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Patchy landscapes support more plant diversity and ecosystem services than wood grasslands in Mediterranean silvopastoral agroforestry systems

Simonetta Bagella [a,b,*], Maria Carmela Caria [b], Giovanna Seddaiu [b,c], Laura Leites [d], Pier Paolo Roggero [b,c]

[a] Department of Chemistry and Pharmacy, University of Sassari, Sassari 07100, Italy
[b] Desertification Research Center, University of Sassari, Sassari 07100, Italy
[c] Department of Agricultural Sciences, University of Sassari, 07100, Italy
[d] Department of Ecosystem Science and Management, The Pennsylvania State University, University Park, PA 16802, United States

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ABSTRACT

Plant diversity and proxy indicators of ecosystem service were assessed for three structural components of Mediterranean silvopastoral agroforestry systems: WL = cork oak woodlands, WG = cork oak wood grasslands, and OG = open grasslands. Our study was conducted in a long-term observatory located in NE Sardinia, characterized by fragmented land ownership and land use, which generates a patchy landscape that is different from the extensively studied large scale Iberian *dehesas* and *montados*. Our research question was focused on assessing whether, a "patchy" land use scenario made of the combination of WL, WG and OG or a "specialized" scenario could provide more plant diversity and ecosystem services than a "*dehesa* type" scenario including only WG under the same overall tree cover. The results showed that γ and β diversity, species unique to a position, C stock, cork and acorn production, Hemicryptophytes cover, nectariferous species cover, decreased and pastoral value, excellent and good forage species and legume cover, increased along the WL➔WG➔OG land use gradient. Isolated trees in WG and clearings in WL highly contributed to achieving mainly high C stock and plant diversity respectively. The results also showed that the "specialized" scenario can support higher biodiversity and better ecosystem services than the "*dehesa* type" scenario, but the "patchy" scenario made of all three components proved to support the highest level of both biodiversity and ecosystem services in Mediterranean silvopastoral agroforestry systems.

1. Introduction

In Mediterranean countries, oak-wood grasslands (WG), oak-dominated woodlands (WL), and open grasslands (OG) are the key structural components of a typical silvopastoral agroforestry farm (García de Jalón et al., 2018). These components correspond to contrasting land uses and management practices of the silvopastoral agroforestry systems that combine extensive livestock grazing and forestry activities (McAdam et al., 2009). WL, WG and OG are spatially clustered ranging from small fragments of WL or WG in between OG and croplands to large areas with homogeneous WG cover (den Herder et al., 2017). In the EU27 (Croatia not yet included) WG cover some 15.1 Mha, i.e. 3.5% of the land, and represent some 15% of the European grasslands and 35% of the grazed land (den Herder et al., 2017). WG in the context of Mediterranean silvopastoral systems are very common in Spain (3.04 Mha; García de Jalón et al., 2018) and Portugal (about 0.7 Mha; AFN, 2010), where they are named *dehesas* and *montados*, respectively. They are also frequent in other Mediterranean areas such as Greece (1.6Mha) and Italy (1.4 Mha; Paris et al., 2019). The tree cover is a key feature of WG and is the outcome of the co-evolution of human activities and the natural ecological processes (Blondel, 2006). Sparse trees in the grasslands are recognized to generate spatially heterogeneous habitats providing a variety of ecosystem services and supporting different levels of biodiversity (Pinto-Correia, 2000; Koniak et al., 2011; Ribeiro et al., 2014; Schippers et al., 2015).

A high proportion (45% of the articles) of the available literature on agroforestry systems in Mediterranean Europe is focused on the spatially extensive WG in Spain and Portugal (Torralba et al., 2016; Moreno et al., 2018; Torralba et al., 2018), whereas little attention has been paid to silvopastoral systems in other Mediterranean regions (e.g.; Rossetti et al., 2015; Seddaiu et al., 2018). According to the 2018 EU CORINE Land Cover (CLC) program (EEA, 2018), silvopastoral land uses of Sardinia covered in 2018 some 167,000 ha (6,9%), with a net change of −17,000 ha (−10%) when compared to 1990 (EEA, 1990). This land use change was mainly due to a − 38,000 ha (−21%) change of the WG converted to croplands (31% of the loss) or

* Corresponding author at: Department of Chemistry and Pharmacy, University of Sassari, Sassari 07100, Italy.
E-mail address: bage@uniss.it (S. Bagella)

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to the natural succession towards WL (60%) following abandonment, and to the +21,000 ha of WG deriving from the abandonment of croplands and grasslands (80% of the gain), or the conversion of WL into WG (20%).

Traditional Mediterranean silvopastoral systems are characterized by low technology levels, low agronomic inputs, and relatively low productivity and profitability (Plieninger et al., 2006; Moreno and Pulido, 2009; Fagerholm et al., 2016). From the livestock farmer's perspective, the forage and the associated animal products such as milk and meat are provisioning services of primary importance of these ecosystems. Other provisioning services, including the production of cork, honey, mushrooms, wild-edible fruits and medicinal and aromatic plants (Moreno et al., 2018) can be easily monetized while supporting (e.g. nutrient cycling and water regimentation) or regulating (e.g. pollination or soil erosion prevention) services are less perceived and rarely identified (Camilli et al., 2017; Kay et al., 2019; Rolo et al., 2020). Comprehensive analyses of the literature on ecosystem services provided by European agroforestry systems were recently carried out by Torralba et al. (2016), Fagerholm et al. (2016) and Kay et al. (2019) who, in particular, assessed the economic performance of marketable and non-marketable ecosystem services, showing the relevance of the latter in the net economic balance, particularly for the Mediterranean agroforestry systems. The sustainability of Mediterranean silvopastoral systems based on wood grasslands has also been seriously questioned because of grassland management constraints, the lack of tree regeneration (Plieninger et al., 2010; Rossetti and Bagella, 2014) and the lower grassland productivity in the areas shaded by the trees when compared to the OG (Seddaiu et al., 2018). A high tree cover in WG is therefore perceived by farmers as constraining if compared to specialized agroforestry land uses, i.e. a combination of WL and OG (Camilli et al., 2017).

The landscape heterogeneity and the ecological functioning of Mediterranean silvopastoral systems are structurally coupled with the spatial distribution of farm management practices and land ownership (Pornaro et al., 2019), that can result into landscapes ranging from homogeneous WG with large scale property estates, to patches of WL, WG and OG (Fig. 1) emerging from fragmented land ownership patterns and diverging long-term trends towards abandonment and intensification (Plieninger, 2006). The spatial distribution of farm management practices is also constrained by environmental factors such as soil fertility gradients, rocky outcrops, and forested habitats areas (Ribeiro et al., 2014). Farm managers may choose diverging approaches, either modifying the landscape through intensive management practices such as tree clearing, fertilization, tillage and seeding to suit primarily the livestock farming goals or adapting farm management to existing landscape constraints through low-intensity management options like weed chopping, conservation tillage or sod seeding. Such choices can have relevant implications on plant diversity and ecosystem services (Ribeiro et al., 2016).

The information on key indicators of the ecological dynamics of Mediterranean silvopastoral systems is relevant for supporting science-based decisions on the design of sustainable development pathways and agri-environment schemes (Santos-Martín et al., 2013; Guerra and Pinto-Correia, 2016; Ribeiro et al., 2016; Moreno et al., 2018). This is particularly important for the site- and time-specific management and for the prevention and mitigation of current and future environmental hazards .

We hypothesized that a silvopastoral agroforestry landscape made of patches of WL, O … n of specialized land uses based on just … ubstantial changes in terms of plant diversity and ecosystem services of large-scale landscapes based on only WG, such as the ones of the Iberian *dehesas* and *montados* (Fig. 1), even if these landscapes share a similar average tree cover. This research was thus designed to improve our understanding of the relationships between contrasting management options, plant biodiversity, and proxy indicators of ecosystem services of silvopastoral systems in a Mediterranean hotspot of biodiversity (Médail and Quézel, 1999; Myers et al., 2000): the Sardinia island (Italy). In the studied area, traditional silvopastoral systems are the key component of the rural economies, which are based on dairy products from sheep and goat grazing or meat or milk from beef cattle grazing (Porqueddu et al., 2016).

