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The endemic vascular flora of Sardinia: analyses, distribution patterns, ecological processes and implications for conservation

Botanica Ambientale e Applicata (BIO/03)

(Environmental and Applied Botany)

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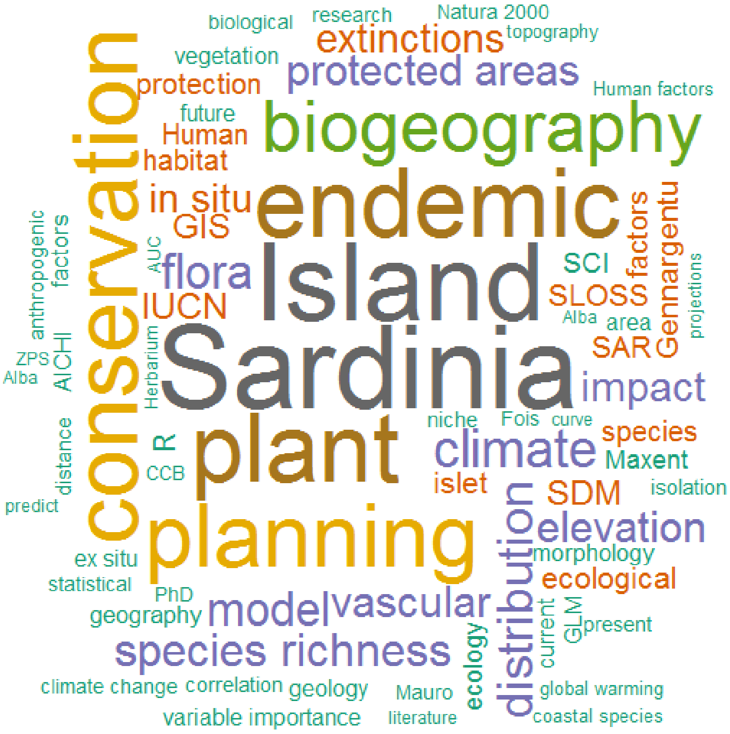

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GENERAL INTRODUCTION

1.1 The study of species distribution and its applications

An integral part of every scientific description of a species is the information about, at least, the geographic provenance of the available type specimen material. The concept of species geographic distribution seems quite straightforward; otherwise, a long chain of events could explain the existence of a data record documenting the species' presence or absence at that site and details can be much more complicated (Peterson et al. 2011). First, when dealing with such kind of data, the possibility of sampling errors and bias should be always taken into account (Rocchini et al. 2011). Besides the mistakes related to the determination of specimens or the registration of coordinate systems, also the data precision and completeness may prejudice the quality of information and subsequent results of distribution analyses. For instance, in many cases collectors accumulate specimens from a broad radius around a field camp, but often use a single locality descriptor for that collecting site. Often, such sampling biases correspond to accessibility that could also cause a bias in environmental space. Ideally, these problems could be avoided by a complete and exhaustive investigation of the study area. Otherwise, this is only feasible in case of a reduced set of species and/or territory; in the rest of cases, which are the majority, uncertainty and precision of data should be considered when analysing results and/or enhanced by a carefully data filtering and extrapolation (Varela et al. 2014).

This said, further elucidations on the definition of species geographic distribution could be provided. Instinctively, ones usually assumes that the information about the species geographic distribution is simply related to the current presence of species. Otherwise, the information about the species geographic distribution could be implemented by many other useful information such as the source (it could be easier revised), date (it could be, for instance, historically present but not currently) and absence (it could provide information about extinctions, unsuitability or dispersal limitations).

Although a certain degree of stochasticity is intrinsic to many natural processes, there are a great deal of examples that found statistical significant relationships among species distribution and many factors. Such information could be thus applied to explain many issues in biological ecology, evolution and conservation. First, the study of patterns of species distribution could allow the regionalization of the global territory at different scales. The so-called biogeography, have a long history that could be dated back to the 19th century when scientists, such as Alfred Russel Wallace (1823–1913) and Charles Darwin (1809–1882), started to systematically explain procedures that causes different biological compositions around the world. According to Bowman (1994), biogeography is the key to read all biological codes contained in the planet. For instance, a knowledge of biogeography was critical in developing the theory of evolution and the Alfred Wegener's theory of continental drift. More locally, it is still important for specific studies on ecology and *in situ* conservation planning (e.g. Saiz et al. 2013; Marignani et al. 2014). In addition, according to growing evidence that global changes and habitat fragmentation are producing unprecedented historical changes in species distributions, the study of regionalization patterns has also great potential for monitoring decreases in beta-diversity and homogenization of biotas (Dapporto et al. 2016).

In this context, plants, and in particular endemic plants, are crucial in making a comprehensive judgment of the environment. Indeed, plants represent a key approach for biodiversity conservation and have been increasingly used as crucial units for inventory, planning and monitoring as they are good indicators of overall biodiversity and they are able to provide information about underlying abiotic components (Damschen et al. 2012; Del Vecchio et al. 2016). In particular, endemic plant species have a crucial role on conservation studies since they are frequently threatened and usually better studied than the complete floras and because endemism-rich areas are also likely to be of conservation interest for other biological groups (Laffan and Crisp 2003; Cañadas et al. 2014).

The determination of biogeographic regions is particularly interesting when the influence of drivers of different nature is also investigated. From a conservational point of view, studies on factors related to areas with an exceptional concentration of endemic and threatened species are particularly relevant.

Firstly, it is important to consider that there are documented examples of the lack of congruence in the spatial pattern between total species richness and richness of endemic species (Orme et al. 2005; Lamoreux et al. 2006; Kier et al. 2009). Main differences have been underlined at both global (e.g. Cabral et al. 2014; Weigelt et al. 2015) and regional scales (e.g. Nogué et al. 2012; Trigas et al. 2013; Iliadou et al. 2014) with respect to altitudinal gradients and islands. Behind these aspects, there is the isolation, a key factor for speciation and colonization rates, which are two of main determining forces that drive a final biodiversity composition (Lomolino 2001; Thompson 2005; Weigelt et al. 2015). For instance, it is generally assumed that total species richness have a decrease close to the highest elevations (Vetaas and Grytnes 2002; Grau et al. 2012; Trigas et al. 2013). This theory also predicts that wide-ranging taxa are more likely to show a lower elevation peak than narrow-ranging taxa, as the latter are assumed to be more influenced by speciation processes (Colwell and Lees, 2000; Gentili et al. 2015a). Accordingly, many researches (e.g. Casazza et al. 2005; Trigas et al. 2013; Cañadas et al. 2014; Steinbauer et al. 2016) underscored a general increase in percentages of endemic species with altitudes, possibly also due to a lower interspecific competition and an increasing habitat diversity (Médail and Verlaque 1997; Casazza et al. 2005; Panitsa et al. 2010). Conservationists underlined that this pattern could be also influenced by a general lower human disturbance and increasing naturalness at higher altitudes (e.g. Lobo et al. 2001; Steinbauer et al. 2013; Trigas et al. 2013).

Similarly to elevation, and sometimes synergistically, also the isolation by sea is another important driving factor of species distribution. The principal effect of isolation on species richness results from an inverse relationship with the probability of dispersal to an island, influencing the chance of colonization (MacArthur and Wilson 1967). Otherwise, it is also assumed the importance of speciation processes in influencing the biodiversity composition of islands, especially in larger and more isolated islands (MacArthur and Wilson 1967). As a consequence, although islands are generally poorer in species than comparable mainland areas, their biodiversity often exhibits unique features and a high degree of endemism (Whittaker and Fernández-Palacios 2007; Kier et al. 2009). Indeed, althrough islands make up only some 5% of the global land area, their endemic biota are estimated to include about 20% of the world's vascular plant species and 15% of all mammal, bird and amphibian species (Mittermeier et al. 2005). Accordingly, hotspots of biodiversity (determined by the percentage of endemic species) include a high percentage of islands (Mittermeier et al. 2005) and the relatively new concept of ‘nano hotspots’ (i.e. biodiversity hotspots at finer scale) were identified on summits, in particular of islands (Cañadas et al. 2014). As well as already discussed for elevation, humans are also considerably influencing the biodiversity patterns in islands. Whilst large and medium islands, as well as coastal areas, generally show a higher human density than inland places, it is also true that many small islets are still uninhabited, with an increasing degree of naturalness directly proportional to distances from continents and/or larger islands (Fernandes and Pinho 2015). This relatively new trend is enforcing the interest of small islets for conservation purposes (e.g. Panitsa et al. 2006; Caujape-Castells et al. 2010). Accordingly, an international program for the promotion, and assistance in the management, of the small islands of the Mediterranean Sea is currently progress, within the framework of the Mediterranean Small Islands Initiative PIM (http://www.initiative-pim.org/en).

All these controversies that has arisen from the ambition to find one single factor that explains the enigmatic gradient in species richness and distribution patterns, demonstrate that different hypotheses are not mutually exclusive and core drivers likely act synergistically (Kreft and Jetz 2007). This confirms the importance of analyses at regional and local scales, which could point out furhter and peculiar conditions that determine the biodiversity of a particular territory.

In order to enhance the knowledge on potential distribution of species at different time points, Species Distribution Models (SDMs) are proposed with increasing frequency throughout ecology and conservation biology. Due to the high correlation between plants and environment, also such recent approaches are in many cases based on the distribution of plants. These methods extrapolate distribution data of species in space and time on the basis of statistical models in order to provide spatial information on the species and other associated elements of biodiversity (Franklin 2009). In particular, the greatest novelty is the possibility of enlarging the information on realized occurrence/absences of species to the potential distribution according to current, past and future conditions determined, for instance, by climate and/or land use change reconstructions/projections. These kind of findings enable conservationists to enhance methods of, for instance, discovering new populations, finding possible causes of extinctions, conservation status assessments, and protected area selection. Because of the free availability of vast amounts of both biological and environmental data at global scale, there is a large literature on the application of SDMs at large scales for macroecological and global change studies (e.g. Sommer et al. 2010; Foden et al. 2013; Nogués-Bravo et al. 2014). Otherwise, studies targeting detailed ecological understanding or conservation planning are enhancing the application of SDM at more local extents by refining the quality of data and implementing methods for small samples (e.g. Pearson et al. 2007; Varela et al. 2014; Proosdij et al. 2016). As far as Sardinia is concerned, most of local researches with the application of SDMs were published on animals. This is in contrast with general trends, where terrestrial vascular plant analyses were prevalent in early years and are still the most common (Elith and Leathwick, 2009). To our knowledge, SDMs were applied in Sardinia to measure the effects of anthropogenic habitat alterations on the distribution of the lizards *Podarcis tiliguerta* and *P. sicula* (Mangiacotti et al. 2015), to provide insights on the habitat suitability of the endemic lizard *Archaeolacerta bedriagae* (Bombi et al.

2009), mouflon (*Ovis aries*) and Corsican deer (*Cervus elaphus corsicanus*) (Puddu et al. 2009; Puddu and Maiorano 2016), and to support the phylogenetic reconstructions of Sardinian Rhinophids (Russo et al. 2014) and the Tyrrhenian tree frog *Hyla sarda* (Bisconti et al. 2011). As far as we know, the only local applications of SDMs for Sardinian plants were very recent and focused on climate change effects for 18 taxa with range restricted to Sardinia (Casazza et al. 2014) and on the Sardinian peripheral populations of yellow gentian (*Gentiana lutea* L.) in order to guide field surveys aimed to discover new populations and to model the vulnerability of this taxon to climate warming (see Fois et al. 2015, 2016, abstracts reported in Annex I and Annex II, respectively). Additionally, further insights on conservation planning in Sardinia will be supported by results obtained by an experimental application of SDMs to evaluate and map at high resolution the field acquisition costs of the entire Sardinian territory (unpublished data; see Annex III for further details).

1.2 The study area: Sardinia

Sardinia Island is located in the middle of the Central-Western Mediterranean Basin at a distance of about 12 km from the Island of Corsica (France), 178 km from Cap Serrat (Tunisia), 275 km from Provence (France) and 340 km from Menorca (Balearic Islands, Spain) (Fig.1). With a surface area of 23,833 km^2 inland, a total of 24,089 km^2 including the minor islands, Sardinia is the is the second largest island in the Mediterranean after Sicily.

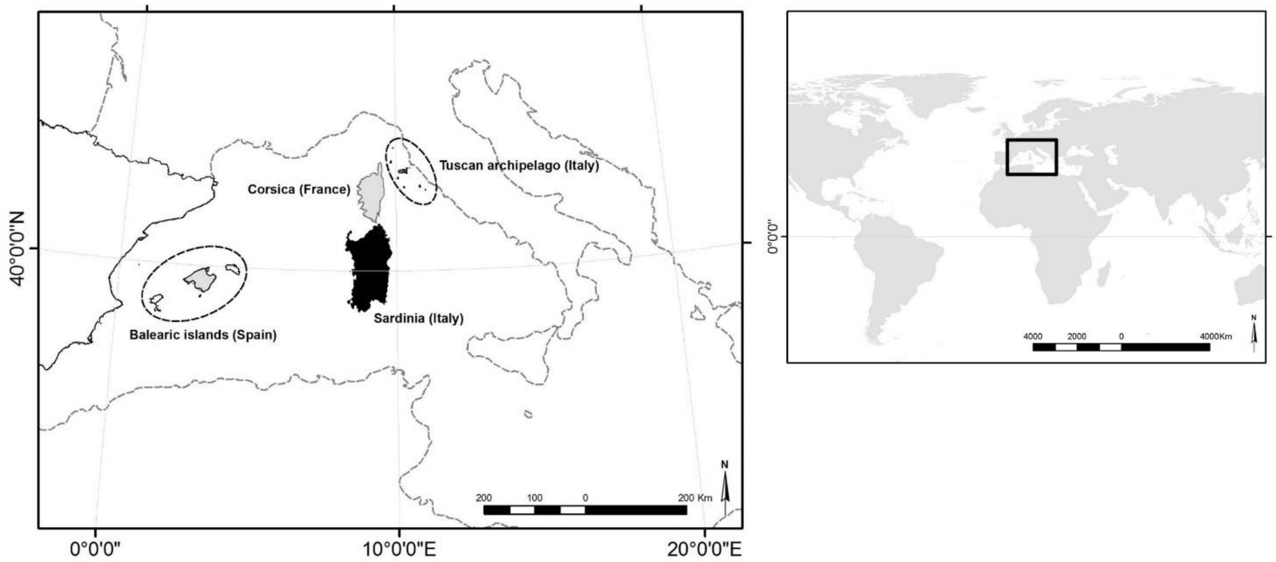

Figure 1. The study area of Sardinia and the closest islands/archipelagos in the Tyrrhenian sea.

According to tectonic reconstructions, the Corsica-Sardinia-Calabria and the Balearic-Kabylies microplates rotated south-eastwards from Iberia and southern Europe at ca. 30 to 28 Ma (Alvarez et al. 1974; Fig. 2). Sardinia and Corsica reached their current position at about 16 Ma (Speranza et al. 2002) while Calabria split off from Sardinia approximately 5 Ma until it collided with the southern part of Apulia (approximately the current Italian peninsula). These movements were accompanied by a Cenozoic to Quaternary marine introgressions and effusive volcanic events which determined sedimentary covers consisting of shallow-water marine carbonates, siliciclastic sediments, continental conglomerates, as well as volcanic rocks represented by a calc-alkaline suite and alkaline basalts.

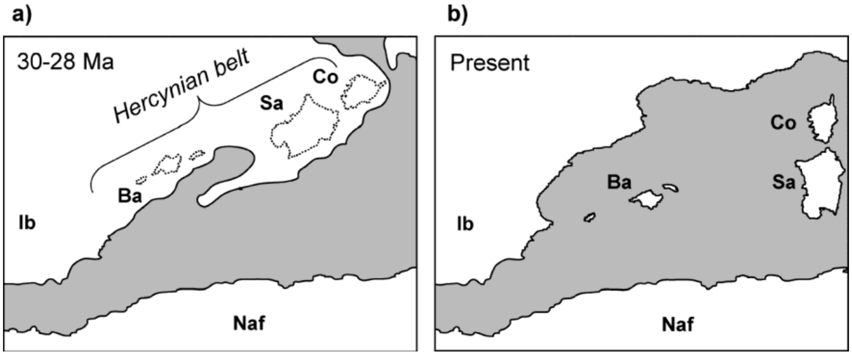

Figure 2. (a) Early Oligocene geography of the western Mediterranean Basin. Corsica, Sardinia, and the Balearic Islands were part of a continuous geological entity—the Hercynian belt—located in the Iberian microplate and (b) Present-day geography of the western Mediterranean Basin (Figure from Mansion et al. 2008).

As a consequence of its geological history, Sardinia represents a puzzle of very different tectonic, stratigraphic and paleontological features that have fascinated geologists (Carmignani et al. 2016). Indeed, it is roughly equally represented by Paleozoic igneous rocks (in common with Provence), Mesosozoic limestones and dolomitic limestones and Cenozoic sedimentary and volcano-sedimentary rocks.

The geological/lithological variability reflects a variability in morphological aspects. Differently from the Italian peninsula and Corsica, Sardinia was only marginally involved in the earlier Cenozoic Alpine orogeny and it is characterised by lower mountains (Punta la Marmora 1834 m; Monte Corrasi 1463 m; Punta Balistreri 1362 m and Punta Perda de Sa Mesa 1236 m), in comparison with the higher peaks of the other Mediterranean islands of Sicily (Etna 3323 m), Corsica (M. Cinto 2706 m), Crete (M. Psiloritis 2456 m) and Cyprus (M. Olimpo 1951 m). Despite of it, the Tertiary tectonic movements of the late Alpine orogeny and the subsequent sea level variation linked to the different phases of the Quaternary (consisting in a regression of up to 120 m below the present sea level) has contributed to the erosion and weathering of all the mountain flanks, with a direct impact on their steepness. Interesting formations are the deep gorges and closed valleys of northern and centre-eastern Sardinia, which resulted from the Tertiary raising of the Mesozoic carbonate covers and successive regressive erosions. In the southern and northern granitic coasts, where the evolution during the regression had not been balanced by the following accumulation of stone and soil, the landscape is still characterized by coasts *a Rias* and Archipelagos, such as La Maddalena (North) and dei Fiori (South-West) (Fig.3).

According to its geographic position, the current climate is typically Mediterranean, with dry and hot summers and relatively rainy and mild winters. Mean annual rainfall ranges from 411 to more than 1215 mm in the inner mountainous regions. Measured mean annual temperature ranges from 11.6°C to 18.0°C (Bacchetta et al. 2009; Canu et al. 2015). Sardinia benefits from a series of microclimates too. Those are related to the different influence of the coast, the degree of exposure to the sun and the protection from the dominant winds of the inner areas (Pungetti et al. 2008). The central mountain area is in itself a source of climate variation because in winter it can be covered by snow for some days, while in the summer the breeze that descends along the valleys mitigates the heat of the coasts. The wind regime is shaped by orographic features, which gather the air coming from different directions. Mistral is the dominant wind and concerns mainly the western coast, more frequently affected by long storms (Pungetti et al. 2008; Bacchetta et al. 2009). Accordingly, two macrobioclimates (Mediterranean pluviseasonal oceanic and Temperate oceanic), one variant of Temperate (Submediterranean), four classes of continentality (from weak semihyperoceanic to weak subcontinental), eight thermotypic horizons (from lower thermomediterranean to upper supratemperate) and seven ombrothermic horizons (from lower dry to lower hyperhumid) were identified (Bacchetta et al. 2009; Canu et al. 2015). Mediterranean types are widespread and occupy an area of about 99%. In the western and southern sectors (along the coastline), oceanic types are common, whereas the semicontinental type characterises only the inner area. Temperate macrobioclimate is confined at higher altitudes in the central and central-western part of the Island, where orographic rainfall values reduce the summer dry period and intensity (Canu et al. 2015).

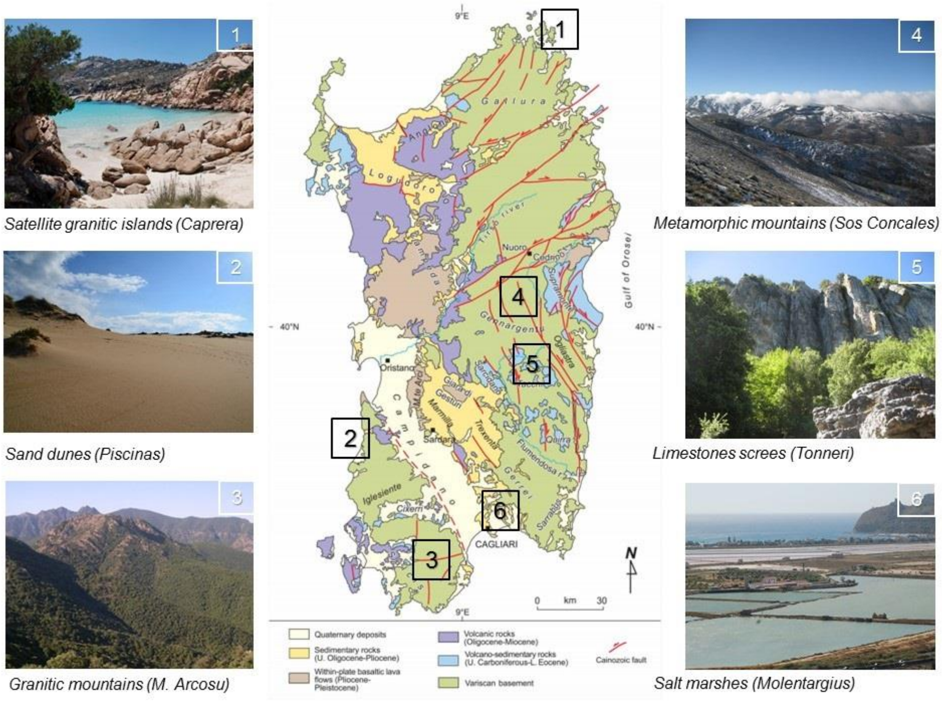

Figure 3. Examples of different environments which are present in the Islands of Sardinia. Numbers on the simplified geological map of Carmignani et al. (2016) indicates locations.

Overall, the complex geological history and paleoclimatology of the Mediterranean Basin are likely to drive biotas to fragment and merge as dispersal barriers appeared and disappeared through time. Within the western Mediterranean, Corsica, Sardinia, the Balearic Islands, the internal parts of the Betic Mountains of Spain and the Rif Mountains of Morocco, the Kabylies (in the Atlas mountains of Algeria), and Calabria (in the southern tip of the Italian peninsula) are all remnants of a Paleozoic mountain chain (the Hercynian belt) that was situated in Iberia and southern Europe during the Early Oligocene. Corsica, Sardinia, and the Balearic Islands are referred here as the "Hercynian Islands," reflecting their geological origin, and their endemic species are designated as "Hercynian endemics", which are often considered palaeoendemic in the broad sense of the term (i.e., ancient or relict taxa often systematically isolated; Greuter, 1995).

With the closure of the Strait of Gibraltar (ca. 5.59 Ma; Garcia-Castellanos et al. 2009) the Messinian Salinity Crisis of the Late Miocene started and some connections were established between North Africa, Corsica, Sardinia, and continental Europe, as well as between the Balearic Islands and Iberia; but no evidence of direct terrestrial corridors between Corsica or Sardinia and Balearic Islands have been documented (Alvarez, 1972; Mansion et al. 2008). During the Messinian, the Tuscan Archipelago may have connected Corsica, Sardinia, and the Italian Peninsula. The cycles of desiccation and transgression of the Mediterranean Sea in this period enabled interchanges of lineages of biota that predated the Messinian Salinity Crisis in all these territories (Salvo et al. 2010; Molins et al. 2011). The subsequent reopening of the Strait of Gibraltar (ca. 5.33 Ma; Garcia-Castellanos et al. 2009) caused partial extinction and isolation of previously connected populations and seems to have promoted vicariant speciation and population divergence at least in some documented cases (e.g., *Quercus ilex* L., Lumaret et al. 2002; *Anchusa* L. Bacchetta et al. 2008; *Aquilegia* L. Garrido et al. 2012). The subsequent establishment of the Mediterranean climate (ca. 3-2 Ma) promoted the expansion of xerophytic elements and typically Mediterranean taxa (Thompson, 2005). Later, the cyclical climatic oscillations of the Quaternary Pleistocene (ca. 1.8-0.01 Ma) also significantly shaped the genetic structure and spatial distribution of the biota, leading to population differentiation and eventually to speciation (Salvo et al. 2010). Particularly, during the Pleistocene glacial maxima the sea level was approximately 120-150 m lower than at present (Yokohama et al. 2000) and the Corsican and Sardinian coastlines were again directly connected by land bridges (Salvo et al. 2010). These connections facilitated exchanges of plant species and have alternatively limited or favoured gene flow between populations of species distributed in both islands and probably also among them and the Tuscan islets.

As other Mediterranean areas, Sardinia has undergone a transformation from the wilderness of the original Mediterranean forest to an agricultural landscape with wheat fields in the plains, vineyards on the slopes, and pastoral land in the highlands (Pungetti et al. 2008). The earliest signs of human presence come from the Mesolithic (9th –8th millennium cal. BC), whereas a stable occupation is documented only during the Early Neolithic (6th millennium cal. BC). The Nuragic civilization started during the Middle Bronze Age when human-ranked communities began to mark their territories by means of monumental tombs, called "Tombe dei giganti", and cyclopean towers, "Nuraghi". These buildings were elements of networks for territorial control and had probably multiple functions (Blake 1998).

Several studies of historical colonization have shown important environmental changes related to initial human settlement (Beffa et al. 2015; Poher 2016), with a rapid deforestation and a subsequent expansion of herbaceous vegetation. Also the presence of commensal mammals (e.g. wild pigs, deers and muflons), which undoubtedly brought profound ecological changes to the vegetation, are likely to be strongly related to human presence (Vigne 1992). Moreover, recent archaeobotanical analyses (e.g. Ucchesu et al. 2015; Buosi et al. 2016), proved the induced spread of many wild crop relatives, such as *Vitis vinifera*, *Prunus* sp.pl. and cereals, since first human colonisations.

More recently, there are many evidences of the human-induced changes on the current landscape. Malaria eradication, river control and irrigation allowed previously underutilized lands to be transformed into intensively cultivated lands. Agricultural development was also one of the goals of the fascist dictatorship (1922–1940). This brought about further changes to the Sardinian landscape, with new drainage channels for land reclamation and the disturbance of wetland ecosystems. As a result, cities, irrigated agriculture and industrial activities are concentrated along the coast and in the alluvial plains. The scattering on the island of wooded pastures, mainly characterised by the dispersion of chestnuts, cork oaks and holm oaks, related to the high landscape diversity, has given rise to a distinct cultural landscape. These traditional agrosilvopastoral systems retain both ecological and economic significance. Ecologically, they host diverse animal and plant communities with many endangered species, and therefore are currently preserved in the protected areas of the Island. Economically, they provide direct benefits derived from cork exploitation and parallel activities carried out under cork canopies, e.g. cropping and grazing, which are of considerable social importance. Although relatively new, since they developed during the 19th and 20th century, these systems represent a sustainable balance between human activities and natural resources, creating landscapes of high heterogeneity and cultural value (Naveh and Lieberman 1994). Nonetheless, the new economy is forcing changes of such overview towards different human pressures and interests. Agriculture and industrial activities and, in particular, agrosilvopastoral rural systems have being abandoned in favour of urban infrastructure expansions and coastal tourism development.

1.3 Floristic discoveries and phytogeographical traits of Sardinia

The floristic inventory of a given area is the very first basis, a prerequisite, and a starting point for assessing plant conservation, management, and ecological restoration, providing information on the need for additional surveys or data collections, as well as establishing the starting point for more detailed studies. It aids in identifying and correctly naming species, essential resources for biodiversity estimates and biogeographic studies (Kier et al. 2005). The floristic knowledge on the Sardinian flora gradually increased from the first researches of Plazza (1759, posteriorly published by Terraciano 1914, 1930) and Moris (1823-59) to the more recent Italian flora (Pignatti 1982) and checklists (Conti et al. 2005, 2007), integrated by new taxonomic descriptions (e.g. Camarda 2013; Bacchetta et al. 2014). In particular, this trend has exponentially increased in last decades (Fig.4) from a total number of 2054 taxa considered by Bocchieri (1995) up to the 2494 taxa reported by Conti et al. (2007); new editions of Italian and Sardinian floras are expected to be published in next years.

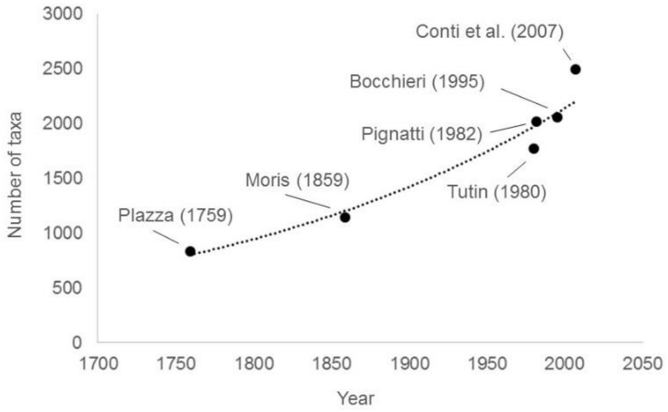

Figure 4. Exponential trend of number of known taxa in Sardinia along the last centuries. The European flora (Tutin 1980) and the last Italian checklist (Conti et al. 2007) are under and above the curve, respectively.

Since first explorations, the singularity of the Sardinian flora, its numerous endemisms, and significant number of rarities, aroused the interests of researchers and collectors; an interesting paper (Arrigoni, 2006) summarised these historical aspects. An important contribute had been the monographic researches of Arrigoni et al. (1977-1992) where all endemic plant species were described and all known occurrence localities reported and mapped. More recently, Bacchetta et al. (2012b) updated the list of all exclusive plant taxa, in the light of more recent discovers.

From a distribution view point, researches were firstly concentrated on the most rich mountainous areas of the centre of the Island, then, the information about the specific findings of many species was enriched by many botanists through collections of herbaria specimens or, more recently, of germplasm materials for *ex situ* conservation in seed banks. In addition, many floras of specific areas, such as the satellite islets of Sardinia (eg. Bocchieri 1992, 1998), mountains and mine areas of the southwestern part of Sardinia, have been published until recent years as scientific articles or master and doctoral theses (e.g. Bacchetta 2006; Iiriti 2006; Pontecorvo 2006). Up to recently, these findings have never been collected into a georeferenced database and comprehensively analysed.

Floristic knowledge, correlated with the above-mentioned environment and geological history, allowed to enhance the phytogeographical regionalization of Sardinia, proposed for the first time by Arrigoni (1983). According to Rivas-Martínez et al. (2004), Sardinia lays in the centre of the Mediterranean biogeographical region and constitutes the Italo-Tyrrhenian province, together with Corsican, Sicilian, and Coastal West Italian subprovinces. However, owing to many similarities, other authors (Bacchetta and Pontecorvo 2005) have suggested the rank of biogeographical province for Sardinia and Corsica, within an Italo-Tyrrhenian superprovince. According to their floristic similarities, Bacchetta et al. (2012a) proposed to include the Tuscan Archipelago within the Sardo-Corsican biogeographical province. More recently, the first version of the georeferenced occurrence localities of the endemic flora allowed to defy six biogeographical sectors and 22 subsectors for Sardinia and the circum-Sardinian small islands (Fig. 5, see Annex IV for further details). All sectors were characterised by the presence of exclusive endemic taxa, ranging from two to 33 taxa. In particular, the sector with the highest number of exclusive entities was the Sulcitano-Iglesiente (33 taxa), followed by the Campidanese-Turritano (31 taxa) and Supramontano (14 taxa) sectors. Besides the presence/absence of endemic plants, the definition of sectors and subsectors was based on the clusterization of homogeneous geomorphological units. These results confirmed the above mentioned strong interaction of plants with the abiotic environment. The reliability of this biogeographical regionalization was proved, for instance, by the according population structure of the endemic *Ribes sandalioticum* from a genetic and biogeographical viewpoint (Gentili et al. 2015b). In line with other examples, further applications of biogeography at regional scales could be applied for practical conservational purposes (e.g. Jelaska et al. 2010; Marignani et al. 2014; Mendoza-Fernández et al. 2015).

