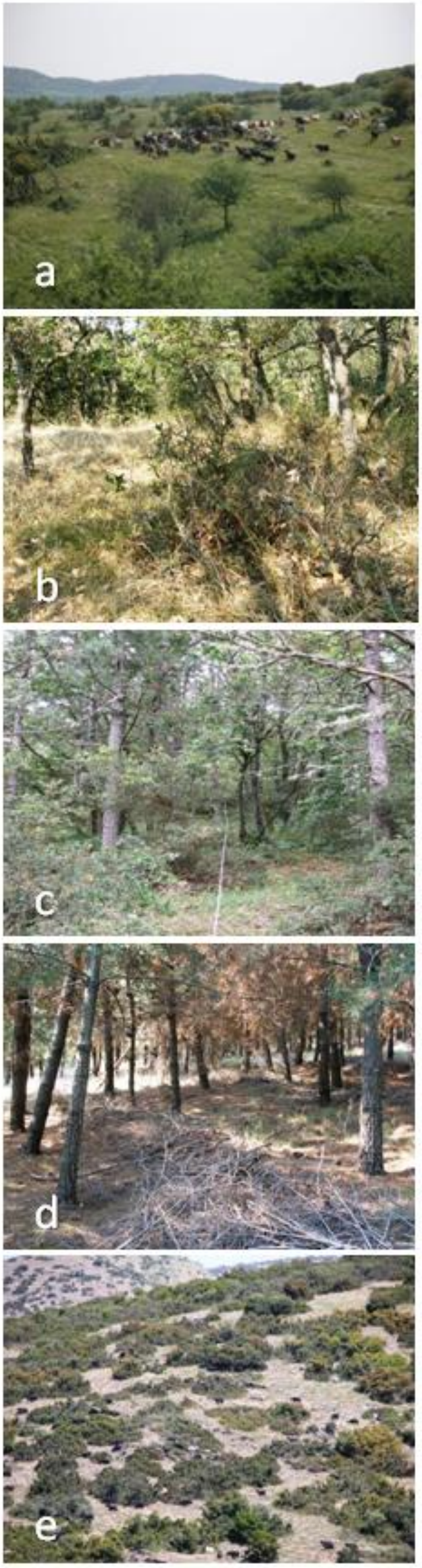

Figure 1. Management actions: a) Moderate grazing, b) Control (no grazing), c) Gap reforestation with *Pinus pinaster,* d) Full reforestation with *Pinus pinaster* and e) Heavy grazing.