More specifically, the research aimed to compare plant diversity and proxy indicators of ecosystem services of silvopastoral systems at the scale of homogeneous management units (WL, WG, OG), or landscape scale, considering contrasting scenarios characterized by different proportions of WG, WL, and OG under the same overall tree cover.

2. Materials and methods

2.1. Study area

The study was conducted in the long term observatory established in 2008 in Berchidda-Monti (NE Sardinia, Italy) at 250–300 m a.s.l. covering an area of approximately 20,000 ha (Fig. 2). Various management units (i.e. farm fields) in the observatory are being surveyed over time to monitor farm practices and indicators of biodiversity and ecosystem services (Bagella et al., 2013a, 2013b, 2014; Seddaiu et al., 2013; Rossetti and Bagella, 2014; Tardy et al., 2015; Bagella et al., 2016; Pulina et al., 2018; Seddaiu et al., 2018).; Bagella et al., 2016; Tardy et al., 2015; Rossetti and Bagella, 2014;;). The average 30-years annual precipitation is 632 mm, with at least 70% of rain falling from October to May. The aridity index (annual rainfall/annual reference evapotranspiration) is 0.53 and the mean annual temperature is 14.2 °C (Bagella et al., 2013a). The soil type, originating from granite, is very homogeneous in the area and was clas-

Fig. 1. WG landscape of the Spanish *dehesa* (left) and patchy silvopastoral agroforestry landscape in Sardinia (right), where the white polygons indicate a sample of WL (up), WG (middle) and OG (down). Images are taken from Google Earth Pro and are 1.5 × 1.0 km in size.

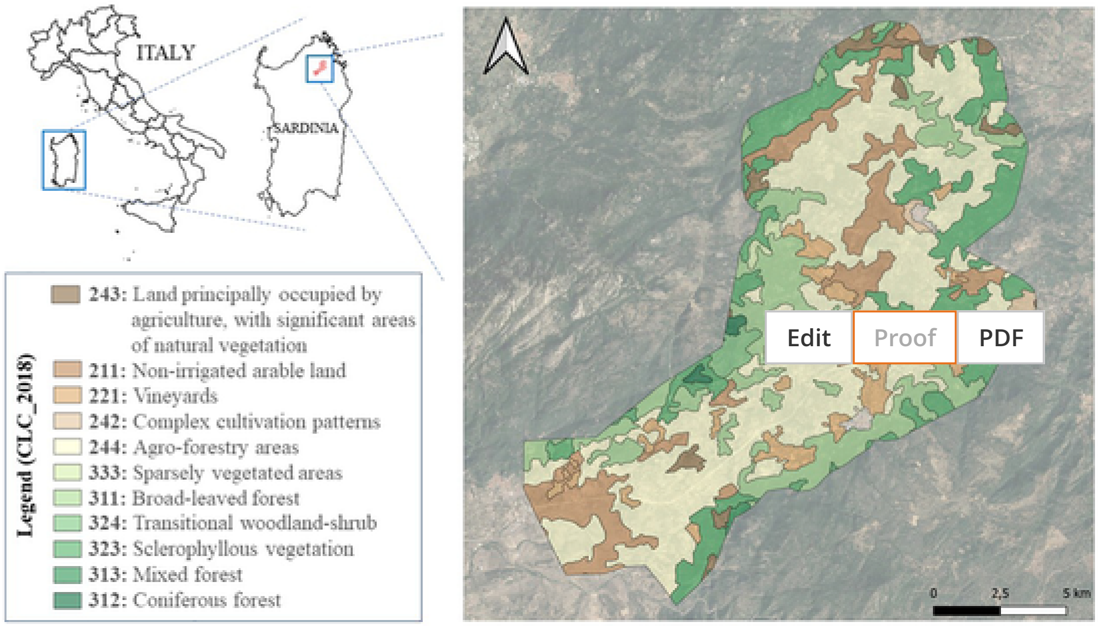

Fig. 2. Land uses according to CLC 2018 in the Long-term Observatory of Berchidda-Monti (NE Sardinia, 40°44 ÷ 56 N; 9° 10 ÷ 24′E). Wood grasslands (WG) and open grasslands (OG) were sampled within the areas classified by CLC 2018 with code 244; woodlands (WL) were sampled within the areas classified by CLC 2018 with code 311 or 323.

sified as Typic Dystroxerept (USDA, 2010). Soil texture in the Ap horizon is sandy loam with an average pH of 5.7, 2.3% organic C and 0.2% total N (Seddaiu et al., 2013). The natural potential vegetation of the studied area is represented by cork oak woods of the *Violo dehnhardtii-Quercetum suberis* association (Bagella and Caria, 2011).

The livestock feeding is mainly based on wood grasslands grazing as the main source: hay crops, fallow grasslands, and occasionally shrubs, trees, and acorns are grazed all year round following a mix of continuous and rotational grazing schemes. Sheep are occasionally sheltered overnight when necessary in winter. Grassland grazing on average contributes to 73% of the total diet of dairy sheep (Sarda breed) and 82% of the diet of the crossbred beef suckler cattle (Bagella et al., 2014). Stocking rates range from 0.7 to 1.5 Livestock Units (LU) ha^{-1} for sheep and from 0.4 to 0.8 LU ha^{-1} for cattle (1 LU = 500 kg cattle live weight = 6.6 dairy ewes) (Bagella et al., 2013a).

According to CLC 2018 (EEA, 2018), agroforestry areas in the long-term observatory (Fig. 2) covered 44% of the total land, with a net change of −717 ha (−1047 ha loss +330 ha gain; −7.6% of agroforestry areas in 1990) when compared to CLC 1990 (EEA, 1990) mainly because of the abandonment of croplands (39% of the loss) or the encroachment of abandoned WG (60% of the loss). In the same period, some new 330 ha of land was converted to WG mainly from the abandonment of croplands (82% of the gain) or the WL selective cut (17% of the gain). Such dynamics are consistent with the ongoing trend of intensification and abandonment that are threatening Mediterranean WG (Pinto-Correia and Mascarenhas, 1999).

Low (WL), medium (WG) or high (OG) land use intensities were attributed to the three structural components of silvopastoral systems in relation to the level of agronomic inputs (e.g. tillage, cutting, fertilization) and grazing management intensity required to maintain their long-term stability. In WL, the woody vegetation is interrupted by small and scattered clearings associated to the cork oak extraction practices, that are occasionally grazed. WL were not fertilized nor tilled for at least 80 years and emerged from the selective cutting of the wood understory to facilitate the access of cork cutters. In WG, the grassland is interrupted by patches of scattered adult cork oak trees. WG areas are grazed and, at 5 to 10 years intervals, they are oversown with annual hay crops (Rossetti et al., 2015). OG are characterized by a continuous herbaceous cover and are grazed and cultivated at 1–5 years intervals with annual hay-crops (Tardy et al., 2015).

2.2. Field experimental layout

The experimental layout was designed to compare WL, WG and OG at the scale of homogeneous management units (field scale). Two sampling positions within WL and WG were considered: open areas (WLc = clearings in the woodland; WGo = outside the tree crowns in the wooded grassland) and areas underneath the crown of the oak trees (WLu for the woodland; WGu for the wooded grassland) (Fig. 3). The WGu positions were selected within the horizontal projection of the tree canopy onto the ground in a North-East to South-west linear transect because the dominant wind in the study area is from South-West and the trees have typically flag-shape crowns.

For each of the five sampling positions under comparison (WLu, WLc, WGo, WGu, and OG), nine replicates were randomly selected. Overall, the surveys were made on a total of 45 sampling units (Table 1). The sampling areas were randomly selected assuming that they were large enough to minimize edge effects and be representative in terms of vegetation of the selected patchy land uses in the landscape.