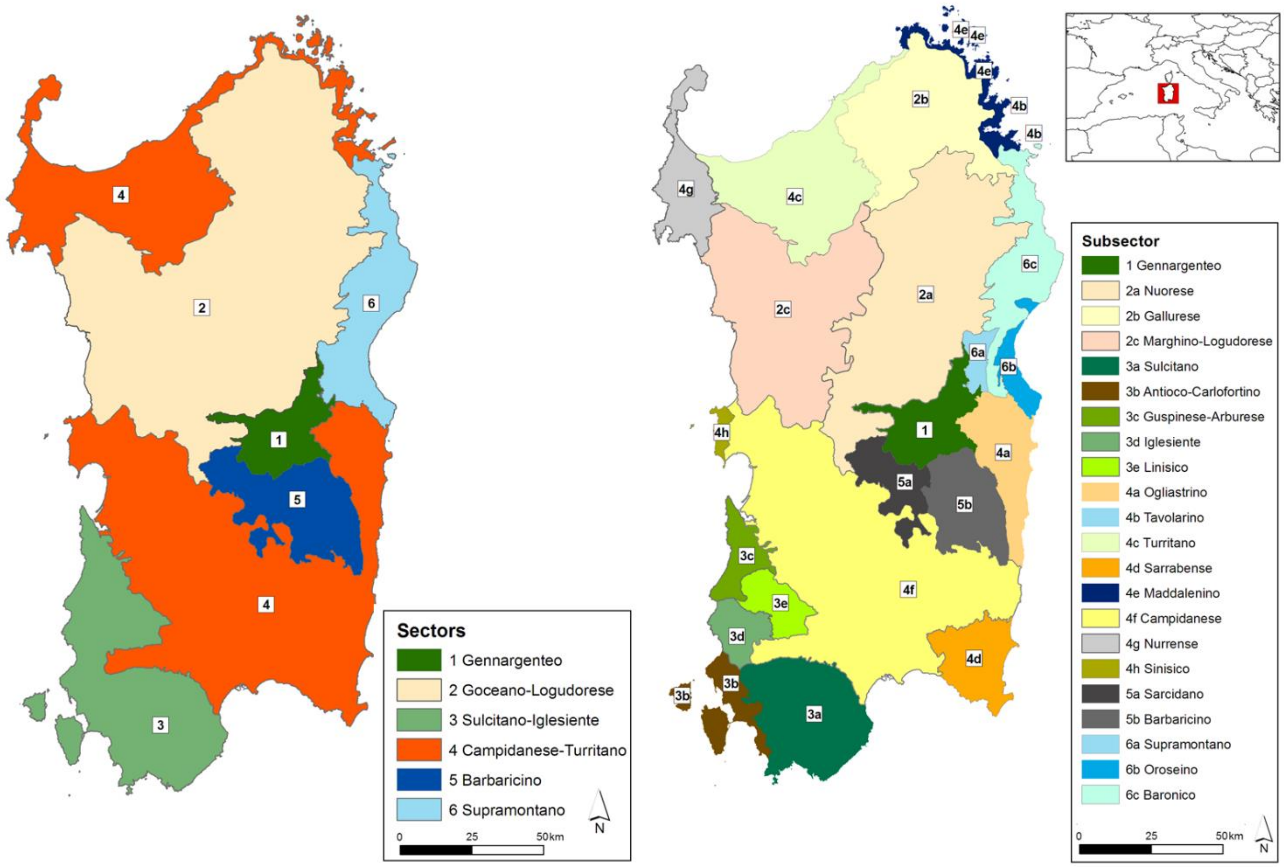

Figure 5. Biogeographical regionalization in sectors (a) and subsectors (b) of Sardinia based on the distribution of endemic vascular plants (Figure from Fenu et al. 2014).

1.4 Research objectives

The objectives of this thesis were to ameliorate the knowledge about the distribution of plant species endemic to Sardinia, and test how and where threats are acting upon this flora to finally combine this information to propose effective *in situ* conservation activities for these species.

This thesis represents a continuum of my Master degree that was focused on the georeferentiation in Geographic Information System (GIS) environment of all known historical and current occurrence localities; such database is continuously being implemented and updated.

During the earliest months of this project, I complete my first research defining the biogeography of the entire territory of Sardinia at sector and sub-sector levels on the basis of endemic plant similarities (Annex I).

In **chapter I**, I tested the trend of the integrated strategy implemented in the last 10 years for the endemic and policy plant species of Sardinia, which includes the following activities: conservation status assessment, *ex situ* conservation, *in situ* monitoring and active protection measures. A gap analysis between endemic species richness and activities efforts allowed to spatially highlight where conservation activities where (or not) adequately implemented. An index (the Area Under the Target) was ideated to measure gaps in achieving the Aichi biodiversity target 12.

In **chapter II**, I performed a regional scale analysis of environmental and anthropogenic correlates for the current endemic species richness to point out how such factors differently contributed to explain the cryptic distribution of the endemic plant species and to show some perspectives that should be considered for concrete conservation activities on endemic plants.

In **chapter III**, I used the documented recent extinction occurrences of threatened and endemic flora to test which variables mainly drove such events and, accordingly, extinction occurrences were spatially modelled to score the potential areas of extinctions.

According to the spatial results obtained in all the three previous chapters**,** in **chapter IV**, I focused my attention on the small satellite islands of Sardinia, trying to find interesting insights for future conservation activities in coastal habitats.

In **chapter V,** I used the distribution of endemic plant richness to identify and to test the efficiency of micro and nano hotspot networks for endemic plant conservation in Sardinia in terms of area, perimeter and connectivity.

Finally, in **chapter VI,** I performed an analysis on the effectiveness of Natura 2000 network under current and future scenarios in order to point out future challenges of improving the current state of the art of Natura 2000 network.

The annexes contain the abstract of researches that can be related with the chapters presented above. Each annex was cited in the main chapters throughout the text.

1.6 References of Introduction

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CHAPTER I

The Aichi Biodiversity Target 12 at regional level: an achievable goal?

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Abstract

The Aichi Biodiversity Target 12 aims to prevent the extinction risk of known threatened species and to improve their conservation status by 2020. We present the integrated strategy implemented in the last ten years for the key stone plant species of Sardinia (Italy, W-Mediterranean Basin) which includes the following activities: conservation status assessment (following the IUCN protocol), *ex situ* conservation, *in situ* monitoring and active protection measures. To date, an average of 51.8% of keystone plant species have been subjected to the latter first three activities while, due to the higher costs, only few active conservation measures have been carried out. Considering the realized activities since 2004, we also predicted the conservation effectiveness toward 2020 and we elaborated an index to evaluate it. Halfway through the strategic plan, we argue that more efforts are needed to guarantee the effective conservation of all threatened plants in Sardinia.

Keywords: Aichi Biodiversity Targets; Conservation planning; Plant conservation; Policy species; Regional Responsibility criterion; Sardinia; Threatened species.

1. Introduction

In 2002, 193 world leaders committed through the United Nation Convention on Biological Diversity (CBD) "to achieve by 2010 a significant reduction of the current rate of biodiversity loss" (CBD 2002). Considering the disregarded expectations and in face of the ongoing biodiversity declines (Butchart et al. 2010), all the parties of the CBD adopted in 2010 a strategic plan for biodiversity and 20 targets (the Aichi Biodiversity Targets, ABTs hereafter) that will constitute the framework until 2020 (CBD 2010).

The 20 ABTs, which provide a coherent guidance on how to achieve a series of targets aimed at improving the status of biodiversity and safeguarding the ecosystems, are organized under five strategic goals; each goal addresses a different challenge related to halting biodiversity loss. Halfway through the strategic plan, it is time to prioritize actions in order to achieve the best possible outcomes for the ABTs in 2020 (Marques et al. 2014); moreover, projected conservation measures are unlikely to achieve notable improvement (Small 2011). One of these strategic goals is focused for the enhancement of the biodiversity status by safeguarding ecosystems, species and genetic diversity. In particular, ABT 12 aims to "prevent the extinction risk of known threatened species and to improve and sustain their conservation status, particularly of those most in decline, by 2020" (CBD 2010). This more specific target of 'preventing' extinction of known threatened species replace the previous vague target of 'decreasing the rate' of biodiversity loss by 2010 (Larsen et al. 2014). Although the majority opinion is that uncommon species should be the focus of conservation, it still unclear which species deserve such attention (Jongman 2013).

The International Union for the Conservation of the Nature (IUCN) Red List provides strong baseline information for this target (www.cbd.int/gspc). Rarity and endemism are often incorporated uncritically as components of conservation status, but do not necessarily equate to vulnerability or threat of extinction (Battisti and Fanelli 2015). According to the regional responsibility criterion, a local priority list should be created in order to better identify the target species for conservation measures (e.g. Gauthier et al. 2010; Bacchetta et al. 2012). In this regard, quantifying endemisms at local scales, such as hotspot areas, can provide useful results for assessing the representation of protected sites, which may have implications for identifying priority areas for conservation (e.g. Laffan et al. 2013; Cañadas et al. 2014).

The aim of this study was to analyse the results obtained in the last ten years via the integrated strategy implemented for the Sardinian plant species with a conservation interest (i.e. key stone species *sensu* Small 2011) in order to verify and to predict conservation effectiveness of the ABT 12 toward 2020; in particular, only the scientific or technical-scientific activities were considered, then the passive way for *in situ* conservation (i.e. protected sites, natural parks) is not analysed in the article, being mainly covered by the ABT 11. Specifically, the following conservations measures carried out in Sardinia were analysed: (1) the conservation status assessment (following the IUCN protocol), (2) the *ex situ* seed conservation, (3) the *in situ* population studies and monitoring and (4) active protection measures.

2. Sardinian case study within the Mediterranean context

The Mediterranean Basin hosts a flora of around 25–30,000 flowering plants and ferns, c. 50% of them are endemic plants; for this reason, it has been identified as one of the 34 global biodiversity hotspots (Mittermeier et al. 2004). Within the Mediterranean Basin, islands and islets constitute the main centres of plant diversity (Médail and Quézel 1997).

Sardinia (Italy) and its ca. 399 satellite islands and islets is located in the Western part of the Mediterranean Basin (Fig. 1) and, covering a surface area of c. 24,090 km^2, is the second largest island after Sicily. From a biogeographical point of view, it is considered a part of the W-Mediterranean biogeographic sub-region (Rivas-Martínez et al. 2007). In particular, owing to the many floristic similarities, Sardinia, Corsica and the Tuscan Archipelago were included in the same independent biogeographical province within an Italo-Tyrrhenian super-province (Bacchetta et al. 2012). On the basis of the endemic plant occurrences and their geomorphological patterns, the sub-province of Sardinia was also recently subdivided into six biogeographic sectors and 22 subsectors (Fenu et al. 2014).

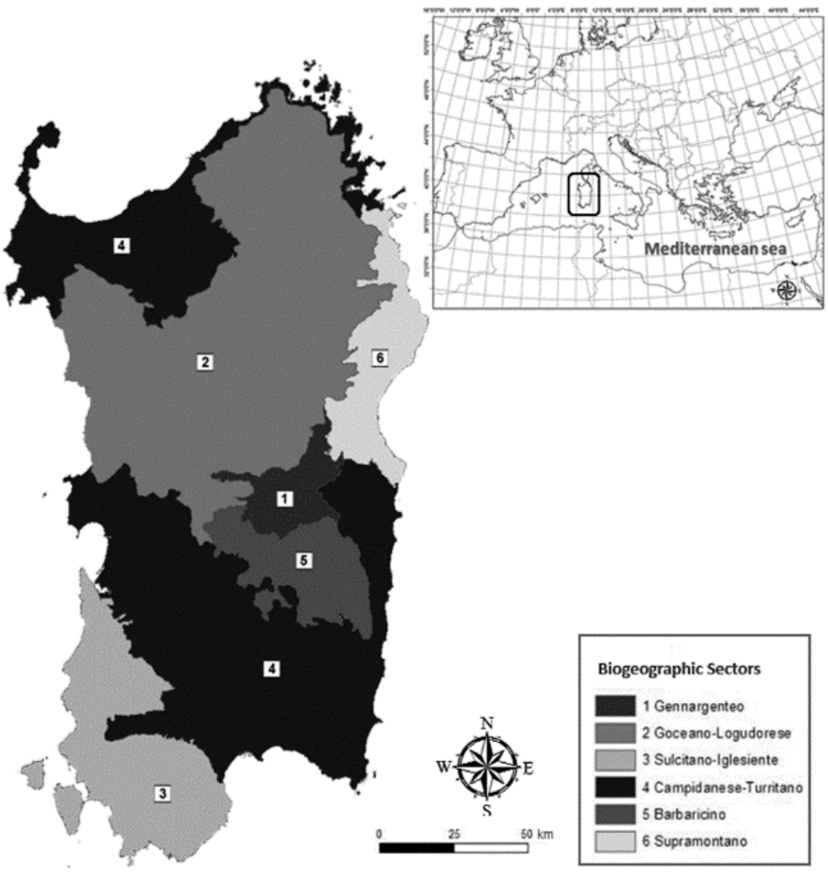

Figure 1. Sardinia study area in the Mediterranean Region. According to Fenu et al. (2014), the six biogeographical sectors are reported. Geographical longitudes and latitudes in minutes and seconds from the meridian of Greenwich are on the top and left side of the geographical background.

Its geographical isolation and high geological and geomorphological diversity contributed to develop a wide range of habitats and a consequent high rate of endemisms (Thompson 2005). Thus, according to the hierarchy proposed by Cañadas et al. (2014), Sardinia could be classified as a meso-hotspot within the Mediterranean mega-hotspot of biodiversity.

The vascular flora of Sardinia consists of 2,494 taxa (Conti et al. 2007), about the 11.6% of them (290 taxa) are considered as Sardinian endemic vascular plants (Bacchetta et al. 2012; Fenu et al. 2014) which are sub-divided into 183 exclusive Sardinian plant species, 90 Sardo-Corsican (including two endemic monotypic genera as *Morisia* Gay and *Nananthea* DC.), and 17 endemics in common with the Tuscan Archipelago (Bacchetta et al. 2012; Fenu et al. 2014).

3. Methodological approach

The methodological approach followed to achieve our target can be schematized through a flow-diagram composed by several stages (Fig. 2). First two steps toward a biodiversity conservation target were to identify the key stone species and to determine their distribution in our study area.

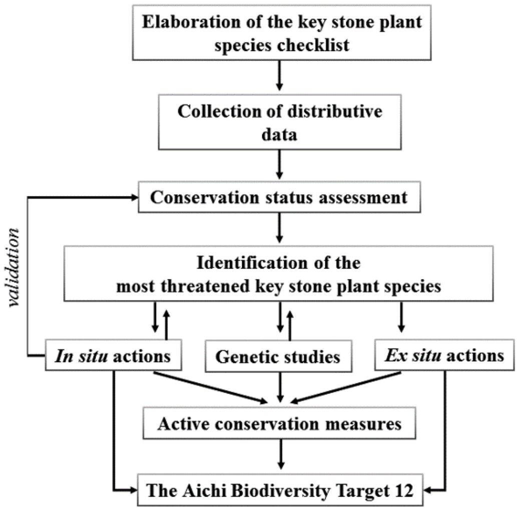

Figure 2. Flow chart summarizing the steps in implementing a methodological approach to conservation for target species. Based on global and regional priority-setting exercise, a comprehensive checklist of all key stone species is compiled and the relative distributive data are collected. From this information, a preliminary conservation status assessment is carried out in order to designate a candidate set of the most threatened key stone species. Through singular and/or simultaneous processes of *in situ*, *ex situ* actions and additional genetic studies, the active conservation measures are planned and then realized for the Aichi Biodiversity Target 12 accomplishment. The *in situ* and genetic actions could also be used for the validation of the first conservation status assessment and for a threatened key stone species updating.

Thus, after elaborating the priority checklist and a rapid assessment of the conservation status, several singular or complementary actions should be focused on each key stone species. In order to rationalize the efforts, a periodical conservation planning was based on the location (e.g. concentrating the efforts on species growing in the same place) and/or on their degree of threat (e.g. starting from the most endangered species); in other words, the conservation status assessment, based on the species distribution and estimated threats, was propaedeutic to the predetermined target. Thus, the most threatened species and/or the species located in an especially biodiverse area (i.e. micro- and nano-hotspots, *sensu* Fenu et al. 2010) were the first to be involved by additional *in situ* (e.g. monitoring) and genetic characterization activities as well as supported by the germplasm storage in seed banks and germination tests. Besides improving the conservation status assessment and knowledge in general, all these actions were crucial for a correct active conservation planning which usually represented the ideal scenario toward the achievement of the ABT 12 (Godefroid et al. 2011a).

The statistical analysis of the current trend of conservation actions and their projection to the expected demanding date could help to evaluate the feasibility of a target. We thus proposed a methodology to supply an analytical and geographical decision tool for the conservation activities planning. In order to predict the gap between the achievement of the ABT 12 by 2020 and what we could achieve maintaining our current trend, we performed a linear regression model of the three main conservation activities (conservation status assessment, *ex situ* and *in situ* conservation activities); starting from the same zero-date-point and reaching the target of both key stone plant groups (32 and 186 for policy and exclusive plant species, respectively), we thus compared the real with the ideal effort trend. A new specific index, defined as the "Area Under the Target" (AUT), was elaborated in order to quantify the distance between the analysed trends of efforts:

$$\mathrm{AUT} = \mathrm{A} - a \qquad (1)$$

where A is the area of the triangle constituted by the intersection of the modelled linear ideal trend and the axes x (years = 2020) and y (number of preserved *taxa* = No. *taxa* = 0), while a is defined as the area of the triangle identified by the modelled linear real trend and the same *catheti*. When the AUT = 0, the target was completely achieved while the AUT = 0.5 meant a totally unreached target; in case of a negative value, the target was achieved before the expected date (i.e. a >A).

Besides the number of taxa, we also tried to focus our attention on the location of our efforts. We first georeferred the occurrences of the analysed key stone groups; secondly, also the localities of the three conservation activities for both groups were georeferred inside a 5 × 5 km grid cell. The geodatabase of the key stone plants, which was used for this goal, is one of the instruments for conservation planning and restoration ecology employed by the Center for the Conservation of Biodiversity (CCB) in collaboration with other centers of research (e.g. Blasi et al. 2011; Fenu et al. 2014; Fois et al. 2015). Using the Geographic Information System (GIS) Software QGIS 1.7.4 (Quantum GIS Development Team 2012), a GAP analysis (Scott et al. 1993) was thus computed for each activity. For each grid cell, the Exclusive Species Richness (ESR hereafter) and Policy Species Richness (PSR hereafter) were calculated and overlaid on the Number of Activities (NA) per taxon which were already done. The GAP value was computed following the equation:

$$\mathrm{GAP} = 3i\mathrm{SR} - \mathrm{NA} \qquad (2)$$

where the PSR and ESR (iSR) multiplied by three (i.e. the number of the analysed activities) stands for the ideal total number of efforts on each grid. Thus, if all the activities for all key stone plants were achieved, the GAP would be zero while an high value of GAP detected the grids where further efforts are necessary.

4. Checklist elaboration and distribution data

Working in species-rich areas, such as the Mediterranean Basin, effective conservation needs the establishment of priorities at finer-scales (Margules and Pressey 2000; Bottrill et al. 2008; Gauthier et al. 2010). In this sense, several authors identified the "regional responsibility" as the first-order of priority at the local level (Gauthier et al. 2010; Bacchetta et al. 2012). Within this context, conservation efforts in the meso-hotspot of Sardinia must be oriented towards the policy species and, according to the regional responsibility criterion, also towards the narrow endemic species (Bacchetta et al. 2012). Thus, species selection was elaborated using both the policy species listed in the international regulations (Bern Convention on the Conservation of European Wildlife and Natural Habitats and the "Habitats Directive" 1992/43/EEC) and the list of exclusive taxa of Sardinia (Bacchetta et al. 2012).

The list of Sardinian policy species was thus obtained from the checklist reported in Rossi et al. (2015) via selecting the plant growing in the Island. The catalogue of the exclusive Sardinian endemic plants, compiled by Fenu et al. (2014), was the other starting point of our work. The updated checklist was obtained considering also all the new taxa described during the last years, as well as all the taxonomic novelties recently published. Finally, the list of all "key stone plant species" was composed by 32 policies plant species and 186 exclusive Sardinian endemics. Due to an overlap of 10 policy species which were also exclusive Sardinian endemics, the compressive lists consist of 208 key stone plant species (Table 1).

Table 1 Checklist of Sardinian key stone plant species. PS and EE indicate Policy Species and Exclusive Endemics, respectively.

N.	Taxon	Family	PS	EE
1	*Anchusa capellii* Moris	Boraginaceae		X
2	*Anchusa crispa* Viv. subsp. *crispa*	Boraginaceae	X	
3	*Anchusa crispa* Viv. subsp. *maritima* (Vals.) Selvi *et* Bigazzi	Boraginaceae	X	X
4	*Anchusa formosa* Selvi, Bigazzi *et* Bacch.	Boraginaceae		X
5	*Anchusa littorea* Moris	Boraginaceae	X	X
6	*Anchusa montelinasana* Angius, Pontecorvo *et* Selvi *ex* Bacch., Coppi, Pontecorvo *et* Selvi	Boraginaceae		X
7	*Anchusa sardoa* (Illario) Selvi *et* Bigazzi	Boraginaceae		X
8	*Anthyllis hermanniae* L. subsp. *ichnusae* Brullo *et* Giusso	Fabaceae		X
9	*Aquilegia barbaricina* Arrigoni *et* E.Nardi	Ranunculaceae		X
10	*Aquilegia cremnophila* Bacch., Brullo, Congiu, Fenu, J.Garrido *et* Mattana	Ranunculaceae		X
11	*Aquilegia nugorensis* Arrigoni *et* E.Nardi	Ranunculaceae		X
12	*Aquilegia nuragica* Arrigoni *et* E.Nardi	Ranunculaceae		X
13	*Aristolochia tyrrhena* E.Nardi *et* Arrigoni	Boraginaceae		X
14	*Armeria morisii* Boiss.	Plumbaginaceae		X
15	*Armeria sardoa* Spreng. subsp. *genargentea* Arrigoni	Plumbaginaceae		X
16	*Armeria sardoa* Spreng. subsp. *sardoa*	Plumbaginaceae		X
17	*Armeria sulcitana* Arrigoni	Plumbaginaceae		X
18	*Asperula deficens* Viv.	Rubiaceae		X
19	*Asperula pumila* Moris	Rubiaceae		X

20	*Astragalus genargenteus* Moris	Fabaceae		X
21	*Astragalus gennarii* Bacch. *et* Brullo	Fabaceae		X
22	*Astragalus maritimus* Moris	Fabaceae	X	X
23	*Astragalus tegulensis* Bacch. *et* Brullo	Fabaceae		X
24	*Astragalus thermensis* Vals.	Fabaceae		X
25	*Astragalus verrucosus* Moris	Fabaceae	X	X
26	*Bellium crassifolium* Moris	Asteraceae		X
27	*Bellium crassifolium* Moris var. *canescens* Gennari	Asteraceae		X
28	*Borago morisiana* Bigazzi *et* Ricceri	Boraginaceae		X
29	*Brassica insularis* Moris	Brassicaceae	X	
30	*Brassica tyrrhena* Giotta, Piccitto *et* Arrigoni	Brassicaceae		X
31	*Buphthalmum inuloides* Moris	Asteraceae		X
32	*Campanula forsythii* (Arcangeli) Podlech	Campanulaceae		X
33	*Carex panormitana* Guss.	Cyperaceae	X	
34	*Centaurea corensis* Vals. *et* Filigh.	Asteraceae		X
35	*Centaurea filiformis* Viv. subsp. *ferulacea* (Martelli) Arrigoni	Asteraceae		X
36	*Centaurea filiformis* Viv. subsp. *filiformis*	Asteraceae		X
37	*Centaurea forsythiana* Lev. *pro hybr*.	Asteraceae		X
38	*Centaurea horrida* Badarò	Asteraceae	X	X
39	*Centaurea magistrorum* Arrigoni *et* Camarda	Asteraceae		X
40	*Centranthus amazonum* Fridl. *et* A.Raynal	Valerianaceae		X
41	*Cephalaria bigazzii* Bacch., Brullo *et* Giusso	Dipsacaceae		X
42	*Cephalaria mediterranea* (Viv.) Szabò	Dipsacaceae		X
43	*Cerastium palustre* Moris	Caryophyllaceae		X
44	*Cerastium supramontanum* Arrigoni	Caryophyllaceae		X
45	*Charybdis glaucophylla* Bacch., Brullo, D'Emerico, Pontecorvo *et* Salmeri	Asparagaceae		X
46	*Clinopodium sardoum* (Asch. *et* Levier) Peruzzi *et* F.Conti	Lamiaceae		X
47	*Clinopodium sandalioticum* (Bacch. *et* Brullo) Bacch. *et* Brullo *ex* Peruzzi *et* F.Conti	Lamiaceae		X
48	*Colchicum verlaqueae* Fridl.	Colchicaceae		X
49	*Colchium gonarei* Camarda	Colchicaceae		X
50	*Cymbalaria muelleri* (Moris) A.Chev.	Plantaginaceae		X
51	*Cymodocea nodosa* (Ucria) Asch.	Cymodoceaceae	X	
52	*Cynoglossum barbaricinum* Arrigoni *et* Selvi	Boraginaceae		X

53	*Delphinium longipes* Moris	Ranunculaceae		X
54	*Dianthus cyathophorus* Moris	Caryophyllaceae		X
55	*Dianthus genargenteus* Bacch., Brullo, Casti *et* Giusso	Caryophyllaceae		X
56	*Dianthus ichnusae* Bacch., Brullo, Casti *et* Giusso subsp. *ichnusae*	Caryophyllaceae		X
57	*Dianthus ichnusae* Bacch., Brullo, Casti *et* Giusso subsp. *toddei* Bacch., Brullo, Casti *et* Giusso	Caryophyllaceae		X
58	*Dianthus insularis* Bacch., Brullo, Casti *et* Giusso	Caryophyllaceae		X
59	*Dianthus morisianus* Vals.	Caryophyllaceae		X
60	*Dianthus mossanus* Bacch. *et* Brullo	Caryophyllaceae		X
61	*Dianthus oliastrae* Bacch., Brullo, Casti *et* Giusso	Caryophyllaceae		X
62	*Dianthus sardous* Bacch., Brullo, Casti *et* Giusso	Caryophyllaceae		X
63	*Dipsacus valsecchii* Camarda	Dipsacaceae		X
64	*Echium anchusoides* Bacch., Brullo *et* Selvi	Boraginaceae		X
65	*Euphrasia nana* (Rouy) Prain	Orobanchaceae	X	
66	*Festuca morisiana* Parl. subsp. *morisiana*	Poaceae		X
67	*Galium glaucophyllum* Em.Schmid	Rubiaceae		X
68	*Galium schmidii* Arrigoni	Rubiaceae		X
69	*Genista arbusensis* Vals.	Fabaceae		X
70	*Genista bocchierii* Bacch., Feoli Chiapella *et* Brullo	Fabaceae		X
71	*Genista cadasonensis* Vals.	Fabaceae		X
72	*Genista ephedroides* DC.	Fabaceae		X
73	*Genista insularis* Bacch., Feoli Chiapella *et* Brullo subsp. *fodinae* Bacch., Feoli Chiapella *et* Brullo	Fabaceae		X
74	*Genista insularis* Bacch., Feoli Chiapella *et* Brullo subsp. *insularis*	Fabaceae		X
75	*Genista morisii* Colla	Fabaceae		X
76	*Genista ovina* Bacch., Feoli Chiapella *et* Brullo	Fabaceae		X
77	*Genista pichi-sermolliana* Vals.	Fabaceae		X
78	*Genista sardoa* Vals.	Fabaceae		X
79	*Genista sulcitana* Vals.	Fabaceae		X
80	*Genista toluensis* Vals.	Fabaceae		X
81	*Genista valsecchiae* Brullo *et* De Marco	Fabaceae		X
82	*Gentiana lutea* L. subsp. *lutea*	Gentianaceae	X	
83	*Glechoma sardoa* (Bég.) Bég.	Lamiaceae		X
84	*Helianthemum caput-felis* Boiss.	Cistaceae	X	
85	*Helianthemum morisianum* Bertol.	Cistaceae		X

86	*Helichrysum montelinasanum* Em.Schmid	Asteraceae		X
87	*Helichrysum saxatile* Moris subsp. *morisianum* Bacch., Brullo *et* Mossa	Asteraceae		X
88	*Helichrysum saxatile* Moris subsp. *saxatile*	Asteraceae		X
89	*Herniaria litardierei* (Gamisans) Greuter *et* Burdet	Caryophyllaceae	X	
90	*Hieracium iolai* Arrigoni	Asteraceae		X
91	*Hypericum annulatum* Moris	Hypericaceae		X
92	*Hypericum scruglii* Bacch., Brullo *et* Salmeri	Hypericaceae		X
93	*Hypochaeris sardoa* Bacch., Brullo *et* Terrasi	Asteraceae		X
94	*Iberis integerrima* Moris	Brassicaceae		X
95	*Lactuca longidentata* Moris	Asteraceae		X
96	*Lamyropsis microcephala* (Moris) Dittrich *et* Greuter	Asteraceae	X	X
97	*Lavatera plazzae* Atzei	Malvaceae		X
98	*Lavatera triloba* L. subsp. *pallescens* (Moris) Nyman	Malvaceae		X
99	*Limonium acutifolium* (Rchb.) Salmon subsp. *acutifolium*	Plumbaginaceae		X
100	*Limonium acutifolium* (Rchb.) Salmon subsp. *bosanum* (Arrigoni *et* Diana) Arrigoni	Plumbaginaceae		X
101	*Limonium acutifolium* (Rchb.) Salmon subsp. *cornusianum* (Arrigoni *et* Diana) Arrigoni	Plumbaginaceae		X
102	*Limonium acutifolium* (Rchb.) Salmon subsp. *nymphaeum* (Erben) Arrigoni	Plumbaginaceae		X
103	*Limonium acutifolium* (Rchb.) Salmon subsp. *tenuifolium* (Bertol. *ex* Moris) Arrigoni	Plumbaginaceae		X
104	*Limonium acutifolium* (Rchb.) Salmon subsp. *tharrosianum* (Arrigoni *et* Diana) Arrigoni	Plumbaginaceae		X
105	*Limonium ampuriense* Arrigoni *et* Diana	Plumbaginaceae		X
106	*Limonium capitis-eliae* Erben	Plumbaginaceae		X
107	*Limonium capitis-marci* Arrigoni *et* Diana	Plumbaginaceae		X
108	*Limonium carisae* Erben	Plumbaginaceae		X
109	*Limonium coralliforme* Mayer	Plumbaginaceae		X
110	*Limonium cunicularium* Arrigoni *et* Diana	Plumbaginaceae		X
111	*Limonium gallurense* Arrigoni *et* Diana	Plumbaginaceae		X
112	*Limonium glomeratum* (Tausch) Erben	Plumbaginaceae		X
113	*Limonium hermaeum* Pignatti	Plumbaginaceae		X
114	*Limonium insulare* (Bég. *et* Landi) Arrigoni *et* Diana	Plumbaginaceae	X	X
115	*Limonium laetum* (Nyman) Pignatti	Plumbaginaceae		X
116	*Limonium lausianum* Pignatti	Plumbaginaceae		X

117	*Limonium malfatanicum* Erben	Plumbaginaceae		X
118	*Limonium merxmuelleri* Erben subsp. *merxluelleri*	Plumbaginaceae		X
119	*Limonium merxmuelleri* Erben subsp. *oristanum* (Alf.Mayer) Arrigoni	Plumbaginaceae		X
120	*Limonium merxmuelleri* Erben subsp. *sulcitanum* (Arrigoni) Arrigoni	Plumbaginaceae		X
121	*Limonium merxmuelleri* Erben subsp. *tigulianum* (Arrigoni *et* Diana) Arrigoni	Plumbaginaceae		X
122	*Limonium morisianum* Arrigoni	Plumbaginaceae		X
123	*Limonium multifurcatum* Erben	Plumbaginaceae		X
124	*Limonium protohermaeum* Arrigoni *et* Diana	Plumbaginaceae		X
125	*Limonium pseudolaetum* Arrigoni *et* Diana	Plumbaginaceae	X	X
126	*Limonium pulviniforme* Arrigoni *et* Diana	Plumbaginaceae		X
127	*Limonium racemosum* (Lojac.) Diana	Plumbaginaceae		X
128	*Limonium retirameum* Greuter *et* Burdet subsp. *retirameum*	Plumbaginaceae		X
129	*Limonium retirameum* Greuter *et* Burdet subsp. *sardoum* (Pignatti) Galasso	Plumbaginaceae		X
130	*Limonium strictissimum* (Salzm.) Arrigoni	Plumbaginaceae	X	
131	*Limonium tibulatium* Pignatti	Plumbaginaceae		X
132	*Limonium tyrrhenicum* Arrigoni *et* Diana	Plumbaginaceae		X
133	*Limonium ursanum* Erben	Plumbaginaceae		X
134	*Limonium viniolae* Arrigoni *et* Diana	Plumbaginaceae		X
135	*Linaria arcusangeli* Atzei *et* Camarda	Plantaginaceae		X
136	*Linaria flava* (Poir.) Desf. subsp. *sardoa* (Sommier) A.Terracc.	Plantaginaceae	X	
137	*Linum muelleri* Moris	Linaceae	X	X
138	*Marsilea quadrifolia* L.	Marsileaceae	X	
139	*Marsilea strigosa* Willd.	Marsileaceae	X	
140	*Micromeria filiformis* (Aiton) Benth. subsp. *cordata* (Moris) Pignatti	Lamiaceae		X
141	*Narcissus supramontanus* Arrigoni subsp. *cunicularium* Arrigoni	Amaryllidaceae		X
142	*Narcissus supramontanus* Arrigoni subsp. *supramontanus*	Amaryllidaceae		X
143	*Nepeta foliosa* Moris	Lamiaceae		X
144	*Odontarrhena tavolarae* (Briq.) L.Cecchi *et* Selvi	Brassicaceae		X
145	*Oenanthe lisae* Moris	Apiaceae		X
146	*Ophrys chestermanii* (J.J.Wood) Gölz *et* H.R.Reinhard	Orchidaceae		X
147	*Ophrys conradiae* Melki *et* Deschâtres	Orchidaceae		X
148	*Ophrys normanii* J.J.Wood *pro hybr.*	Orchidaceae		X
149	*Ophrys ortuabis* M.P.Grasso *et* Manca	Orchidaceae		X

150	*Ophrys panattensis* Scrugli, Cogoni *et* Passei	Orchidaceae		X
151	*Ophrys subfusca* (Rchb. f.) Batt. subsp. *liveranii* Orrù *et* M.P.Grasso	Orchidaceae		X
152	*Orchis provincialis* Balb. *ex* Lam. *et* DC.	Orchidaceae	X	
153	*Orobanche australis* Moris *ex* Bert.	Orobanchaceae		X
154	*Orobanche denudata* Moris	Orobanchaceae		X
155	*Phleum sardoum* (Hackel) Hackel	Poaceae		X
156	*Pilularia minuta* Durieu *ex* A.Braun	Marsileaceae	X	
157	*Pinguicula sehuensis* Bacch., Cannas *et* Peruzzi	Lentibulariaceae		X
158	*Platanthera kuenkelei* H.Bauman subsp. *kuenkelei* var. *sardoa* R.Lorenz, Akhalkatsi, H.Baumann, Cortis, Cogoni *et* Scrugli	Orchidaceae		X
159	*Polygala sardoa* Chodat	Polygalaceae		X
160	*Polygala sinisica* Arrigoni	Polygalaceae		X
161	*Portulaca sardoa* Danin, Bagella *et* Marrosu	Portulacaceae		X
162	*Posidonia oceanica* (L.) Delile	Posidoniaceae	X	
163	*Ptychotis sardoa* Pignatti *et* Metlesics	Apiaceae		X
164	*Pulicaria vulgaris* Gaertn. var. *sardoa* Fiori	Asteraceae		X
165	*Quercus ichnusae* Mossa, Bacch. *et* Brullo	Fagaceae		X
166	*Ranunculus breyninus* Crantz subsp. *sardous* Em.Schmid	Ranunuclaceae		X
167	*Ranunculus cymbalarifolius* Balbis *et* Moris	Ranunculaceae		X
168	*Rhamnus persicifolia* Moris	Rhamnaceae		X
169	*Ribes multiflorum* Kit. *ex* Roem. *et* Schult. subsp. *sandalioticum* Arrigoni	Grossulariaceae		X
170	*Ribes sardoum* Martelli	Grossulariaceae	X	X
171	*Romulea bocchierii* Frignani *et* Iiriti	Iridaceae		X
172	*Romulea limbarae* Bég. *pro hybr.*	Iridaceae		X
173	*Rouya polygama* (Desf.) Coincy	Apiaceae	X	
174	*Rubus arrigonii* Camarda	Rosaceae		X
175	*Rubus limbarae* Camarda	Rosaceae		X
176	*Rumex pulcher* L. subsp. *suffocatus* (Moris *ex* Bertol.) Nyman	Polygonaceae		X
177	*Ruscus aculeatus* L.	Asparagaceae	X	
178	*Ruta lamarmorae* Bacch., Brullo *et* Giusso	Rutaceae		X
179	*Salicornia veneta* Pignatti *et* Lausi	Amaranthaceae	X	
180	*Salix arrigonii* Brullo	Salicaceae		X
181	*Salvia desoleana* Atzei *et* V.Picci	Lamiaceae		X
182	*Santolina insularis* (Gennari *ex* Fiori) Arrigoni	Asteraceae		X

183	*Scorzonera callosa* Moris	Asteraceae		X
184	*Scrophularia morisii* Vals.	Scrophulariaceae		X
185	*Sedum villosum* L. subsp. *glandulosum* (Moris) P.Fourn.	Crassulaceae		X
186	*Selaginella denticulata* (L.) Spring	Selaginellaceae	X	
187	*Senecio squalidus* L. subsp. *sardous* (Fiori) Greuter	Asteraceae		X
188	*Senecio vulgaris* L. var. *tyrrhenus* Fiori	Asteraceae		X
189	*Sesleria insularis* Sommier subsp. *barbaricina* Arrigoni	Poaceae		X
190	*Sesleria insularis* Sommier subsp. *morisiana* Arrigoni	Poaceae		X
191	*Silene beguinotii* Vals.	Caryophyllaceae		X
192	*Silene ichnusae* Brullo, De Marco *et* De Marco f.	Caryophyllaceae		X
193	*Silene martinolii* Bocchieri *et* Mulas	Caryophyllaceae		X
194	*Silene morisiana* Bég. *et* Rav.	Caryophyllaceae		X
195	*Silene rosulata* Soy-Will. *et* Godr. subsp. *sanctae-theresiae* (Jeanm.) Jeanm.	Caryophyllaceae		X
196	*Silene valsecchiae* Bocchieri	Caryophyllaceae		X
197	*Silene velutina* Loisel.	Caryophyllaceae	X	
198	*Spiranthes aestivalis* (Poir.) Rich.	Orchidaceae	X	
199	*Taraxacum barbaricinum* Arrigoni	Asteraceae		X
200	*Taraxacum garbarianum* Peruzzi, Aquaro, Caparelli *et* Raimondo	Asteraceae		X
201	*Taraxacum genargenteum* Arrigoni	Asteraceae		X
202	*Taraxacum sarcidanum* Arrigoni	Asteraceae		X
203	*Taraxacum sardomontanum* Arrigoni	Asteraceae		X
204	*Thesium italicum* A.DC. *in* DC.	Santalaceae		X
205	*Verbascum plantagineum* Moris	Scrophulariaceae		X
206	*Vinca difformis* Pourr. subsp. *sardoa* Stearn	Apocynaceae		X
207	*Viola corsica* Nym. subsp. *limbarae* Merxm. *et* Lippert	Violaceae		X
208	*Zostera marina* L.	Zosteraceae	X	

Distribution data, obtained from available literature, *herbarium* specimens and implemented with the authors' field researches (including unpublished data), was georeferenced by the Geographic Information System (GIS) Software QGIS 1.7.4 (Quantum GIS Development Team 2012), and organized in a geodatabase. Each georeferenced record was reported into a 5 × 5 km regular grid of Sardinia and the number of *taxa* in each cell was thus computed.

