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Original Research Article

The role of demography and grazing in the patterns of endangerment of threatened plants

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ABSTRACT

This study explores the effect of two types of threat on a group of rare plants at risk of extinction in Spain, distributed across altitudinal and latitudinal gradients, under a scenario of rapid changes in land use. On the one hand, we focused on livestock herbivory as one source of risk for these plants, also taking into account factors such as altitude and the protection status of the area. On the other hand, we explored a threat which has been little documented so far, namely, rapid changes of the dynamics of plant communities.

Field data from 54 populations of 37 different threatened plant species were analyzed using contingency tables and linear models (ANCOVA and GLM) to study the effect of livestock and herbivory. In addition, we used an asymmetry index of population structure to look for deviations and asynchronies with respect to general vegetation dynamics.

Our results point to an overestimation of grazing as a threat to these plants, its effect having declined as a result of rapid changes in land use. Altitude was found to provide a dual conservation shield, preventing grazing for different reasons. Hence, we recommend that herbivory to threatened plants should be reported by accurately quantifying individual losses and damage. We found no relationship between population structure and plant conservation status. In addition, we detected no asynchronies between threatened plant population structure and community dynamics.

New indicators related to size and distance between populations as well as density distribution within patches should be explored.

1. Introduction

Assessing threats is of paramount importance in conservation practice. A set of tools exist in conservation science in order to ascertain as best as possible the true status of endangered species, see among others: (Subcommittee, IUCN Standards and Petitions Subcommitte, 2017). Furthermore, identifying and quantifying threats is vital to gain a clearer understanding of the threat process, that is, the path of a species from becoming rare to extinction. In particular, batch assessments, or large groups assessments for Red listing allows patterns of endangerment and broad changes in biodiversity to be identified (Lughadha et al., 2005; Brummitt et al., 2008; Mace et al., 2008; Butchart et al., 2010).

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Our main goal was to gauge the contribution of some major sources of threat for wild plants in the context of recent, fast moving land conversions in Spain. For that, we use grazing and demographic asymmetry as indicators of the effects of rapid anthropogenic changes on endangered plant populations. Rapid anthropogenic changes have a greater effect on biodiversity, but also complicate conservation status assessments. Moreover, there is a direct relationship between rapid anthropogenic change and habitat loss due to urban sprawl and associated land use changes (McKinney, 2002; Aronson et al., 2014; Jones et al., 2018).

Grazing is an anthropogenically-mediated process. Thus, Mediterranean ecosystems are associated to a wide use of livestock herding as an economic activity (Clergue et al., 2005), as well as part of traditional culture (Caballero et al., 2007, 2008). However, grazing of livestock and, to a lesser extent that of wild animal species, poses one of the most widespread threats reported for plants in the Mediterranean basin (Domínguez Lozano et al., 2013). Moreover, some authors point to specific examples of how problematic herbivory can be for endangered plants (Herrera, 1989; Blanca et al., 1998; Simon et al., 2001). In spite of this, evaluating the effects of grazing is not straightforward since both anthropogenic but also natural factors have contributed to the current situation of threat from grazing (Marañón et al., 2009; López-Sánchez et al., 2016).

Furthermore, rapid land use changes and biota community dynamics may underlie the negative demographic response of rare and threatened plants, which in turn makes them less able to cope with this rate of change in vegetation communities. Recent studies have shown a relationship between demographic traits of species and environmental changes (Adler et al., 2014), and some authors point to habitat changes affecting endangered plants and altering demographic structures (Kienberg and Becker, 2017).

Hence, in the case of threatened plants, demographic structure, i.e. plant size or stage class frequency distributions, may serve as an indicator of asynchrony between the demographic dynamic of rare plant populations and community dynamism. In other words, slow demographic response of some populations of threatened species may lay behind the overall rapid demographic response of major components of the plant community where they are in.

Taking these issues into account, two indicators of endangerment were measured in several endangered populations: herbivore predation and population structure. We sampled these variables for a range of different threatened plant species and habitats across a large area, collecting field data in populations.