2.3. Field measurements

Plant diversity and ecosystem services were assessed based on tested and comparable indicators. Plant assemblage composition, α, β, and ɣ diversity were used as biodiversity indicators. A set of proxy indicators for ecosystem services were selected following the Common International Classification of Ecosystem Services (Haines-Young and Potschin, 2018): Pastoral Value (PV) was used as a proxy indicator of forage production; cork and acorn production as a proxy indicator of productions of the oak tree component; the cover of excellent and very good forage plant species as a proxy indicator of high-value genetic resources conservation; legume cover as a proxy indicator of external natural nitrogen input to the ecosystem; the cover of hemicryptophytes as a proxy indicator of soil protection; the plant nectariferous value as a proxy indicator of potential honey production and pollination service; the soil Carbon stock (SOC), live tree Carbon stock (LTC) and total Car-

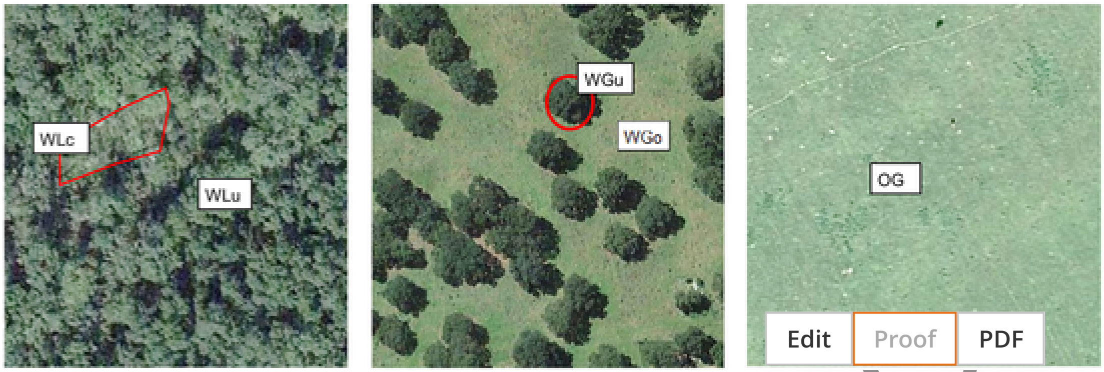

Fig. 3. Areal photographies illustrating the three land use types in the study area and the relative sampling positions. WLu = Underneath tree canopy in the woodlands; WLc = Clearings in the woodlands; WGu = Underneath tree canopy in the wood grasslands; WGo = Open areas in the wood grasslands; OG = Open grasslands. Images are taken from Google Earth Pro.

Table 1
Experimental design.

Land use	Sampling position	No. of sampling units
Woodlands (WL)	Underneath tree canopy in the woodlands (WLu)	9
	Clearings in the woodlands (WLc)	9
Wood grasslands (WG)	Underneath tree canopy in the wood grasslands (WGu)	9
	Open areas in the wood grasslands (WGo)	9
Open grasslands (OG)	Open grasslands (OG)	9

bon stock (TOC) as proxy indicators of regulatory ecosystem services (Layke et al., 2012).

Vegetation surveys were repeated in spring 2011 and spring 2012 at each sampling position. Data were taken inside 2 m × 2 m sampling units randomly located in the herbaceous or understory layer within each sampling position, except for the WLu sampling position, were a 10 m × 10 m sampling unit was used for the sampling of the understory tall shrubs. All plant species rooted inside the sampling units were identified and assigned a cover score (Braun-Blanquet, 1951). Plant identification and nomenclature follow the free online application for plant identification http://dbiodbs.units.it/carso/chiavi_pub21?sc=624. The average yearly species cover in each sampling position was used for building a species/cover matrix for data analysis.

The soil was sampled in May 2011 at 20 cm depth with a soil corer (diameter 68 mm). At each sampling position, a soil bulk was obtained by pooling five randomly chosen soil cores: one central core and four cores taken 1 m around. The soil samples were oven-dried at 40 °C and sieved at 2 mm to remove rocks, large roots, and organic debris before analysis. A LECO CHN 628 elemental analyzer was used to determine the soil organic C content. Inorganic C content was negligible. The SOC ($t\ ha^{-1}$) in the 0–20 cm soil layer was calculated as the product of soil organic C content ($g\ kg^{-1}$), soil bulk density ($kg\ dm^{-3}$), and soil thickness (dm). To measure the bulk density of the fine earth, soil cylinders of 493 cm^3 (height: 10.6 cm; diameter: 7.7 cm) were used to collect horizontal soil cores from the soil layer (Seddaiu et al., 2013).

To quantify LTC, ten circular plots of 20 m of radius were identified using systematic sampling. In each plot, all trees having a diameter above 5 cm at breast height, i.e. at 1.3 m (dbh) were measured. Above (AGB) and below ground (BGB) dry biomass was estimated using allometric equations developed by Montero et al. (2005):

$$\text{Biomass} = \text{CF} \times \beta_1 \ \text{dbhib} \ \beta_2$$

where Biomass is dry biomass (kg), CF is a correction factor obtained from the allometric equations developed by the National Institute of Agricultural Research and Technology and Food of Spain (Montero et al., 2005), β1 and β2 are parameters for either AGB or BGB, and dbhib is the dbh inside the bark. LTC was then calculated by summing AGB and BGB and by multiplying by the C content in the tree dry matter that was assumed to be 47.2% as indicated by Montero et al. (2005) for cork oak. All data were expressed as $t\ ha^{-1}$.

Annual cork production was estimated on single oak plants in WL (n = 36) and WG (n = 108) based on the number of cork oak plants ha^{-1} (N), the average perimeter over bark at breast height (pdh = dbh × π), the actual debarking height (dh) considering an average bark thickness of 0.035 m, a cork density of 210 $kg\ m^{-3}$ and a harvest interval of 10 years, for all sampled plants with a dbh > 20 cm:

$$\text{Cork}\ (kg\ ha^{-1}\ year^{-1}) = N \times pbh \times dh \times 0.035 \times 210 \times 10^{-1}$$

The acorn production was monitored from September 2013 to August 2014 using litterfall plastic traps (Pitman et al., 2010). Four traps (0.7 m × 0.7 m) were placed in six trees randomly chosen in the WL and WG sampling units at 1.0 m height from the soil level. Three of the traps were placed starting from the trunk following the cardinal NE direction: the first trap was placed within the space from the trunk to half of the crown's radius length; the second trap was placed within the space of the second half of crown's radius; the third one was placed immediately outside the crown. The fourth trap was placed in the orthogonal direction to the first three ones, in the homologous position of the first trap. The collected samples were dried in a ventilated oven at 65 °C until achieving constant weight, after their separation in wood, leaves, flowers, and acorns components.

2.4. Data analysis

Plant cover-abundance data from vegetation surveys were transformed into numeric values according to Van der Maarel (1979) cover scale before statistical analysis. The presence/absence data for the assessment of biodiversity indices were derived from the species/cover matrix.

Plant assemblage composition was compared between the five sampling positions (WLu, WLc, WGo, WGu, OG) using a one-way permutational multivariate analysis of variance, PERMANOVA (Anderson, 2001), with the sampling positions and plant assemblages in the five sampling positions as variables. The Bray–Curtis dissimilarity measure was used on square-root transformed data for calculating a distance matrix between pairs of samples. Non-parametric multidimensional scaling (nMDS) was used as the ordination method for illustrating differences in the plant assemblage composition (Clarke and Gorley, 2006). Taxa responsible for differences among plant assemblages in different habitat types, as indicated by *a-posteriori* tests using PERMANOVA, were identified by similarity percentages for species contributions analysis, SIMPER (Clarke, 1993). Species contributing at least 5% dissimilarity for any comparisons were considered important discriminators. The PRIMER statistical package version 7, with the PERMANOVA add-on (Anderson et al., 2008) was used to conduct the analysis.

The average plant α diversity was quantified as the average number of plant species for each land use type and sampling positions, the γ diversity as the cumulative number of species for land use type and position.

The β diversity was quantified following (Whittaker, 1972), i.e. $\beta = \gamma/\alpha$.

