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Beautiful agricultural landscapes promote cultural ecosystem services and biodiversity conservation

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Abstract –Agriculture, during its millenarian history, had contributed to shape impressive cultural landscapes; however, in recent decades, many of these have been lost or degraded because of widespread intensification or abandonment. Low-intensive agricultural landscapes are of utmost importance for biodiversity conservation and the delivery of cultural ecosystem services.

We worked in a cultural landscape shaped by viticulture (in Trentino, Italy), which recently underwent a widespread intensification. We explicitly quantified two cultural services (aesthetic and cultural heritage values), and the biodiversity (bird species richness) associated with this landscape at 24 sampling sites. We then related these variables with the territory density of an indicator/flagship bird species, the common redstart. Finally, we assessed redstart ecological requirements at the territory scale. We aimed to define an appealing strategy combining biodiversity conservation and ecosystem service delivery in the cultural landscapes.

Redstart density was positively related with avian species richness and landscape aesthetic value, the latter being related with the cultural heritage value. Redstart occurrence was positively associated with hedge and tree rows, dry-stone walls, marginal habitats, and the compositional diversity of the land-cover.

We concluded that managing the agricultural landscape to maintain aesthetic and heritage values, which primarily means conserving and enhancing its key ‘traditional’ traits, would favour an indicator/flagship species and likely the wider bird diversity. It will also promote the heritage and recreational value of the landscape itself, underlining the importance of the synergistic integration of multiple conservation targets into a combined strategy.

1. Introduction

Cultural landscapes result from long-term, complex human-nature interactions (Tieskens et al. 2017) and stand “at the interface between nature and culture, tangible and intangible heritage, biological and cultural diversity” (Rössler 2006). They are characterized by distinctive biophysical features, including substantial amounts of natural/semi-natural habitats, land-cover heterogeneity (Plieninger et al. 2006), relatively low nutrient inputs and low outputs per hectare (Bignal & McCracken 1996; Kleijn et al. 2009). These characteristics make cultural landscapes pivotal for biodiversity conservation (Antrop 1997; Fischer et al. 2012).

Cultural landscapes also promote the delivery of cultural ecosystem services (Schaich et al. 2010; Tengberg et al. 2012), i.e. the “non-material benefits people gain from ecosystems”, such as spiritual, religious, aesthetic and cultural heritage values, recreation, and ecotourism (Millennium Ecosystem Assessment 2005). Cultural ecosystem services are fundamental for human life quality, but due to their intangible nature are often difficult to quantify and to incorporate into economic assessments and landscape planning (Daniel et al. 2012; Plieninger et al. 2013, 2015).

In its millenarian history, agriculture has contributed to create distinctive cultural landscapes (Zimmermann 2006), to the point that ‘low-intensity farmland’ has been used as a synonym of cultural landscapes (Tieskens et al. 2017). In those areas, both biodiversity and cultural services have been favoured by a prolonged low-density settlement and low-intensity land use (Schaich et al. 2010; Gatzweiler & Hagedorn 2013). However, in the recent decades those systems and the species and services they harbour collapsed, because of the intensification of cultural practices and the abandonment of marginal and less productive areas (Tscharntke et al. 2005; Beilin et al. 2014).

Conservation approaches based on ecosystem services delivery could broaden and deepen supports for biodiversity protection, potentially aligning conservation and production issues (Goldman et al. 2008). However, most of the current conservation practices and legislations are tightly focused on protecting species and habitats (Maes et al. 2012). This separation is problematic because, even if the relationship between biodiversity conservation and ecosystem service delivery is often positive (Harrison et al. 2014), some negative impacts of biodiversity conservation programs on wider ecosystem services were reported (Austin et al. 2016). Thus, the integration of these two conservation approaches into landscape planning, based on the possible synergies between biodiversity conservation and the delivery of a wider bundle of ecosystem services, should be pursued, also because it was proven to be more appealing for a variety of stakeholders (Ekroos et al. 2014; Brambilla et al. 2017). In fact, cultural services can help to raise public support for protecting ecosystems (Gobster et al. 2007; Schaich et al. 2010), and thus constitute an ideal framework to integrate ecosystem service delivery and biodiversity conservation into synergistic strategies (Mace et al. 2012).