A specific aim is to analyze whether livestock are grazing on threatened plant populations at country scale. First, we want to determine the effect of grazing over different environments and threatened plants. Among the many factors (habitat structure and function, community composition, or species dispersion capacities) shaping grazing and livestock management response, orography and conservation practices are major contributors in terms of modulating the effect of herbivores. Thus, to address a second question, we analyze how the distribution and altitude of protected natural areas shape the response of threatened plant species to livestock presence and grazing.

Finally, we investigate whether departures for expected demographic structures of the population may serve as a demographic signal that threatened populations are unable to thrive under present community dynamics. Firstly, we measure the present population structure of the selected populations and check for possible deviations from a well-structured, J-shaped population model. Additionally, under the assumption that endangered plants participate in the dynamics of the community of which they are part, we test whether some of them are not keeping track with the present community changes. Our aim was to explore the risk of disappearance resulting from inability to follow changes in their community dynamics, denoting by an asynchrony between population structure curves of the target plant and its community.

2. Methods

2.1. Study area

Situated in the western part of the Mediterranean Basin, Spain is located in a biogeographical and human crossroad. It hosts some major plants hotspot in the Mediterranean (Médail and Quézel, 1997), and also possess relatively well studied endangered flora and fauna (Pineda et al., 2002). In addition, the tourism industry has targeted Spain since the middle of last century, contributing not only to a growth in road infrastructure and urbanization but also indirectly to rapid land use change from agriculture to service sectors as well as to regional demographic changes (Del Molino, 2016).

2.2. Species selection

Based on present Spanish Red List (Moreno Sáiz and coord, 2008), we selected 37 threatened species (we attempted to locate populations of another two species unsuccessfully). Species were selected according to their threat status, biotype (life form) and rough large-scale biogeographical affinities (Online Resource 1, Table 1-S1). For these 37 plants, we located 54 populations across a geographical gradient 1000 km wide, and 3000 m high (coast to peak), considering both coastal and inland locations (see Fig. 1 and Online Resource 1, Table 2—S1).

2.3. Field data collection

We established 85 plots, the size and shape of which depended on species life form and habitat. According to this, highly aggregated species with small biotypes were sampled using small plots (usually less than 20 m^2 in area) and larger plants