PV was assessed according to Daget and Poissonet (1971) based on the specific indices (Is) and of the CSP_i, i.e. the percentage contribution of each species to plant cover (Appendix 1):

$$PV = 0.2\sum_{i=1}^{i=n} CSP_i * Is_i$$

CSP_i was calculated as the percentage ratio between the species cover assigned to each species and the sum of the cover of all species. *Is* indicates the agronomic value of the plant (Bagella et al., 2013a) and ranges between 5 and 0 in relation to the forage value of each plant species (5 = excellent; 4 = very good; 3 = good; 2 = poor; 1 = occasionally grazed; 0 = ungrazed or toxic plant).

The nectariferous value was assessed as the sum of products between the specific nectariferous indices indicating the quality/quantity of nectar produced by each species by its specific cover (Bagella et al., 2013b). The nectariferous specific index of each plant species ranges between 3 and 0 (3 = excellent nectariferous plant, 2 = good nectariferous plant, 1 = poor nectariferous plant, 0 = non nectariferous plant).

The cover of perennial herbs (Hemicryptophytes) was assessed considering the sum of the cover of herbaceous perennial species.

The legume species cover was assessed considering the cover of the plant species belonging to the Fabaceae family.

Data on α diversity, PV, nectariferous value, Hemycriptophyte cover, legume cover, SOC, LTC, and TOC were used to compare, following a one-way ANOVA at a critical *P*-value of 0.05, five sampling positions (WLu, WLc, WGo, WGu, OG). A one-way ANOVA was also followed to test differences between the land uses (WL, WG, OG) by aggregating cover-weighted data of the sampling positions within WL and WG or to test differences between WG and WL for cork and acorn production. Prior to the ANOVA, the homogeneity of variance was tested by the Cochran's test (Winer, 1971) and data were transformed whenever necessary. When ANOVA identified a significant difference between treatments, the post hoc Student–Newman–Keuls test was used for multiple comparisons (Underwood, 1997). ANOVA was performed using the GMAV5 software (University of Sydney, Australia). Differences between LTC averages between the three land uses were analyzed with a Friedman test followed by a Conover test, due to the non-equality of variances tested with the Bartlett's test.

The data obtained from the sampling positions and land uses were used to compare three different scenarios at the landscape scale: (i) a "*dehesa* type" scenario including only WG having an average tree cover of 27%; (ii) a "specialized" scenario, that is composed by distinct areas of OG (73%) and WL (27%); (iii) a "patchy" scenario, similar to the current patchy land uses in the landscape of the study area, represented by 50% of the land covered by WG (with 27% tree cover), 13.5% of the area covered by WL and 36.5% covered by OG. The three landscape scenarios were identified considering the perspectives of the many farmers interviewed over the years in the long-term observatory, who showed a general preference for specialized land uses (WL + OG) within the same farm rather than WG, as the sparse trees in the grasslands are perceived as constraints for forage crop management and production (Rolo et al ... income from cork harvest is constantly de... k price.

The scenario analysis relies on the assumption that the management units where the data were collected large enough (WL > 10 ha; WG > 5 ha; OG > 3 ha) to minimize edge effects and hence be fully representative of each land use type. To facilitate the multidimensional comparisons of the different proxy indicators between land uses, the results were standardized as follows:

$$M_i = \frac{(\bar{y}_i - \bar{\bar{y}}_i)}{s_i}$$

where: M_i is the overall mean of variable i (indicator or proxy indicator of biodiversity or ecosystem services); $\bar{y}_i$ is the treatment mean of variable i; $\bar{\bar{y}}_i$ is the grand mean of each variable; s_i is the standard deviation of treatment means for each variable.

3. Results

3.1. Plant diversity

Overall, 202 plant species were detected across the three land use types.

The outcomes from nMDS and PERMANOVA pointed out significant differences between all the plant assemblages of the three land uses: Pperm = 0.001 for all a posteriori comparisons (Fig. 4). Sharp differences in terms of plant assemblages also emerged between sampling positions within WL and WG land uses. The highest similarity was observed between OG and WGo.

The main discriminating plant species among assemblages, as identified by SIMPER, were 15 (Fig. 5). The most abundant understory species in WLu were *Cytisus villosus* and *Arbutus unedo*, while *Asphodelus microcarpus*, *Pulicaria odora* and *Cistus monspeliensis* were most abundant in WLc assemblages. The most abundant species of WGo were *Avena barbata* and *Trifolium subterraneum*, while WGu assemblages were dominated by *Hordeum leporinum* and *Briza maxima*. The most abun-

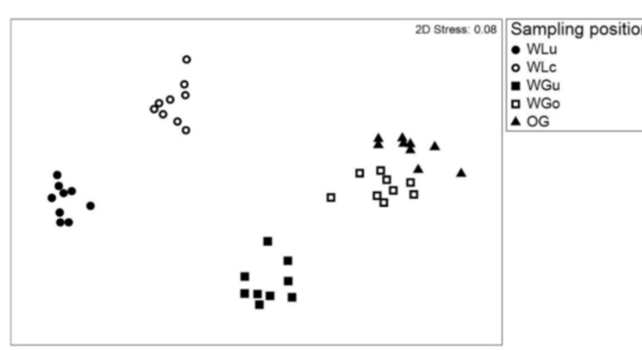

Fig. 4. Two-dimensional nonmetric multidimensional scaling ordination (NMDS) of plant assemblages composition in the five sampling positions. WLu = Underneath tree canopy in the woodlands; WLc = Clearings in the woodlands; WGu = Underneath tree canopy in the wood grasslands; WGo = Open areas in the wood grasslands; OG = Open grasslands.

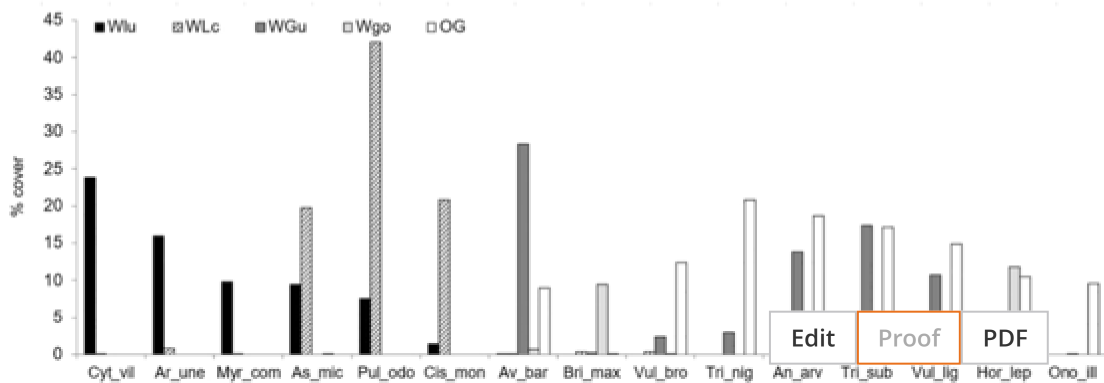

Fig. 5. Abundance of the 15 species contributing at least 5% to the dissimilarity for any comparison between plant assemblages in the five sampling positions according to SIMPER.WLu = Underneath tree canopy in the woodlands; WLc = Clearings in the woodlands; WGu = Underneath tree canopy in the wood grasslands; WGo = Open areas in the wood grasslands; OG = Open grasslands.Cyt_vil = *Cytisus villosus*, Ar_une = *Arbutus unedo*, Myr_com = Myrtus communis, As_mic = Asphodelus microcarpus, Pul_odo = Pulicaria odora, Cis_mon = *Cistus monspeliensis*, Av_bar = *Avena barbata*, Bri_max = *Briza maxima*, Vul_bro = Vulpia bromoides, Tri_nig = *Trifolium nigrescens*, An_arv = Anthemis arvensis, Tri_sub = *Trifolium subterraneum*, Vul_lig = Vulpia ligustica, Hor_lep = *Hordeum leporinum*, Ono_ill = Onopordum illyricum.

dant species in OG were *T. subterraneum*, *T. nigrescens* and *Anagallis arvensis*.