As a consequence of the high percentage of policy species strictly linked to coastal and marine habitats (17 *taxa* corresponding to the 51.5% of the total), a high PSR was detected along the Sardinian coastal areas (Fig.

3a). In inland territories, high concentrations of policy species only occur in narrow areas of the Gennargentu and Sulcitano-Iglesiente biogeographic sectors (see Fig. 1 for the location of each biogeographic sector), located in the Central and South-Western part of the Island, respectively (Fig. 3a).

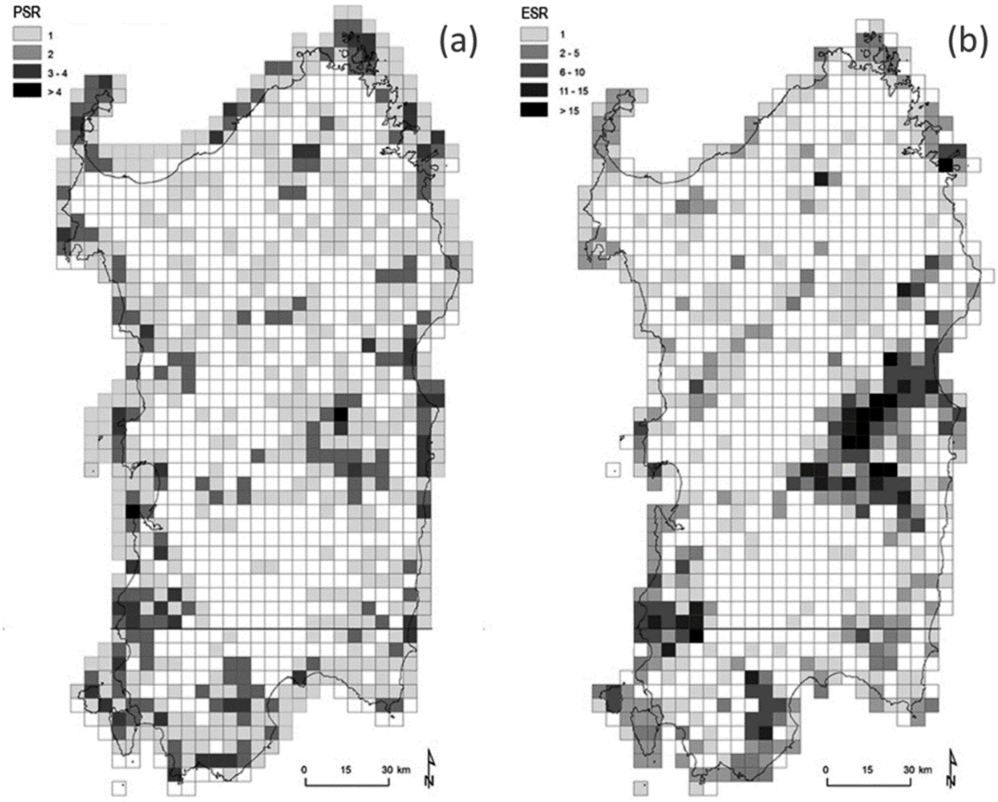

Figure 3. Distribution of (a) Policy Species Richness (PSR) and (b) Exclusive Species Richness (ESR) in Sardinia inside the 5 × 5 km grid cells.

A similar pattern to the PSR was found for the ESR. As previously highlighted (e.g. Fenu et al. 2010; Cañadas et al. 2014), the endemic plant richness is not uniformly distributed, but depends largely on the environmental conditions which also brought to the identification of a series of endemism-rich areas for Sardinia (Cañadas et al. 2014). Such micro- and nano-hotspots (*sensu* Fenu et al. 2010), which not only host more than 20% of regional endemics but also a high number of narrow endemics, mainly correspond to peaks of mountains, rocky places (Thompson 2005; Fenu et al. 2010; Cañadas et al. 2014), and small islets (Arrigoni and Bocchieri 1995; Fenu et al. 2014). Our analysis on a 5 × 5 km grid (Figure 3a, b) revealed high values of both PSR/ESR on the mountainous systems of the Gennargenteo and Goceano-Logudorese sectors (e.g. Gennargentu and Limbara massifs), on the carbonates outcrops of the Barbaricino and Supramontano sectors (e.g. Tacchi and Supramontes regions) and on the paleozoic authocthnous siliceous rocks of the Sulcitano-Iglesiente sector (Fig. 1). A high rate of key stone species was also depicted on the satellite islands, principally concentrated in the Campidanese-Turritano and Sulcitano-Iglesiente sectors (e.g. Tavolara and S. Pietro islands).

5. Conservation status assessment

While the conservation status assessment of the policy species is mandatory for all EU member states, the assessment of exclusive endemics, based on the regional responsibility criterion, represents the first step to develop a conservation strategy at local level.

The criteria established by the IUCN (2001) are widely employed as the gold standard for information on the conservation status of species (e.g. De Grammont and Cuarón 2006; Rodrigues et al. 2006). The IUCN red listing procedure represents a suitable and rapid method for verifying the effectiveness of conservation policies and highlights threatened taxa, providing an assessment of relative extinction risk (IUCN 2012). Globally, this procedure constitutes the most used red listing protocol (Rodrigues et al. 2006) because it allows objective, replicable, and flexible risk assessments (De Grammont and Cuarón 2006). Although the IUCN red listing system was originally developed for global assessments, it is widely used at regional or subglobal scales (Gärdenfors et al. 2001), also considering the biogeographic level (Gentili et al. 2011). The regional assessment of species extinction risk provides a basis for planning conservation actions and allocation of economic resources. For these reasons, updated red lists represents an important starting point for further conservation measures and may provide information useful for monitoring changes of species conservation status (Rossi et al. 2014).

To date, the only comprehensive Italian Red Lists of threatened plants (Conti et al. 1992, 1997) are based on an older version of the Red List system, different from the current IUCN standards (IUCN 2012). Recently the Italian Ministry for the Environment, Land and Sea (MATTM) promoted "the New Red List of the Italian Flora" project based on the current IUCN standards (IUCN 2012) as a first step towards the Red List of the whole Italian flora and it represents the starting point of a process of conservation of the national biodiversity (Rossi et al. 2014). The assessment activities carried out in Sardinia since the last years are a part of this national project.

The conservation status of all Sardinian policy species and numerous exclusive endemic plants was assessed following the standardized procedure adopted at national level (Rossi et al. 2014). As a result of this approach, the assessment of policy species is made at the Italian level, unless for the policy species that are also Sardinian exclusive endemics.

Data on species distribution were gathered from field surveys, *herbarium* specimens, and published/unpublished data. With a special attention for the historical localities, all distribution data were updated to the current occurrences and organized in a geodatabase, in which each georeferenced record was reported into a 2 × 2 km regular grid in order to ensure the standardized calculations of the Area Of Occupancy (AOO), required under the criterion B2 (IUCN 2012). The assessment procedure was mainly based on the criterion B, related to the plant distribution; only in a few cases, when reliable data on population trends or population size were available, other criteria were applied.

In 2014, the evaluation of all Italian policy species was completed (Rossi et al. 2015), as a consequence of a significant effort made in 2012 and 2013; in addition, 91 exclusive endemic plants, corresponding to the 48.9% of the total Sardinian endemics, were assessed.

In the last years, one regional policy species was no retrieved in the field in Sardinia (*Marsilea quadrifolia* L.), 15 were listed in a risk category (including CR, EN and VU plant species) corresponding to the 46.9% of the total and for two additional marine plants [*Cymodocea nodosa* (Ucria) Asch. and *Zostera marina* L.] no data is available and then considered as data deficient (DD). Regarding the exclusive endemics, 57 taxa are listed in a risk category corresponding to the 63.6% of the total and only for two plant species (*Colchicum verlaqueae* Fridl. and *Orobanche australis* Bertol.) no data is available and, then, considered as data deficient (DD).

The data from Sardinian policy species is consistent with those obtained at Italian and European levels, in which 41.9% (Rossi et al. 2015) and 44.9% (Bilz et al. 2011) of vascular plants were listed in a threat category. Similarities between Sardinia, Italy and Europe are also evident for threat type, as human-mediated habitat loss, degradation, and fragmentation are also among the most prevalent threats (Bilz et al. 2011; Rossi et al. 2015). Considering last reports at EU level, only 17% of the species and habitats protected by the Habitats Directive 1992/43/EEC have a favourable conservation status (Condé et al. 2010), thus, the conservation through the attenuation of most relevant threats is still a problematic issue.

The assessment of the Sardinian exclusive endemics still remains incomplete; therefore, it is imperative to increase the efforts in order to identify the risk status and provide suggestions on the plants needing a direct conservation actions.

6. *Ex situ* conservation actions

Although *in situ* conservation measures are the best methods for preserving plant diversity (UNEP 2002), *ex situ* conservation provides an alternative and complementary method for preventing immediate extinction (Godefroid et al. 2011a). As a precautionary measure, more importance should be given to *ex situ* conservation in seed banks (Mattana et al. 2012; Krigas et al. 2016), which may support further interventions, such as translocations (e.g. Godefroid et al. 2011b; Cogoni et al. 2013). One of the most effective ways to preserve the plant diversity through *ex situ* conservation is the storage in seed banks, which allows conserving large amounts of genetic material in a small space and, under suitable conditions, for a long time with minimum risk of genetic damage (Godefroid et al. 2011a; Mattana et al. 2012; Krigas et al. 2016). This is reflected in the Target 8 of the Global Strategy for Plant Conservation for 2020 (GSPC 2020) asking for "at least 75% of threatened plant species in *ex situ* collections, preferably in the country of origin, and at least 20% available for recovery and restoration programmes". Botanical gardens and seed banks play a major role in achieving this goal (Maunder et al. 2001; Sharrock and Jones 2011).

The Sardinian Germplasm Bank (BG-SAR), located in the Botanical Gardens of the University of Cagliari, has undertaken the *ex situ* long term conservation of the plant diversity of Sardinia. Several collecting trips, covering the whole island, were carried out during the period 2004-2014. For each seed lot, a data form concerning both the sampling method and the ecology, demography, phenology and conservation status was drafted and seed collection in field was made according to scientific criteria (Bacchetta et al. 2006, 2008). Until now, germplasm of ferns, marine plants and orchids has not been conserved due to the difficulties related to the seed management and their long-term conservation (e.g. Seaton et al. 2010).

The seed conservation processes carried out at BG-SAR followed internationally recognized protocols and guidelines following the genebank standards (Bacchetta et al. 2006, 2008). With the aim of a long-term conservation, all seed lots were dried at 15°C and 15% of relative humidity (R.H.), to reduce the internal seed moisture content at ca. 3-5%, and stored at - 25°C (as base collections under long-term conservation) and/or at 5°C (as active collections under medium-term conservation).

So far, BG-SAR preserved the germplasm of 15 policy species (46.9%) and 77 exclusive Sardinian endemics (41.4%) for their long-term conservation. The propensity of many plants (policy and/or endemic species) to grow in remote and/or inaccessible locations, and the lack of low seed production are two of the main obstacles inherent in harvesting seeds (Godefroid et al. 2011b). This is coherent with many Sardinian policy and/or endemic species, such as *Ribes sardoum* Martelli for which the number of available seeds was always insufficient to allow an effective seed collection (Fenu et al. 2012). However, despite these objective problems, our purpose is to guarantee further efforts for the long-term conservation of the maximum possible number of the Sardinian key stone plant species. By excluding the plants which are difficult to preserve with the standard *ex situ* conservation methods, the percentages of policy and exclusive endemic species of Sardinia effectively stored in BG-SAR are 65.2% and 42.8%, respectively.

From 2004 to date, germination tests conducted at BG-SAR investigated the seed ecology of 43.5% of the policy species, and of 15.6% of exclusive Sardinian plant species (e.g. Porceddu et al. 2013; Santo et al. 2014), which could be useful for *in situ* conservation actions, such as plant reintroduction or population reinforcement programs.

7. *In situ* conservation actions

The best methods for preserving plant diversity are the *in situ* conservation measures (UNEP 2002). Furthermore, monitoring and reporting the conservation status of species listed in the Habitats Directive is mandatory for all of the signatory EU Member States. In particular, as a first step, monitoring data related to plant populations can be used to predict the effects of various management practices on population size, condition, stage distribution (i.e., seed production and/or seedlings development) and demographic processes, including survivorship and seedling recruitment (Fenu et al. 2015). The parameters selected and the intensity of measurement will depend on specific management or conservation objectives, on initial assessment of threat or need (Kentel et al. 1989), on the biology of the species, and on available resources (Menges and Gordon 1996).

A multilevel monitoring scheme, with three different levels corresponding to an increasing intensity approach following Menges and Gordon (1996) and adapted to our insular situation, was applied in Sardinia in the last decade (Fenu et al. 2015). In these 10 years of field work, a total of 39 plant species have been periodically surveyed; in particular, 12 policy species, corresponding to the 37.5% of the total, have been monitored. In addition, 34 exclusive endemic plants (seven of them are also policy species) are still monitored, corresponding to the 18.3% of the total.

Although much of work is needed, the policy species monitored in Sardinia, represent the 7.6% and the 1.8% of the Italian and European policy species, respectively. Threatened plant monitoring faces two main practical difficulties: the need to maintain a sustained effort of monitoring across years in order to ensure the collection of relevant monitoring time series and the need to obtain precise monitoring data which allow the detection of significant changes across space and time in biodiversity. These needs come into conflict with the usually limited amount of available financial and human resources (Schmeller et al. 2009; Fenu et al. 2015).

Considering the scarcity of economic resources for Sardinia, conservation strategies must be focused on small areas that represent a maximum diversity and/or endemicity (Murray-Smith et al. 2009; Fenu et al. 2010; Cañadas et al. 2014). Therefore, besides implementing the priority lists, the geographical identification of the areas with a high endemic/key stone plant richness, such as the nano-hotspots (Fenu et al. 2010; Cañadas et al. 2014), could be also considered as a crucial target.

8. Active conservation measures

The importance of *in situ* conservation of endangered plant species has been highlighted by the Target 7 of the GSPC for 2020 which scheduled that at least 75% of known threatened plant species should be conserved *in situ* (GSPC 2020).

To prevent the extinction risk of known threatened species and to improve their conservation status, translocations have become increasingly important in management worldwide (e.g. Maunder 1992; Godefroid et al. 2011a) and they represent the ideal scenario, although it is not often practicable (Godefroid et al. 2011a). In this way, translocations (population reinforcement, reintroduction and introduction) aim to enhance population viability, for instance, by increasing population size, genetic diversity, or by the representation of specific demographic groups or stages (Godefroid et al. 2011a). The potentiality of translocations to contribute to the recovery of threatened species is significant when is a part of integrated conservation activities (Cogoni et al. 2013). However, many limits remain in the implementation of these conservation actions, such as the high both economic and time costs, the availability of the optimal site, the difficulties on the implementation of these actions in private areas and the high uncertainty of success principally connected to natural stochastic events. Thus, considering all these limitations, it is often necessary to identify other active measures, such as fence erections, to protect small population or to clone in a local nursery the entire population. To date, only eight active conservation measures were carried out in Sardinia. Besides the high economic and time costs, the genetic studies should be carried out before the improvement of any active conservation measure. Indeed, preparatory genetic studies were done for several threatened key stone species, such as *Lamyropsis microcephala* (Moris) Dittrich *et* Greuter (Bacchetta et al. 2013), *Rhamnus persicifolia* Moris (Bacchetta et al. 2011), *Ribes sardoum* and *Ribes multiflorum* Kit. *ex* Roem. *et* Schult. subsp. *sandalioticum* Arrigoni (Gentili et al. 2015).

A low-cost translocation project for *Dianthus morisianus* Vals., which achieved a positive result, was started in 2010 (Cogoni et al. 2013) and, more recently, a translocation for *Gentiana lutea* L. subsp. *lutea* by selecting the suitable reintroduction sites through Species Distribution Models (Fois et al. 2015) was started in 2014.

Fences to prevent nomadic grazing and, in particular, to protect the most critical life-cycle stage for population survival, were erected for the principal populations of *Ribes multiflorum* subsp. *sandalioticum* of Supramontes and Gennargentu mountains (Fenu et al. 2012), for the largest population of *Gentiana lutea* L. subsp. *lutea* and, more recently, for the most threatened population of *Cynoglossum barbaricinum* Arrigoni *et* Selvi. Finally, all known populations of the exclusive endemic *Silene ichnusae* Brullo, De Marco *et* De Marco f., *Rhamnus persicifolia* and *Centranthus amazonum* Fridl. *et* A. Raynal were multiplied and cultivated in small nurseries located in mountainous areas near to the wild populations.

Although the active actions in the field are the best way to conserve natural plant populations, very little has been done compared to what is necessary to prevent the risk of extinction of many plant species. Thus, taking into account the limited available economical funds and human resources, the implementation of the active conservation measures shold be the first purpose at regional level.

9. Towards the 2020, where and how much efforts we need?

For both key stone plant species groups, the most distant activity to be reached by 2020 was the *in situ* conservation (AUT = 0.23 and 0.37, for policy and exclusive *taxa*, respectively; Fig. 4c, 4f) while the *ex situ* conservation of the exclusive Sardinian endemics (AUT = 0.12; Fig. 4e) and the conservation assessment of policy plant species resulted the most achievable activities; the latter case of the conservation status assessment for the policy taxa has been already achieved (AUT = -0.29; Fig. 4a).

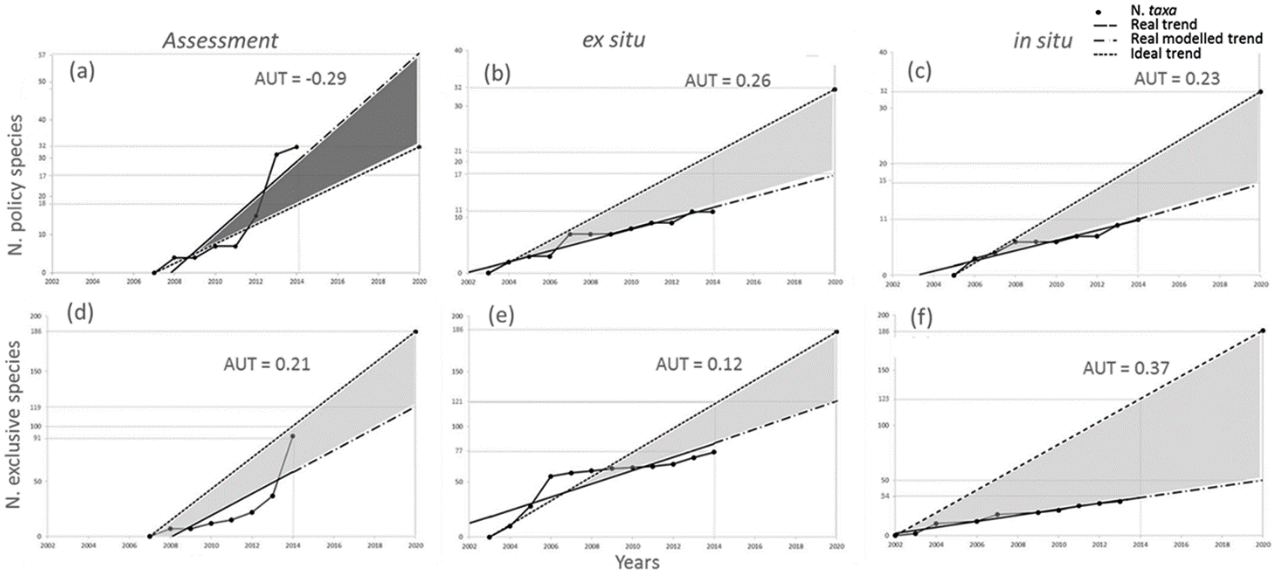

Figure 4. Modelled linear trends of the real (continue line) and ideal (dashed line) efforts in order to achieve the conservation of all Sardinian policy species (a-c) and exclusive Sardinian endemics (d-f). Values on the vertical exes (No. policy species/exclusive species) are referred to the ones of both ideal and real preserved *taxa* by the years 2014 and 2020. The AUT values are both represented numerically and graphically (the area in light grey when A > *a* and dark grey when *a* > A). Graphs are referred to the status assessment (a), *ex situ* (b), and *in situ* (c) conservation activities.

Following the PSR distribution pattern (Fig. 3a), the conservation activities were mainly intensified along the coast and in some areas of the Gennargenteo and Sulcitano-Iglesiente biogeographical sectors (Fig. 5a). Otherwise, a high GAP value depicts the necessity to intensify the efforts in the inland Barbaricino and Supramontano sectors (Fig. 5b). Because of their logistic difficulties and the high economic costs (Fenu et al. 2015), also marine, capes with military installations and insular contexts were not adequately protected up to now (Fig. 5b).

On account of their high number, the achievement of all target purposes will be generally more difficult for the exclusive species. Thus, the high GAP values were consistent with both ESR (Fig. 3b) and intensity of the activities (Fig. 5c). As discussed by Fenu et al. (2015) and also confirmed by the GAP analysis for the policy species, the increasing transportation costs represented a further impediment which biased the conservation activities through the easier to reach areas (Fig. 5d).

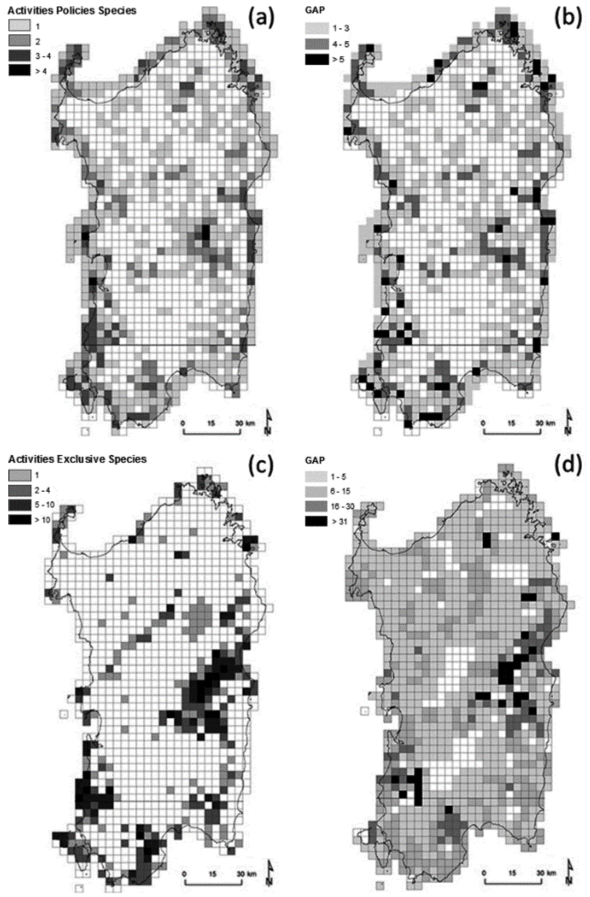

Figure 5. Distribution and concentrations of the activity carried out for the Sardinian policy species (a) and exclusive plant species (b). Following the equation 2, also the distribution of GAPs values of exclusive (c) and policies plant species (d) are reported

10. Conclusions

Due to the impellent necessity to halt biodiversity loss and ecosystem degradation (Butchart et al. 2010; Marques et al. 2014), we consider the 20 ABTs as an important step forward from the generic 2010 target of "achieving a significant reduction of the current rate of biodiversity loss". However, tackling 20 targets simultaneously may represent an extraordinary and ambitious burden for some Countries. In particular, also the achievement of the single ABT 12 at regional level could be ambitious: in key stone species-rich areas such as Sardinia, the achievement of the target requires special efforts which conflict with the usually limited amount of available financial and human resources.

A trade-off solution is to involve volunteers in conservation activities; for example, Schmeller et al. (2009) demonstrated that the volunteers involvement is a good solution and extremely important and valuable for biodiversity monitoring because data from participatory monitoring networks are not less useful, and may be more informative, than those collected in professional schemes. Nevertheless, although volunteers can help in some conservation activities (Fenu et al. 2015), it is unreasonable to plan all the conservation efforts exclusively relying upon volunteers. In fact, a successful conservation strategy should also involve the participation of Public Institutions (at all levels), research centres and also volunteers.

Despite this, some activities could be completed by 2020. In particular, the more optimistic perspectives are deserved for the status assessment and the *ex situ* seed storage (overcoming the current technical difficulties). Conversely, despite the importance of the *in situ* measures (plant population monitoring and active conservation actions), their full application remain far from being achieved by 2020. These activities are a straightforward instance of how good purposes seldom bump into the reality; for example, the reasons which drive the choice of the optimal location are seldom influenced by external causes (e.g. private areas, local interests) which sometimes even reverse the decision from the site with the optimal ecological conditions. In addition, the lack of a specific law on the flora conservation in Sardinia makes difficult the implementation of conservation programs.

In general, although global-scale conservation planning can serve as a guideline for conservation actions, national- and regional-scale planning are both necessary and useful. We thus argue that the ten-year Aichi framework for action by all Countries to save biodiversity, and in particular ABT 12, should be necessarily adapted to the specificities of the territories (e.g. hotspot areas); moreover it should be supported by adequate economic resources, an optimal strategy and a clear support in policy choices.

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CHAPTER II

Disentangling the influence of topography, climate, and human factors on the endemic vascular plant richness at regional scale: the cryptic and peculiar distribution pattern of narrow species

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This research is currently under review

Contribution to the project: I leaded the analyses and preparation of manuscript. I collected the available data about the distribution of species that was revised and implemented by G. Fenu and G. Bacchetta. All the authors contributed for the study design and revision of all parts of the manuscript.

Abstract

Due to the impelling urgency of plant conservation and the increase of available spatially interpolated data (e.g. climate variables) and categorical data (e.g. land cover and vegetation type) at high resolution, many papers have recently dealt with relationships among plant species distributions and a diversified set of factors; nevertheless, global and regional patterns of endemic plant richness remain in many cases unexplained. The richness of 294 endemic vascular plant taxa in the environmentally very heterogeneous island of Sardinia (W-Mediterranean Basin) was recorded inside 36,235 grid cells with a resolution of 1-km resolution grids. Sixteen predictors, including topographic, climatic and anthropogenic aspects, were considered. Generalized Linear Models were performed in order to evaluate the deviance explained by each model and the relative importance of each variable. Elevation contribution was the most explicative parameter, while the influence of climate and humans were less significant. Additionally, comparisons of endemic flora against the relative elevational frequencies highlighted that endemic plant richness was related to infrequent (i.e. peculiar) environmental conditions. This work represented a necessary step further to analyse which and how drivers influence the richness distribution pattern of an interesting group of plant species in Sardinia. The importance of topography, as well as the environmental specificity, were here underlined, but further species specific and more local studies should be developed in order to determine additional explanations and a more effective conservation management on the cryptic vascular endemic species.

Keywords: Species richness; Environmental drivers; Habitat specificity; Island flora; Mediterranean Basin; Sardinia.

1. Introduction

The fascinating question of how plant diversity is distributed on Earth has inspired many biogeographers and ecologists. Due to the impelling urgency of plant conservation, and yet the increase of available data at high resolution, many researches have recently dealt with relationships among plant species richness and several anthropogenic and environmental factors, emphasising that distribution pattern was the result of interactions among several aspects which principally comprise topography, climate and anthropogenic factors (e.g. Araújo 2003; Fløjgaard et al. 2011; Trigas et al. 2013). Nevertheless, global and regional patterns of plant richness are still in many cases unresolved.

Since narrow endemic plants are frequently threatened, they constitute a pivotal group for conservation (Laffan and Crisp 2003; Orme et al. 2005; Ohlemüller et al. 2008). Despite their importance, the ecological investigation on these plants highlighted controversial results, mainly due to their high degree of habitat specialization; as a consequence, in-depth studies on endemic species at very fine scales are particularly appropriate (Pausas and Austin 2001; Harrison et al. 2008; Caperta et al. 2014). Most of researches on this issue were carried out on islands (e.g. Kallimanis et al. 2010; Trigas et al. 2012; Irl et al. 2015), having been considered global centers of plant endemism richness (Kier et al. 2009). Apart from the area *per se*, in continental and/or larger islands, plant diversity was analysed at increasingly fine scales, founding further influential factors. Generally, the elevation gradient and habitat diversity were the most important drivers (Trigas et al. 2013; Cañadas et al. 2014), however, it is also true that human beings are nowadays considered one of the most novel forces in the evolution of life (Concepción et al. 2015).