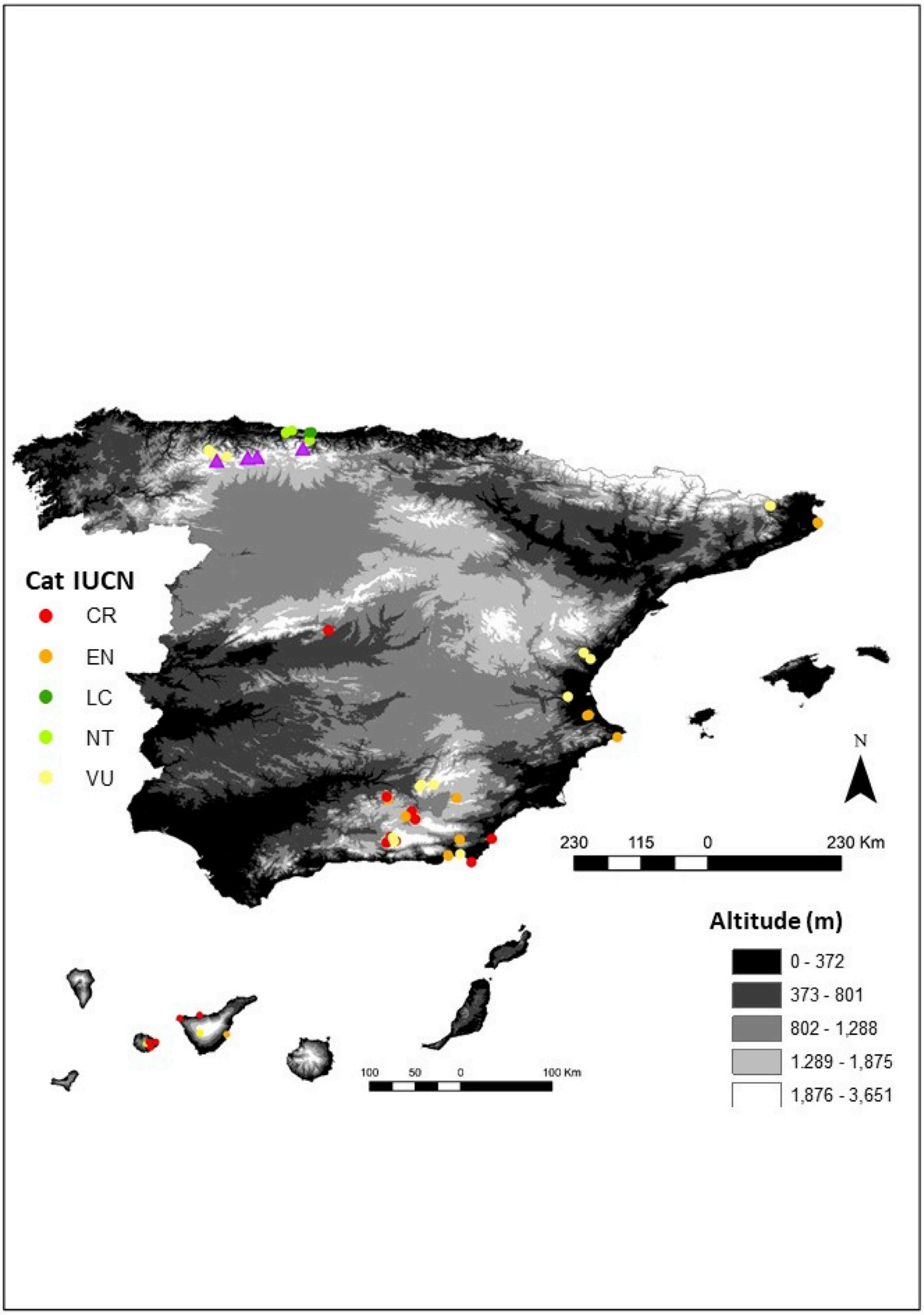

Fig. 1. Location of populations in mainland Spain and Canary Islands, indicating threat level according to UICN categories. CR: critically endangered, EN: endangered, LC: lower concern, NT: near threatened, VU: vulnerable. Note the four triangles indicating populations either extinct or not found.

with scattered individuals or uneven habitats were sampled with larger plots (such are the cases of *Antirrhinum charidemii*, *Bencomia exstipulata* or *Sideritis marmorea*). Size, census and other properties of each plot are shown in Table 3–S1 of Online Resource 1.

Secondly, for every plot, we recorded livestock presence. Depending on location, several animals may be involved, usually sheep but also cows, goats, and finally, in some instances, wild herbivores. Signs or tracks of herbivory were recorded at the time of the visit but no previous data were recorded about past or present grazing regimes. We also registered other variables such as plot trampling (human and non-human), slope, cover, aspect, soil and altitude.

Additionally, we also recorded signs of predation at species level in the plot based on observed plant damage. For each individual of the target species and plot, we registered signs of herbivory consisted mainly of leaf and twig browsing but also broken branches and trampling. The effect of invertebrate herbivory; terrestrial mollusks was also taken into account. 2994 plants were sampled for these purposes in all plots.

2.4. *Analyses*

Contingency tables (contrasted with Fisher's Exact Test for Count Data) were employed to study the effect of livestock presence on signs of predation at both plant and plot level.

Plant density in plots may affect the level of herbivory, as plots with more individuals could attract more predation. Hence, we use analysis of covariance (ANCOVA) to better explore the effect of herbivory, factoring out plant density in plots, using number of eaten plants as a response variable, and livestock presence as an independent variable for this analysis.

Two important factors may contribute to explaining the effects of grazing on populations: protected status of the land and altitude. The binomial variable "livestock presence" can be explained using either of these variables independently or a combination of both. Therefore, generalized linear models (GLM) were employed to model the effect of both protected area status and altitude (independent variables) in the distribution of livestock presence (dependent variable) in the plots. We used the family of binomial curves and the default logit link function.