The ANOVA of biodiversity indices pointed out several significant differences between sampling positions (Table 2) and between land uses (Table 3). WLu showed the highest ß diversity and species unique to a sampling position and the lowest average α diversity; WLc showed the highest γ diversity and number of species unique to a position and an α diversity not significantly different from that of OG; WGu showed the lowest ß diversity and species unique to a sampling position and an average α diversity not significantly different from WGo and OG; WGo showed the highest γ diversity and average α diversity; OG showed a α diversity not significantly different from that of WGo and WGu, the lowest ß diversity and number of species unique to a sampling position (Table 2).

Among the species unique to WLu (Table 2), we found species typical of Mediterranean forests, such as *Cyclamen repandum*, *Lonicera implexa, Luzula forsteri*, *Melica ciliata*, *Smilax aspera*, *Tamus communis*, *Viburnum tinus* and *Viola alba* subsp. *dehnhardtii*. Among the species unique to WLc we found typical species of ephemeral canopies such as *Centaurium maritimum*, *Isoetes histrix*, *Orchis longicornu*, *Romulea ramiflora* and *Tuberaria guttata*. WGu was characterized by mesophilos species such as *Agrostis stolonifera* and *Arisarum vulgare*. Several plant species unique to WGo were ruderal e.g. *Echium vulgare* or *Scandix pecten-veneris*, but there was also the Sardinian spiny endemic *Dipsacus ferox* (Bagella et al., 2019). Among the species unique to OG we observed mainly ruderal species such as *Raphanus raphanistrum* and *Spergula arvensis*. The differences observed between sampling positions within each land use explain the differences among the three land uses: γ diversity, ß diversity and the number of species unique to a position decreased along the WL➔WG➔OG intensification gradient, while average α diversity increased was lowest in WL (Table 3).

Table 2
Plant diversity and proxy indicators of ecosystem services in the five sampling positions. WLu = Underneath tree canopy in the woodlands; WLc = Clearings in the woodlands; WGu = Underneath tree canopy in the wood grasslands; WGo = Open areas in the wood grasslands; OG = Open grasslands. Average values with different letters significantly differ for $P < 0.05$.

Land uses:	~~WL~~		WG		OG
Variable *Sampling positions:*	WLu	WLc	WGu	WGo	
Area covered	89%	11%	27%	73%	100%
γ diversity	76	98	86	99	78
Average α diversity	19.4c	37.0b	40.7ab	47.0a	41.6ab
β diversity	3.9a	2.6b	2.1bc	2.1bc	1.9c
Total no. of species unique to a position	20	23	7	14	6
Pastoral value (PV)	13.7c	9.0c	21.5b	37.8a	40.4a
Excellent and very good forage plant cover %	0.3d	1.7d	12.6c	36.7b	60.1a
Legume cover %	24.9b	3.1c	2.4c	28.9b	56.9a
Hemicryptophyte cover %	21.6b	56.3a	11.8c	22.2b	13.6c
Nectariferous value	23.3a	15.4b	4.7c	12.9b	15.6b
Soil organic Carbon stock (0–20 cm), t ha $^{-1}$ (SOC)	75.0a	64.2b	74.5a	57.0bc	55.4c
Live tree Carbon stock, t ha $^{-1}$ (LTC)	29.4b	–	61.1a	–	–
Total organic C stock t ha $^{-1}$ (TOC)	104.4a	64.2b	135.6a	57.0bc	55.4c

Table 3
Plant diversity and proxy indicators of ecosystem services as influenced by land use. WL = Woodlands, WG = Wood grasslands, OG = Open grasslands. For each variable, mean values with different letters significantly differ for P < 0.05.

Variable *Land uses:*	WL	WG	OG
γ diversity	129	128	78
Average α diversity	28.2b	43.9a	41.6a
β diversity	4.6a	2.9b	1.9c
Total no. of species unique to a land use type	43	21	6
Pastoral value (PV)	13.2c	33.4b	40.4a
Excellent and very good forage plant cover %	0.5c	30.2b	60.1a
Legume cover %	22.5b	21.7b	56.9a
Hemicryptophyte cover %	25.4a	19.4b	13.6c
Nectariferous value	22.4a	10.7b	15.6b
Acorn production (kg ha $^{-1}$ year $^{-1}$)	1037a	643b	–
Cork production (kg ha $^{-1}$ year $^{-1}$)	214a	89b	–
Soil organic Carbon stock (0–20 cm), t ha $^{-1}$ (SOC)	73.8a	61.7b	55.4c
Live tree Carbon stock, t ha $^{-1}$ (LTC)	26.2a	16.5b	–
Total organic C stock t ha $^{-1}$ (TOC)	100.0a	78.2b	55.4c

3.2. Proxy indicators of ecosystem services

The ANOVA of the proxy indicators of ecosystem services pointed out significant differences between sampling positions (Table 2) and land uses (Table 3).

The highest average PV was observed in OG as a consequence of the high cover of *Trifolium subterraneum* and *T. nigrescens,* similarly to WGo, where the high PV was associated with the high cover of *Avena barbata*.

OG also showed the highest cover of excellent and very good forage and legume plant species cover. Overall, excellent plant species were five, the most abundant being *T. subterraneum* and very good species were 12, the most abundant being *T. nigrescens.* Overall, 35 herbaceous legume species were identified, and the permanent shrub *Cytisus villosus* in WLu.

Hemicryptophytes cover showed the highest value in WLc where the dominant species (50% of the cover) was *Pulicaria odora*.

The nectariferous species showed the highest cover in WLu. Overall 21 excellent and 27 very good nectariferous plant species were detected. The most abundant were *Asphodelus microcarpus* among the excellent and *Chondrilla juncea* among the very good nectariferous species.

SOC was similar in the WGu and WLu and significantly lower in WLc and WGo (−14% and – 24% respectively) and OG (−26%). No significant differences were observed in terms of SOC between WGo and OG. The LTC stock was influenced by the smaller tree size in WL than in WG, whose C biomass per plant was about double than that of the plants in WL. The TOC of WLu and WGo was between 1.6 and 2.4 times higher than that in WLc, WGo, and OG.

Considering the three land uses, on average, the number of cork oak tree density was 308 ± 175 adult plants ha^{-1} in WL and 48 ± 43 plants ha^{-1} in WG. WL trees had significantly lower average pbh (0.78 ± 0.32 m std. dev) than WG trees (1.47 ± 0.34 m). The actual average debarking height was on average 1.22 ± 0.68 m in WL and 1.50 ± 0.54 m. Annual cork production of WG was lower (−59%) than that of the WL (Table 3) but annual cork production per plant of cork oak trees in the WG (1.83 ± 0.55 kg $plant^{-1}$ $year^{-1}$) was significantly higher than in the WL (0.69 ± 0.51 kg $plant^{-1}$ $year^{-1}$). However, the average annual acorn production per hectare of cork oaks in the WG was only 38% lower than in the WL as the average acorn production per plant was much higher in the sparse trees of the WG than in the dense WL stand.

In synthesis, WLu showed the highest nectariferous values and SOC and the lowest PV and excellent and good forage plant cover; WLc showed the highest Hemicryptophyte cover and the lowest PV and excellent and good forage plant cover; WGu showed the highest SOC, TOC and LTC stock and the lowest nectariferous value; WGo and OG showed the highest PV and the lowest LTC and TOC; OG showed the highest excellent and good forage plant cover and legume cover.

In the comparison along the WL➔WG➔OG land use intensity gradient, PV, excellent and good forage plant cover and legume cover increased while Hemicryptophytes cover, cork and acorn production, SOC, LTC, and TOC decreased.

3.3. Scenario analysis

The two alternative scenarios to a large scale "*dehesa* type", "specialized" and "patchy", showed higher γ and ß plant diversity and number of species unique to a land use type than each of the three land use types taken separately (Fig. 6). The γ diversity was about 3 and 2 standard deviations (SD) higher in the "patchy" and "specialized" scenarios respectively than the average γ diversity of the three land uses, while in WL and WG it was only +0.6 SD and in OG −1.1 SD with respect to the overall mean. The highest average α diversity was observed in the "*dehesa* type" scenario (+0.71 SD) while in the "patchy" and "specialized" scenarios α diversity was similar to the overall mean.