The five Mediterranean-climate regions have been one of the main subjects of studies about endemic plant richness (e.g. Thompson 2005; Vogiatzakis et al. 2008; Ackerly 2009). Reasons are related to their high climatic and edaphic diversity, which led this biome to be one of the richest in geographically restricted plant species (Médail and Quézel 1997; Harrison et al. 2008; Ackerly 2009). In particular, the diversification of several endemic plants across the islands of the Mediterranean Basin was substantially originated via processes of land migration/vicariance driven by connections/disconnections between micro-plates (Mansion et al. 2008). Such colonization/expansion events, followed by successive fragmentation episodes, were also associated with the aridification of the climate starting from the last glaciations (Suc 1984). Thus, the diversification of the Mediterranean flora, besides the geographical isolation, was also strengthened by progressive climatic modifications related to the onset of the Mediterranean climatic regime during the Pliocene (ca. 3.2 kya; Suc 1984). These processes explained the current pattern of endemic taxa that are particularly concentrated in stressing habitats, often characterized by a low interspecific competition (e.g. psammophilous and halophytic places and peaks of mountains; Cogoni et al. 2012; Caperta et al. 2014; Gentili et al. 2015). In addition, endemic plants in the Mediterranean Basin have historically been subjected to intense disturbances, such as wars, deforestations (Ajibilou et al. 2006), fires (Céspedes et al. 2014), human trampling and overgrazing (Pisanu et al. 2012; Fenu et al. 2013, 2015). Apart from the ecological implications, the definition of species distribution and ecological niches is thus also a needful tool for conservation planning which, according to the regional responsibility criterion (Bacchetta et al. 2012), should be principally concerned at local scale (Bacchetta et al. 2012; Diekmann et al. 2015; Rossi et al. 2015).

This research was focused on the island of Sardinia that could be considered a representative case of study of endemic plants, being the second largest island of the Mediterranean Basin. In particular, we present a regional-scale analysis on the Endemic Vascular Plant Richness (EVPR) inside a 1-kilometer grid covering all the surface of Sardinia. Our main aim was to investigate how differently climate, topography, geology and human influence differently contributed to explain the cryptic distribution of the EVPR and to highlight difficulties and perspectives that should be considered when concrete conservation activities on endemic plants are developed.

2. Materials and Methods

2.1 Study area

Sardinia (Italy) and its ca. 399 satellite minor islands are located in the central part of the Western Mediterranean and cover a surface area of c. 24,090 km^2. In the Mediterranean biogeographic region, it is particularly related to Corsica and the Tuscan Archipelago that all together constitute an independent biogeographical province (Fenu et al. 2014). The island is mainly mountainous (Fig. 1a) with several isolated groups of mountains or massifs such as Limbara, Sette Fratelli, Monti del Sulcis and Gennargentu, the highest of all at 1,834 m, but also with hilly lands, plateaus and a few plains; coast is marked by a variety of landscapes, such as cliffs, sandy dunes and beaches. Substrata and related environments are very heterogeneus and mainly composed of metamorphites and batholiths of the Palaeozoic Era, a volcano-sedimentary lithostratigraphic complex related to a Mesozoic marine transgression and more recent marine and volcanic depositions related to the opening Tyrrenian Basin (Carmignani et al. 2016; Fig. 1b).

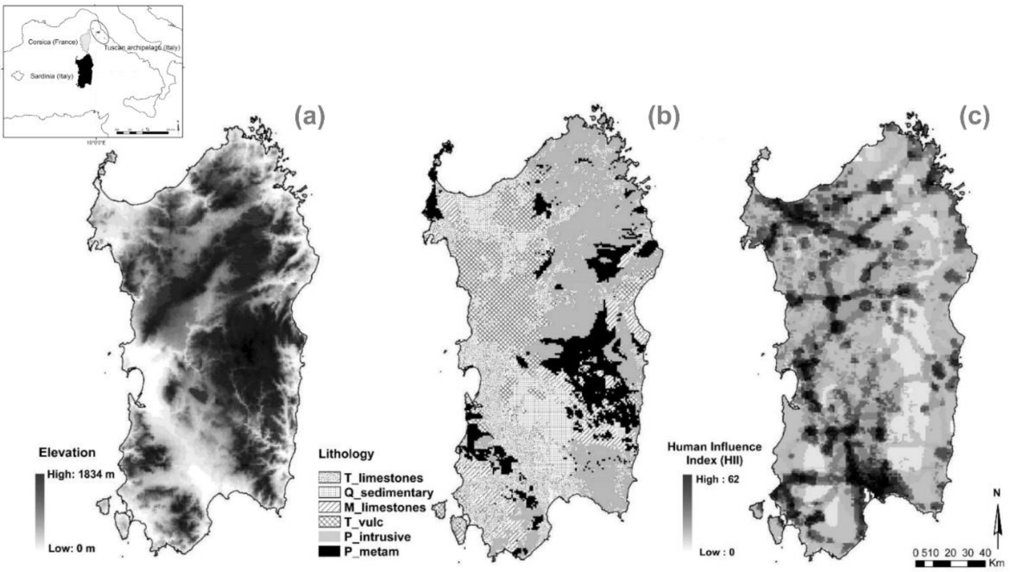

Figure 1. Maps on the spatial distribution of (a) elevation, (b) the simplified lithology subdivided into six categories: Quaternary sedimentary outcrops (Q_sedimentary), Tertiary limestone outcrops (T_limestones), Tertiary volcanic outcrops (T_vulc), Mesozoic

limestone outcrops (M_limestones), Paleozoic metamorphic outcrops (P_meta) and Paleozoic intrusive outcrops (P_intrusive) and the (c) Human Influence Index (HII; WCS and CIESIN, 2005).

Bioclimatically, two macrobioclimates (Mediterranean pluviseasonal oceanic and Temperate oceanic), four classes of continentality (from weak semihyperoceanic to weak subcontinental), eight thermotypic horizons (from lower thermomediterranean to upper supratemperate) and seven ombrothermic horizons (from lower dry to lower hyperhumid) have been identified (Bacchetta et al. 2009; Canu et al. 2015). A long presence of humans on the island (since the Lower Palaeolithic; Vogiatzakis et al. 2008) has been pivotal in shaping the current landscape. In recent decades, the inland mountain villages have continued to lose population, while the largest towns have expanded due to economic development. Lowland plains and coastal zones have also grown rapidly due to agricultural and touristic development (Fig. 1c). This is a common trend in Mediterranean islands, which has caused relevant changes in their landscapes (Vogiatzakis et al. 2008).

2.2 Floristic data

From the 2494 taxa present in Sardinia (Conti et al. 2007), the checklist of the Sardinian Endemic Vascular Plants (hereafter, EVP) was based on the updating list in Fenu et al. (2014) by selecting the 294 endemic taxa (total EVP), which were in turn sub-divided into exclusive to Sardinia (187 taxa; exclusive EVP), and plant taxa also present in Corsica and the Tuscan Archipelago (107 taxa; spread EVP).

The geodatabase of all EVP database was assembled from information obtained from literature, *Herbaria* collections (CAG, CAT, FI, RO, SASSA, SS, TO) and unpublished field survey records of the authors. We obtained 60,301 occurrence records that were carefully revised in order to avoid the potential large errors due, for example, to the approximation of the collection locations and the inclusion of extinct localities (Feeley and Silman 2010). Subsequently, from the 60,301 EVP occurrence data, we built a species richness matrix in a 1-kilometer grid cell for all the Sardinian territory in order to constitute our three response variables: (1) the Endemic Vascular Plant Richness of exclusive EVP (hereafter, exclusive EVPR), the Endemic Vascular Plant Richness of (2) spread and (3) total EVP (hereafter, spread and total EVPR, respectively). From an overall of 36,235 cells, we reduced our analyses to 2466, 34,375 and 34,603 grid cells with a minimum of one exclusive, spread and total endemic EVP respectively.

2.3 Explanatory variables

All explanatory variables used for this study were derived from high-resolution free datasets. A total of 16 predictors were subdivided into three groups: topography and geology (five variables), climate (six variables), and human influence (five variables).

Topography and geology

We used two variables (elevation and slope) strictly associated with topography and three further variables related to geology: number of geological units, number of land units and lithology. Elevation and slope were computed by averaging values of a 10 m Digital Terrain Model (DTM; available from the institutional Sardinian geoportal, http://www.sardegnageoportale.it) while the number of geological and land units per cell were, respectively, accounted from the 1:25,000 map of geology (available from the same institutional Sardinian geoportal) and from the land units map of Italy (kindly provided in raster format by the authors; Smiraglia et al. 2013). Lithology was elaborated simplifying the same 1:25,000 map of geology into six categories: (1) Quaternary sedimentary outcrops, (2) Tertiary limestone outcrops, (3) Tertiary volcanic outcrops, (4) Mesozoic limestone outcrops, (5) Paleozoic metamorphic outcrops and (6) Paleozoic intrusive outcrops.

Climate

Six bioclimatic variables from the WorldClim database version 1.4 (years 1950–2000; Hijmans et al. 2005) with a spatial resolution of 30 arc second (~1 km^2) were used. They included information about both temperature and precipitation. In particular, we selected the following variables: annual mean temperature (Bio1), minimum temperature of the coldest month (Bio6), annual range of temperature (Bio7), annual precipitation (Bio12), precipitation seasonality (Bio15) and precipitation of driest quarter (Bio17).

Human influence

We used five variables related to the human influence. Besides one variable (Human Influence Index), all the rest of information to define variables was downloaded from the institutional Sardinian geoportal (http://www.sardegnageoportale.it). Specifically, the human influence variables were: (1) Roads, calculated summing the kilometers of roads per grid (from the shapefile of the road network). (2) Number of buildings, calculated from a shape point file obtained by extrapolating local landscape maps. (3) Fires, an index (i) computed from the shapefiles of the burned areas (2005-2013) and taking into account which grid were never influenced by fire ($i = 0$) or one up to nine times ($1 \leq i \leq 9$). (4) Land use ratio was computed from the CORINE land use map and represents the proportion of the area covered by units belonging to the 1-2 Land Use first levels (i.e. anthropogenic uses) against the total surface. High Land use ratio values (i.e. near to one) were accounted for highly anthropogenic areas, while lower values were assigned for areas that were more natural. (5) Human Influence Index (HII) was obtained from WCS and CIESIN (2005), a free worldwide dataset of 1-kilometer grid cells created from nine global data layers covering human population pressure (population density), human land use and infrastructure (built-up areas, nighttime lights, land use/land cover), and human access (coastlines, roads, railroads, navigable rivers).

2.4 Statistical analyses

Methods to perform variable reduction in order to avoid collinearity were carried out following Irl et al. (2015). First, linear relationships between response and explanatory variables were assessed by bivariate correlations; we used polyserial correlations (‘polycor’ R package; Fox 2010) which enabled to include also categorical variables. Explanatory variables with correlations $-0.1 \leq r \leq 0.1$ were excluded due to weak explanatory power (Fløjgaard et al. 2011). In a second step, collinearity was addressed by testing correlations for each possible pair of explanatory variables. If |r| > 0.7, the explanatory variable performing poorer with the response variable was excluded. This resulted in a reduction of used variables reported in Table 1.

Variance partitioning for Generalized Linear Models (GLMs) was implemented to assess the overall importance of climate, topography human influence following Legendre (2008). This approach quantifies the independent and/or joint explanatory power of different groups of explanatory variables by estimating the proportion of variation between a focal set of explanatory variables and the residuals of a regression of the dependent variable and all other explanatory variables (Irl et al. 2015). This procedure allows reducing overestimations by avoiding to sum up to the pure deviance explained all the mixed variation attributable to space and shared among predictors (Legendre 2008; Fløjgaard et al. 2011). Accordingly, the independent and total contribution of each set of variables (i.e. topography, climate or human influence) was based on the amount of deviance accounted for (D^2; Guisan and Zimmermann 2000) and computed by the *Dsquared* function in the 'modEvA' package for R (Barbosa et al. 2014).

In addition, the relative contribution of each response variable was calculated using bootstrapping (n = 1000) with all possible model combinations and using the method *lmg* implemented by the *booteval.relimp* function in the 'relaimpo' R package (Grömping, 2006). This procedure have been proposed as a promising method to decompose the variance of final models among different predictors and interactions (Millington and Perry 2011; Irl et al. 2015). As a result, the explained importance and the relative percentage contribution of each explanatory variable was given.

Partial linear regressions are generally used to quantify by adjusted R^2 (Weisberg 1980) the independent and/or joint explanatory power of different groups of variables (Legendre 2008; Tanaka and Sato 2014; Irl et al. 2015). For GLMs, the adjusted R^2 equivalent is measured by the amount of deviance accounted for (D^2; Guisan and Zimmermann, 2000). The equivalence between adjusted R^2 (proportioned by 'relaimpo' R package) and D^2 values was confirmed comparing the results of *Dsquared* function in the 'modEvA' R package (Barbosa et al. 2014) with the ones proportioned by 'relaimpo'.

Once found, relationships between the most significant factors (with the greatest percentage of relative importance) and EVPR was visually inspected subdividing the predictor into equal intervals and contrasting their frequencies with the boxplot of the EVPR inside the respective environmental ranges.

3. Results

After excluding collinear and weak explanatory predictors, residual variables with significant relationships were nine for total EVPR, ten for spread EVPR and four for exclusive EVPR (Table 1). All predictors related to human influence, as well as the number of geological and land units, and precipitation seasonality (Bio15), revealed a negative correlation with all groups of EVPR. Contrarily, EVPR increased with elevation, slope and annual ranges of temperatures (Bio7). Nonetheless, most of variance still unexplained (Fig. 2); this was more evident for the exclusive EVPR (88.2%) than for total and spread ones (71.1% and 67.3%, respectively).

Table 1. Bivariate correlation between total, spread and exclusive Endemic Vascular Plant Species Richness (EVPR), and the explanatory variables, as well as between each pair of explanatory variables. Values are given as the correlation coefficient r resulting from polyserial test. Values of variables with explanatory power (r > 0.1) are highlighted in bold and were used for further analyses. Each variable was assigned to three general categories: Human influence (H), Topography and geology (T) and Climate (C). In this table are reported only the variables which were not excluded for high collinearity (HII, Human Influence Index; Fires, index of fires occurred among the years 2005-2013; LU_ratio, 1-2 Land Use first levels (i.e. anthropogenic uses) and the total surface; Roads, kilometres of roads per grid; N_Geol, number of geological units; N_ Land, number of land units; Elev, elevation; Bio7, annual range of temperature; Bio15, precipitation seasonality.

		EVPR					LU		N	N				
		Tot	Spread	Excl	HII	Fires	ratio	Roads	Geol	Land	Elev	Slope	Bio7	Bio15
HII	H	**-0.22**	**-0.24**	**-0.11**	1.00									
Fires	H	-0.09	**-0.10**	-0.05	0.28	1.00								
LU_ratio	H	**-0.29**	**-0.31**	**-0.16**	0.53	0.33	1.00							
Roads	H	**-0.11**	**-0.13**	-0.04	0.30	0.29	0.39	1.00						
N_Geol	T	**-0.12**	**-0.13**	-0.08	0.09	0.18	0.22	0.37	1.00					
N_Land	T	**-0.16**	**-0.18**	-0.08	0.24	0.08	0.30	0.08	-0.02	1.00				
Elev	T	**0.54**	**0.58**	**0.30**	-0.36	-0.18	-0.50	-0.21	-0.14	-0.24	1.00			
Slope	T	**0.24**	**0.27**	0.10	-0.47	-0.26	-0.63	-0.28	-0.03	-0.31	0.39	1.00		
Bio7	C	**0.14**	**0.17**	0.04	0.06	0.11	0.06	-0.03	-0.09	0.25	0.41	-0.07	1.00	
Bio15	C	**-0.27**	**-0.30**	**-0.11**	0.29	0.09	0.40	0.13	0.03	0.60	-0.50	-0.42	-0.04	1.00

Inside the variance explained, contribution of topography was in all cases the most explicative variable and, specifically, elevation alone accounted always more than the sum of all other variables (Fig. 2).

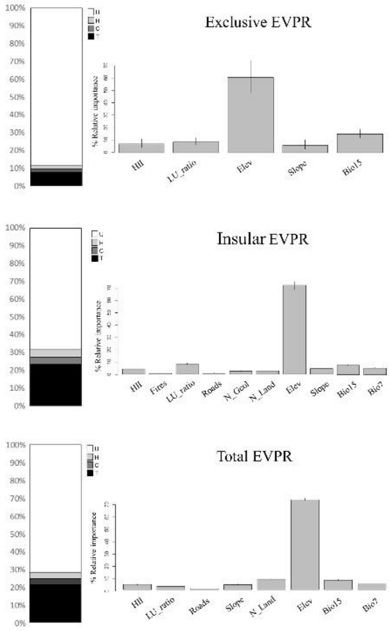

Figure 2. Variation partitioning based on the GLM results for total EVPR, spread EVPR and exclusive EVPR. The Unexplained variation (U), and the explanatory power (in terms of adjusted R2) of each group of explanatory variables [Human influence (H), Climate (C), Topography and geology (T)] are on the right. Figures on the left display the relative importance of each explanatory variable calculated as the normalized per cent contribution to the adjusted R2 for the respective response variable.

Comparisons among exclusive, spread and total EVPR against the relative elevation frequencies stressed that all EVPR were related to infrequent (i.e. peculiar) environmental conditions (Fig. 3). The pattern of such peculiar endemic rich areas were very similar among the three analysed groups (exclusive, spread and total); in particular, the Gennargentu massif (Center of Sardinia) was likely to be the richest area in endemic taxa. Otherwise, further small mountainous areas were depicted (e.g. Limbara and Sulcitan Mountains in northern and southwestern parts, respectively) and some coastal spots, especially in the southwestern and northeastern parts.

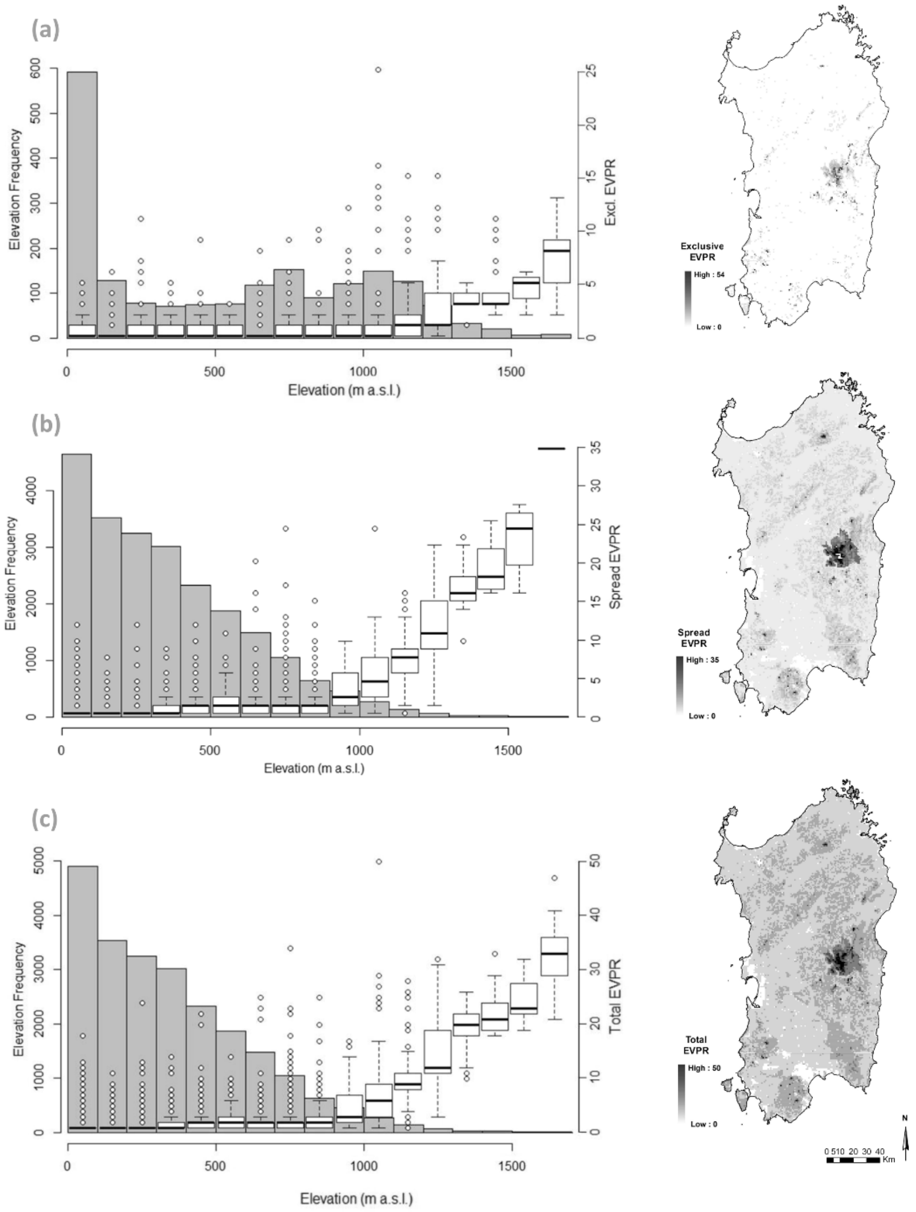

Figure 3. EVPR distribution pattern and boxplots of exclusive (a; N = 2466), spread (b; N = 34375) and total EVPR (c; N = 34603) against the frequencies histograms of elevation.

4. Discussion

As previously found for other Mediterranean continental islands (e.g. Thompson 2005; Trigas et al. 2013), and also for Sardinia (Cañadas et al. 2014), the elevation was the most important factor explaining EVPR. In particular, the highest numbers of EVP was found at highest elevations. These results support authors who previously highlighted the mountainous Gennargenteo and Supramontano biogeographic sectors as important areas for the conservation of plant diversity (Fenu et al. 2010; Cañadas et al. 2014; Fenu et al. 2014) and of other organisms, such as butterflies and amphibians (Grill et al. 2002; Lecis and Norris 2004). Species composition and richness of these mountainous areas were also related to the ancient traditional land use of ecosystems (Vogiatzakis et al. 2008), characterised by the exploitation of lowlands, leaving the higher slopes for cattle raising and thus conserving their more natural state. According to our results, also previous studies on narrow endemic plant species in the Mediterranean context found a richness increase until the maximum elevation (e.g. Trigas et al. 2012; Cañadas et al. 2014); differently, other researchers reported an increase in the endemic plant species richness at intermediate altitudes in islands with mountain systems reaching higher elevations (e.g. Trigas et al. 2013; Birnbaum et al. 2015). Such mid-domain trend is thus likely to be less evident in case, like in Sardinia, of absence of a three-line limit above whom usually occur a hump in species richness. In addition, mountainous areas in Sardinia are characterised by a complex paleogeography (these areas have long been a landmass) and by the absence of quaternary glacial perturbations which led to define these areas as southern putative refugia (*sensu* Tzedakis et al. 2002). Assuming that speciation through adaptive radiation is faster than random drift (Steinbauer 2013), elevational trend in Sardinian endemic richness can be interpreted by an increase of speciation rate with an increasing environmental specificity (Stuessy et al. 2006). The negative relationship between EVPR and the precipitation seasonality (Bio15) and the positive one with the annual range of temperatures (Bio7) suggest a possible correlation between these variables and elevation, being this trend very common in other areas (Körner 2007) and having, in our case, a relative high collinearity. On the other hand, the precipitation and temperatures are *per se* crucial factors in plant species richness (e.g. Harrison et al. 2008; Cañadas et al. 2014) and evolution (e.g. Mansion et al. 2008; Gentili et al. 2015), and their importance have been also underlined from a conservational perspective, especially related to climate changes (e.g. Ohlemüller et al. 2008; Vásquez-Morales et al. 2014). Despite it, our results did not determine their influence and we argue that only specific (and maybe expensive) *in situ* and *ex situ* empirical researches could allow to substantially improve the knowledge on these topics.

Although it is a widespread idea that humans acted as major extinction filters (e.g. Balmford 1996; Araújo 2003; Rossi et al. 2015), we found a weak relationship between EVPR and human influence. These results could be ostensibly interpreted as a lack of human threats; however, further considerations should be weighted. Firstly, analyses of human presence as a determining factor of current Mediterranean landscape and biodiversity patterns have faced several shortcomings principally related to difficulties in accurately evaluating consequences of such long-term presence and several indirect factors triggered by it. Furthermore, if present-day biodiversity should be biased toward species that are generally more tolerant of humans (Araújo 2003), it would be unsurprising to find a correlation between people and extant biodiversity because species that were intolerant of humans are either extinct, or persist at low abundances in the most undisturbed areas (Araújo 2003). Additionally, several authors remind that the high elevations in the Mediterranean context usually imply a low human presence (e.g. Steinbauer et al. 2013; Fenu et al. 2014). Accordingly, a deeper attention on the current negligible factors related to human influence have to be paid in order not to underrate them. In general, the high rate of unexplained variance could also be due to an insufficient quality and quantity of data; however, in order to dispel doubts on data quality, we underline that such difficulties were also found for endemic plant species in other similar insular contexts (e.g. Trigas et al. 2012; Irl et al. 2015). Thus, we could assume that such cryptic distribution pattern was something intrinsic to the endemic species and just maybe a change to finer-scale studies could contribute to find stronger relationships among predictors and EVPR, especially of exclusive EVP. According to their lower relative importance, this is more necessary for climate and human factors rather than for topographic ones that were in this case extrapolated from data at a higher resolution (from DTM with 10 m grid size).

Comparisons among exclusive, spread and total EVPR against the relative elevation frequencies produced a further demonstration of the strictly circumscribed ecological niche of many endemic taxa. Although this concept has been generally accepted (e.g. Thompson 2005; Harrison 2008; Fenu et al. 2010), these sort of studies are not frequently found in the literature with local and concrete examples (Caperta et al. 2014).

Concluding remarks

Similarly to the definition of macroecology explained by Brown (1995), this study on the distribution of endemic plants could be also conceived as a step away from the objects which facilitates the recognition of general patterns. On the other hand, this conjecture also indirectly assumes that even a step further is possible to gain different and more specific understandings. In spite of concealing the difficulties we had encountered, we tried to learn about them and we gained useful outcomes that could be the basis for future studies. Without underrating the importance of results at global and, as in this case, at regional scales that provide a general-picture perspective, we argue that finer analyses (e.g. at biogeographical subsectors and biodiversity nano-hotspots scales), such as ecological analyses and identification of cost effective conservation areas, should be recommendable in some studies on extremely narrow taxa.

According to the idea that all researches on threatened and/or uncommon species distribution patterns along environmental gradients have the ultimate goal of conservation, we emphasise the importance of the following points:

1. Spatial scales should be also related to the distributional range of the study species. Indeed, although studies in Sardinia are usually conceived at ‘regional scale’, this research on endemic species, which often occur only in few 1-km grid cells, could be otherwise defined at ‘macro-scale’.
2. According to the first point, our results are more similar to global scale results (that provide a general-picture perspective) rather than results at local scale (more useful for concrete conservation activities).

3. Although we underline the relevant continuous improvement in the availability and quality of human-related and environmental data, we argue that for some specific issues, such as the conservation planning of narrow species, the empirical researches are the best solution, despite of their costs.
4. According to previous authors (e.g. Bacchetta et al. 2012; Diekmann et al. 2015; Rossi et al. 2015), political decisions on conservation planning should be taken at regional level. However, target species have in most cases such a restricted distribution that very detailed studies, involving local researchers, authorities and stakeholders, are necessary for an effective conservation management.

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CHAPTER III

Using extinctions in species distribution models to evaluate and predict threats: a contribution to the plant conservation planning in the Island of Sardinia

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Abstract

Recent extinction rates suggest that humans are now causing the sixth mass extinction, and the Mediterranean islands are at the forefront of many of the environmental issues involved. This study provides an alternative approach for investigating documented local plant extinctions that occurred in Sardinia (W-Mediterranean Basin) during the last half century. A total of 190 local extinctions of 62 plant species were used to investigate the independent effects of eight ecological and anthropogenic variables and to model the areas of potential extinctions where plant conservation efforts could be focused. In most of cases, both anthropogenic and environmental factors explained local extinctions. Furthermore, areas potentially rich in species with conservation interest, but which are particularly anthropized, were highlighted by our analysis. Building upon this practical case study in Sardinia, this paper suggests a reproducible, operational framework to analyse which extinction factors may play an important role in similar contexts and where they might occur.

Keywords: Drivers of extinctions; Generalized Linear Models; Human influence; Mediterranean island flora; Random Forest

1. Introduction

Although extinction is a recurrent evolutionary phenomenon, it does not proceed at the same pace at all times (Nakamura et al. 2013). Whilst a relatively low number of species usually become extinct during any given time span (background extinctions), there are periods during which a large proportion of biota is exterminated in a very short period in a geological timescale (mass extinctions; Nakamura et al. 2013). In terrestrial groups, estimates of recent extinction rates are between 100 and 1000 times greater than the long- term global average derived from geological records (May et al. 1995). In addition, numbers of documented extinction are likely to be serious underestimates because most species are still unknown (Joppa et al. 2010; Barnosky et al. 2011). Several researchers (e.g. May et al. 1995; Joppa et al. 2010; Nakamura et al. 2013) thus suggest that humans are now causing the sixth mass extinction through co-opting resources, fragmenting habitats, introducing non-native species, polluting, killing species directly and inducing climate change (Barnosky et al. 2011). In this context, there is an increasing need to find innovative tools to improve the effectiveness of biodiversity conservation.

One of the most common approaches in this sense is to model the presence or range of key species using remote data. These methods are widely used for a variety of reasons, including the high availability of remote sensing data and because these can be used to predict how target species may respond to changes in climate or land use (Buckland and Elston 1993). Furthermore, predictions of Species Distribution Models (SDMs) can sometimes reveal additional populations of threatened species (e.g. Alfaro-Saiz et al. 2014; Fois et al. 2015) or guide the management of protected areas or other environments (e.g. Fois et al. 2016; Kaky and Gilbert 2016). Increasingly, there is a need to use environmental data to identify those areas that might be candidate locations for species translocations (e.g. López-Tirado and Hidalgo 2015; Fois et al. 2016). However, despite their usefulness and large applicability, to our knowledge there are no examples of distribution models that use local extinctions in the Mediterranean territories. This is mainly due to a general lack of detailed information about where these extinctions have occurred (Greuter 1994; Domina et al. 2015).

Mediterranean islands provide a fascinating framework for studying the impact of human activity on biodiversity. With about 10,000 islands and islets, 244 of which are inhabited (Pons et al. 2013), the Mediterranean Basin encompasses one of the world's largest archipelagoes (Pons et al. 2013). Some eastern Mediterranean Countries such as Greece and Croatia with ca. 1150 islands and islets (Nikolic et al. 2008; Iliadou et al. 2014), encompass a significant number of these islands; however, the western side includes the largest Mediterranean islands of Sicily, Sardinia and Corsica, as well as around 1100 islets (Pons et al. 2013). For historical and geographical reasons, but also due to the particular biotic interactions among species, the particular Mediterranean insular conditions determine specific plant diversities and assemblages (Pons et al. 2013). Plant endemism in Mediterranean islands often reaches high levels, generally comprising between 10–12% of the total vascular flora (Pons et al. 2013; Fenu et al. 2014). In particular, the plant endemism rate is generally higher in mountain ranges and in satellite uninhabited islets of Sardinia, Corsica and Crete, where endemics represent about 35–40% of the vascular flora (Bocchieri 1998; Iliadou et al. 2014).

Plant diversity in Mediterranean insular territories shares its heritage with several human activities that have had profound consequences — often negative — on plant distribution and dynamics (Lavergne et al. 2005; Pungetti et al. 2008). In the Mediterranean Basin, climatic anomalies (e.g. López-Tirado and Hidalgo 2015; Kaky and Gilbert 2016) and human-related factors, such as human trampling and land use change (e.g. Lavergne et al. 2005; Fenu et al. 2013), have been identified as important drivers of local extinctions or population decreases in narrowly distributed plants; however, several data gaps still exist (Greuter 1994; Domina et al. 2015).

This study focused on local extinctions of vascular plants that occurred since 1960 on the island of Sardinia (W-Mediterranean Basin). An experimental approach was possible due to the unusual long-term, documented investigations of the island's flora. Indeed, many regional floras were published when environmental conditions were different; this has allowed authors to already discuss the local extinctions of specific areas such as the small satellite islets (Bocchieri 1998) and the north-western part of Sardinia (Bagella and Urbani 2006). The main aims of this study were: (1) to identify extinction locations of plant species of concern on Sardinia; (2) to investigate how important each considered variable was in determining local extinctions by measuring the relative influence of anthropogenic factors in relation to ecological constraints; (3) to explore the extinction pattern and to identify, by a novel application of SDMs, areas where plant extinctions may potentially occur. The utility of this approach was tested for localizing and mapping areas where anthropogenic and ecological drivers of local extinctions were most influential and where further investigations on extinction threats could be focused.

2. Methods

2.1 Study area

Sardinia and its ca. 399 satellite small islands cover 24,090 km^2 with a coastline of about 1900 km. The island is characterised by complex orography with plain, hilly and mountainous landscapes on different geological substrates. Due to its geographic position, it has a typically Mediterranean climate with dry and hot summers and relatively rainy and mild winters, with a temperate bioclimate only on the higher summits. These traits, in conjunction with prolonged isolation, are the main factors that promoted the speciation of endemic plants (Cañadas et al. 2014). The subsequent high proportion of endemic taxa (about 13% of the entire flora; Fenu et al. 2014) considerably increases, up to about 35%, on mountain peaks and uninhabited islets (Cañadas et al. 2014).

Sardinia is underpopulated compared to other Italian (and European) regions: it has a demographic density of 66 inhabitants per km^2, compared with an average of 194 persons per km^2 in the whole of Italy (ISTAT 2001). Nonetheless, the long human habitation has been pivotal in shaping the landscape and its plant diversity (Pungetti et al. 2008). Plant extinctions in Sardinia, as well as in the entire Mediterranean region, are bound to have occurred in historical times with the massive development of agriculture and the related significant environmental transformations. In particular, the island has undergone a transformation from the wilderness of its original Mediterranean habitats to an agricultural landscape with wheat fields in the plains, vineyards on the slopes and pastoral land in the highlands (Pungetti et al. 2008). As on many other Mediterranean islands, industrial and seasonal (summertime) and local (coastal) tourist activities have also grown rapidly in recent decades in lowland plains and coastal areas (Pungetti et al. 2008).