2.5. *Deviated demographic structures*

The demographic structure of the population is useful to characterize certain demographic properties (Hara, 1988; Lesser and Brewer, 2012). For example, this analysis approach has been widely used in research concerning forest dynamics, where detailed studies have shown that specific size structures can be important indicators for conservation purposes (Uotila et al., 2002; Westphal et al., 2006). Plant size is usually the state variable for developing population structures as it is a rapid and easy variable to record. We measured plant size for every individual of the threatened species within the plots.

As plant size can be defined differently depending of life forms, trees are not sized in the same manner than annuals for example. In accordance with these differences, we used several methods to size plants using a size-related state variable specific to life forms or biotype (See Online Resource 1 for detailed explanations in each life form type).

We removed annual plants (6 species with 10 plots), and seedlings of the rest from this demographic analysis since our one visit sampling does not capture the true demographic dynamic of these stages. 7 additional plots were removed from the analysis, as there were insufficient individuals (less than 2) to construct a proper demographic structure. Thus, we employed the remaining 2891 individuals in 60 plots. For each, we produced histograms, or population structures, using size class distribution (Online Resource 2, Fig. S1). Histogram classes were selected as the default setting in R function hist, using the Sturges formula (Sturges, 1926).

We calculated the skewness index for each plant size distribution for comparing population structures across a whole range of populations and environments. Skewness can be calculated in several ways. We used the Fisher standardized skewness, corrected for small sample sizes (G1) (Cain et al., 2017), and available under the R package "moments" (Komsta and Novomestky, 2015):

$$G_1 = \frac{\sqrt{n(n-1)}}{n-2}\left[\frac{\frac{1}{n}\sum_{i=1}^{n}(x_i-\overline{x})^3}{\left[\frac{1}{n}\sum_{i=1}^{n}(x_i-\overline{x})^2\right]^{3/2}}\right]$$

Table 1
Definition of the five community types.

Early annuals: composed of fast growing early successional species, mainly annual graminoids or herbs. Soil type differs, from sand to gravel but commonly with low level of nutrients and water retention. Dwarf shrubs are occasionally common in some plots, especially at the base of large walls and cliffs.

Semi-desert perennials: open formations with dominant small size shrubs and tussock grasses. This category also includes sun facing, water stressed rock plant formations.

Secondary scrublands: intermediate successional Mediterranean type formations. Common, fire follower shrubs and fire prone trees (pines).

Late successional: Shade tolerant formations under dense tree canopy, shade and humid conditions, deciduous, evergreen or pine forests with understory herbs and forbs.

Mountain perennials: Cold tolerant, mountain communities, composed of slow growing perennial species at high elevations, usually above the tree line, with rocky soils and a short growing season.

where $\bar{x}$: is the mean of the state variable for each sample unit, and n is the number of individuals in the sample unit. In our case, the sample unit consists in the plots of each species, therefore 60 units.

Assuming inverted j-shaped population structures to be optimal, we used Chi square tests to identify differences in plot skewness values from the expected mean of all plots. In this way we were able to identify deviated population structures not following the j-shape model.

2.6. Community dynamic

Urbanization has been blamed for large scale, rapid changes in vegetation and habitat conversion (Underwood et al., 2009). This fast change of plant communities has been termed hyperdynamism, which involves the substitution of certain mature plant communities for simpler and less structured communities over a relatively short period of time (Laurance, 2002; Hahs et al., 2009). Hyperdynamism may poses a threat to endangered species not able to cope with fast vegetation movements.

To detect asychrony, measuring population structures for all species in the plant community of each plot was not feasible in our study, so we produced a simple approach to catalogue community dynamism for each sampled plot. Based on species composition and cover (online resource 1 and Table 4–S1), we assigned one of five simple assemblages or community types described in Table 1 to each plot. We assumed that these represent the dominant life forms (biotypes) and specific life cycle strategies in each plot, providing information in simple terms on community dynamism (i.e.: fast vs. slow dynamic).