PV, excellent, and good forage species and legumes cover were always less than 1 SD below the grand mean in the "patchy" and "specialized" scenarios and close to 1 SD units over the grand mean in OG. Nectariferous species and Hemicryptophyte cover were close to the grand mean in the "specialized" scenario and close to −1 SD in the "patchy" scenario. TOC, live tree Carbon and SOC were clearly higher in WL (1 SD over the grand mean) while they were close to the grand mean in the "patchy" scenario and 0.5 SD over the grand mean in the "specialized" scenario. For the proxy indicators of organic C stock, the best situation was represented by WL.

4. Discussion

The results demonstrate that a patchy arrangement of land uses within Mediterranean silvopastoral systems, corresponding to different management practices, can enhance plant diversity and ecosystems services with respect to large scale *dehesa* type, and can contribute in many ways to the conservation of ecological processes and services. The Long-term Observatory was chosen in an area characterized by homogeneous environm[illegible] phology) and climatic conditions, resulting [illegible] mixed *Quercus suber* and *Quercus ilex* forest (Bagella et al., 2016); therefore, the differences found in plant diversity and ecosystem services can be attributed to the different land uses (Rossetti et al., 2015). The patchy landscape of the cork oak silvopastoral systems of Sardinia is generated by a relatively fragmented and small size land ownership, with most pure cork oak WL in the range of 10–20 ha (Sedda et al., 2011), even if the average utilized agricultural land per farm in Sardinia (24.5 ha) is the highest among the Italian regions, the national average being 11.0 ha (ISTAT, 2016). The patchy land use structure of Sardinian silvopastoral systems is therefore very different from that of the Spanish *dehesas*, dominated by large farms and homogeneous management units associated with large fields with wood grasslands (Milán et al., 2006). We discuss results on plant diversity and ecosystem service proxy indicators by land use type as structural components of silvopastoral systems in Sardinia, then we compare contrasting scenarios deriving from a range of combinations at landscape scale of the three land uses.

4.1. Plant diversity

The floristic richness of the sampled areas represents about 10% of total γ plant diversity of Sardinia (Bagella et al., 2020). Plant assemblage composition was distinct in the five sampling positions across the land use intensification gradient, revealing a sharp response of vegetation to human activities, as already observed in different biogeographical contexts (del Arco Aguilar et al., 2010; De Keersmaeker et al., 2013; Torralba et al., 2016). The main contribution to γ diversity and to the variety of plant species was provided by the low (WL) and the medium (WG) intensity management levels, that generated the microhabitats associated with WL clearings and WH isolated trees (Rossetti et al., 2015).

The *Q. suber* woodland (WL), the lowest intensity land use under comparison, was well structured and rich in shrubs, lianas and herbaceous species in the understory, with many characteristic elements of the *Violo dehnhardtii-Quercetum suberis* association to which it was referred (Bacchetta et al., 2009). The lower total γ diversity of WLu in comparison to that observed under the scattered tree canopies of WG should not be surprising since the plant diversity of the understory of Mediterranean evergreen forests is constrained by the low solar radiation due to the full radiation interception of the top tree layer. Nevertheless, the total number of species found in WLu was relatively high if compared for instance with the data reported for the *Q. suber* forests of south-eastern Spain, where only 64 species ha^{-1} were found (Díaz-Villa et al., 2003). The highest β diversity of WL reflects the high spatial variability of *Q. suber* forest, in part generated by local disturbances caused by the extraction of cork and shrub clearing around the oaks (Díaz-Villa et al., 2003), and the presence of one/few strongly dominant species. Within WL, WLc provided a substantial contribution to total plant diversity, due to the high number of unique species and almost the highest values of biodiversity indicators. Díaz (2008) and Marañón et al. (2009) reported a significantly higher diversity of vascular plants for Iberian *dehesas* compared to other adjacent land

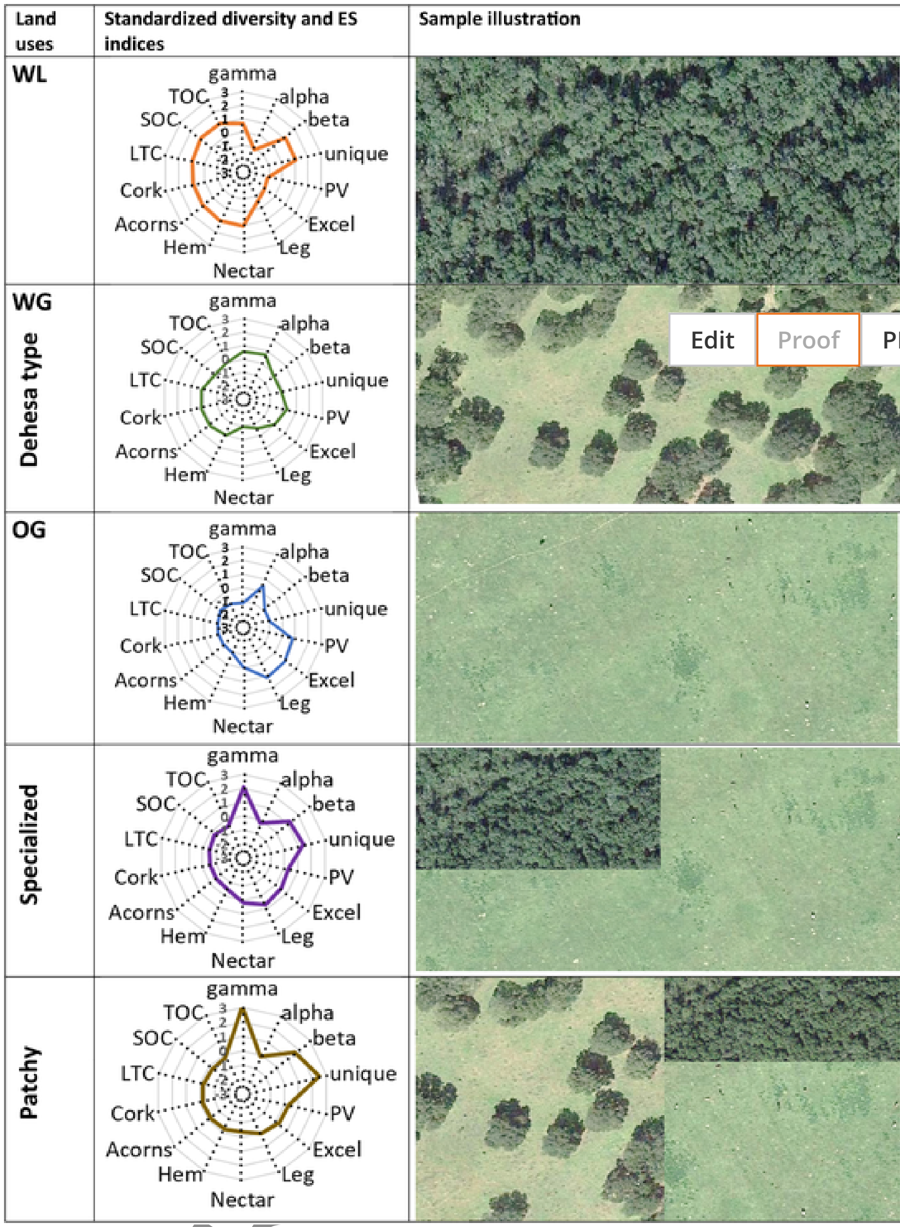

Fig. 6. Spider diagrams of the standardized values of plant diversity indicators and ecosystem service proxy indicators in relation to land uses and the different scenarios. OG = 100% open grassland; WL = 100% Woodlands; WG (“*dehesa* type”) = 100% wood grasslands with 27% tree cover; “specialized” = 27% WL and 73% OG; “patchy” = 50% WL with 27% tree cover, 13,5% WL and 36.5% OG. The scale of the diagram expresses the deviation from the average value of the three land uses.Abbreviations: gamma = γ diversity; alpha = average α diversity; beta = ß diversity; unique = Total no. of species unique to a a land use type; PV = Pastoral Value; Excel = Excellent and very good forage plant cover %; Leg = Legumes cover%; Nectar = Nectariferous value; Hem = Hemicryptophyte cover; Acorns = annual acorn production; Cork = annual cork production; SOC = Soil organic Carbon stock; LTC = Live tree Carbon stock; TOC = Total organic C stock.