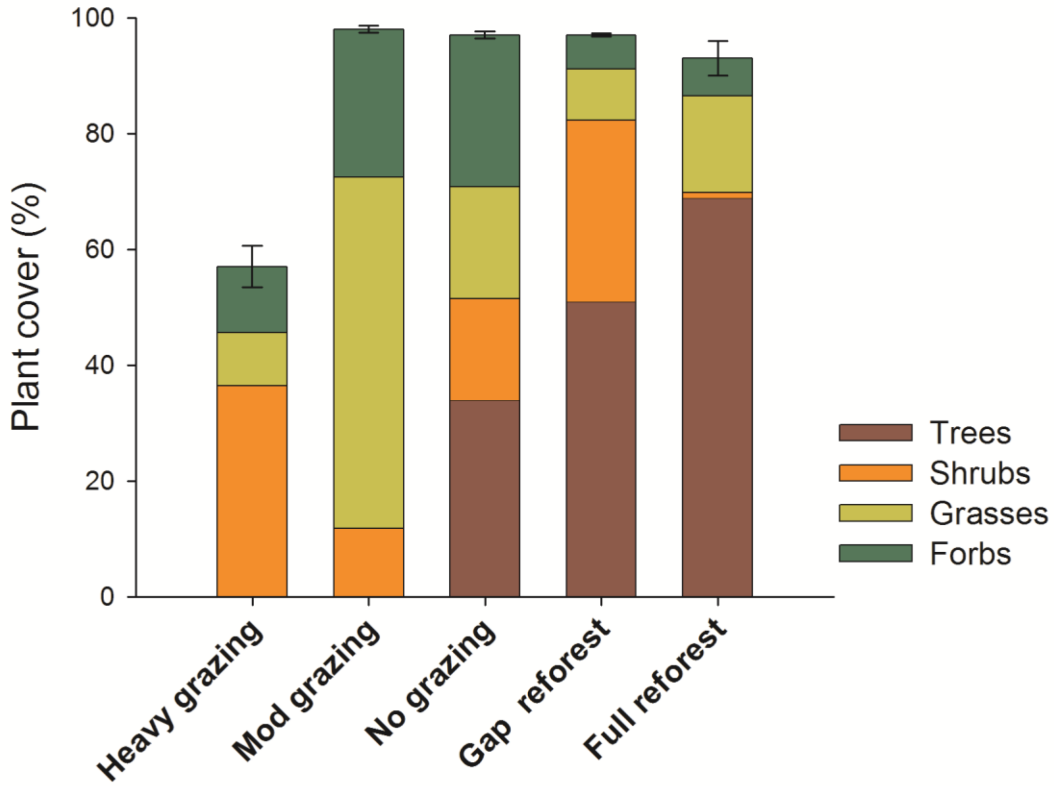

Figure 2. Total plant cover (mean values ± 1 SE) and proportion of trees, shrubs, grasses and forbs cover for each type of management action: heavy and moderate grazing, no grazing, and gap and full reforestations.

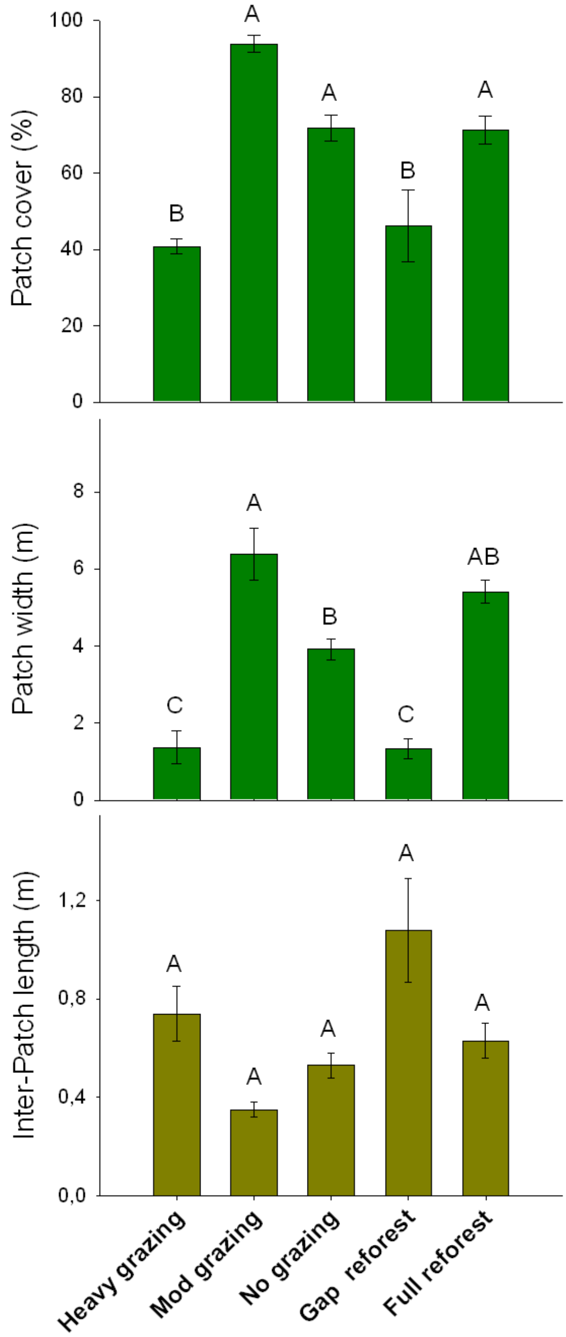

Figure 3. Landscape pattern metrics (mean values ± 1 SE) for each type of management action (labels as in Figure 2). Different letters denote significant differences between the actions (p<0.05; Tukey test).