These five types allow us to test the hypothesis that some communities contain threatened species with population structures not matching the general dynamism of the site where they are found. If detected, these deviations may have certain negative effects in the long term due to asynchronization of our sampled populations with the rest of the plant community. Differences may be associated with rapid, fast growing vegetation, or to stress tolerant, slow growing communities. Skewness index was the response variable used to determine the relationship among threatened plant population structures and community types. We used ANOVA models to check for demographic associations between population structure and community types.

3. Results

3.1. Herbivory signs

We recorded a very low rate of herbivory in our samples (14.1%, 12 out of 85 plots). In addition to this finding, we also detected no relationship between livestock presence and herbivory (F value: $p = 0.5399$, odds ratio $= 1.5096$). Herbivory effect is low regardless of whether livestock presence is detected or not in our plots (Table 2).

The results of the ANCOVA model showed no significant relationship between any variable and number of predated individuals. Neither grazing effect nor total plants in plots contribute to explaining the distribution of predated individuals. Moreover, herbivory in these species is not related to livestock grazing or abundance (plant density) in plots (F-statistic: 0.8184 on 3 and 81 DF, p-value: 0.4874).

As regards our next question concerning whether the protected area network contributes to modulating livestock response, no close relationship was detected between livestock presence and protected area status in our plots. We may conclude that conservation areas do not explain livestock presence around threatened populations (Table 3, Fisher's Exact Test for Count Data-: p-value $= 0.6385$).

Hence, livestock presence is not related with the level of protection of the land. In some protected areas endangered plant populations and livestock are found together in the same space while in other equally protected areas they are not.

As plots were sampled across an elevation gradient, altitude may contribute to explaining these results. Therefore, we tested whether the livestock response variable was related not only with area status but also with altitude. The GLM results shows two main relationships (Table 4). The strongest effect relates livestock presence and altitude in our sample $p = 0.00319$, revealing that there is an increase in livestock presence in the plots with altitude.

Secondly, absence of livestock and low protection status in the area are also significantly related ($p = 0.00499$).

The deviance values show that altitude is strongly associated with a reduction in residual deviance (altitude difference deviance from the null model: 11.617, d.f.: 82, $p = 0.0006$), and altitude mediates the presence of livestock grazing and area protection in the sample (Online Resource 2, Fig. 2-S2).

These results are reinforced by the effect of altitude on the protected area variable. Maximum area protection is significantly related to higher elevations for our sample of threatened plants (ANOVA, $F = 77.62$, $p < 0.0001$). However, according to

Table 2
Number of plots with and without observed herbivory for individuals and with presence or absence of livestock signs.

	No herbivory	Herbivory
No livestock	44	6
Livestock	29	6

Table 3
Livestock presence in plots according to protected area status. 0: no protected area, 1: National parks, 2: Natural park, and 3: other type of protection.

Protected area	No livestock presence	Livestock presence
0	13	8
1	20	14
2	4	6
3	13	7

Table 4
Results of GLM analysis. Distribution model: binomial, using logit as link function. In this case the variable related to area protection status is grouped in two factors (National Park, NP, and no or less area protection, nNP).

	Estimate	Std. error	z value	P
nNP	−1.0622500	0.3783218	−2.808	0.00499
NA	−1.7366241	0.7669303	−2.264	0.02355
Altitude	0.0015257	0.0005175	2.949	0.00319

the contingency table and GLM results there is no clear contribution of this relationship to the exclusion of livestock presence in plots from protected areas located at higher elevations.

3.2. *Demographic signal*

Skewness values were bounded between [-2, 6] in our results. There is a general trend towards positive skewness, indicating that inverted j shape population curves are dominant in our sample (Online Resource 2, Fig. 1-S2), showing an overall representation of young states in the populations structures. In addition, only two populations experience skewness significantly different from the mean expected values using Chi square tests, one positive, ViolaPuertodelasPalomas1A, and more importantly, another with a negative skewness: VerbascumLloma1D (Online Resource 1, Table 6-S1).

Linear regression models using several independent variables (community type and some demographic traits; life span and seed set) revealed no relationships with demographic structures (data not shown). Only when skewness values were transformed (absolute values) did community type become a significant descriptor. We found that the group threatened late successional shade tolerant populations was notably different from the rest of plots (ANOVA F-statistic: 2.706 on 4 and 55 DF, p-value: 0.03949, Online Resource 2, Fig. 3-S2).