uses, confirmed by Bergmeier et al. (2010) for other European countries. On the contrary, our study shows that if the plant diversity assessment of cork oak WL also includes the clearings (WLc), the total plant richness is similar to that found in WG. Indeed, WLc plays a mirror role of that of scattered trees (WGu) in WG in supporting spatial heterogeneity. We here highlight for the first time the relevance for plant diversity of the clearings in cork oak woodlands generated by the traditional practices of cork extraction, which are rarely if ever considered in the assessment of biodiversity of agroforestry systems (Díaz, 2008; Bergmeier et al., 2010; Moreno et al., 2016)

The biodiversity relevance of WL and WG is also supported by the correspondence of the vegetation types with habitat types listed in Annex I of the Habitat Directive (Gigante et al., 2019). Indeed, WLu corresponds to the habitat 9330-*Quercus suber* forests, WLc to the priority habitat 6220* Pseudo-steppe with grasses and annuals of the *Thero-Brachypodietea* and WG to habitat 6310 Dehesas with evergreen *Quercus* spp. supporting high plant and animal diversity levels and serving as habitat for several threatened species of the European fauna (Gigante et al., 2016, 2018).

In WG, species unique to shaded positions contribute to enhancing the total species richness of WG (Manning et al., 2006; Rossetti et al., 2015; Seddaiu et al., 2018). The variability of the plant assemblage composition and richness under scattered trees can be dependent not only on the understory management but also on the livestock selective use as shelter (Olea and San Miguel-Ayanz, 2006).

The OG and WGo grasslands were similar in terms of plant assemblage composition, with a high specific cover contribution of *T. subterraneum* in both sampling positions. Nevertheless, the biodiversity indicators were significantly different: total and average richness and spatial variability were lower in OG than in WGo. This was interpreted as the consequence of the more frequent application of selective agronomic practices (plowing, cutting, fertilization, etc.) in OG than in WGo. However, the management practices applied in OG allowed to maintain a plant species-rich sward including excellent and very good annual self-reseeding forage legumes. The relationships between grassland management practices, biodiversity, and ecosystem services and the corresponding values depend on local conditions as described by other authors in the same area or in different contexts (e.g. Bagella et al., 2013a; Probo et al., 2016; Čop and Eler, 2019).

4.2. Ecosystem service proxy indicators

The three land uses under study showed a substantial variation of almost all the selected ecosystem service proxy indicators, providing also a clear evidence of the complementarities of services between land uses. As expected, OG were best in providing forage production and quality, while WL showed higher values for supporting and regulatory services. WG showed intermediate values for almost all the proxy indicators, with WGo showing very similar values to OG and WGu showing significantly lower values than WLu for all indicators but SOC. These results have direct implications on the outcomes of the scenario analysis (see 4.3) as none of the three land uses or sampling positions showed optimal values for all proxy indicators.

The PV of the pastoral vegetation as a proxy indicator of the provisioning of forage quantity and quality, increased along the WL➔WG➔OG intensification gradient, as well as the contribution of excellent and very good forage species cover. However, the provisioning of feeds from oak trees to domestic animals, besides grass and shrubs, also includes a substantial acorn production in both of Sardinian WL and WG, which is consistent to what reported by other authors in Spain (Cañellas et al., 2007; López-Díaz et al., 2015). Acorns provide supplementary feed in autumn and early winter, which is particularly important when the summer dry season extends until the end of October-mid November, and grasslands regrowth is delayed (Caballero et al., 2009; Pardini, 2009; Porqueddu et al., 2017). The forage provisioning of oak tree branches of WL and WG can also represent a strategic pastoral resource either through direct grazing or through pruning (Scarascia-Mugnozza et al., 2000) but were not considered in our assessment, therefore the contribution of the wood component to the provisioning of animal feed is here underestimated. Pardini (2009) estimated a forage production up to 4 t ha^{-1} from the palatable shrubs of the oak understory. In the warmest months, cattle livestock prefer to graze woodlands than open grasslands, as woodlands provide also protection to the grazing cattle from flies such as *Hypoderma bovis* (Karatepe et al., 2013). Nevertheless, scattered trees in the WG also represent a shelter in the warm season for livestock, which however can damage the herbaceous vegetation under the trees by trampling and smearing with dung and urine favoring the presence of nitrophilous species (De Miguel et al., 2013; Rossetti et al., 2015) characterized by low palatability (Roggero et al., 2002; Bagella et al., 2013a). These results are also consistent with the findings of López-Carrasco et al. (2015) that evidenced the lack of legumes and prevalence of grasses under the tree canopies in the Spanish *dehesas*.

According to the Daget and Poissonet (1971), WG and OG could potentially support some 0.6–0.8 livestock units ha^{-1}, which is consistent to what observed in the farms included in the long-term observatory (Bagella et al., 2013a) and indicates substantial self-sufficiency and environmental sustainability of the current stocking rates. Such moderate grazing pressure in the long-term has generated the ideal habitats for in situ conservation of genetic resources that have become an important ecosystem service of silvopastoral systems of Sardinia. The germplasm of many annual self-reseeding legume species collected also in the study area has specific adaptive traits (e.g. soil acidity tolerance, long-term persistence, tolerance to pest and pathogens) that made it suitable to investments in plant breeding programs (e.g. *T. subterraneum*, *Medicago* sp.pl., *T. michelianum, T. incarnatum, Ornithopus* sp.pl. and many others) by researchers and private enterprises of southern Australia since the 1950s (Nichols et al., 2013a, 2013b, 2016; Porqueddu et al., 2016). A sustainable germplasm resource management model which do not cause the decline of biodiversity and does not jeopardize the capability of future generations to meet their needs is still lacking (Perrino et al., 2006) despite the objectives of the UN Convention on Biological Diversity.

The legume spec[illegible]roxy for the nitrogen fixation ecosystem s[illegible]WL was mainly represented by the understory of *Citysus villosus* that establishes N_2-fixing symbiosis with the slow-growing bacteria of the genus *Bradyrhizobium* (Rodríguez-Echeverría and Pérez-Fernández, 2005). The high legume cover observed in OG represents a very important alternative source of combined nitrogen for these oligotrophic grasslands, where nitrogen leaching is favored by the Mediterranean rainfall regime and the very permeable sandy soils. The low fertility is the main driver for triggering the encroachment of grasslands with shrubs such as *Cistus monspeliensis* L., a pyrophytic species which is favored by low pH and soil fertility (Bagella et al., 2017). Therefore, in these silvopastoral systems, annual legumes play a fundamental role in supplying proteins to the grazing animal, improving soil fertility with natural N fixation (Lucas et al., 2010) and hence supporting C cycling and the productivity and persistence of secondary grasslands.

The nectariferous value is another important proxy of ecosystem services in the Mediterranean silvopastoral systems. WL represents a great resource for pollinators and honey production, a traditional, renewable, and therefore sustainable resource from an economic perspective (Merlo and Croitoru, 2005). Honey characteristics are mainly influenced by the composition of feed resources for bees and flowering phenology (Floris et al., 2016) and in these contexts, foraging for valuable unifloral honey is also possible (i.e. *Arbutus unedo* and *Asphodelus microcarpus*). Beside honey production, another service is related to the pollination of self-incompatible grassland species such as *T. nigrescens*, which is among the most important species contributing to the overall PV of OG. Hence the chosen proxy indicator (nectariferous value) is relevant per se but is also important considering that in a patchy silvopastoral landscape pollinators can find in the WL a source of feed during the dry season when OG vegetation is almost completely dry.