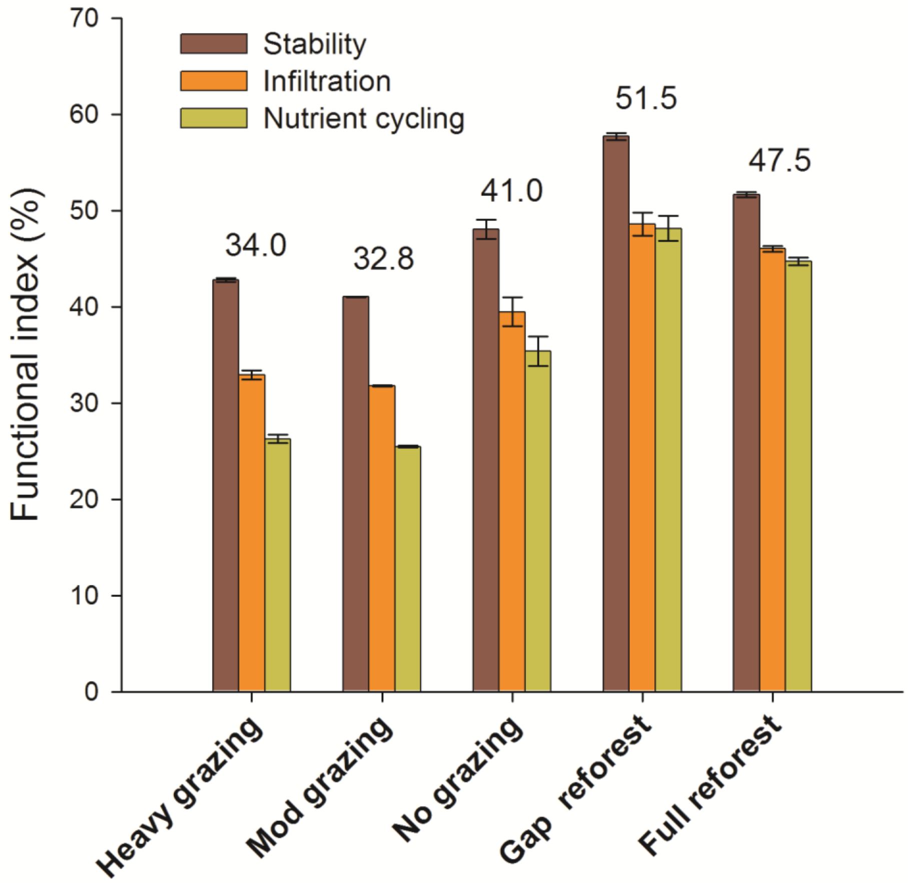

Figure 4. Stability, infiltration, and nutrient cycling indices (mean values ± 1 SE) for each type of management action (labels as in Figure 2). On top of the bars: Average functional index (mean values per action type of the three functional indices).