4. Discussion

How do two broad patterns of risk, grazing and demographic asymmetries, shape the response of threatened plants? In the current Spanish scenario, our results do not support the idea that grazing is a risk factor for endangered plants. Most importantly, we found that direct predation was rare in our sample (14.1%). Furthermore, in spite of the fact that we detected a relatively high presence of livestock (mainly herds of sheep but also cows in northern mountain areas) human-mediated grazing was not significantly related to herbivory in our selected populations. This finding does not appear to be affected by resource availability to grazers, since the quantity of individuals grazed was not related to plant density.

Secondly, we discovered that altitude contributes to explain the effect of grazing in sampled plots, acting as a conservation shield for several reasons. We identified the absence of two counteracting conservation factors in plots at low elevations; on the one hand threatened plants in lowland areas receive little exposure to livestock activity, which is good, but at the same time, coastal plants tend to be outside protected areas, which is bad. Our interpretation of the situation is that low altitude excludes both livestock and area protection under the current land use scenario in Mediterranean Iberia. Sheep herding and livestock farming have all but disappeared in these coastal areas, where tourism is at its maximum. The demanding and fast growing urbanization process which has taken place over recent decades in Spain: "*la burbuja inmobiliaria*" (the real estate bubble) has displaced both activities from low elevation areas in favor of rapid economic revenues derived from land urbanization (Obiol Menero and Pitarch Garrido, 2011; Requejo Liberal, 2014). In addition, there are scarce large protected areas (i. e. National and Regional Parks) in lowland areas in Iberia. The establishment of large conservation areas in zones of rapid economic growth seems unlikely, and there are several problems associated with protected areas and urbanization (McDonald et al., 2008; Aronson et al., 2014). In Spain, the most successful approach to protection of areas under rapid urbanization has been the micro-reserve system. This system comprises a network of very small reserves of land (less than 0,2 km^2) which are mainly privately owned and managed through periodical conservation agreements with public conservation agencies (Laguna Lumbreras et al., 2004).

The outcomings of some conservation measures to protect threatened plant species may also contribute to explain our findings about why at higher elevations, in the mountain regions of the Iberian Peninsula, strict area conservation is not necessarily at odds with the presence of domestic herbivores (Fig. 2). In this sense, intensive management strategies such as