2.2 Local extinctions and occurrence data

The study focused on a selected group of vascular plants with local biogeographical and/or conservation interest in Sardinia. In particular, plants of biogeographical interest were those endemic to the Sardo-Corsican biogeographical province (Fenu et al. 2014) and/or those plants that, for their geographical disjunction, are proved to be ecologically and/or genetically isolated. Plants of conservation interest were all those plant species listed, at least, as "Endangered" at regional and/or global level according to the IUCN classification (2016), and/or listed in International protection conventions (see Fenu et al. 2015 for the full list).

Information about both present occurrences and extinction localities was obtained from herbarium collections (CAG, CAT, FI, RO, SASSA, SS, TO) and available literature and was confirmed and implemented by unpublished field survey records of the authors. While the creation of the occurrences dataset consisted of updating datasets used in previous studies (e.g. Fenu et al. 2014; Fois et al. 2015), the extinction localities dataset was constructed *ad hoc* for this research. All reported extinctions of local floras (e.g. Bacchetta 2006; Pisanu et al. 2014), researches on floristic changes (e.g. Bocchieri 1998; Bagella and Urbani 2006) and conservation status assessments (e.g. Fenu et al. 2012; Fois et al. 2016) were taken into account. Further extinctions were directly recorded by the authors during the last 10 years through revisiting localities with reports of threatened plants. Using the framework of specific demographic studies (e.g. Morris and Doak 2002), localities with fewer than 20 reproductive individuals were considered to be sites of extinction and were included in these analyses. All distribution data were recorded within a grid of 1 × 1 km in GIS environment (Quantum GIS Development Team 2014). In the final database, the plants were categorised according to the Raunkiaer's life-form classification system (1934). The subdivisions of the Raunkiaer system are based on the place of the plant's growth-point during seasons with adverse conditions, reflecting the adaptation of plants to surviving unfavourable seasons (cold or dry seasons), and correlated with growth forms: therophytes (annual plant species), hemicryptophytes (perennial forbs and grasses), geophytes (perennial plants with bulbs, corms, or rhizomes), chamaephytes (semi-shrubs) and nanophanerophytes/phanerophytes (shrubs and trees).

Because altitude was one, if not the main, factor related to the distribution of several plant species in Sardinia (e.g. Cañadas et al. 2014; Fois et al. 2015), another subdivision was implemented according to altitudinal range, obtained using extrapolated mean values per 1 km^2 grid-cell: coastal (0–150 m asl), plains and hilly (10–800 m asl), mountainous (> 800 m asl) or widespread (with an altitudinal range larger than 1000 m).

2.3 Ecological and anthropogenic factors

Data used as explanatory variables in the extinction models were subdivided into two main categories: ecological and anthropogenic variables (see Table 1 for a detailed description). The first group included two monthly mean climatic datasets (Bio7 and Bio15) for current conditions (~1950–2000) and three geomorphological variables (Elev, Slope and Lith), extrapolated using a Digital Terrain Model (DTM) and from a simplified geological map of Sardinia (Fenu et al. 2014; Table 1). The anthropogenic factors were compiled from a free worldwide raster dataset called Human Influence Index (HII; WCS and CIESIN 2005), the summarised meters of streets per grid (Street) and the number of fires that occurred over a nine-year period (2005–2013; Fires). All variables were converted in raster format at the same 1 km^2 resolution of species data. Multicollinearity problems were tested by computing Variance Inflation Factors (VIFs; Marquardt 1970), which measure how strongly each predictor can be explained by the rest of the predictors, is based on the square of the multiple correlation coefficient (R^2) resulting from regressing the predictor variable against all other predictor variables (Naimi and Araújo 2016). As a rule of thumb, a VIF > 10 signals that the model has a collinearity problem (Chatterjee and Hadi 2006). We used a stepwise procedure, implemented through the 'sdm' package (Naimi and Araújo 2016) in the R environment (version: 3.1.1; R Development Core Team 2014) to remove all variables with VIFs > 5, which was imposed as a precautionary threshold (Table 1).

Table 1. List of variables subdivided into Ecological (E) or Anthropogenic (A) and their respective source are reported. Problems related to collinearity were avoided removing factors with Variance Inflation Factors (VIFs) > 5.

	E/A	*Information*	*VIF*	*Source*
Bio7	E	Temperature Annual Range	1.38	Hijmans et al. 2005
Bio15	E	Precipitation Seasonality	1.63	See above
Elev	E	Average of values obtained by a 100m DTM	1.97	http://www.sardegnageoportale.it
Slope	E	Average of values obtained by a 100m DTM	1.61	See above
Lith	E	Six classes of geological types adapted from the 1:25,000 map of geology according to the plant-soil requirements: (1) Quaternary sedimentary, (2) Tertiary limestone, (3) Tertiary volcanic, (4) Mesozoic limestone, (5) Paleozoic metamorphic outcrops and (6) Paleozoic intrusive outcrops	1.23	See above
HII	A	Created from nine global data layers covering human population pressure (population density), human land use and infrastructure (built-up areas, night-time lights, land use/land cover), and human access (coastlines, roads, railroads, navigable rivers)	1.42	WCS & CIESIN 2005
Street	A	Computed by a shapefile of streets networks	1.17	http://www.sardegnageoportale.it
Fires	A	Computed by nine shape polygon files (one per year) of burned areas	1.18	See above

2.4 Evaluation of variable importance

The effect of each independent variable was tested by means of Generalized Linear Models (GLMs) using the logistic (0: extant locations; 1: extinct locations) as a link function and the binomial as an error distribution (Carrete et al. 2007). We performed a hierarchical partitioning analysis (package 'hier.part'; Walsh and Mac Nally 2013) in R environment (version: 3.1.1; R Development Core Team 2014) to estimate the independent effect of each factor on determining local extinctions. This process involved computing the increase in the fit (measured as deviance explained) of all models with a particular variable compared with the equivalent model without that variable. In this way, multicollinearity problems that are effectively ignored by using any one-model technique are likely to be alleviated (Mac Nally 2000). The size of the individual effect of each variable (percentage of independent effect) was used as a criterion for ranking and deriving conservation extinction risks. We assumed only variables with a percentage of independent effect > 10% (Mac Nally 2000) as possible cause of extinctions. Analyses were repeated for each taxon and then averaged according to life form and altitudinal ranges classifications. If no variables satisfied this criterion, local extinctions were considered stochastic. After selecting the most important factors driving the extinctions of each taxon, the respective range values of occurrences and extinctions were compared.

2.5 Procedures, evaluation and ensemble of distribution models

The same binary form of extinct (1) and extant (0) records was applied to predict potential areas where local extinctions may occur. Species with only one extinction event and/or less than three occurrence records were excluded from these analyses due to their low reliability (van Proosdij et al. 2015). In addition, only extinction causes highlighted according to their variable importance (independent effect > 10%) were employed in each species-specific model.

Generalized Linear Model (GLM) and Random Forest (RF) presence-absence methods were used to model plant extinctions as the basis of a final mean ensemble method (Araújo and New 2007; Marmion et al. 2009). These techniques, widely used to model species distributions and capable of modelling nonlinear functions (Franklin 2010), were implemented by using the ‘sdm’ package (Naimi and Araújo 2016). This package is an integrated framework that enables multiple modelling techniques to be fitted and compared simultaneously (Naimi and Araújo 2016). Settings implemented by the ‘sdm’ R package were applied by default. In particular, GLM with logistic link function and RF models with 500 trees were used.

For each model, we used 10-fold cross validations to give a more robust estimate of predictive performance. For each cross validation iteration, 70% and 30% of the data were randomly selected for use as training and testing datasets, respectively (Elith et al. 2011). Model performance was determined by calculating the Area Under the Curve of a receiver operating characteristic plot (AUC; Fielding and Bell 1997) and the True Skill Statistic (TSS; Allouche et al. 2006) using the model validation data set. Results were averaged among 10 replicates per modelling technique.

The final extinction model for each species was calculated as the mean value of the outputs of all single runs. This consensus approach has recently been applied in broad-scale conservation studies and is based on the idea that different predictions are copies of possible states of real distributions and their ensemble will result in a more accurate prediction (Marmion et al. 2009). In addition, this allows a comparison of the methods’ predictive ability and a quantification of uncertainties deriving from the choice of the modelling approach (Marmion et al. 2009; Naimi and Araújo 2016). The type of output provided varied in relation to the modelling technique used. The ‘sdm’ package enabled different predictions to be standardised in a continuous index of probability, ranging from 0 (low probability of extinction) to 1 (high probability of extinction). The outputs used for the ensemble were at least satisfactorily (AUC > 0.7 and TSS > 0.3; Heikkinen et al. 2012) predicted by the models.

The final result was thus obtained by merging all species-specific outputs (if satisfactory) with the 'merge' function of raster R package (Hijmans et al. 2015) and plotted in GIS environment (Quantum GIS Development Team 2014) in order to graphically depict the areas of conservation interest.

3. Results

A total of 62 vascular plant species (for 190 extinction events and 2357 occurrence records) were analysed; 39 of these plant species were considered to have both conservation and biogeographical interest, while only 10 and 13 plant species were considered to have only biogeographical or conservation interest, respectively. These 62 plant species included 10 therophytes, 19 hemicryptophytes, nine geophytes, 14 chamaephytes and 10 nanophanerophytes/phanerophytes. Regarding their altitudinal range, the plant species were subdivided into coastal (18 species), plains and hilly (12 species), mountainous (four species) and widespread (14 species).

3.1 Evaluation of variable importance

Both ecological and anthropogenic factors explained the local extinctions of 34 plant species, whereas in only 14 and five cases, respectively, did ecological or anthropogenic factors exclusively explain extinctions. Six cases were assumed to be stochastic.

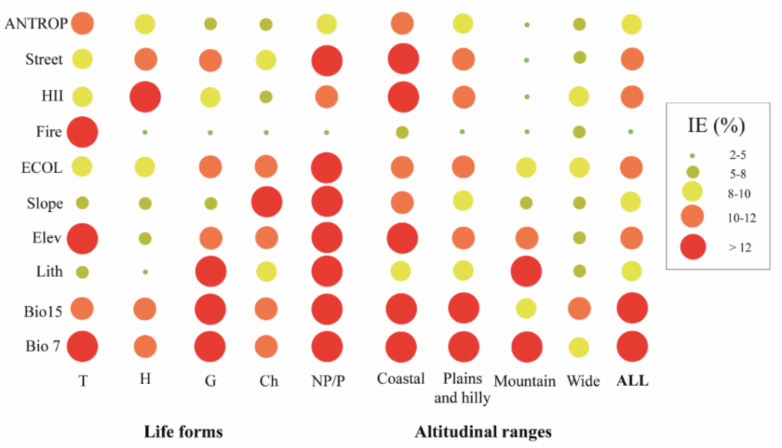

Figure 1 Scatterplots of the percentage of independent effect of each variable (BIO7 = temperature annual range; BIO15 = precipitation seasonality; Lith = lithology; Elev = elevation; Slope = morphological steepness; Fire = number of fires occurred from 2005 to 2013; HII = Human Influence Index; Street = meters of streets per grid; see Table 1 for further details) and the mean of ecological (ECOL) and anthropogenic (ANTROP) groups. Values were the average of each model obtained from the 62 vascular plants analysed. These were considered all together (ALL) and also subdivided according to the life forms proposed by Raunkiaer (1934) and altitudinal ranges.

Ecological factors fairly equally influenced extinctions among life forms and altitudinal ranges. The only exceptions were plants with a wide altitudinal range, the influence of which was generally considered to be stochastic (Fig. 1). Otherwise, the influence of anthropogenic factors was greater for therophytes, chamaephytes and nanophanerophytes/phanerophytes, especially in coastal and plains/hilly localities (Fig. 1). If all analysed plant species were considered together, ecological factors (in particular temperature annual range and precipitation seasonality) explained local extinctions more than anthropogenic factors (Fig. 1).

3.2 Model evaluation and ensemble forecasting

Extinction distribution models were implemented for 101 extinction localities of 32 plant species. Model performances were generally high both in terms of AUC and TSS values. Nonetheless, the ensemble of RF and GLM algorithms were not used in nine cases because one of these two approaches did not satisfy the AUC (> 0.7) and TSS (> 0.3) thresholds; in only two cases did neither algorithm satisfy the criteria.

The final map, obtained by merging the 30 species-specific outputs, highlighted the areas where drivers of plant extinctions should be more influential. The probability of extinction was higher (> 0.4) along the coast and in plains areas (Fig. 2).

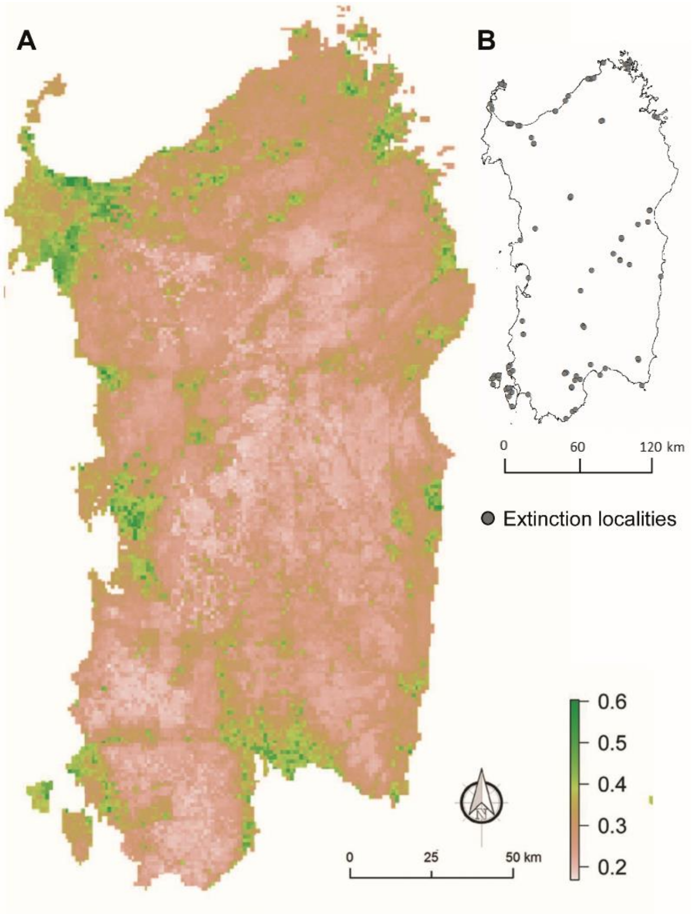

Figure 2. Average map of 101 local extinction cases per 32 singularly modelled plant species (A). Values from 0 to 1 measured the probabilities of extinction in all the Sardinian territory. Localities of extinction used are reported on the right (B).

4. Discussion

Despite the importance of disentangling the causes of recent extinctions, researchers have faced many difficulties in finding correlations between extinction events and anthropogenic/ecological factors. This is mainly due to the unavailability of historical information on plant occurrences and/or extinctions (Greuter 1994) and because extinctions are sometimes caused by stochastic events and/or to the combined effects of many factors (Renton et al. 2014). For these reasons, large-scale researches on extinctions are barely feasible, but similar studies can be carried out only for small and well-delineated areas, such as Mediterranean islands, which are historically well known from a plant diversity and floristic viewpoint. Nonetheless, we argue that the results obtained for our specific case could reflect the extinction patterns of similar contexts, such as other coastal areas of the Mediterranean or other regions with high endemicity combined with long histories of human-induced transformations.

According to other studies in the Mediterranean context (e.g. Lavergne et al. 2005; Fenu et al. 2013), both anthropogenic and environmental variables concurred to explain extinctions, and they also explain the distribution of endangered coastal plant species in Sardinia. Although ecological factors generally explained local extinctions more than anthropogenic factors, the independent effect of each factor considerably varied among life forms and altitude ranges. Such differences confirmed the necessity of supporting general overviews by detailed studies, since species- or habitat-specific results are sometimes in contrast with the general trend.

Regarding life forms, the effect of anthropogenic factors was less strong for geophyte and chamaephyte extinctions than for therophytes, hemicryptophytes and nanophanerophytes/phanerophytes. Geophytes and chamaephytes have previously been recognised as two of the plant forms most resistant to fires, trampling and grazing (Pignatti et al. 2002). Although the degree of sensitivity to anthropogenic factors was difficult to determine in our analyses, the independent effect of each variable implicitly suggests different degrees of sensitivity between life forms. The HII, which is a sum of many anthropogenic factors, was likely to be a finer measure of even low intensity disturbances than the summarised meters of streets per grid (Street), which accounted for a more destructive level of disturbances. This could explain why even a low intensity disturbance, such as the human trampling, was an influential factor for many coastal endangered hemicryptophytes, such as *Anchusa crispa* (Bacchetta et al. 2008) and *Astragalus maritimus* (Bacchetta et al. 2011), while coastal nanophanerophytes and phanerophytes were only influenced by more intense disturbances, often connected to the development of infrastructure (Tzanopoulos et al. 2005). The frequency of fires (Fires) only explained extinction events for therophytes. Although covers of annual plant communities generally increase with human disturbance (Pignatti et al. 2002), Mediterranean endemic and specialist therophytes are likely to have a lower tolerance to competition than endemic and specialist perennial life forms (Imbert et al. 2011; Fenu et al. 2013). This competition is particularly important for annual species in soil seed banks, where burning events often cause a reduction in seed bank diversity, benefiting perennial life forms and widespread and/or pioneer plants (Torres et al. 2012). Hence, we suggest that the fire could have a particular negative influence on less competitive species, such as the endemic and specialist therophytes.

The influence of ecological factors, which are often complementary to and/or consequences of anthropogenic activities (Lavergne et al. 2005; Renton et al. 2014) was similar among species altitudinal ranges, with the exception of wide-ranging species. According to previous research (e.g. Imbert et al. 2011; Renton et al. 2014; Kaky and Gilbert 2016), plants with a wide distribution and ecological range are less prone to suffer from climatic changes than those which occur only in specific environments such as coastal environments and Mediterranean mountains. This discussion regarding species with narrow ecological requirements could also be applied to explain the influence of lithology (Lith), which is especially characteristic in geophytes, including orchids (Djordjević et al. 2014) and many perennial endemics such as *taxa* belonging to the genus *Ribes* in Sardinia (Fenu et al. 2012).

The areas highlighted by averaging all modelled extinction cases were characterised by land use change from semi-natural into urbanised landscapes that occurred in recent decades (Zoppi and Lai 2012). This was mainly a consequence of industrial settlements (e.g. in the north-western and south-western coasts of Porto Torres and Portoscuso, respectively) and increasing tourism development along the rest of the coast. Otherwise, most similar neighbouring environments were considered important areas for plant conservation, owing to their high number of endemic and/or threatened plant species (Fenu et al. 2014). This result aligned with our expectations due to the implicit ecological information contained in the modelled extinction localities of the analysed endemic and endangered species.

The idea of this research was to treat occurrence data in an experimental way. Indeed, the common usage of presence data in SDMs was in this case replaced by extinction occurrences. Therefore, this approach could be defined as an “extinction” (and not “species”) distribution model that underscored “potential threatened areas” instead of “potential niches”. Therefore, the species-specific results usually obtained by environmental modelling could be extended in this case to more generalized results for potential areas of extinctions and thus threats, which refer not only to each singular specific case but to all taxa with a similar pattern and ecology. This potential is strengthened by averaging many cases that occurred in a diversified environment (from coastal to mountainous and from rural to semi-urban areas). To our knowledge, no scholars have used a similar methodological approach; hence, at this stage, comparisons with our results are not possible.

Although correlation does not imply causation, our stepwise procedure of investigating drivers of extinctions has suggested further insights regarding how and where extinctions may occur in Sardinia. Instead of representing the mere pattern of recorded extinctions, our resulting map allowed to be highlighted also those vulnerable areas that are probably not sufficiently investigated until now and where further populations that are extinct or at the brink of extinction could be thus found. In other words, the threatened areas identified by our study should not simply be considered “loser zones”. Conversely, these should be considered areas where much more interesting work, such as ecological analyses and conservation activities, could be focused.

Because part of our analysis used an experimental approach, we have a special interest in sharing the results of this research in order to compare our results with those of different species and/or in other environmental conditions. As information about even recent extinctions is seldom reported in the literature, only further investigations and comparison could enhance the current state of the art about the main reasons for recent extinctions, which are often based on suppositions and underestimations (Joppa et al. 2010; Barnosky et al. 2011). Furthermore, this research provides a general perspective that should be implemented through more focused investigations of each analysed species.

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CHAPTER IV

Global analyses underrate part of the story: finding applicable results for the conservation planning of small Sardinian islets' flora

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Abstract

Although many studies have addressed island biogeography, the biodiversity of very small islets has mostly been ignored due to the lack of high-resolution data. We analysed the vascular plant taxa distribution pattern of 81 satellite islets scattered around Sardinia, the second largest island in the Mediterranean Basin. Power models were used to determine the influence of environmental and anthropogenic factors on both endemic and non-endemic species richness. Analyses of the distribution pattern highlighted the high endemism rate in these islets. In addition, reliable results for conservation, such as correlations with land use diversity, climate and human presence, were found only when categorising the islets by their environmental characteristics instead of their geographical position. This study of a representative set of Mediterranean islets contributes to the knowledge base about the controversial role of very small islets in biological conservation. Large-scale analyses have often underrated their importance and we therefore suggest that local studies about their conservation planning are needed worldwide.

Keywords: Island biogeography; Biodiversity hotspots; Endemic plants; Species richness; Species-area relationships, Western Mediterranean Basin.

1. Introduction

Due to their apparent simplicity and discrete nature, islands have often been considered “natural laboratories” for studies on the genetics, biology, ecology and conservation of species (Greuter 1995; Henderson and Whittaker 2003). If the aim is conservation planning, preliminary studies focused on island species richness are appropriate ways to help maximize the effectiveness of conservation investments (Brooks et al. 2006). According to the theory of island biogeography (MacArthur and Wilson 1967), area is *per se* the most important explicative factor in species richness. Based on a very broad consensus of ecologists (e.g. MacArthur and Wilson 1967; Lomolino and Weiser 2001; Triantis and Sfenthourakis 2012), the basic species–area relationship illustrates that species richness increases with increasing island area but that the rate of increase slows for the larger islands. This relationship is plotted in arithmetic (un-transformed) space but can be linearised using log transformations to facilitate statistical analyses and to help visualise relationships over a broad range in island area (Lomolino and Weiser 2001). Following pioneering studies on island biogeography (Preston 1962; MacArthur and Wilson 1967), the so-called small island effect (SIE) was also depicted and described (Lomolino 2000). This theory highlights the vulnerability of small island species populations that tend to be composed of only a few individuals to stochastic extinctions. This partially explains the decline in the genetic variability of populations on small and isolated islets (Thompson 2005; Mameli et al. 2008; Herradine et al. 2015). Although some impacts on island biodiversity, such as human pressure (e.g. Caujapé-Castells et al. 2010; Fenu et al. 2013), climate change (e.g. Ricklefs and Lovette 1999; Bellard et al. 2014) and land use changes (e.g. Mangiacotti et al. 2013; Proença and Pereira 2013), have been widely documented, predictions about changes in species richness often reveal a degree of uncertainty related to the tendency to under-sample relatively small islands (Greuter 1995, 2001; Lomolino and Weiser 2001). For instance, Weigelt et al. (2013) found no environmental information for 1,509 of the 19,392 islands across the world with a surface area greater than 1 km^2. Regarding the Mediterranean Basin, the vascular plant species diversity of about 430 eastern Adriatic islands was until recently unknown (Nikolić et al. 2008). Otherwise, several studies in this area have disentangled these aspects at fine scale due to the exceptional concentration of endemic and endangered species (Médail and Quézel 1997) in a conspicuous and heterogeneous set of archipelagos and scattered islands. In particular, interesting findings were obtained in analyses of the Eastern Mediterranean islands. For instance, the ancient Greek word “*choros*” was used to define models of species richness improved by the incorporation of habitat diversity (Triantis et al. 2003). The validity of these models was also confirmed by further analyses of the plant diversity of very small Aegean islands (Panitsa et al. 2006). In addition, the influence of other aspects, such as elevation, distance to inhabited islands and the number of geological substrata, have been studied in several other analyses of the biodiversity of Aegean archipelagos (e.g. Kallimanis et al. 2010; Trigas et al. 2013). Other important aspects of island biogeography (e.g. SIE, human pressures and climate change effects) are more critical for conservation planning than the surface characteristics of islands (e.g. area, elevation) and the degree of isolation (e.g. distances from mainland, degree of nestedness), which are more applicable to disentangle phylogenetic theories. Otherwise, as Fernandes et al.

(2015) noticed, there are only a few references to biodiversity conservation applications in small islands compared to the number of ecological studies.

In this paper we analysed the species richness of vascular plants for 81 small satellite islets around Sardinia, the second largest island in the Mediterranean Basin. According to the European PIM initiative (http://www.initiative-pim.org), the Mediterranean Basin (2,969,000 km^2) contains approximately 15,000 islands and islets. The Eastern side contains more islands than the Western side, but the latter contains the three bigger islands (Sicily, Sardinia and Corsica) and approximately 1100 minor islands. According to Bocchieri (1993), many of these minor islands (37% or 399 of the 1100 islands of Western Mediterranean Basin) are located off Sardinia. Therefore, this area can serve as a representative case of study of the Western Mediterranean Basin. Within the Mediterranean Basin biodiversity “mega hotspot” (Médail and Quézel 1997; Cañadas et al. 2014), some nested “macro hotspots” (e.g. the Tyrrhenian Islands) have been identified, which in turn contain the so-called “meso hotspots” (e.g. Sardinia; Cañadas et al. 2014), “micro hotspots” (e.g. the Supramontes region; Fenu et al. 2010) and “nano hotspots” (e.g. La Marmora Peak; Cañadas et al. 2014). The endemic vascular flora found there are mainly concentrated in scrubs, screes and rocky habitats (Thompson 2005; Fenu et al. 2014; Fois et al. 2015). Based on the latest assessment (Bocchieri 1998), the floristic diversity of the satellite Sardinian islets include approximately 1,200 vascular plant taxa distributed over a surface exceeding 275 km^2 (approximately 1% of the Sardinian surface). Thus, according to the last checklist of the Sardinian vascular flora (Conti et al. 2007), approximately 49% of the total flora is concentrated in only 1% of the overall Sardinian area. Regarding this high biodiversity richness (not only from a floristic perspective), approximately 75% of the surface of the satellite islets is rightly included as protected areas (Natural Parks and Sites of Community Importance). Nonetheless, local extinctions of 191 vascular plant taxa (including 10 endemics) on 18 islets belonging to three Sardinian sub-archipelagos (La Maddalena, Sulcis and Tavolara; see Bocchieri 1998 for details) have been documented in the last century.

Due to the conservation importance of these satellite islets, including very small ones, we wanted to investigate their biogeography of the Sardinian satellite islets by exploring the influence of both environmental and anthropic factors on the pattern of the Endemic Vascular Plant Richness (EVPR) and Total Vascular Plant Richness (TVPR). First, we subdivided the islets according to their surface characteristics instead of their geographical position (e.g. archipelagos). In this way, we tried to de-emphasise the “biogeographical factors”, such as the area and degree of isolation, in order to obtain more applicable results for conservation planning, such as those relating to climate and human disturbance. In addition, we deepen our investigations of the conservation importance of these satellite islands by reporting which endemic plant taxa would suffer the most in terms of hypothetical extinctions on satellite islets.

2. Materials and methods

2.1 Study area and floristic data

According to Bocchieri (1993), there are 399 satellite islets off the coast of Sardinia (40°54’N and 9°35’E; Fig. 1a, b). This study informs about their highly diversified insular conditions resulting from different geographical patterns and the wide ranges of area (from the biggest islet of Sant’Antioco that is 109 km^2 to small ones that are only of few dozen square meters), elevation (from Tavolara at 565 m a.s.l. to flat islets of only a few meters), shape and lithology (intrusive and effusive igneous, sedimentary and metamorphic substrata). Because they lie on the same Cyrno-Sardinian tectonic microplate, they can all be considered continental islets. In addition, they can be subdivided into two categories according to their geological history: islets that were still connected to Sardinia during the late Pleistocene (with lower distance and elevation) and islets formed before 12,000 years ago (Ulzega 1995). Currently, the human population density in these islets is higher than in Sardinia (about 106 people/km^2; data available at http://www.sardegnastatistiche.it) but is mainly concentrated on three islets (La Maddalena, Sant’Antioco and San Pietro). Traditional agricultural and pastoral activities are quite intense only on a few of the islets. Otherwise, fishing activities and tourism in particular are prevalent where legally permitted.

The comprehensive geodataset of all 2,494 Sardinian vascular plant taxa (including native and non-native species and subspecies; Conti et al. 2007) was divided into two groups: endemic and total, which determined our EVPR and TVPR, respectively. The checklist of vascular plant taxa endemic to Sardinia was based on Fenu et al. (2014) by selecting the 290 exclusive taxa of the biogeographic province. These species and subspecies are categorised into Sardinian (183 taxa), Sardo-Corsican (90 taxa), and plants also present in the Tuscan Archipelago (17 taxa). Information about the richness of this latter group (EVPR) on the islets was obtained from a geodatabase already developed by the same authors for the entire Sardinian territory that consists of approximately 60,000 data records obtained from seven herbarium collections (CAG, CAT, FI, RO, SASSA, SS, TO), the literature and unpublished field surveys sighting records based on fieldwork by the authors, between 2002 and 2015. Specifically, the dataset on satellite islets consists of 1,279 records obtained from the literature (768 records), herbarium collections (116 records) and field surveys sighting records (395). The richness of all vascular plants was recorded as number of taxa obtained using two steps: firstly, all bibliographic information was checked, and then all unpublished information obtained from the above-mentioned sources was registered to update and integrated. Due to the complexity and lack of data, we chose to consider for our analyses only the 81 satellite islets that are home to a minimum of one endemic plant (Table 1).

Table 1 Richness of the Total (TVPR) and the Endemic Vascular Plants (EVPR) are listed for the 81 analysed islets that are home to a minimum of one endemic vascular plant taxon. Additional information is referring to the coordinates in meters (Univerlsal Trasverse Mercator system, ellipsoid WGS84) and the parameters used for the PCA analisys: Area (Area), Elevation (E), Steepness Index (SI) in angular degrees units and environmental diversity computed averaging the number of land uses and land units (H).