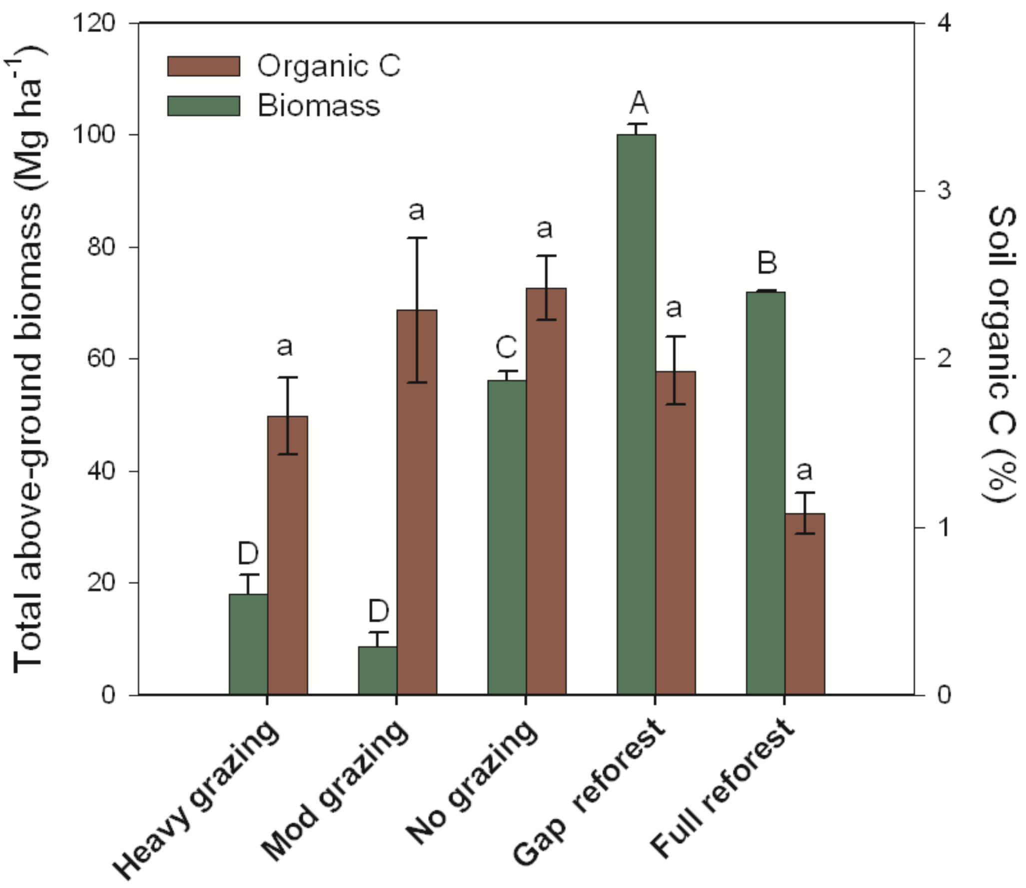

Figure 5. Total aboveground biomass and soil organic carbon (mean values ± 1 SE) for each type of management action (labels as in Figure 2). Different letters denote significant differences in aboveground biomass between the actions ($p<0.05$; Tukey test).