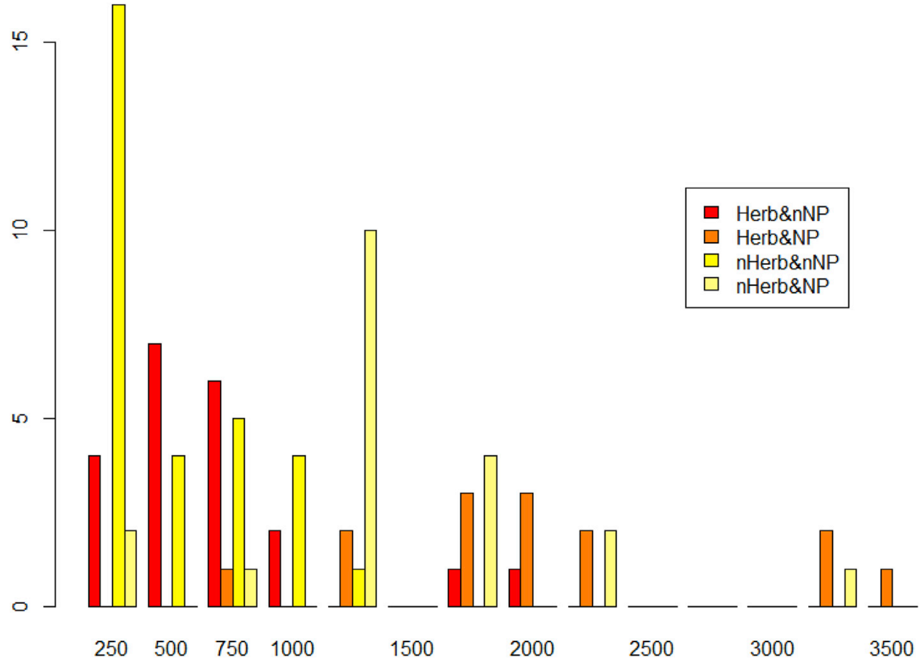

Fig. 2. Histogram for the frequency of plots according to altitude in 250 m intervals. Bar colors differentiate classes according to decreasing level of herbivory and increasing area protection, from red to light yellow. Herb: presence of livestock, NP: presence of National Park. nHerb & nNP: no presence of livestock and no National Park. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

livestock exclosure and agreements with sheep owners and shepherds help to guarantee the current coexistence of threatened plants and livestock. This is evident in certain conservation areas in the southeast of Spain with the highest levels of land protection, such as Cazorla natural park and Sierra Nevada national park. These two parks are well-known for their richness in endangered plant species, while at the same time, high levels of herbivory pressure exist (Herrera, 1989; Blanca et al., 1998; Luque Moreno and Sánchez Morales, 2001). The mentioned conservation measures, where enforced, may partly explain the fact that scarce signs of herbivory were recorded in our plots in these two areas.

In contrast to the herbivory signal, the demographic signal shows no particular relationship with the community dynamism threat. The majority of populations display a generally left skewed population structure, most plots reflecting the larger proportion of earlier demographic states. This finding is independent of community type (ANOVA results). The only exceptions are those plants developing in late successional, shade tolerant communities where a more balanced population, less skewed structure seems to occur more often. Plants in this group include ferns, large forest shrubs and trees (see supporting information Table 4–S1), so it might be expected that the population structures will show a lower proportion of young classes derived from a natural forest dynamic promoting persistence rather than colonization. However, in this regard we should mention the case of *Silene diclinis*, a forb present in two sampled plots in this group. The rest of the studied populations of this plant are catalogued as “early successional”, as the species is more frequent in forest clearings (Table 1S). Low skewness values (SilenePlà2A in Online Resource 2, Fig. 1) may denote lack of recruitment rather than synchrony with the shade tolerant community dynamism, hence, this may indicate risk of local extinction. The presence of this particular population (Plà de Mora) in a vegetation community formed by a pine forest could be related to the recovery of woodlands and forests in the area as a consequence of the decrease in human activities in rural areas and the loss of traditional practices, shepherding included (Montesinos et al., 2006). In addition, Chi square tests revealed a single but interesting case of asynchrony between the demography of the target species and the rest of the plant community for VerbascumLloma1D (*Verbascum fontqueri*). *Verbascum* is a short lived perennial, mostly biannual (Benedí, 2009) appearing in a secondary shrubland community (see Online Resource 2, Fig. 4-S2). It was growing in a plot which had been burned by wildfire almost 4 and half years prior to the time we recorded its presence. In this case, skewness values denoted an old population structure with risk of disappearance due to community competition.

In accordance with general demographic patterns (Bernucci Virillo et al., 2011; Castillo-Lara et al., 2017), our study found no relation of inverse J-shape population structures to conservation problems. Moreover, endangered flora is often more affected by extrinsic threats from human activities at regional scales without the need to alter demographic rates (Zeigler et al., 2013), and our results seem to support this.

Finding links between demographic response and environmental change is by no means an easy task. Recent studies have shown that the demographic response to variability differs and that not only negative trends (low skewness values for demographic structures in the case of our study) are to be expected in the event of rapid habitat perturbation (Boyce et al., 2006;

Lawson et al., 2015; McDonald et al., 2017). These diverse demographic responses, may include tracking (Jongejans et al., 2010) or buffering the impact (Csergo et al., 2017). Moreover, existing records of this variation in demographic rates suggest that it does not necessarily mean lower population viability (Compagnoni et al., 2016).