Group 1	X	Y	TVPR	EVPR	Area (km^2)	E (m)	SI(°)	H
I. S. Pietro	437398	4332624	580	60	51.0	211	3.0	21.5
I. S. Antioco	439574	4545181	672	42	109.5	273	2.6	31.0

I. Maddalena	534371	4564696	645	33	19.7	156	3.6	16.5
I. Caprera	538997	4562180	602	29	15.9	212	5.4	13.5
I. Asinara	439574	4545181	616	18	51.7	408	5.7	14.0
I. Budelli	529268	4570228	262	17	1.7	88	6.8	4.0
I. Spargi	528961	4565570	387	17	4.2	153	7.5	5.0
I. S. Stefano	534504	4560688	390	15	3.0	100	5.8	6.5
I. Razzoli	528990	4572230	190	14	1.7	65	5.1	2.5
I. Molara	561313	4524427	384	17	3.5	161	8.7	4.0
I. S. Maria	531271	4571864	277	12	1.9	49	3.6	6.5

Group 2	X	Y	TVPR	EVPR	Area (km^2)	E (m)	SI(°)	H
I. Tavolara	554126	4528534	502	49	6.0406	565	22.2	7.5
I. Figarolo	554126	4536785	210	4	0.2201	139	27.6	1.5
Pan di Zucchero	448290	4354001	37	4	0.0514	133	46.5	1.5
I. Campionna	479675	4495133	61	2	0.0101	16	19.6	1.0
I. dei Porri	433747	4525793	40	2	0.0478	63	28.9	1.5
I. d'Ogliastra	560173	4425227	138	2	0.0614	47	18.1	2.0
I. Pecora	560173	4559135	117	2	0.0714	16	6.0	1.5
I. del Toro	448853	4304580	33	3	0.1301	112	28.4	1.0
I. E di Stramanari	530575	4570290	6	2	0.0020	9	24.0	1.0
I. Molarotto	530575	4525116	54	2	0.0356	51	28.0	1.0
I. della Vacca	452305	4309855	63	2	0.0914	94	28.7	1.5
I. Piana di Capo Caccia	427322	4309855	85	1	0.1311	105	26.9	1.0
I. S. Macario	502792	4717180	98	1	0.0201	29	19.9	1.0
I. Foradada	428190	4491440	71	1	0.0540	131	44.9	1.0
I.tto di Buggerru	488645	4362198	10	1	0.0002	9	42.6	1.0

Group 3	X	Y	TVPR	EVPR	Area (km^2)	E (m)	SI(°)	H
I. Mortorio	550775	4547392	284	16	0.633	77	9.7	3.0
I. Serpentara	552342	4332428	136	13	0.379	54	8.8	2.5
I. dei Cavoli	546018	4326493	223	11	0.427	40	6.2	1.5
I. delle Bisce	543879	4557144	209	8	0.296	16	3.0	2.0
I. su Cardolinu	490524	4305426	168	8	0.018	14	10.5	1.5
I. Rossa di P. Niedda	475420	4307335	206	7	0.108	43	13.0	2.5
I. Piana dell'Asinara	434243	4536287	250	7	0.121	23	2.1	2.0
I. Giardinelli	536979	4564619	n.d	7	0.470	16	2.4	1.5
I. a N di Paduleddi	530697	4570626	82	6	0.019	12	8.9	1.0
I. Tuarredda	483792	4304763	149	6	0.045	32	15.0	1.0
I. Piana di Tavolara	554832	4526554	250	6	0.138	14	3.8	1.5
I. Barca Sconcia	554519	4529818	60	5	0.007	6	7.2	1.5
I. Barrettini	533650	4570297	79	5	0.107	39	11.9	1.0
I. Corcelli	533516	4571556	93	5	0.128	32	9.0	1.5
I. Abbatoggia	533463	4566942	21	5	0.021	14	9.7	1.0
I. Proratora	560974	4521964	156	4	0.049	27	12.2	1.5
I. Capicciolu	530904	4573133	32	4	0.017	2	3.1	1.0
I. Cavalli	553945	4526254	178	4	0.023	5	3.3	1.5
I. la Presa	531441	4573243	58	4	0.290	2	0.2	2.5
I. Mal di Ventre	440686	4426933	187	4	0.885	18	1.9	1.5

I. di mezzo Stramanari	530475	4570276	20	3	0.003	5	9.2	1.0
I. dei Porri	556281	4527145	16	3	0.004	6	10.1	1.0
I. Piccolo Porco	538534	4558209	37	3	0.004	7	11.5	1.5
I.tto S Italiani	538182	4563918	37	3	0.009	8	8.5	1.0
I. Cappuccini	542123	4555883	56	3	0.031	23	13.0	1.0
I. Piana di Maddalena	533949	4571566	208	3	0.041	10	5.0	1.5
I. Porco	538667	4558206	80	3	0.054	25	10.8	1.0
I. Rossa di Teulada	488492	4540329	40	3	0.064	29	11.5	1.0
I. Spargiotto	527145	4566607	76	3	0.103	47	14.5	1.5
I. Carpa	530354	4570744	54	2	0.004	9	13.8	1.0
I.tto N Italiani	538149	4564234	8	2	0.004	5	7.6	1.0
I. S Paduleddi	530591	4570517	56	2	0.005	7	10.2	1.0
I. le dei Monaci	543429	4562976	8	2	0.018	12	8.9	1.0
I. del Cavallo	544297	4529697	15	2	0.015	4	3.3	1.0
I. W Camere	548882	4546486	120	2	0.038	22	11.4	1.5
I. Chiesa	535023	4562511	n.d.	2	0.071	16	6.1	1.0
I. Piana di S. Pietro	441304	4338247	51	2	0.220	19	4.1	2.0
I. Soffi	548301	4546095	216	2	0.454	30	4.5	2.5
I. dei Topi	556860	4527513	9	1	0.003	5	9.7	1.0
Barrettinelli di Fuori	533577	4572298	n.d.	1	0.005	11	15.1	1.0
I. del Coltellazzo	502038	4315093	30	1	0.005	11	15.0	1.0
I. Cana	557053	4523933	30	1	0.006	6	7.8	1.0
I. Cavalli	526669	4561274	n.d	1	0.007	2	5.9	1.0
S. di P.ta Stagnali	537108	4561252	31	1	0.008	5	5.7	1.0
I. Rosso	556593	4525402	115	1	0.025	11	7.1	1.5
I. E Camere	549265	4546690	94	1	0.052	16	7.1	1.0
S. S di Abbatoggia	533309	4566545	9	1	0.012	2	4.7	1.0
I. W Stramanari	530376	4570357	10	1	0.002	3	7.8	1.0
I. Portisco	544442	4543538	87	1	0.013	5	4.4	1.5
I. Fico	536355	4559707	17	1	0.003	2	13.9	1.0
S. di P. Abbatoggia	533568	4567930	1	1	0.004	2	6.6	1.0
I. S dei Poveri	548642	4549155	14	1	0.023	3	2.7	1.0
I. dei Garofani	553363	4526308	36	1	0.007	5	5.9	1.0
S. di Cala Giorgio Marino	530685	4571089	41	1	0.002	2	13.4	1.0
I. Piccola Pecora	540912	4559284	2	1	0.004	5	8.4	1.0

2.2 Islets' attributes

We analysed several variables that we subdivided into five main groups related to: (1) surface characteristics, (2) choros, (3) isolation, (4) climate, and (5) human access and presence. All basic cartographic data was downloaded from the official Sardinian geo-portal (http://www.sardegnageoportale.it). When not already available, variables relating to extent (area, perimeter) were derived from aerial orthophotos taken in 2006 and available from the same Sardinian geo-portal. The islet elevation dataset was based on the Digital Elevation Model (DEM) from the regional Light Detection and Ranging (LIDAR) elevation dataset. Where elevation data was not available or had low precision (e.g. for very small islets), it was corrected using 1:25,000 Military

Geographic Institute (Istituto Geografico Militare or IGM) maps or by bibliographic information (e.g. Arrigoni and Bocchieri 1995; Poggesi et al. 1995). In addition to the area and perimeter, we also computed the perimeter to area ratio (PAR; Yo et al. 2012) and a Steepness Index (SI). The SI is a measure of islet steepness (in angular units) and was calculated by simplifying the three-dimensional geometric shape of each islet into a cone:

$$\mathrm{SI} = Arctan\left(\frac{R}{E}\right) \qquad (1)$$

where R represents $\sqrt{\frac{A\ island}{\pi}}$, E the Elevation and A the Area.

Three measures of environmental heterogeneity (i.e. number of units per islet) were obtained from geological maps (Carmignani et al. 2001), land use maps (Regione Autonoma della Sardegna 2003) and land units maps (Smiraglia et al. 2013). We multiplied the respective number of land types by the area in order to constitute the choros variables group (chorosG; chorosLUse and chorosLUnits, respectively). In addition, three types of geographical isolation were computed following Weigelt and Kreft (2013): the nearest distance to the mainland (Sardinia and, occasionally, Corsica) and to the closest islet and the surrounding land mass (SEA). This latter metric was recently conceived by Weigelt and Kreft (2013) and consists of the proportion of land area in the surroundings of the target islet within a buffer distance (in our case 5 km in order to include the complexity of archipelagos). With the aim of including a major climatic complexity, we used four bioclimatic indices found in Pesaresi et al. (2014) instead of simple commonly used climatic parameters collected directly from the Worldclim database (Hijmans et al. 2005). In particular, we applied four indexes that comprised both temperatures and precipitation variabilities: (1) the ratio of the mean temperature of the warmest month and the mean temperature of the coldest month (i.e. simple continentality index), (2) the ratio of the annual positive precipitation and yearly positive temperatures (i.e. annual ombrothermic index), (3) the total average precipitation of months with an average temperature higher than 0 °C (i.e. annual positive precipitation) and (4) the sum of mean annual temperatures plus the average of the minimum temperatures of the coldest month and the average of the maximum temperatures of the coldest month (i.e. thermicity index). Human influence was measured as the distance from the nearest port and the number of moorings (within a radius of 1 km from the islet) obtained from the atlas of the Sardinian beaches (Di Gregorio et al. 2000). The concentration of human-built structures was computed from the database of the official Sardinian geo-portal (http://www.sardegnageoportale.it) and carefully revised in order to include all artefacts. All digitalisations and measurements were executed using Quantum GIS software (QGIS Development Team 2014).

2.3 Environmental and endemic floristic analyses

All analyses were conducted in R environment (R Development Core Team 2013). In agreement with our purpose of evaluating the differences among the islets with different surface characteristics (area, elevation, slope and habitat diversity computed averaging the number of Land Uses and Units), the overall complexity of islets was categorised by visually inspecting the two-dimensional plot of the two Principal Components

(PC) from multivariate analysis (Principal Components Analysis, PCA). We used the *envfit* function of the *Vegan* package (Oksanen et al. 2013) to calculate the multiple regression of each variable with ordination axes. Significance (P value) was tested by permutation test. The variables were automatically standardised (mean centred and scaled) by the package to give them all the same importance.

Once the main islet groups were determined, the importance of satellite islets for conservation was corroborated by hypothesising the extinction of islets' endemic plants. To do this, the extent of the Minimum Convex hull Polygon (MCP) of each taxon was calculated applying the method implemented by GEOCAT (Bachman et al. 2011) and differences among complete MCP extents (Sardinia plus its satellite islets) and the MCPs without record occurrences of endemics in satellite islets were compared. This analysis was repeated, omitting all islets' occurrences and in turn, all occurrences of groups obtained by the PCA.

In addition, we analysed the differences among these groups in terms of correlation with all the variables. We particularly investigated whether surface characteristics, choros, degree of isolation, climatic factors or factors related to the human disturbance were the most influential drivers for both EVPR and TVPR. We thus fitted the linear version of the power function (log-log model; Arrhenius 1921) to each version of the datasets, after log-transforming both variables, using linear regression:

$$\mathrm{Log}S = c + z\mathrm{Log}X \qquad (2)$$

where S = species richness, X = factor, and z and c are the slope and intercept, respectively. According to Preston (1962), quantitatively rigorous studies of the species–area relationship have largely become an exercise of analysing trends in just z-values. These trends can be viewed as a scaling factor, rather than just the 'slope' of the species–area relationship and enables searches for central tendencies in the parameters and statistical comparisons of differences in this value among species groups or types of archipelagos (Lomolino and Weiser 2001). However, interpreting z and c parameters for applications involving conservation targets is rarely straightforward (Matthews et al. 2014). We thus evaluated the reliability of all explanatory variables in each power model by comparing the adjusted coefficients of determination (R^2_{adj}) of only significant results ($P < 0.01$). Due to the high correlation between habitat diversity, human presence and area that could result in imprecise parameter estimates (Triantis and Sfenthourakis 2012), we considered the factors obtained by the combination of area (i.e. choros and the concentration of human presence) only when their relative R^2_{adj} was higher than the area *per se* one.

3. Results

3.1 Islets' classification and endemism analysis

The analyses of the distribution pattern highlighted a high endemism rate in Sardinian islets. Indeed, the 81 analysed islets are home to 105 out of the 290 vascular plant taxa endemic to Sardinia and occupy a whole surface area of about 275 km^2 (approximately 1% of the total Sardinian surface).

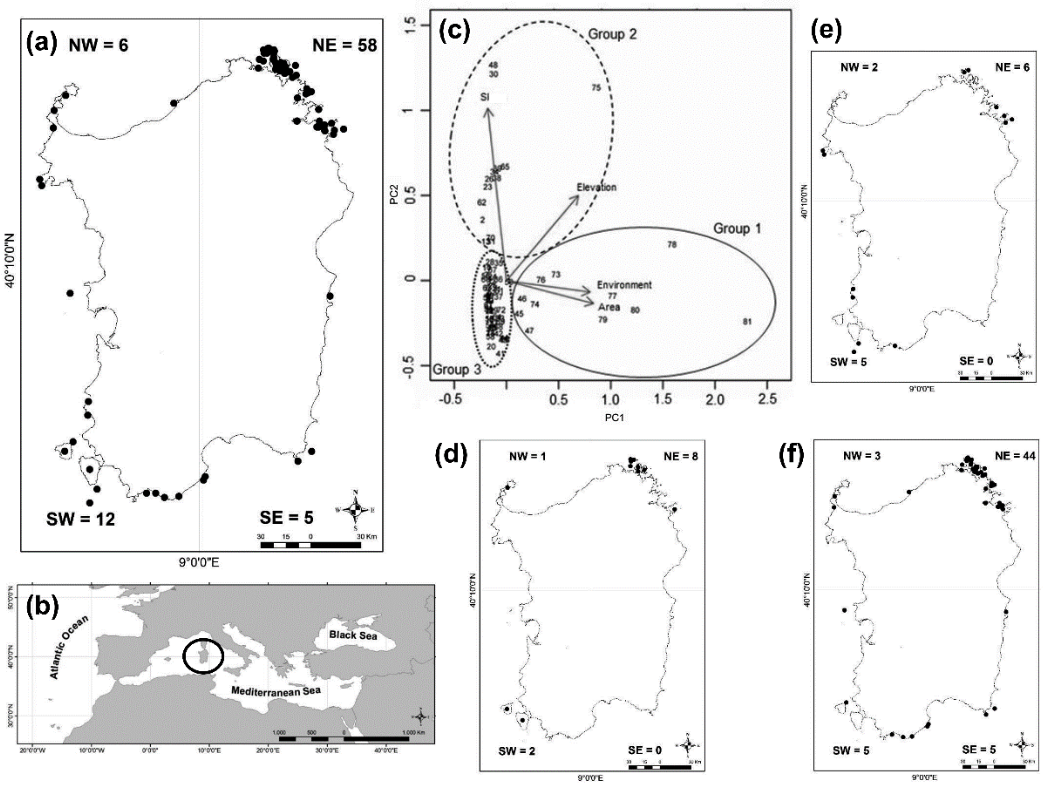

Figure 1. Distrubution of all Sardinian satellite islets (a) and position in the Mediterranean context (b). For each group depicted by PCA analyses (c), specific distrubution of relative islets are also reported for Group 1(d), Group 2 (e) and Group 3 (f). The number of islets in each quadrant (northwest, NW northeast, NE; southwest, SW and southeast, SE) and in each specific group is also reported.

The first two PCA axes that together explained 70.8% of the variance allowed identifying the groups of islets (Fig. 1c). The first axis (PC1) was strongly influenced by area and environmental heterogeneity, while the second axis was principally influenced by the SI (Table 2).

Table 2 Simple statistics (minimum, maximum and average values) and scores of the parameters fitted onto the PCA ordination are shown in Fig. 1. The direction cosines of the vectors, squared correlation coefficient (R^2) and Pearson correlations are reported (N = 81 = islets that are home to a minimum of one endemic plant). P-values are based on 999 random permutations of the data. Environmental descriptors are ordered from higher to lower ordination PC1 axis scores. Environmental heterogeneity (Environment) was computed by averaging the number of land uses and land units. The letter acronyms stand for Area (A; km^2), Elevation (E; m) and Steepness Index (SI; angular units).

Variables	Min-Max	Average	PC1	PC2	R^2	P-value
A	2×10^{-3}-109.5	4.1	0.99488	-0.10102	0.8335	0.001
Environment	1-38	3.2	0.98500	-0.17256	0.9115	0.001
E	1-565	52.7	0.78845	0.61510	0.7941	0.001
SI	0.2-46.5	11.3	-0.15035	0.98863	0.8824	0.001

The three groups of islets were therefore defined as (1) flat and big islets (hereafter Group 1), (2) islets with high slope (hereafter Group 2) and (3) flat and small islets (hereafter Group 3). The general distribution pattern

(Fig. 1a) was reflected by each group (Fig. 1d-f). Approximately 70% of the islets are situated off the northeast coast of Sardinia (Fig. 1d), approximately 15% are situated off the southwest Sardinian coast (Fig. 1e) and the remaining islets are equally divided between the other two quadrants (Fig. 1f). The greatest number of endemic plants (77 taxa; Table 3) is present in Group 1, while the highest concentration of endemic plants is in Group 3; the islets belonging to this last group are also characterised by a very low average elevation (Table 3).

Table 3 Main characteristics of the three PCA groups. The overall area (A), average of Elevation (E), number of endemic plants (End) and their concentration (End/A) are reported.

Group	n	A (km^2)	E	End	End/A
1	11	263.87	170.5	77	0.29
2	15	6.96	101.3	56	8.04
3	55	4.10	8.1	49	12.19

n = number of islets of each group; A = total surface Area of each PCA group; End = number of endemic plant in each PCA group; End\A = density of endemic plants per each group.

A number of 32 endemics would suffer a reduction of extent greater than 50% and of these, 10 *taxa* would become extinct in Sardinia if the populations of the satellite islets were to disappear (Table 4). Among the islets groups, the endemics that would suffer the most drastic reduction are those connected with the omission of occurrences on islets belonging to Group 1 (11). The persistence of seven and eight vascular plant *taxa* is strictly due to the conservation of the populations of islets belonging the Groups 2 and 3, respectively (Table 3). Only in four cases (e.g. *Silene martinolii* and *Narcissus supramontanus* ssp. *cunicularium*), reductions were connected with the omission of occurrences belonging more than one group of islets (Table 4).

Table 4 List of endemic vascular plant *taxa* that would suffer a reduction greater than 50% due to a hypothetical extinction of islets localities with their respective chorological forms (C) and protection by the European Habitats Directive (*). Percentages of reductions were computed omitting all islets *taxa* occurrences and, in turn, occurrences on the islets belonging to Group 1, Group 2 and Group 3. The cases of *taxa* that are not present in some islet groups or that have an irrelevant percentage of reduction values are not reported (-).

Scientific name	C	Without Islets (%)	Without Gr1 (%)	Without Gr2 (%)	Without Gr3 (%)
Astragalus maritimus	SA	-100.0	-100.0		
Carduus fasciculiflorus	SA-CO	-100.0	-	-58.5	-
Centaurea forsythiana pro hybr.	SA	-100.0	-	-100.0	-
Colchicum verlaqueae	SA	-100.0	-96.5	-	
Limonium articulatum	SA-CO-AT	-100.0	-	-	-65.2
Limonium cunicularium	SA	-100.0	-88.0	-	-
Limonium strictissimum	SA-CO*	-100.0	-100.0	-	-
Senecio vulgaris var. *tyrrhenus*	SA	-100.0	-79.4	-	-
Silene martinolii	SA	-100.0	-	-96.0	-82.9
Silene valsecchiae	SA*	-100.0	-52.0	-	-72.3

Silene velutina	SA-CO	-99.8	-	-	-99.8
Narcissus supramontanus ssp. *cunicularium*	SA	-99.7	-76.5	-	-99.3
Centaurium erythraea ssp. *rhodense* var. *sanguineum*	SA-CO	-99.3	-89.7	-	-89.9
Clinopodium sandalioticum	SA	-99.3	-	-99.3	-
Artemisia gallica ssp. *densiflora*	SA-CO	-99.0	-	-	-
Buphthalmum inuloides	SA	-98.7	-	-	-
Centaurea horrida	SA*	-95.2	-	-91.3	-
Stachys salisii	SA-CO-AT	-91.1	-91.1	-	-
Orobanche australis	SA	-90.9	-90.9	-	-
Nananthea perpusilla	SA-CO	-85.1	-	-	-
Ranunculus cordiger ssp. *cordiger*	SA-CO	-84.6	-	-	-84.6
Limonium hermaeum	SA	-83.0	-	-	-
Limonium protohermaeum	SA	-82.8	-	-	-
Romulea revelieri	SA-CO-AT	-82.4	-82.4	-	-
Anchusa crispa ssp. *maritima*	SA*	-76.7	-	-	-76.7
Lavatera triloba ssp. *pallescens*	SA	-75.5	-	-75.5	-
Odontarrhena tavolarae	SA	-75.4	-	-75.4	-
Erodium corsicum	SA-CO*	-70.0	-	-	-
Limonium insulare	SA	-69.2	-69.2	-	-
Saxifraga corsica	SA-CO	-63.0	-	-	-
Ferula arrigonii	SA-CO	-60.3	-	-	-
Charybdis glaucophylla	SA	-59.5	-59.5	-	-

3.2 Drivers of species richness

For all 81 islets, similar results were detected for TVPR (Fig. 2) and EVPR (Fig. 3). Measures of surface characteristics (area) and isolation (SEA) were the most significantly correlated among both TVPR and EVPR and all the 81 islets. Further differences were also found between the islet groups. Due to the previous grouping exercise, the area factor lost its importance in relation to other surface parameters, such as the perimeter (TVPR of Group 2), PAR (TVPR of Group 3 and EVPR of Group 1) and choros for the remaining cases. Apart from the TVPR of Group 2, the SEA measure of isolation was another determinant factor. We found significant correlations between climate and factors relating to human presence only for the EVPR of Groups 1 (annual positive precipitation) and 3 (thermicity index and concentration of human presence).

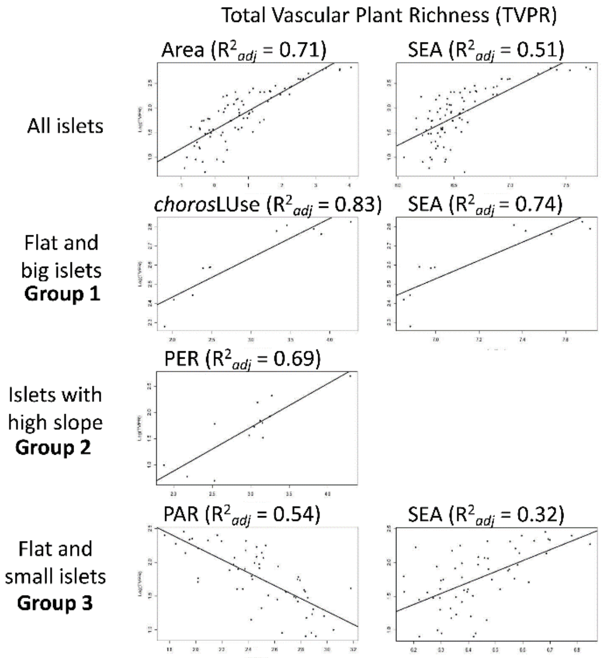

Figure 2. Best-fit power models among variables and the TVPR of all Sardinian satellite islets and the three PCA groups. The adjusted R-squared (R^2_{adj}) of each variable is reported on the plots. The results of all models reveal highly significant relationships (P<0.001).

The highest positive z-values were found for SEA and the thermicity index, while negative z-values were detected only for PAR and the concentration of human presence.

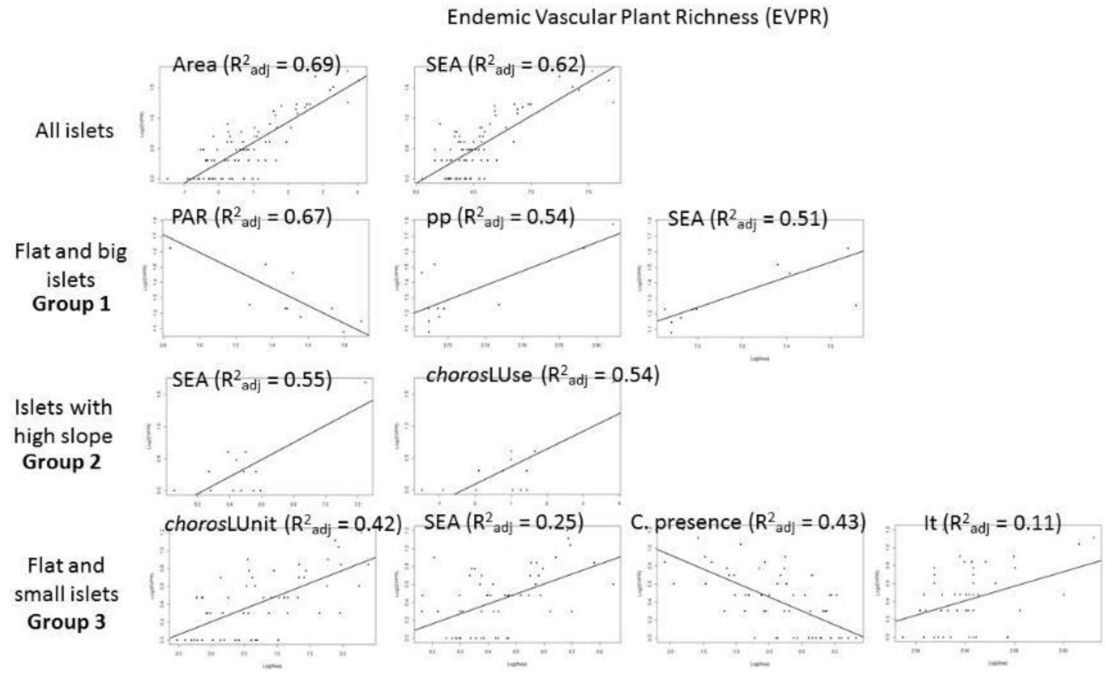

Figure 3. Best-fit power models among variables and the EVPR of all Sardinian satellite islets and the three PCA groups. The adjusted R-squared (R^2_{adj}) of each variable is reported on the plots. The results of all models reveal highly significant relationships (P<0.001).

4. Discussion

4.1 Islets' classification and endemism analysis for PCA groups

The importance of island biodiversity, as declared by the United Nations (SCBD 2014), was confirmed by our analyses of the endemic plants patterns on Sardinian islets. As 36% of the plants endemic to the biogeographical province are present on the 81 satellite islets (covering 1% of the overall Sardinian area), the endemism rate is high; therefore we recognise them as nano hotspots of biodiversity (*sensu* Fenu et al. 2010). In particular, the exiguity of their overall surface (only 4.1 km^2; Table 3), the 49 endemic vascular plant taxa (16% of the total amount of Sardinian endemic plants) found of the islets of Group 3 results in them being 'more than exceptional' centres of endemism richness. In addition, their environmental traits (small and with a low hill slope) makes them particularly susceptible to the extinction process due to a rise in the sea level and genetic drift (Bellard et al. 2014; Harradine et al. 2015); nevertheless, they should be considered as 'modern refugia' from the human pressure. This is based on findings on taxa highlighted in Table 4. Indeed, mainland extinctions (e.g. of *Carduus fasciculiflorus* and *Silene valsecchiae*) and pressures (e.g. suffered by *Astragalus maritimus* and *Centaurea horrida*) have been induced by human activities (e.g. Pisanu et al. 2009; Bacchetta et al. 2011), which are usually more intense around the coast of Sardinia and the bigger islets (Group 1). An additional straightforward example is the Cardulinu Islet (SW Sardinia), which harbours seven endemic vascular plant taxa in an area of 1.79 km^2. Two of these are exclusive to Sardinia (*Bellium crassifolium* and *Romulea requienii*) and two (*Genista valsecchiae* and *Limonium tigulianum*) are also exclusive to the Sulcitan biogeographic subsector (Fenu et al. 2014). Such floristic heritage can be explained by the concomitance of low human pressures and the short distance from an endemic-rich main coast.

The importance of satellite islets for conservation was also emphasised by analysing the consequences of hypothetical endemic vascular plant taxa extinctions. Indeed, about 30% of the 105 endemics analysed could suffer an extent reduction greater than 50%. According to IUCN criteria (2014), these taxa are at least Endangered at the regional level. Considering that five of these taxa are also listed in the European Habitats Directive (92/43/EEC; Table 4) and that their conservation is therefore mandatory for all EU members, the protection of these areas should be a priority.

4.2 Drivers of species richness

As generally asserted (e.g. Preston 1965; MacArthur and Wilson 1967; Lomolino and Weiser, 2001), the main factors that influence~~s~~ the TVPR/EVPR on the Sardinian satellite islets are the metrics of extent that are irregularly related to habitat heterogeneity and isolation. Following the suggestions of several authors (e.g. Ricklefs and Lovette 1999; Triantis et al. 2003), we also applied different measures of extent (area, perimeter and PAR) and found that each one fit differently in each group of islets. As recently found for the Corsican flora (Jeanmonod et al. 2015), endemic and total plants are found in all substrata in similar proportions, while, as we found on the islets in Group 3, land unit diversity has a stronger influence on the endemic taxa richness. The law of proximity in terms of neighbouring large islands that serve as major sources for colonisation and maintenance of species richness (MacArthur and Wilson 1967) is confirmed for our small islets dataset. Indeed, significant correlations with SEA measure of isolation was in all cases characterised by positive z-values (i.e. more nested islets contained a higher plant richness). This is also in line with the evidences found for the continental islands of the Mediterranean Basin (Steinbauer et al. 2013) and for the particular case of the Ionian islets (Iliadou et al. 2014). The isolation of continental islands (i.e. located on the shelf close to the continent and consisting of continental parent material and often older than oceanic islands) is less effective for inducing speciation than in the case of oceanic islands (volcanic islands on the oceanic crust; Cody 2006), and the probability for an island to be part of a dense agglomeration of neighbouring islands of comparable age is rather high on the shelf. Furthermore, the speciation processes on the continental islands are associated with random rather than adaptive differentiation (Georghiou and Delipetrou 2010). In our particular case, we thus argued that a positive relationship between the degree of nestedness (SEA) and plant richness could be interpreted as a symptom of a higher capacity to disperse into these islets and, due to the lower human presence compared to the Sardinian coast, to preserve the colonising species.

4.3 Small islets' conservation management planning

Our method of subdividing the set of islets according to their environmental characteristics allowed us to point out further reliable factors for conservation planning. First, we determined that land use factors, and thus land use changes, have a significant influence on the TVPR of Group 1 and the EVPR of Group 2. If TVPR can be also influenced by some human-induced colonisation, the influence of land use diversity on EVPR must be related to other factors. Although the intensification of agriculture, which tends to increase the size of production units for these areas, is counterproductive for the conservation of their plant diversity (Green 1990; Grill et al. 2005), our findings confirmed that even the complete abandonment of traditional and sustainable practices could have the same effect of decreasing the land use diversity (Alard and Poudevigne 1999; Amici et al. 2015). No significant relationships were found among the overall complex and climatic drivers, but it was possible to determine two significant relationships for the Groups 1 and 3. As demonstrated for the oceanic islands (Harter et al. 2015), despite the commonly assumed influence of climate change on species diversity, significant uncertainty and knowledge gaps exist and preclude distinct statements about islands' vulnerabilities. The reasons for this are linked to difficulties in predicting the relative influence of climate changes on such small areas that are also characterised by a heterogeneous set of environmental conditions. However, the decrease of such environmental variability through a PCA subdivision allowed to depict some significant influences of climatic factors on the EVPR of Groups 1 and 3. Considering that a conservation planning should be designed with a low degree of uncertainty due to the high economic costs (McShane et al. 2001; Fenu et al. 2015), we agree with Willis et al. (2015) and suggest that such practices based on predictions of climate changes should be carried out at the regional rather than the global level, trying to correlate very specific causes. Furthermore, e found a significant relationship between human concentration and the EVPR. Despite the large body of literature on the negative human effects upon the plant diversity in the Mediterranean Basin (e.g. Greuter 2001; Lavergne et al. 2005; Fenu et al. 2013), no statistically significant response tendency has emerged for such small islet hotspots of biodiversity. The EVPR of small and flat islets (Group 3) was negatively influenced by the human presence; otherwise, due to their exiguity, the economic interest in these areas is irrelevant and the human presence is only occasional. Thus, despite their plant diversity relevance, their conservation could be without many problematic implications and they could represent a viable opportunity to conserve at least part of the populations of many coastal vascular endemics.

5. Conclusions

According to several authors (e.g. Brooks et al. 2006; Cañadas et al. 2014), the identification of very small areas with an exceptional concentration of conservation-priority species is an essential way to maximise conservation efforts. Plant biodiversity nano hotspots in Sardinia have only been found in the mountain complexes (Fenu et al. 2010; Cañadas et al. 2014). These findings consequently omitted the identification of the most relevant territories for the conservation of coastal plants. Due to tourism pressures, it is currently almost impossible to find and develop effective conservation measures for such nano hotspots around the main Sardinian coast. We therefore propose the conservation of the satellite islets that share the same environmental conditions and have the advantage of a more available and cost-effective conservation management. Small islets and mountainous complexes have one thing in common: they are both isolated; therefore, the endemic species richness there could be supported by genetic drift events and low human disturbance.

Our research shows that even in an apparently homogenous set of islets, drivers of species richness could act differently. These findings suggest caution should be used when global island patterns are analysed. We therefore suggest that researchers at local scale can more effectively design tools for the conservation. Specifically, we argued that, besides the most commonly known drivers of island species richness, such as the area and isolation, specific and more reliable drivers for the conservation planning can be found by analysing sub-groups of islets categorised according to their environmental drivers rather than their geographical position. Even if bigger islets contain a greater number of endemic vascular plant taxa, our results regarding hypothetical extinctions underscored that the conservation of each set of islets is crucial for different sets of endemic plants. In addition, it became possible to identify the particular floristic relevance and their practicability of conservation on the very small satellite islets of Sardinia. Besides the human influence, which should also be considered (especially for TVPR) a positive factor in increasing the number of land uses, the nestedness among islets and the consequences of local climate changes should be studied in order to optimise biodiversity conservation efforts. More research is needed for a more exhaustive representation of reality. We highlighted the uncertainties in slope extrapolations and the difficulties in acquiring the necessary very high spatial resolution data in general. Additional analyses and comparisons of the island biogeography are also needed—particularly for small islands—to establish whether the findings reported herein are representative of other Mediterranean islands. Our results illustrate possible direction of the effects of principal drivers on such cryptic small units of biodiversity trying to highlight the importance of very small islands in biodiversity conservation. All considerations regarding the conservation relevance of Sardinian islets are similar in the overall Mediterranean context (e.g. Lavergne et al. 2005; Iliadou et al. 2014; Jeanmonod et al. 2015), and the suggestions could be therefore considered for all the Mediterranean biodiversity hotspots.

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CHAPTER V

Identifying and assessing the efficiency of micro and nano biodiversity hotspots networks at regional level

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This research is under review

Contribution to the project: I leaded the analyses and preparation of the manuscript. All the authors contributed for the study design and revision of the manuscript.

Abstract

The island of Sardinia (Western Mediterranean Basin) is considered a meso hotspot within the Tyrrhenian macro hotspot and the Mediterranean mega hotspot of biodiversity. In this context, a finer hierarchy of micro and nano hotspots was proposed in order to concentrate active conservation measures.

In this chapter, the main aim was to identify and to test the efficiency of micro and nano hotspot networks for endemic plant conservation in Sardinia. To do it, we set networks of micro, nano and integrated hotspots for the entire territory of Sardinia. The richness of Endemic Vascular Plant Species (EVPS), area, perimeter, connectivity and surplus costs for the protection of all endemic plants were used as ranking criteria.

Eight micro, 82 nano and an integrated network of eight micro and 24 nano hotspots were identified; these respectively include the 82%, 87% and 89% of all EVPS. The identification of hotspots networks at regional scale could allow to set a priority list of areas, also already protected, where to primarily allocate economical resources and conservation efforts for endemic plants. After considering all pros and cons, the integration of micro with nano hotspots resulted the most forward-looking plan from a both conservational and economical perspective.

Keywords: Endemic Vascular Plant Species; Mediterranean Basin; Species richness; Protected area selection; Conservation planning; Island flora; Areas of endemisms.

1. Introduction

The greatest challenge in environmental management is to preserve enough areas in order to conserve as many species as possible (Tjørve 2010). Although the 13% of Earth's land is already protected (Bertzky et al. 2012), targets adopted by the Convention on Biological Diversity (CBD) aim to expand the protected area network to at least 17% of the terrestrial world and to prevent the extinction risk of all known threatened species by 2020 (CBD 2010). Otherwise, the conservation of a large portion of globe implies high costs in economic and social terms, and strategies should thus be focused on areas that represent a maximum of the biome's diversity and endemism (Margules and Pressey 2000; Cañadas et al. 2014). In addition, protected areas alone do not guarantee the species conservation and their designation often represents a first step of further positive activities that would ensure, for instance, the maintenance of natural, semi-natural or agricultural ecosystems (Heywood 2015; Fenu et al. 2016).