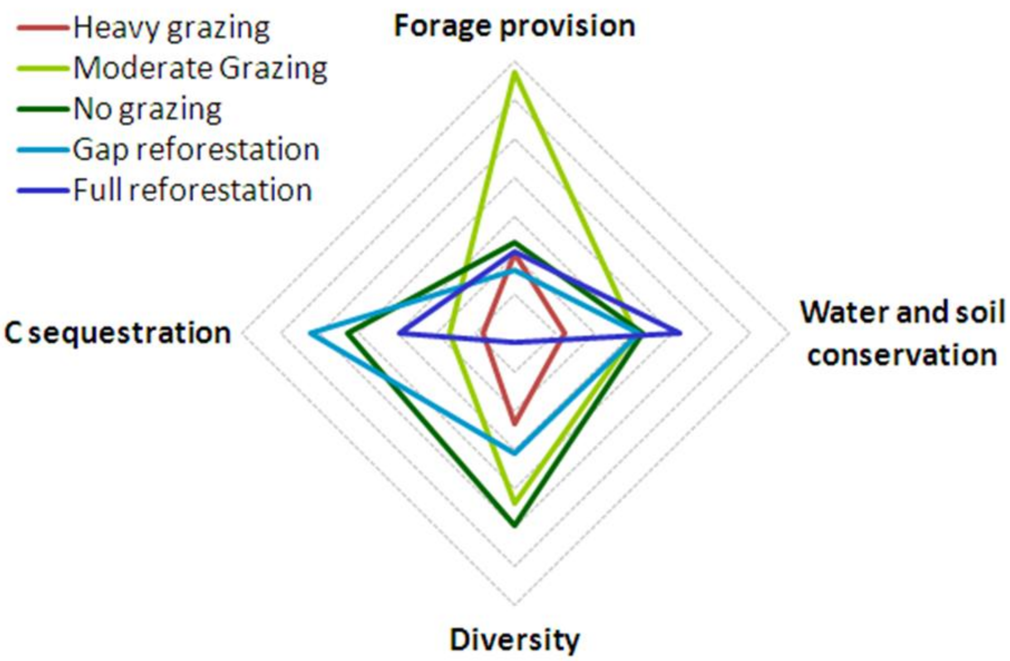

Figure 6. Relative contribution of each type of management action to the provision of ecosystem services (water and soil conservation, C sequestration, and forage) and plant diversity. Contribution data are based on the averages of standardized values for the various indicators considered.

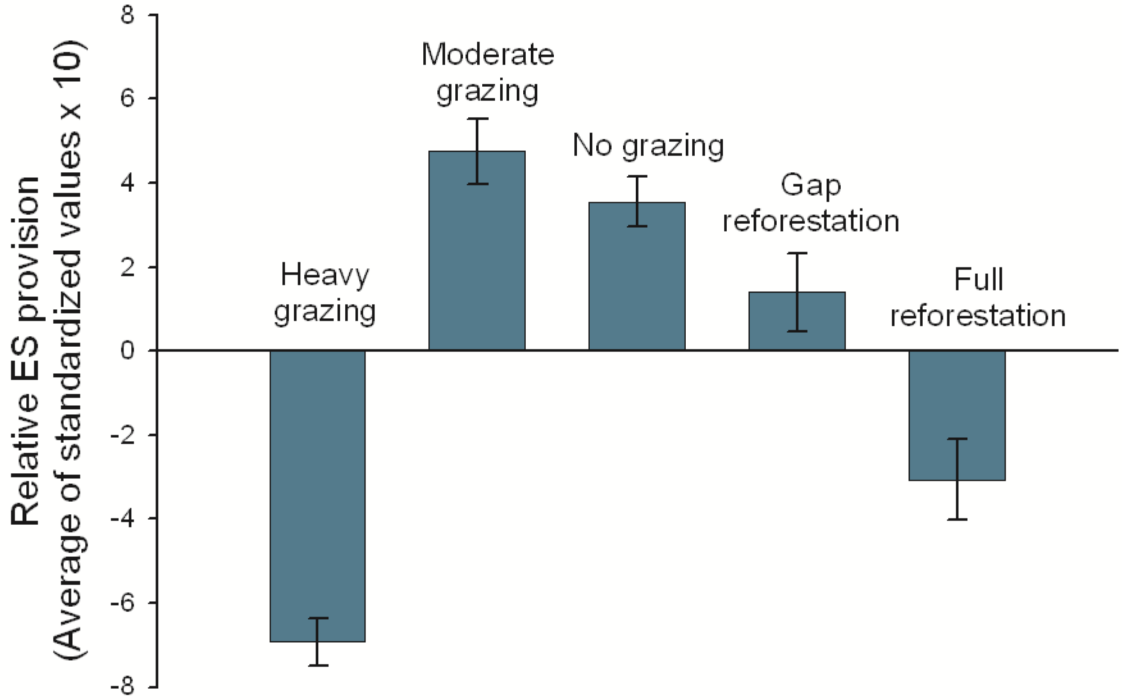

Figure 7. Relative overall provision of ecosystem services by each type of action. Bars represent averages (± 1 SE) of the standardized values for each major type of ecosystem services considered (water and soil conservation, C sequestration, forage, and plant diversity).