4.1. *Global conservation strategies*

As regards the interpretation of herbivory, several detailed studies focusing on particular species have pointed to negative effects (Herrera, 1989; Blanca et al., 1998; Simon et al., 2001), although results obtained by others are more balanced (Pfab and Witkowski, 1999; Domínguez Lozano et al., 2011). There are also authors who have reported positive interactions (Kaye et al., 2001; Correia and Freitas, 2002), as well as absence of any interaction with herbivory (García et al., 2002). When herbivory is unequivocally detected, for example in areas with a high animal stocking rate, conservation measures should be considered urgently because grazing above some thresholds undoubtedly poses a threat for plants in Mediterranean ecosystems. Similar warnings should be heeded in the case of the demographic results. Inverted j-shaped population structures are common in plants provided that no demographic problems arise. However, cases exist in Spain (two in this study), where community displacement of an endangered plant is interpreted as a response to extreme climatic events (Cursach et al., 2018). Elsewhere, certain rare plants are associated with clearings or high perturbation regimes, where altering these dynamic conditions leads to negative demographic consequences (see for example (Menges, 1990; Jones et al., 2017). Therefore, demographic threats in the form of deviated populations structures are a possibility for several populations, so corresponding conservation measures should be applied in these cases too.

Clearly, the rapid change scenarios identified should oblige us to reconsider the role of analyses of herbivory and demography in the conservation assessments of endangered plant populations. In the case of herbivory, monitoring is clearly important, although we should take into account the need to record the precise effect of the threat on individuals. In other words, herbivory reporting should include quantification of individual losses and damage, not simply describing the effect of grazing on surrounding vegetation. Secondly, we need to be aware of the fact that some threats which are strongly related to economy, rather than biology, may change more rapidly than the biological response to them. In contrast to the herbivory signal, widely used demographic data, censuses, population structures and demographic parameters do not usually provide an indication of plant conservation status. Thus, some other demographic indicator should be used to make sure that those responsible for conservation programs are aware of rapid land use changes. In this regard, from our perspective, the most relevant are those which link individuals with spatial analysis and biogeographical settings: size and distance of populations along with density distribution within patches. Finally, we recommend that threat analysis for red listing and environmental impact assessment should be based on up-to-date detailed field prospecting. Recommended reevaluations can be implemented on a representative sample of the complete set of endangered species using minimum sample size techniques (Domínguez Lozano et al., 2012).

4.2. *Some limitations of large scale sampling for threatened species*

Measuring broad patterns of endangerment for plants suffers from at least two drawbacks, plants response is slow and threat factor variation is usually large (both in space and time). Actually, grazing varies at all level in our study, over time, over spatial scales and over species. Similar variation can be expected for demographic structures. Present databases of endangered species (UICN, GIBIF or Spanish national databases) are useful for setting priorities for most endangered plants but, from our point of view, fail to register the most updated status of populations and species. Thus, field sampling is needed to record how rapid changes in risk factors may contribute to the decline of rare populations. But how could we be sure that one single visit over a reduced sample of threatened plants can be informative?

To answer this question it is important to notice that the aim of this study does not pursue the accurate measure of the amount of variation of any threat factor on a species basis, but rather to detect extreme values i.e., those that results in a risk of extinction as a general trend. In other words, when recording grazing for our study, we relay on the fact that grazing could be recurrent, fully predictive or on the other side, less common, difficult to detect or even stochastic. We believe that the former can be detected by one single visit, using plots widely spread over the whole studied territory.

On the other hand, we are fully aware that our sample only represents a small fraction of the total endangered flora in Spain. Only increasing the number of species sampled will show the consistency of our results in the future. Nevertheless, we are reasonably confident that our plots represent fairly well the overall situation of the species considered (with an average ratio of plots per UTMs of 0.897297+-1.150907, see table S1-2).

5. Conclusions

Some general conclusions can be drawn from our work. Firstly, signals of endangerment may render different results. In our case, herbivory predation point to an overestimation of grazing as a threat factor. The impact of grazing has been reduced in areas where rapid changes in land use have taken place thanks to an ‘altitude shield’; altitude as a factor of passive protection in lowland areas.

In contrast, our study shows that deviated demographic structures seem not to be a valid indicator of threat. In addition, the demographic structures of endangered populations are in most occasions following general community dynamism.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.gecco.2020.e01151.

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