While several priority areas have been proposed at global and regional scales (e.g. Mittermeier et al. 2004; Mendoza-Fernández et al. 2015; Koch et al. 2016), methods and solutions are still extensively debated. A number of systematic conservation planning tools are available to aid in making prioritization decisions (e.g. ResNet, Kelley et al. 2002; Zonation, Moilanen 2007; Marxan, Ball et al. 2009); otherwise, expert-based assessment and scoring procedures have been widely used to provide a method that can easily be understood by stakeholders, conservation practitioners and politicians involved in conservation (Burke et al. 2008; Marignani and Blasi 2012). Among these methods, areas of endemisms (*sensu* Richardson 1978) has become an increasingly common and irreplaceable surrogate for identifying protected areas (e.g. Burke et al. 2008; Huang et al. 2011; Mendoza-Fernández et al. 2015). Indeed, endemic plant species are potentially threatened because they often distribute only in restricted areas (Linder 1995; Laffan and Crisp 2003) and endemism-rich areas are also likely to be rich in other species of conservation concern (Whittaker et al. 2001; Landi and Chiarucci 2014). In this context, the term of 'biodiversity hotspot' was coined by Myers in the late 1980s (Mayers 1988) in order to identify priority conservation regions with a large number of endemic species. Since its introduction, the concept of hotspots was used as a key strategy for global conservation actions and it has become the principal global conservation-prioritization approach, attracting over $1 billion in conservation investment (Marchese 2015). As the size of the sample unit increases, the concepts of 'micro' (a maximum extent of 500 km^2) and 'nano hotspots'(a maximum extent of 10 km^2) were conceived at regional and local levels in order to encompass as much as possible Endemic Vascular Plant Species (EVPS hereafter) in a reasonable extent of territory (Fenu et al. 2010; Cañadas et al. 2014).

The identification of narrow hotspots could be useful to find gaps in the protected-area networks and may be helpful for future implementations. In the European Union, one of the key tools for biodiversity conservation is the Natura 2000 network based on the Birds and Habitats Directives (79/409/EEC and 92/43/EEC). This system is not a network of strict nature reserves, but is adopting a wider approach of promoting sustainable land uses, largely centred on people working with nature rather than against it (European Commission 2016). Inside and outside this network, the aim of a hotspots hierarchy proposal was instead to set priority areas where to primarily allocate economical resources and monitoring efforts (Cañadas et al. 2014; Kougioumoutzis and

Tiniakou 2015). According to the regional responsibility criterion (Mendoza-Fernández et al. 2009; Bacchetta et al. 2012a, 2012b), the identification of hotspots at lower scales makes nature-conservation planning more efficient with a more direct involvement of local stakeholders (Feng et al. 2011; Cañadas et al. 2014).

With the aim of enhancing the effective management and protection of biodiversity, in particular of all EVPS, two micro and seven nano hotspots were proposed in part of the territory of Sardinia (Fenu et al. 2010; Cañadas et al. 2014). Accordingly, several *in situ* and *ex situ* conservation activities and researches (e.g. Fois et al. 2015; Gentili et al. 2015; Cuena-Lombraña et al. 2016) have been then concentrated in these hotspot-areas. Nevertheless, further Sardinian territories, and EVPS hosted by them, are still not underscored by these kind of studies.

In order to fill gaps in the current hotspots network at micro and nano scale levels, we did an extensive survey on the possibilities for the prioritization of areas in Sardinia. The main aims of this research were to (1) identify all micro and nano hotspots for the entire territory of Sardinia and (2) to evaluate the efficiency, in terms of number of EVPS, extents (area and perimeter) and connectivity, of three network solutions composed by only nano hotspots, only micro hotspots or by the integration of micro and nano hotspots. Finally, (3) the EVPS that were still unprotected by each solution were localised and the surplus costs for the protection of all EVPS in Sardinia were also evaluated.

2. Materials and Methods

2.1 Study area

Sardinia is the second largest island in the Mediterranean Basin after Sicily, with a main inland surface area of 23,833 km^2 and a total of 24,089 km^2 including minor islands and islets (Fig. 1). The island is mainly composed of several isolated groups of mountains such as Limbara, Supramontes and Gennargentu (the highest of all at 1834 m a.s.l.). The about 1900 km of coast are marked by a variety of landscapes, such as cliffs, sandy dunes and both long and pocket beaches. From a biogeographical point of view, it is considered part of the Western Mediterranean biogeographic sub-region (Fenu et al. 2014). In particular, owing to many floristic similarities, Sardinia, Corsica and the Tuscan Archipelago were included in the same independent biogeographical province within an Italo-Tyrrhenian superprovince (Fenu et al. 2014). Moreover, the sub-province of Sardinia was also recently subdivided into six biogeographic sectors and 22 subsectors on the basis of the EVPS occurrences and their geomorphological patterns (Fig. 1; Fenu et al. 2014). Such geographical isolation and high geological and geomorphological diversity contributed to characterise a wide range of habitats and a consequent high rate of endemisms (e.g. Médail and Quézel 1999; Thompson 2005; Cañadas et al. 2014). Sardinian flora consists of 2,494 vascular plants (Conti et al. 2007). At the current state of the art, the flora limited to the Tyrrhenian island territories (i.e. endemic flora *sensu lato*) comprises a high proportion of exclusive Sardinian endemics (189 taxa), 90 Sardinian–Corsican endemics, and 16 taxa also present in the Tuscan Archipelago (Fenu et al. 2014).

According to its peculiar flora, Sardinia has been classified as a meso hotspot within the Tyrrhenian islands macro hotspot and the Mediterranean mega hotspot of biodiversity (Cañadas et al. 2014). Additionally, a network of ‘micro’ and ‘nano hotspots’ were hypothesised (Fenu et al. 2010) and three micro (Supramontes, Iglesiente and Gennargentu) and seven nano hotspots belonging to the micro hotspots of Supramontes and Gennargentu were already identified (Cañadas et al. 2014).

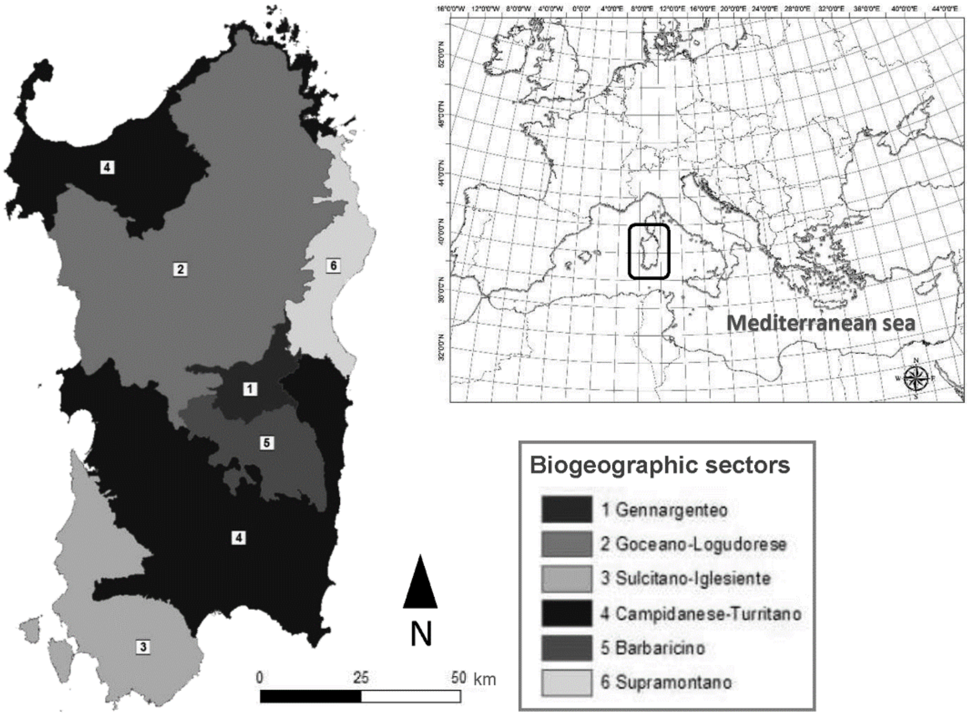

Figure 1. Study area of Sardinia in the Mediterranean context and its subdivision into the six biogeographical sectors (Fenu et al. 2014).

2.2 Distribution of Endemic Vascular Plant Species (EVPS)

Distribution data of all the 295 EVPS was obtained from available literature, Italian herbarium collections (CAG, CAT, FI, RO, SASSA, SS, TO *herbaria*) and unpublished field surveys records of last 20 years reported by the authors of this work. The spatial distribution was digitised by a Geographical Information System (GIS) software (Quantum GIS Development Team 2012) and organised in a geodatabase. An overall number of 60,309 occurrence points were carefully revised in order to avoid the potential large errors due, for example, to the approximation of the collection locations and the inclusion of extinct localities (Feeley and Silman 2010). A previous version of the same database was already used for the definition of the biogeography of the main island of Sardinia and its satellite small islands (Fenu et al. 2014; Fois et al. 2016b). In this case, each record was reported into 36,235 cells following the UTM 1 × 1 km grid. This map resolution was chosen to ensure consistency with the resolution of the floristic dataset and to generate a sufficiently high number of cells to challenge the practicability of the protected area prioritization proposed herein (Fig. 2).

2.3 Micro and nano hotspots selection

We used the same approach of Cañadas et al. (2014) and we based the hotspot selection on the total number of EVPS per each grid cell. Specifically, we selected a given floristic territory with a maximum extent of 500 km^2 as a micro hotspot whenever it accounted for more than 20% (≥ 58 EVPS) of the total Sardinian EVPS. Additionally, we selected a given area with a maximum extent of 10 km^2 as a nano hotspot, whenever it accounted for more than 5% (≥ 14 EVPS) of the total EVPS. In order to avoid biological complexities inside each hotspot, we encompassed only areas belonging to the same biogeographical sector, recently defined on the basis of the same EVPS distribution (Fenu et al. 2014). Once micro and nano hotspots were identified, an additional solution was also developed by integrating all micro hotspots with the non-overlapping nano hotspots (Fig. 2). All these procedures were carried out in the QGIS environment (Quantum GIS Development Team 2012).

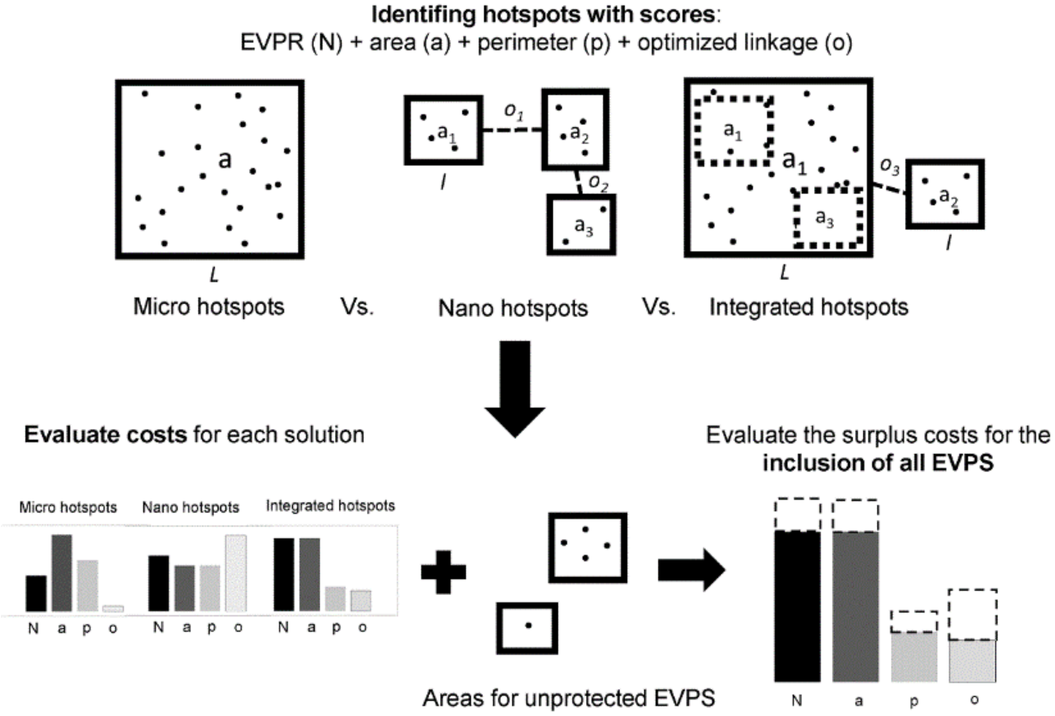

Figure 2. Schematic framework of the area prioritization procedures performed in this study.

2.4 Assessing the efficiency of micro and nano biodiversity hotspots networks

Four measures were used in order to evaluate the efficiency of each solution: (1) the number of EVPS, (2) the summarised areas, (3) perimeters and (4) the optimized linkages. This latter measure was computed using the ‘Type by Type’ algorithm implemented by the MulTyLink software (Brás et al. 2013). MulTyLink assumed that the study region is divided into cells and constructs a graph for each group (of “similar”) cells, taking into account the areas acting as barriers and the dispersal capacities of these species. When selecting areas in a graph, MulTyLink deems the possibility of using these areas for other groups, thus reducing costs and the number of selected areas (Brás et al. 2013). The functionality of this algorithm was previously tested in other

Mediterranean contexts (e.g. Alagador and Cerdeira 2007; Alagador et al. 2012) to identify cost-efficient linkages between environmentally similar protected areas looking for optimal feasible linkages like stepping stones (Brás et al. 2013). In this case, we connected the identified hotspots belonging to the same biogeographical sector using, when necessary, other hotspots as stepping stones. Because the dispersal capacity significantly differ among species of each hotspot, only sea was considered as a barrier.

Finally, we identified the location of all EVPS that were still unprotected by each solution. We then delimited further areas for these taxa, limiting as much as possible their extents and distances from the preselected ones and yet maintaining the same root criteria (i.e. preferring few big areas for micro hotspots and yet several small for integrated and nano hotspots). Thus, we evaluated the full conservation costs in terms of area, perimeter and optimized linkage surpluses (Fig. 2).

3. Results

3.1 Micro and nano hotspots selection

Eight floristic territories were designated as micro hotspots since each one includes more than 20% of the 295 Sardinian EVPS while 82 areas were selected as nano hotspots, since they host more than the 5% of the Sardinian EVPS (Fig. 3). The integrated network proposal was consisting of all the eight micro hotspots plus 24 nano hotspots. Further 14, 26 and 22 hotspots were respectively added to the micro, nano and integrated hotspot networks in order to include all the remaining EVPS (Fig. 3).

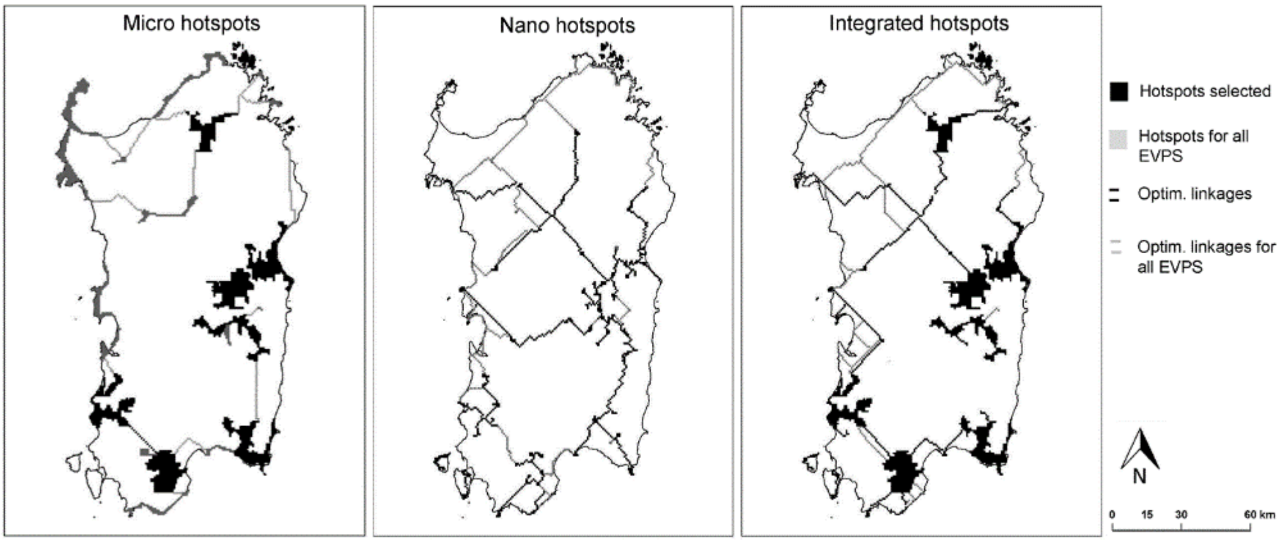

Figure 3. Micro, nano and integrated hotspots selected and their relative optimized linkages. Additional surplus hotspots and optimized linkages for the inclusion of all Endemic Vascular Plant Species (EVPS) are reported in grey.

3.2 The efficiency of micro and nano biodiversity hotspots networks

The eight micro hotspots identified for the entire Sardinian territory comprehensively contain the 82.6% (243 EVPS) of Sardinian endemics and occupy the 12.1% (2905 km^2) of the Sardinian surface area (Fig. 4). Area and species richness thresholds influenced the designation of some non-optimal shapes that, in concomitance with the proportion of the designed territory, were the cause of a high perimeter cost (1999 km; Fig. 4). Besides the case of two micro hotspots belonging to the same biogeographical sector (Barbaricino), each hotspot was selected in different biogeographical areas; this permitted to account a very low optimized linkage cost of 21 km.

In comparison to the micro hotspot network, the 82 nano hotspots included a higher percentage of EVPS (87.1%, i.e. 256 EVPS; Fig. 4) in a considerably lower proportion of surface area (458 km^2 corresponding to 1.9% of the Sardinian surface area). This lower surface-area cost was also reflected by a slightly diminished perimeter cost (1032 km); otherwise, optimal linkage costs considerably increased up to 769 km (Fig. 4). The integrated network proposal covered a surface of 3054 km^2 (12.6% of the Sardinian territory) which permitted to increase the number of protected EVPS up to 262 (89.1% of the Sardinian EVPS; Fig. 4). The increment in surface area and perimeter costs (2337 km) was partially compensated by a reduction of the optimized linkage costs (455 km).

The micro hotspot network resulted to have the less spread distribution pattern through the territory, but it encompassed the smallest quantity of EVPS. For these reasons, the increases in area (from 2905 km^2 to 3767 km^2), perimeter (from 1999 km to 3115 km) and optimized linkages (from 21 km to 262 km) were consistent when all the remaining EVPS were included (Fig. 4). Conversely, small increases in area (ca. 1%) and perimeter (ca. 4%) were registered for both nano and integrated networks while a considerable increases in optimized linkage costs were accounted for the integrated solution (from 455 km to 630 km) and yet especially for the nano hotspot network (from 769 km to 1064 km; Fig. 4).

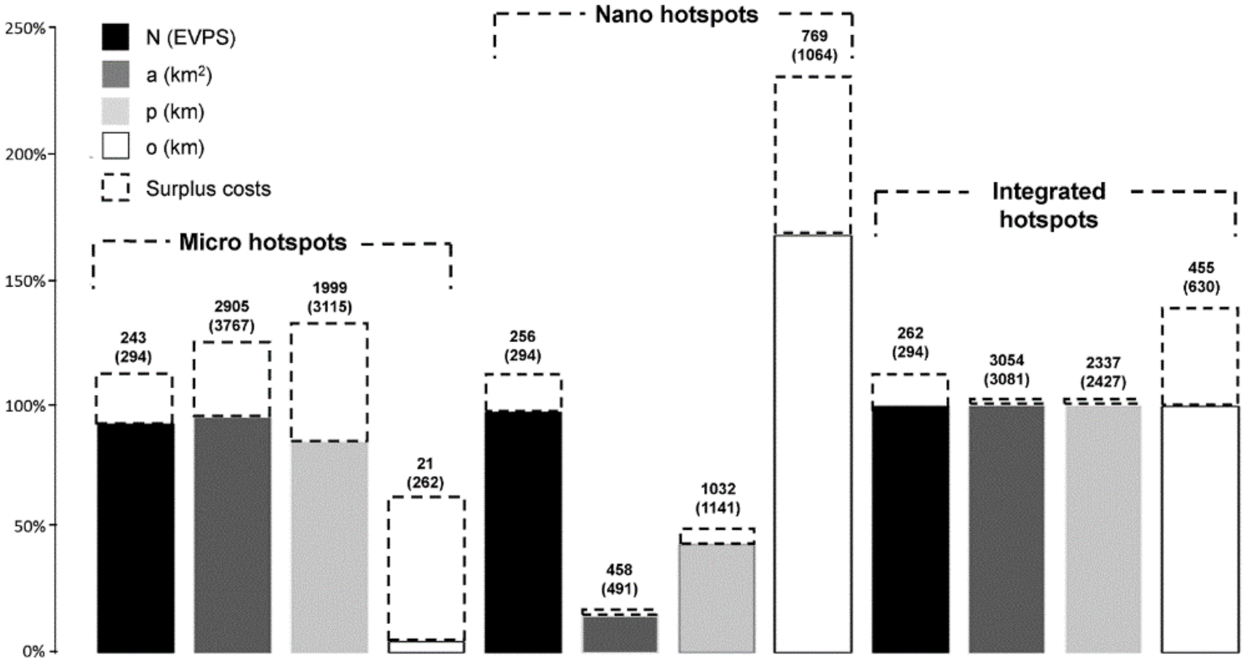

Figure 4. Percent of overlap among the integrated hotspots and micro and nano hotspots in terms of summarized Number (N) of Endemic Vascular Plant Species (EVPS), area (a), perimeter (p), optimized linkages (o) and surplus costs for the inclusion of all EVPS. Relative values of each parameter is also reported with the surplus ones between braklets.

4. Discussion

4.1 The importance of identifying micro and nano biodiversity hotspots at regional level

This research represents the first step towards a complete identification of areas of conservation interest at increasing scales. While this method was previously separately applied for the identification of only a part of the Sardinian territory (e.g. Fenu et al. 2010; Cañadas et al. 2014; Fois et al. 2016b), a complete geodatabase on the distribution of endemic plants allowed to fill gaps in knowledge on Sardinian micro and nano hotspots. These new and more comprehensive results were consistent with the already mentioned studies on particular Sardinian biogeographic sectors. Indeed, the predefined micro hotspots of Gennargentu massif, Supramontes and Iglesiente (Fenu et al. 2010; Cañadas et al. 2014), as well as the seven nano hotspots highlighted by the same authors (Fenu et al. 2010; Cañadas et al. 2014), were also here identified as part of most rich areas in EVPS. Otherwise, we achieved to complete this state of the art by identifying further four micro and 78 nano hotspots that comprised also portions of territories that are still poorly studied and protected (Fenu et al. 2015). An examination of hotspots locations enables the detection of specific threats, which is a priority in conservation (Brooks et al. 2006); the fact that the identified micro and nano hotspots often coincide to areas rich in endangered species (Fenu et al. 2016), confirms the importance of these small areas for plant conservation. In particular, endemic plants concentrated in high mountain peaks are especially vulnerable to climatic change as, being already at the elevational limit, they cannot adopt the strategy of vertical migration (Cañadas et al. 2014; Fois et al. 2016a; Speziale and Ezcurra 2016). According to a recent review on future threats in Mediterranean islands (Vogiatzakis et al. 2016), also the hotspots highlighted in coastal areas would face a particular decrease in species richness due to consequences related to sea level changes. Such threats are enforced by an increasing human presence along the same areas (Fois et al. 2016b). For these reasons, if protected area designation should reconcile the needs of nature conservation and socioeconomic development (Vergílio and Calado 2016), micro and nano hotspots should be areas where conservation activities are mandatory.

The identification of narrow hotspots could be useful to find gaps in the Natura 2000 network and may be helpful for their future new definition; otherwise, they would not represent an alternative to the already designed protected areas due to their different conservation focus and perspective. Indeed, the Nature 2000 network was based on the identification, monitoring and protection of habitats and species of community importance which are listed in the Habitats and Birds Directives and were not conceived as a system of strict nature reserves from which all human activities would be excluded (European Commission 2016). Differently, our hotspots of EVPS were conceived for planning active conservation efforts, also considering strict limitations on human activities in some restricted area.

4.2 Micro, nano or an integrated biodiversity hotpots network?

As often it happens, it is difficult to find a universal answer on a conservation debate, which often could be argued from different perspectives. The proposal of a micro hotspot network could achieve to preserve the majority of EVPS within a small territory and a low cost in terms of optimized linkages among the protected areas. Otherwise, the network of nano hotspots would host a higher percentage of EVPS in a minimum extent of area and perimeter but a higher cost in optimized linkages. In addition, as several small areas (i.e. nano hotspots) showed a wider-spread and diversified distribution pattern, they diminished risks related to stochastic and/or catastrophic events which could otherwise rapidly destroy few and more connected areas such as the micro hotspots (Cox and Underwood 2011). In terms of differences in monetary terms, area and perimeter are the two most related factors with the economic costs of a protected area in terms of acquisition and management (Frazee et al. 2003). Hence, we could assert that the nano hotspot network is in this case the most economical solution. On the other hand, optimized linkages are related to the habitat fragmentation and edge effects. These factors, are considered one of most prevalent threats for the Mediterranean flora (Cox and Underwood 2011; García-Camacho et al. 2012; Fenu et al. 2016) exposing protected areas to undergo changes in populations, communities and ecological processes and making them more susceptible to biological invasions than the core area of isolated remnants (Mendonça et al. 2015; Celesti-Grapow et al. 2016). Despite of its monetary effectiveness, the high costs in connecting the nano hotspots would thus expose these areas to a conservation failure. In comparison to the other networks, the integrated hotspots network would be an intermediate solution, as the protection of more EVPS would be achieved by increasing a reasonable portion of surface and perimeter and by significantly decreasing risks related to habitat fragmentation and edge effects (i.e. optimized linkages costs). In addition, it would permit to consider many territories that would not be otherwise protected inside micro hotspots. Indeed, the majority of the surplus areas suggested for the protection of all EVPS by micro hotspots are situated in proximity of touristic and urbanized areas of Sardinia and concrete conservation measures would be thus concretely unattainable because of the high interests revolving around these areas.

The integrated hotspots solution is *per se* suggesting a stepwise procedure to design protected areas. This is particularly clear when analysing surplus costs for the inclusion of all EVPS. When economic resources are limited, it could be reasonable to plan a nano hotspot network; on the other hand, if few areas are considered to deserve a particular attention, micro hotspots could be designed. Besides the already mentioned limits which could provoke the irremediable extintion of some species, if the advocated target of protecting all endemic species will be improved, drastic increases in connectivity and financial costs would be necessary if starting from an already-designed micro or nano hotspots network. Conversely, the integrated solution, would be a valid mid-step plan toward a gradual inclusion of all targeted species.

This paper showed an overview at regional scale on the areas and possibilities for plant conservation with the aim of enhancing the implementation of active conservation programs. In comparison with the Natura 2000 network, which comprises 18% of the Sardinian territory (Fois et al. unpubblished data), our integrated network of micro and nano hotspots would be a reasonable proposal of conservation at increasing protection level, as all the 295 endemic plants are included in less territory. Following this example, an integrated network of the Mediterranean micro and nano hotspots can be designed in order to more precisely and comprhensively face threats and to allocate effective protection efforts. This said, one must be very careful in choosing few large instead of many small patches, and decisions should be wisely taken at regional scale in order to conceive as many as possible aspects of each particular territory.

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CHAPTER VI

Current and future effectiveness of the Natura 2000 Network for protecting plant species in Sardinia: a nice and complex building in its raw state?

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This research is under review

Contribution to the project: I leaded all the analyses and the preparation of the manuscript. Giuseppe Fenu and I conceived the study design. All the authors contributed for the revision of the manuscript.

Abstract

Sardinia hosts 186 endemic plant species and represents an important centre of Mediterranean biodiversity. According to the threats acting upon its flora, 27 terrestrial vascular plants were listed in the international regulations and 124 sites for species and habitat conservation were designated. Here, analyses of gaps in the Natura 2000 network and current and future distribution of four representative plant species were carried out. Each plant population was georeferenced and the effectiveness of Natura 2000 network was compared according to the conservation status and distribution. Future species distributions were modelled by considering current climatic conditions and future scenarios. In apparent discordance with other results, we found that Natura 2000 network well represents most of plant species. This research shows a looking-forward survey on the regional effectiveness of protection measures which led us to confirm the necessity to enhance the current state of Natura 2000 network mainly by implementing local legislation and regulation.

Keywords: Conservation planning; Gap analysis; Maxent; Policy plant species; W-Mediterranean Basin.

1. Introduction

The increasing environmental degradation by human activities can no longer be ignored and immediate actions for the conservation of plant species are needed (Van Stichel 2008; Rossi et al. 2016). Accordingly, the development of large-scale networks of protected areas is one important tool for counteracting biodiversity loss and its negative impacts on ecosystems (Rodrigues et al. 2004; Orlikowska et al. 2016).

In the European Union (EU), the key biodiversity conservation tools are the Directives 2009/147/EC (Birds Directive), 92/43/EEC (Habitats Directive, HD hereafter) and the Natura 2000 network of protected areas (Maiorano et al. 2007; Pullin et al. 2009; Orlikowska et al. 2016). The Natura 2000 network promotes the conservation of biodiversity maintaining, or restoring, natural habitats and species in a favourable conservation status by means of cogent protection policies (European Commission 1992). All EU member states had to draw up lists of national conservation sites to be included in the European Natura 2000 network and to establish management plans and conservation measures to ensure the protection of all plant and animal species listed in the annexes to the HD, with particular regard to endemic species (Fenu et al. 2015a; Rossi et al. 2016). As required under the Article 17 of the HD, each EU member states is also in charge of monitoring the conservation status of habitats and species and reporting the results to the Commission every six years.

Currently, the Natura 2000 network encompasses more than 27,000 protected sites and over 788,000 km^2 (18%) of Europe’s landmass and almost 360,350 km^2 (6%) of the marine environments across all 28 EU member states and it is considered the largest coordinated multinational network of protected areas in the world (European Commission 2016).

While many conservation activities were carried out in order to enhance the efficiency of this network, some pitfalls were highlighted in recent years. For instance, implementation of conservation policies was often hampered by inadequate and/or obsolete scientific knowledge (e.g. Maiorano et al. 2007; Kati et al. 2015; Rossi et al. 2016) or the selection of the areas was biased by an incomplete representation of all target species’ requirements (e.g. Vellak et al. 2010; Bagella et al. 2013; Maiorano et al. 2015). In addition, the global current protected areas are traditionally determined spatially and environmentally under the assumption of relatively low changes in species distribution in the future (Araújo et al. 2011; Leach et al. 2013). While large-scale networks are likely to become particularly important in the face of ongoing climate change that threatens species’ abilities to adapt to geographical shifts in the distribution of suitable habitats (Araújo et al. 2011; Papanikolaou et al. 2014), Natura 2000 network is considered a relatively rigid network since the possibility of adding new sites or changing the location of existing sites is rather limited (Orlikowska et al. 2016). It is important, therefore, to be able to assess the likely effectiveness of a protected area network under future scenarios of climate change (Vos et al. 2008; Picketts et al. 2014; Papanikolaou et al. 2014). In this sense, Species Distribution Models (SDMs) have been widely used to estimate the potential impacts of climate change on species distributions and ecosystems (Franklin 2009). In particular, these models help to manage species facing possible future threats by identifying biological corridors for dispersal, determining sites for reintroduction and areas requiring protection measures (e.g. Fois et al. 2015; Ferrarini et al. 2016; López-Tirado and Hidalgo 2016).

In this paper, we present a comprehensive analysis on the effectiveness of Natura 2000 network under current and future climate scenarios in Sardinia (Western Mediterranean Basin). Using a representative study case of the Mediterranean Basin, we analysed how terrestrial plant species listed in the international regulations (Bern Convention and Habitats Directive) are protected in order to point out strengths and weaknesses of the current Natura 2000 network.

First, to gather the necessary information, all point occurrences of 27 terrestrial plant species were organized in a geodatabase. Then, a gap analysis was carried out to estimate the population percentages inside Natura 2000 sites. Last but not least, SDMs for four representative species, with different distribution patterns and conservation status, were applied in order to evaluate how such percentages of protection would change under future climate scenarios.

2. Material and methods

2.1. Study area and plant taxa selection

Sardinia (Italy) has been widely considered a representative center of Mediterranean plant endemisms (i.e. meso hotspot of plant diversity *sensu* Cañadas et al. 2014), being the second largest island of the Mediterranean Basin (covering 24,090 km^2) and hosting a peculiar biodiversity and environment (Fenu et al. 2014). The high proportion of endemic plants (approx. 13% of the total flora; Fenu et al. 2014) considerably increases up to about 35% in mountain peaks and uninhabited islets (Cañadas et al. 2014).

Because of the interest in the Sardinian biodiversity conservation, about 18% of its terrestrial area (aaprox. 4,523 km^2) has been included in 124 Natura 2000 sites (MATTM 2016). However, there are several examples of how some plants that should need active protection measures are still not represented in the Natura 2000 network (e.g. Bagella et al. 2013) or still experiencing an ongoing decline (e.g. Fenu et al. 2015b).

Species used for our study were all the 27 terrestrial plant species listed in the Bern Convention and in the HD annexes, which are present in the Sardinian territory. According to the terminology used in official Italian reports of the Joint Research centre (Ispra; e.g. Ercole and Giacannelli 2014) we refer to these species as “policy plant species”. The information about the number of populations were obtained from the dataset of the endemic and threatened flora of Sardinia developed throughout several years by the authors, while the conservation status at regional level was obtained from Rossi et al. (2016).