The Hemicryptophyte cover, a proxy for soil protection from erosion and water runoff (Naveh, 1974), was highest in WLc, which were never cultivated, confirming that the pressure of recurrent tillage selects annual with respect to perennial species in Mediterranean grasslands (Tuesca et al., 2001).

The observed cork productivity and the biometric characteristics of the oaks are consistent to what found by Sedda et al. (2011) in Sardinia, that had already observed a relatively low debarking coefficient (debarking height/plant perimeter at breast height), much lower than the maximum prescribed by law (2 for virgin cork and 3 for mature cork extraction). This is probably one of the main reasons why the cork productivity observed in the study area is relatively low when compared with the cork productivity observed in Iberian cork WL and WG (Pereira and Tomé, 2004). Such non-wood products of cork-oaks in WL and WG used to be one of the main sources of income for the silvopastoral farmers of Sardinia, until the raw cork price declined in the international market, thus threatening cork production system sustainability (de Almeida Ribeiro et al., 2011).

The TOC in topsoil and woody vegetation observed in WL and WG was consistent with what found by Oubrahim et al. (2015). The lower TOC in WG (−22%) than in WL, was a consequence of a substantially similar SOC value in the topsoil of WLu and WGu but a much higher (×6) tree density and cover (×3) in WL than in WG and a × 2 average tree size in WG than in WL.

The SOC of OG was consistent with what found by Cappai et al. (2017) and Seddaiu et al. (2013) in the same area and was 25% or 10% lower than the C stock per unit area measured on WL and WG respectively.

WL showed the highest TOC, of which about 26% was related to the living tree component. While the TOC is consistent with similar cork oak WL in Spain, the proportion of LTC assessed in Sardinia was slightly lower the tree C stock of Iberian cork oak WL being about 35% of the LTC + SOC measured in the 0–60 cm soil horizon (Correia et al., 2016). The TOC in the WL was underestimated as we did not quantify the contribution of the understory woody plants, which can be substantial in the patches surrounding the cork oaks that were not cleared for the cork harvest, differently to what assessed in Spain, where anyway the contribution of the understory and herbaceous layers to TOC was as low as 1%. The areas underneath the tree canopy showed similar SOC in the topsoil layer in both WLu and WGu sampling positions, having on average 18% more SOC than in the open areas and in the OG. However, Seddaiu et al. (2013) in the same environmental and management conditions highlighted that when SOC is calculated for specific soil horizons (e.g., Ap) rather than for fixed soil depths as in the present study, the similarities between WLu and WGu were not evident and WGu showed up to 80% more SOC than WLu. These results are associated with the lower thickness (about 5 cm) of the A horizons in the WLu compared to that of the Ap horizons (about 20 cm) in the WGu. Overall, SOC in the 0–20 cm soil depth decreased from WL to WG and to OG and WG showed higher values (62 Mg ha^{-1}) than those found in the central-western Spanish *dehesas* with cork oaks by Howlett et al. (2011). Our data, however, confirm the results obtained by Cappai et al. (2017) in a Sardinian study area. The estimates of the C stocked in the oak biomass are consistent with what found by Oubrahim et al. (2015) on cork oaks in Morocco.

4.3. Scenario analysis

The patchy landscape generated by the combination of the three land uses, simulated in the "patchy" scenario, showed the highest levels of plant diversity among those under study and offers niches for all species unique to a single position.

The "specialized" scenario, consisting of WL and OG with an overall tree cover comparable to that of a large scale WG, would not provide niches for less than 5% of the total plant species identified across all land uses and sampling positions, mostly herbaceous species typical of the WGu, characterized by low PV. Rossetti and Bagella (2014) showed in the same area that under the business-as-usual management system, where tree regeneration in the open grassland is not actively sought by farmers, WG of pastoral farms would be progressively converted into open grasslands over the decades as adult plants die, either naturally or because of the recurrent damage to the root system during tillage for forage crops seedbed preparation. An alternative possibility is the evolution of the WG towards a mature WL caused by the abandonment of pastoral activities, providing that wildfires are prevented during the natural ecological succession through shrublands (Bagella et al., 2017). Similar problems were described for two decades in the Portuguese montados by Pinto-Correia and Mascarenhas (1999). The isolated trees in the pasture are perceived by farmers as an obstacle to farm mechanization while WL are perceived as a strategic resource for livestock farming systems. In the Sardinian context, the "specialized" scenario then corresponds to the expectations of many farmers who would prefer to manage OG and WL as two distinct land uses on the same farms.

The "patchy" scenario was characterized by highest legume and Hemicryptophyte cover and nectariferous value, while the "specialized" scenario was always better than the "*dehesa* type" scenario for all ecosystem services with the exception of SOC. These results indicate that a homogeneous WG, that proved to be more balanced in terms of variety of ecosystem service provisioning when compared to WL or OG, is not necessarily more desirable in terms of biodiversity and ecosystem services than a patchy landscape combining small woodlands and open grasslands and that a diversified landscape made of a balanced proportion of all three land use types (WL, WG, and OG), even if maintaining the same total tree cover of WG, would be a more desirable than a large scale WG both in terms of plant diversity and ecosystem services. The current Sardinian silvopastoral landscape is well representative of the "patchy" scenario, while the more regular landscape of *dehesas* and *montados* in Spain and Portugal respectively are more like a uniform WG and are threatened by lack of regeneration in the high-intensity land uses corresponding to the gentle slopes and frequently plowed areas for cropping (Sevillano et al., 2017). In Sardinia, the current patchy landscape might evolve into the "specialized" if the WG are intensively managed ignoring the regeneration of the tree component (Rossetti and Bagella, 2014) or abandoned to the natural ecological succession towards a WL, providing that they are preserved from wildfires.

5. Conclusions

The results presented in this study highlight that the patchy landscape and the spatial heterogeneity within land uses of the Mediterranean silvopastoral agroforestry systems of Sardinia are a relevant source of plant diversity and multiple ecosystem services. The assessment made in this study, following a decade of research in a long-term observatory, provides substantial evidence on the structural coupling between farming practices, biodiversity, and ecosystem services in Mediterranean silvopastoral systems.

Among the three main structural components of these systems, WL ensured the highest C stock, non-wood products, and strategic forage resources for the grazing livestock. The cork oak clearings (WLc) proved to be an important source of plant diversity in cork oak WL and are associated with the traditional cork oak extraction management practices, which have been rarely if ever considered in the assessment of biodiversity of Mediterranean agroforestry systems. OG and WGo showed a relatively high PV, when considering the soil fertility and climatic constraints, and are in balance with the current livestock stocking rate, which mostly relies on the grazing of natural vegetation. In the context of the patchy Sardinian silvopastoral landscape, the role of the tree cover in WG appears to be not as relevant as it is for the Iberian *dehesas* and *montados*, where the scattered trees represent one of the few sources of landscape heterogeneity, given the large farm size and homogeneous management systems.

Sardinian silvopastoral systems, combined with cork oak WL and the fragmented land ownership, provide a range of different land management practices which interact with ecological processes generating valuable ecosystem services including forage and animal feed production, in situ conservation of valuable genetic resources, pollination and honey, cork, soil protection from erosion, N fixation and nutrient cycling, organic C stock.

The hypothesized "specialized" and "patchy" scenarios both showed higher plant diversity and similar or better ecosystem services than large scale "*dehesa* type" scenario. As forage provisioning is currently the primary marketable product of these agroforestry systems, the farming intensification goals are currently directed to improve grassland productivity and quality, while biodiversity conservation and the provision of the other ecosystem services are indirectly related to the farm management practices. An advantage of these systems is that the farm manager is also the main beneficiary of several ecosystem services deriving from the different land use options which support the environmental heterogeneity. A critical issue is that this virtuous cycle of benefits could be interrupted or degraded by changes in farm structure, which is influenced by external factors including normative, social, and economic drivers.

Declaration of Competing Interest

None.

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