2.2. Gap analysis

In order to test how the policy plants of Sardinia were represented in Natura 2000 network, we applied a gap analysis (Scott et al. 1993). First, we overlapped the shapefile of the distribution data of each plant species to the shapefile of the Natura 2000 network (MATTM 2016) using Quantum GIS Development Team, version 1.7.4 (QGIS 2014). Second, the percentage of sites inside/outside the Natura 2000 network was evaluated for each plant species. Once groups of policy species were defined according to their number of populations and conservation status, differences among percentages were evaluated by Kruskal-Wallis Test performed in R environment (R Core Team 2014).

2.3. Species Distribution Modelling and variables selection

Species Distribution Models (SDMs) have been widely used to estimate the potential impacts of climate change on species distributions and estimate potential future extinction risks (Franklin 2009).

In our case, SDMs were assessed by using the MaxEnt algorithm and application version 3.3.3 (Phillips et al. 2006), developed to model species distributions with presence-only data, applying the principle of maximum entropy (Elith et al. 2006). Presence-only SDMs, as used here, are the best available option when absence data is not available and the number of data points is reduced (Vasconcelos et al. 2012; Vergílio et al. 2016). Data points were split into training and testing datasets, with 80 and 20% of the points, respectively. The accuracy of the resulting models was then assessed using the area under the curve (AUC) of the receiver operating characteristic (ROC; Elith et al. 2006). An AUC value of 1.0 indicates perfect discrimination ability and a value of 0.5 or less indicates a prediction no better than random (Phillips et al. 2006). In particular, Swets (1988) suggested the following interpretation: AUC>0.9: excellent agreement between observed and predicted distribution; 0.8<AUC<0.9: good; 0.7<AUC<0.8: fair; 0.6<AUC<0.7: poor; 0.5<AUC<0.6: fail. According to Vergílio et al. (2016), all final models of each species were also analysed by an expert researcher, based on their knowledge of the species. The remaining parameters of the MaxEnt application were maintained at default settings (Vasconcelos et al. 2012; Vergílio et al. 2016).

For the characterisation of the target plant species, all bioclimatic (11 layers of temperatures and eight precipitation indices) at 30 arc-seconds layers (~0.7 km of resolution at Sardinia’s latitudes) implemented by WorldClim (Hijmans et al. 2005) were considered. We only used bioclimatic variables rather than other physical environmental or land cover variables in the model in order to provide an indication of changing climatic suitability assuming all else remains equal (Porfirio et al. 2014; Bosso et al. 2016). In order to avoid the inclusion of highly correlated variables and to minimize overfitting, we computed the Variance Inflation Factor (VIF) values (Marquardt 1970). The VIF is based on the square of the multiple correlation coefficient resulting from regressing the predictor variable against all other predictor variables in order to measure how strongly each predictor can be explained by the rest of predictors (Marquardt 1970). We used the vifstep function from R package *usdm* (Naimi et al. 2014), which first calculated VIF values for all variables and then automatically excluded which has the greater VIF until no variable is correlated with another. The automated exclusion criterion was based on the precautionary threshold of VIF > 5 (Chatterjee and Hadi 2006). Consequently, the five selected bioclimatic variables were used to construct SDMs and defined according to WorldClim: Isothermality (BIO3), Temperature Annual Range (BIO7), Mean Temperature of Wettest Quarter (BIO8), Precipitation Seasonality (BIO15); Precipitation of Coldest Quarter (BIO19).

The same variables were used for SDMs under future scenarios of the most extreme Representative Concentration Pathway (RCPs) 8.5 proposed by the Intergovernmental Panel on Climate Change (IPCC) over the time period 2070 (average for 2061–2080). Among the 19 Global Circulation Models (GCMs) presented by IPCC for its fifth Assessment Report in 2014 (Rogelj 2013), we choose the GCMs implemented by the

Met Office Hadley Centre (UK, HadGEM2-ES) which offered the best performance for the Mediterranean Basin (Brands et al. 2013; Al-Qaddi et al. 2016; Bosso et al. 2016).

In order to investigate the present and future representativeness of Natura 2000 network, we selected four plant species with different conservation status, ecology and distribution. In particular, these taxa were: (1) *Euphrasia nana* (Rouy) Prain, an orophilous annual plant, endemic to Sardinia and Corsica, that occurs at altitudes from 1200 to 1800 m a.s.l., it was assessed as Critically Endangered (CR) in Italy under the IUCN criteria; (2) *Linaria flava* (Poir.) Desf. subsp. *sardoa* (Sommier) A. Terracc., an annual psammophilous plant, endemic to Sardinia and Corsica, that occurs at altitudes from 0 to 200 m a.s.l., it was assessed as Endangered (EN) in Italy under the IUCN criteria; (3) *Brassica insularis* Moris, a rupestrian perennial plant that occurs in Southwestern Mediterranean areas at altitudes from 0 to 1200 m a.s.l., it was assessed as Near Threatened (NT) in Italy under the IUCN criteria; finally, (4) *Ruscus aculeatus* L. a widespread perennial plant that occurs in Sardinia at altitudes from 0 to 1200 m a.s.l., it was assessed as Least Concern (LC) in Italy under the IUCN criteria (Rossi et al. 2016).

2.4. Present and future effectiveness of Natura 2000 network

Species Distribution Models (SDMs) were repeated for the four species under current and future scenarios. The final potential species distribution maps had a range of values from 0 to 1; a maximum training sensitivity plus specificity threshold (Liu et al. 2005; Riordan et al. 2015) was used to convert the logistic suitability maps to binary potential current and future species distribution maps for each species (1 = suitable, 0 = unsuitable). Maps were therefore plotted in GIS environment (QGIS 2014) and percentages of current and future potential distribution represented in Natura 2000 network were computed for the four analysed plant taxa.

3. Results

3.1. Gap analysis

The spatial analysis revealed that Natura 2000 network of Sardinia encompasses most of the richest areas in policy species (Fig. 1). All taxa concerned resulted having almost one locality inside this network. No statistical differences ($P > 0.05$) were pointed out by Kruskal-Wallis tests among taxa with a different conservation status and number localities. Indeed, an average percentage higher than 60% of populations of each species resulted inside the Natura 2000 network among all groups defined by conservation status and number localities. Nonetheless, two critically endangered (CR, *sensu* IUCN) plants (*Astragalus verrucosus* Moris and *Helianthemum caput-felis* Boiss.) and the widespread and less endangered (i.e. least concern, LC) *R. aculeatus* stood out as having more locations outside than inside the Natura 2000 network (Table 1). On the other side, the rest of CR taxa resulted completely inside protected areas (Table 1).

Table 1 List of all policy plant species considered for this study. The percentage of populations inside (In) or outside (Out) the Natura 2000 network are reported for each plant species. All species were classified according the range number of known populations (No.) and the respective IUCN category (IUCN; published by Rossi et al. 2016) and the Conservation Status (CS) at Italian level based on third reports of the European Environmental Agency (EEA, 2013).

Plant species	In (%)	Out (%)	No.	IUCN[1]	CS[2]
Astragalus verrucosus Moris	30	70	<5	CR	U1
Helianthemum caput-felis Boiss.	35	65	<5	CR	U1
Ruscus aculeatus L.	49	51	>20	LC	FV
Limonium insulare (Bég. *et* Landi) Arrigoni *et* Diana	54	46	<20	EN	U1
Linaria flava (Poir.) Desf. subsp. *sardoa* (Sommier) A.Terracc.	66	34	>20	EN	U1
Spiranthes aestivalis (Poir.) Rich.	66	34	>20	EN	U1
Selaginella denticulata (L.) Spring	77	33	>20	LC	NV
Marsilea strigosa Willd.	70	30	>20	LC	U1
Rouya polygama (Desf.) Coincy	70	30	<20	EN	U1
Orchis provincialis Balb. *ex* Lam. *et* DC.	72	28	>20	LC	NV
Anchusa crispa Viv.	63	27	<20	EN	U1
Carex panormitana Guss.	75	25	<20	EN	U1
Limonium pseudolaetum Arrigoni *et* Diana	75	25	<5	VU	U1
Brassica insularis Moris	77	23	>20	NT	FV
Linum mulleri Moris	67	23	<20	EN	U1
Pilularia minuta Durieu *ex* A.Braun	84	16	<10	VU	NV
Centaurea horrida Badarò	95	5	<20	EN	U1
Astragalus maritimus Moris	100	0	<5	CR	U2
Centranthus amazonum Fridl. *et* A.Raynal	100	0	<5	CR	NV
Euphrasia nana (Rouy) Prain	100	0	<10	CR	U1
Gentiana lutea L.	100	0	<20	NT	FV
Herniaria litardierei (Gamisans) Greuter *et* Burdet	100	0	<5	LC	FV
Lamyropsis microcephala (Moris) Dittrich *et* Greuter	100	0	<5	CR	U2
Limonium strictissimum (Salzm.) Arrigoni	100	0	<5	VU	U1
Ribes sardoum Martelli	100	0	<5	CR	U2
Salicornia veneta Pignatti *et* Lausi	100	0	<5	LC	FV
Silene velutina Loisel.	100	0	<5	NT	FV

[1] IUCN acronyms: CR, Critically Endangered; EN, Endangered; VU, Vulnerable; NT, Near Threatened; LC, Least Concern

[2] CS acronyms: U2, Unfavourable-Bad; U1, Unfavourable-Inadequate; FV, Favourable; NV, Not Valuated

3.2. Present and future effectiveness of Natura 2000 network

Predictions of the potential habitat suitability over current and future climatic conditions showed from excellent to fair (>0.7; Table 2) scores of AUC in three cases, whereas the most widespread plant species (*R. aculeatus*) scored a poor value (0.6<AUC<0.7; Table 1). In one case (*L. flava*. subsp. *sardoa*) the percentage of suitable areas will decrease and, in the case of *E. nana*, will approximately be the same (Fig. 2; Table 2). On the other hand, *E. nana* and *L. flava*. subsp. *sardoa* will be the two cases of increasing percentages of potential distribution covered by Natura 2000 sites in the future (Fig. 2; Table 2). The less-threatened species, according to their large number of populations and wide ecological and altitudinal ranges (*B. insularis* and

R. aculeatus), will also gain in future suitable areas under climate scenarios for 2070 (Table 2). Thus, reductions in Natura 2000 coverages are in these cases irrelevant.

Table 2 Data of the models for the four policy species selected: Number (No.) of points used, conservation status (IUCN) assessment obtained from Rossi et al. (2016) and discrimination ability scores (AUC). Reductions (-) or gains (+) of future suitable areas (Suit. Area) and their percentages inside Natura 2000 network (% inside) in comparison to the current values are also reported.

Policy Plant Species	**No.**	**IUCN Category**	**AUC (present)**	**AUC (future)**	**Future reduction(-) / gain(+)**	
					Suit. Area	**% inside**
Euphrasia nana	5	CR	1	0.999	+1.7%	+9.8%
Linaria sardoa	30	EN	0.925	0.972	-37.2%	+2.1%
Brassica insularis	26	NT	0.780	0.739	+63.5%	-9.7%
Ruscus aculeatus	305	LC	0.628	0.604	+79.1%	-2.1%

4. Discussion

The results of our research on policy plant species pointed out that the existing Natura 2000 network in Sardinia provides a high degree of coverage under current and future climatic conditions; however, this finding is controversial. With some exceptions (e.g. Mazaris et al 2013; Kallimanis et al. 2015), this disagrees with many other studies evaluating the effectiveness of the Natura 2000 network (e.g. Dimitrakopoulos et al. 2004; Maiorano et al. 2007; Mendoza-Fernández et al. 2009; Jantke et al. 2011; Bagella et al. 2013) that essentially found important gaps in where and how conservation areas were located.

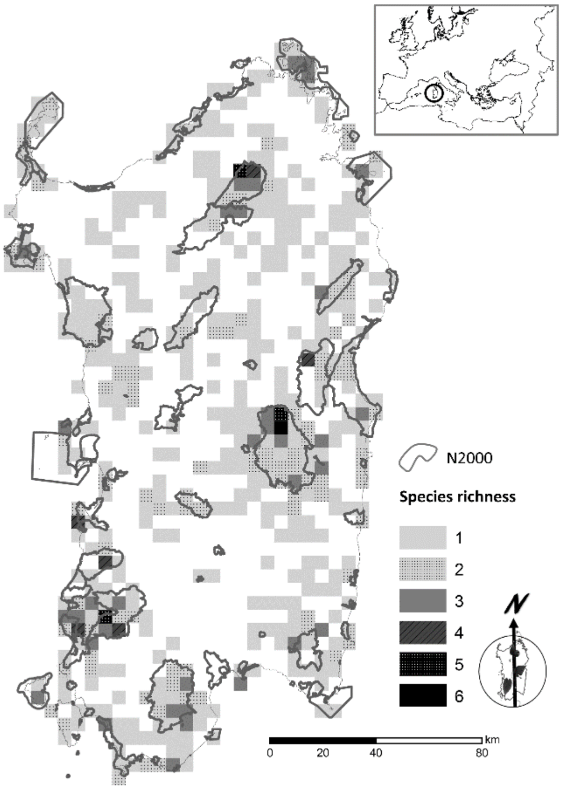

Figure 1. Policy plant species richness per grid cell (5×5 km) and the Natura 2000 network in Sardinia.

Also, the results of third Italian assessment of the conservation status of all policy plants (reported by the European Environmental Agency; EEA 2013) found that only 34% of plant species were in a favourable conservation status, while 50% were unfavourable and 16% unknown (Ercole and Giacannelli 2014). However, other recent reviews of gaps in conservation studies at European (Orlikowska et al. 2016) and Sardinian level (Fenu et al. 2015b) found that policy species, and especially plants, have been largely studied and monitored.

There are some points which deserve specific mention in order to explain such apparent controversy: first, the expected relationship between IUCN Red Lists and conservation measures (active and legal) is often not consistent (Mendoza-Fernández and Mota 2016). Second, most of studies that found a poor Natura 2000 network effectiveness (e.g. Dimitrakopoulos et al. 2004; Maiorano et al. 2007, 2015; Mendoza-Fernández et al. 2009; Jantke et al. 2011; Bagella et al. 2013) were based on species which are not listed in EU directives. These choices had the strong and right aim to highlight that most of endemic and/or threatened species are not currently listed in the most important directives for nature conservation in Europe. The first issue may thus be the updating of the lists to capture the changing species taxonomy, distribution and the description of new species (e.g. Bacchetta et al. 2012b; Hochkirch et al. 2013*;* Maiorano et al. 2015). According to the regional responsibility criterion (Bacchetta et al. 2012a), these problems are more related to a lack of local legislations rather than European (Rossi et al. 2016). In addition, the introduction in the listing process should strictly follow more objective criteria to identify species (and habitats) with higher risk of extinction (Bacchetta et al. 2012a; Hochkirch et al. 2013; Rossi et al. 2016). For instance, only 60 over the 151 Italian policy plant species were categorised as critically endangered or endangered at Italian level; this means that more than 60% of all legally protected plant species in Italy are essentially least or not threatened species. Similar problems were also previously highlighted by analysing, for example, all Sardinian endemic plants (Bacchetta et al. 2012a), lichens (Rubio-Salcedo et al. 2013), saproxylic beetles (D'Amen et al. 2013), and terrestrial vertebrates (Beunen 2006; Maiorano et al. 2015).

Besides these argumentations, many authors (e.g. Hochkirch et al. 2013; Fenu et al. 2015b; Rossi et al. 2016) have recently upraised another important problem: is legal protection sufficient to ensure plant conservation? All these authors agree that at the state of the art legal protection is often not followed by effective conservation measures; this is mainly due to a lack of a systematic conservation planning and financial resources (Kati et al. 2015). In Sardinia, some straightforward examples could be the population extinctions occurred in the last 20-years period within the Natura 2000 network [i.e. the cases of *Silene velutina* Loisel. (Bocchieri 1998), *Centranthus amazonum* Fridl. *et* A.Raynal (Fridlender 2006) and *Anchusa crispa* Viv.(Bacchetta et al. 2008)]. In line with all these statements, also predictions of plant responses to future climate-change conditions underscored that not all plant species will face a reduction in habitat suitability and Natura 2000 coverage. This is particularly clear for mid-altitudes and/or policy plants with a wide ecological and altitudinal range (e.g. *B. insularis* and *R. aculeatus*) which, according to the general expectation of increase in temperatures and decrease in annual rainfalls in Southern Europe (Giorgi and Lionello 2008; IPCC 2007), will colonise further areas towards higher elevations. As a consequence, being mountain systems the most covered areas by Natura 2000 network in Sardinia, also due to a lower economical interest

(Fenu et al. 2015b), global warming would sometimes even imply a concentration of potential distribution covered by Natura 2000 network for narrow species such as *E. nana* and *L. flava* subsp. *sardoa*.

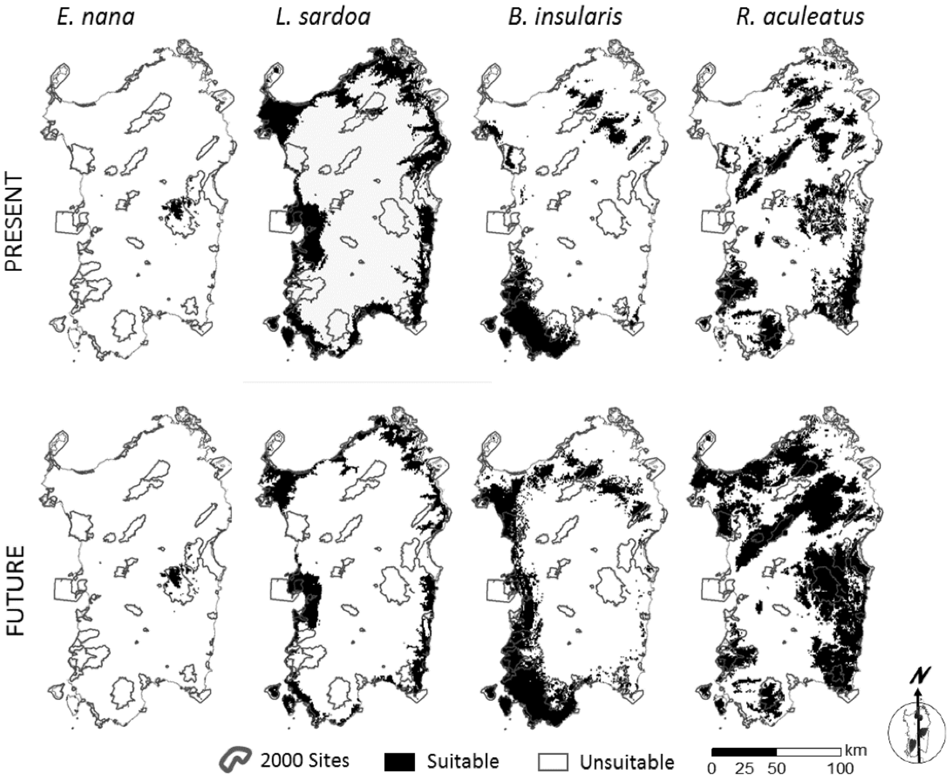

Figure 2. Present (1960-1990) and future (2061–2080) suitable and unsuitable areas according to the results of species distribution models. The same five climatic variables (Isothermality, Temperature Annual Range, Mean Temperature of Wettest Quarter, Precipitation Seasonality, Precipitation of Coldest Quarter) and plant species (*Euphrasia nana*, *Linaria flava* susp. *sardoa*, *Brassica insularis* and *Ruscus aculeatus*) were used for both time windows. See Table 2 for specific values.

Nonetheless, our cases confirmed the general expectation that climate changes will particularly determine a distribution reduction of orophilous species (like *E. nana*), which are already in the altitudinal/ecological extremes of the Sardinian territory (Gentili et al. 2015; Fois et al. 2016), and coastal/psammophilous species (like *L. flava* subsp. *sardoa*), which would face morphological and climatic limits of dispersion enforced by sea level changes (Bellard et al. 2014). In particular, these latter species seem to deserve a particular attention highlighted by a general underrepresentation coastal species (see also the cases of the coastal species *A. verrucosus*, *H. caput-felis* and *Limonium insulare;*Table 1) and an overexploitation of coastal habitats mainly due to the tourism-related economic development and the consequent high human pressures (Fenu et al. 2015a). In this sense, further predictions including mechanisms related to land use change, such as grazing pressures or land abandonment, would enhance the predictability of future conservation status trends. Indeed, many studies (e.g. Lehsten et al. 2015; When and Johansen 2015) demonstrated that also land use change may have an effect (not always negative) on species diversity.

5. Conclusions

Although Natura 2000 network coverage may be satisfactorily considered for covering the localities of the Sardinian policy flora, many pitfalls were also found in our case. To our opinion, the necessary process of enhancing the effectiveness of Natura 2000 network should consider the limited financial resources (Fenu et al. 2015b). Indeed, it is demonstrated that many already designated areas for legal protection in the Mediterranean Basin are not effectively well managed (e.g. Mendoza-Fernández et al. 2009; Fenu et al. 2015b; Pellegrino et al. 2016).

As an opposition from various stakeholder groups was experienced in the selection of first protected areas (Blicharska et al. 2016; Geitzenauer et al. 2016), the selection and management of Natura 2000 protected sites resulted in a reduction in area and less stringent implementation measures, which allowed to increase their representativeness by including lands with high economic values (Kati et al. 2015). In comparison to non-Natura 2000 sites, this important achievement was also observed in Sardinia. Indeed, our analyses found that only few Natura 2000 sites along the Sardinian coast would be necessary to enhance the current network.

Instead of consistently increasing the area covered by protected areas, we therefore propose to better focus the limited resources towards most threatened species/habitats. To do that, the species and habitats listed in international directives should be supplemented by regional lists (at national or regional administrative level) and based on a complete and objective conservation status assessment (Mendoza-Fernández et al. 2009; Bacchetta et al. 2012a).

Secondly, a preliminary protected area designation should imply a detailed presentation of all specific goals and the methodologies and economic resources that will be applied to achieve such goals (Gil et al. 2011). In order to reduce costs and enhance the effectiveness, local stakeholders should be largely informed and population acceptance and participation should be of pivotal importance (Gil et al. 2011; Kati et al. 2014; Blicharska et al. 2016).

In resume, it seems that Natura 2000 network is a nice, big and complex building in its raw state; to our opinion, this structure should be improved by making EU directives more pliable and by the implementation of local legal measures. However, we believe in the great potential of Natura 2000 network and we hope that our results and suggestions will be of help to enhance such important network, rather than suggesting the development of a completely new strategy.

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SUMMARY

This thesis includes six manuscripts with methodological, empirical and theoretical contributions that aim to enhance the understanding of endemic vascular plant species distribution patterns and their underlying drivers. Being Sardinia a representative study case, Mediterranean islands systems are in the special focus of this work. Due to their ecological and distributional constraints, as often it happens, also many endemic vascular plants are of conservational interest in Sardinia. Consequently, all the here exposed researches had the final aim of providing tools for a more effective conservation planning.

In **chapter I**, we analytically and spatially considered all the activities (*in situ* and *ex situ*) implemented for the conservation of exclusives and policy plant species of Sardinia during ten years (2004-2014); this was the first steppingstone towards further analyses. In particular, we found that while conservation status assessments and *ex situ* conservation activities are likely to be applied for all exclusive and policy plants by 2020, *in situ* conservation activities, such as fencing and monitoring, are still far from this target, especially due to the limited financial resources. This was particularly clear for the 186 exclusive plants that, besides their greater number compared to the 32 policy species, are often distributed in difficult and expensive to reach localities (e.g. satellite small islands and screes). This was confirmed by spatial gap analyses that suggested to intensify conservational efforts in satellite small islands and mountains.

These first findings led us to deepen on drivers of the peculiar endemic plant species distribution in **chapter II**. In particular, are the current areas rich in endemic species the result of evolutionary/ecological or human-driven processes? Results obtained by analysing all the Sardinian territory did not completely answered this question as elevation explained alone more than all the rest of the ecological and anthropological factors considered. Indeed, the endemism richest areas at high elevations are characterised by isolation, which could facilitate evolutionary processes but also a lower intraspecific competition and human presence. In addition, most of the variance was still unexplained at regional scale level. Further analyses in this sense were thus focused on more specific issues.

In **chapter III**, we used historical information to disentangle the drivers of plant extinctions occurred from 1960' up to more recent years. Besides the confirmation of elevation as an important driver of the distribution pattern of plant species (also exinct), further interesting factors were outscored. Indeed, extinctions were in many cases recorded in localities occurring in species-specific ecological limits, highlighting the importance of ecological studies, such as the climate change effects, for biological conservation. Additionally, we found significant correlations among species extinction and human factors. For instance, the extinction localities of therophytes were correlated with fire frequency or the presence of streets were more important in determining extinctions of coastal than mountain species. In this chapter, we also presented an experimental approach for species distribution models using extinction occurrences as presence data. We believe these results may be interesting as they highlighted areas where species would be present if causes of extinctions did not occurred.

In **chapter IV**, we were inspired by the previously highlighted insights on the conservational importance of satellite islands. Fist, we subdivided the set of 82 satellite small islands of Sardinia into three groups (relatively big islands, islands with high slope and small and flat islands) in order to “mask” the area and elevation effects in determining endemic species richness; this allowed to find further useful insights for conservation planning.

Indeed, we highlighted that all satellite islands, and yet especially islands with high slope and small and flat islands, represent a great opportunity where effective conservation practices could be placed, as these are usually richer in endemics than the main island coast, and have a lower economic interest and human accessibility.

According with previous chapters' findings, the micro and nano hotspots identified in **chapter V** were mainly concentrated in areas at higher elevations and in the satellite islands. Nonetheless, it could be also necessary to find a compromise in order to cover all the biological diversity, also in places with high economical interests. To do that, we demonstrated that an integrated network of micro and nano hotspots could be a feasible solution. Finally, further insights on conservation planning were highlighted in **chapter VI** by analysing the effectiveness of the Natura 2000 network in representing policy species under current and future conditions. Like the integrated network of micro and nano hotspots, Natura 2000 sites resulted well distributed across the territory and representing the analysed species also in the future. Nonetheless, policy species are only a part of the species that should deserve protection. In this sense, legislations at local level (regional and national) should integrate European laws in order to enlarge the list of policy species (including, at least, endemisms) and, accordingly, to rearrange the limits of protected sites in line with the areas highlighted in chapter V. In addition, as discussed in chapter I, the designation of protected sites is not always accompanied by the effective protection of the species inside. Further investments should be thus targeted in this sense.

This project represents only a step further towards a more comprehensive analysis on the distribution pattern of endemic plant species in Sardinia. Many other researches based on the geodatabase of plant species are currently in progress. For instance, the use of species distribution models could guide investigations on new plant species occurrences, as demonstrated with the study case of *Gentiana lutea* (Annex I). Additionally, conservation status assessments may be integrated by the analyses of the potential reductions under future climate changes, following the example provided in Annex II. In conclusion, an effective conservation planning should take as much as possible into account the economic and social aspects. To do that, land values could be approximated in monetary terms through the modelization of selling offers provided by real estates (Annex III). This output could provide a basis for further researches for more effective conservation planning at regional scale.

ANNEXES

ANNEX I - A practical method to speed up the discovery of unknown populations using species distribution models

Mauro Fois, Giuseppe Fenu, Alba Cuena-Lombraña, Donatella Cogoni, Gianluigi Bacchetta

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Abstract: Species Distribution Models (SDMs) could be an important tool to limit search efforts by selecting the areas where field surveys are to be carried out; due to the constant decrease of financial funds, this challenging purpose is particularly necessary. In particular, these methods are useful when applied to endangered and/or rare species with a poor known distribution area, especially due to difficulties in plant detection and in reaching the study areas.We hereby describe the development of maximum-entropy (Maxent) models for the endangered yellow gentian *Gentiana lutea* L. in Sardinia with the aims of (i) guiding survey efforts; (ii) estimating SDMsutility by post-test species current/extinct localities through the Observed Positive Predictive Power(OPPP) values; and (iii) evaluating the influence of sample data addition. Besides the Area Under Curve(AUC) values, we used the OPPP (observed/modelled positive localities ratio) to compare results from eight, 24 and 58 presence-only data points. Even with the initial small and biased sample data, we found that surveys could be effectively guided using such methods, whereby the focus of our research was on 48% of our initial 721 km^2 study area. The high OPPPs values additionally proved the reliability of our results in discovering 16 new localities of *G. lutea*. Nevertheless, the predictive models should be considered as a complementary tool rather than a replacement for expert knowledge.

Keywords: Field efforts; *Gentiana lutea*; Maxent; Positive Predictive Power; Sardinia; Threatened vascular flora

ANNEX II - The reliability of conservation status assessments at regional level: Past, present and future perspectives on *Gentiana lutea* L. ssp. *lutea* in Sardinia

Mauro Fois, Alba Cuena-Lombraña, Giuseppe Fenu, Donatella Cogoni, Gianluigi Bacchetta

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Abstract: Regional assessments could be more reliable than global ones, especially for species in peripheral parts of their range. In this sense, distribution criteria proposed by the IUCN, expressed by the Extent of Occurrence (EOO) and the Area of Occupancy (AOO), are of critical importance to evaluate the risk category at any scale. Our study was focused on the yellow gentian (*Gentiana lutea* L. subsp. *lutea*), a rhizomatous mountain plant of central and southern Europe, included in the Annex V of the Directive 92/43/EEC. The exhaustive and updated regional distribution and the recently published extinct localities, allowed for review of option to increase the number of criteria used for its regional conservation status assessment. In particular, the current restricted EOO and AOO extents and reductions occurred during the last three generations led this plant being considered as Endangered (EN), according to the IUCN sub-criteria A2, B1 and B2. Additionally, Species Distribution Models based on the extant localities and the respective current and future climate scenarios (2050 and 2070) predicted a decrease of its environmental niche reducing its altitudinal range towards higher elevations. This reduction was greater than 50% and confirmed the regional conservation status of EN, also according to the sub-criterion A3.

This paper shows the reliability of regional risk assessment of peripheral and/or narrow populations, which is an effective tool for helping the development of the necessary conservation management, underling a diversified set of solutions for each specific past, present and future threat that could occur in each locality.

Keywords: Climate change, Gentianaceae, IUCN, Maxent, Mediterranean Basin

ANNEX III- Using a Species Distribution Model approach to map land acquisition costs: a supporting method for implementing the economic complexities in spatial conservation planning

Mauro Fois, Giuseppe Fenu, Gianluigi Bacchetta

This research is currently under review

Abstract: There is a growing urgency for integration and coordination of environmental, biological and anthropogenic indicators required to assess a cost-effective conservation planning. While cost estimation is a very positive and constructive contribution and most common tools for spatial conservation prioritisation are conceived to use it, there are a handful of examples where costs in monetary terms are applied as limiting targets. In this study, we presented a practical and repeatable method to model the acquisition field cost using common ecologists' techniques. For each one of the eight Sardinian administrative provinces, we modelled information obtained by several real estate's agents (N = 337) with a variegate set of eleven environmental, climatic and anthropogenic variables. Finally, we obtained an acquisition cost map of the entire island of Sardinia (Western Mediterranean Basin) with a resolution of 1 km^2. In most of cases, land cover was the main influencing factor, but further variables were driving costs according to each socio-economic specificity, which characterise every single administrative province. People and their economic interests could be modelled as though another species in order to apply such information for applicative researches in biodiversity conservation planning. As any model's result, this also showed uncertainty; otherwise, our method offers an alternative perspective on the challenges of adopting a species distribution modelling approach to extrapolate useful information, which otherwise is often unobtainable or expensive to achieve. We thus invite any biologist concerned to apply our output for further researches or to replicate this experiment in other areas.

Keywords: Sardinia, Land prices modelling, Generalized Linear Models, Spatial conservation prioritisation, Cost-effectiveness, Decision making

ANNEX IV - Using endemic-plant distribution, geology and geomorphology in biogeography: the case of Sardinia (Mediterranean Basin)

Giuseppe Fenu, Mauro Fois, Eva M. Cañadas, Gianluigi Bacchetta

This research has been published in 2014
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Abstract: The importance of robust systems for classifying biogeographical patterns has been emphasized for its usefulness in designing conservation strategies. For such purposes, the distribution patterns of the endemic flora have often been used. Several studies have identified phytogeographical units within Sardinia (western Mediterranean); however, the main part of the island remains unstudied. Thus, the aim of this study is to lay out a comprehensive biogeographical scheme for Sardinia based on endemic vascular plant distributions, together with geological and geomorphological units. We georeferenced, in a 1-km^2 grid cell, the presence of 290 vascular endemic taxa from the literature, herbarium specimens and field investigators' research. Sardinia was subdivided into 31 homogeneous units through the integration of geological and geomorphological maps and, subsequently, a presence–absence matrix of endemic taxa in each unit was built. Hierarchical cluster analysis was performed to define two levels of biogeographical units (i.e. sectors and subsectors). For each unit the exclusive and differential endemic taxa were identified. For sectors, indicator species were explored by the Indicator Value (Ind Val) analysis and relationships were analysed by quantitative interaction web. A total of six sectors and 22 subsectors were identified. The highest endemic plant richness was found in the Campidanese-Turritano, Sulcitano-Iglesiente and Supramontano sectors, and in the Gennargenteo, Barbaricino, Iglesiente and Sulcitano subsectors. All sectors were characterized by the presence of exclusive, differential and indicator taxa. The interaction analysis showed the highest uniqueness in endemic flora in the Supramontano and Sulcitano-Iglesiente sectors, which hosted a high number of exclusive endemic species. Mostly mountainous sectors/subsectors had higher endemic-species richness compared with lowland ones. The study showed the relevance of geology and geomorphology, together with accurate data on endemic distribution, to define consistent phytogeographical units. Furthermore, the biogeographical scheme presented here helps to define area-based conservation strategies in Sardinia.

Keywords: Conservation, Continental island, Endemic plant richness, Geology, Geomorphology, Mediterranean vascular flora

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