In this work, we quantified two cultural services identified in the MEA (2005), the aesthetic value and the cultural heritage value, provided by a traditional viticultural landscape characterized by various levels of recent intensification. Vineyards have been selected as a case study because viticulture contributed to model impressive cultural landscapes in the Mediterranean basin, such as the terraced vineyard systems supported by dry-stone walls (Petit et al. 2012), which support high levels of biodiversity and threatened species (Assandri et al. 2016; Guyot et al. 2017). In parallel, vineyard-dominated cultural landscapes (at least, the non-intensive ones) provide a variety of cultural ecosystem services, including aesthetic values, cultural heritage values and recreational and ecotourism opportunities (Winkler et al. 2017).

On the other hand,viticulture intensification and expansion, favoured in Europe by the Common Agricultural Policy (CAP) and by the huge economic value of wine, are nowadays resulting in homogeneous monocultures, landscape simplification (Martínez-Casasnovas et al. 2010) and loss of natural habitats (Viers et al. 2013), with strong impacts on biodiversity and ecosystem service delivery (e.g. Caprio et al. 2015; Assandri et al. 2017c; Winkler et al. 2017).

In addition to the ecosystem service assessment, we quantified bird diversity in the same cultural landscape and related these benefits with the density and the ecological requirements of a flagship/indicator species for vineyards, the common redstart *Phoenicurus phoenicurus* (Aves: Muscicapidae) (Assandri et al. 2017b). We expected that landscape traits associated with extensive agriculture, which qualify the landscape as cultural (and likely affect its aesthetic and heritage value), could be the same factors promoting redstart occurrence and the maintenance of the wider bird diversity.

Our eventual goal was therefore to suggest an appealing strategy combining biodiversity conservation and ecosystem service delivery into an integrated plan for the cultural landscape.

2. Materials and methods

2.1 Study area and design

This study was performed in Trentino (south-eastern Alps, Italy; Fig. 1a), working along a landscape gradient, defined by environmental and agricultural management characteristics. Lowland areas (200-230 m a.s.l.) are mostly covered by intensive vineyards and infrastructures, which eroded the most of natural and semi-natural habitats (Assandri et al. 2017e). On mountainsides, specifically in Cembra Valley, the high acclivity limited mechanization and vineyards (still the dominant land-use up to 900 m a.s.l.) are grown thanks to a system of terraces supported by dry-stone walls. Natural (e.g. woodlands) and semi-natural habitats (e.g. hedgerows, natural field margins) regularly occur, resulting in a relatively high landscape heterogeneity. These characteristics contribute to qualify the valley as a cultural landscape included into the National Register of Historical Rural Landscapes (Agnoletti 2013).

Within this area and along this gradient, we selected 24 sampling sites, for a total of 400 ha (mean extent ± SD: 15.8 ± 3.4 ha; range: 10.8-22.8 ha; Fig. 1b).

2.2 Model species

Common redstart is widely distributed in Europe, where it underwent a sharp decline until the end of the last century, followed by a strong recovery (Birdlife International, 2004).

During the breeding season, it is mainly found in semi-open areas with sparse vegetation and mature trees, and increasingly in urban areas, where it easily finds cavities for nesting (Cramp 1988; Droz et al. 2015). During the breeding season it is territorial, defending territories ranging from 0.14 up to 1 ha (Menzel, 1971; Glutz von Blotzheim, 1988). It is an insectivorous bird hunting from vantage points and catching about 50% of its prey (mainly Lepidoptera, Coleoptera, Diptera, Hymenoptera and Arachnida) on the ground (Martinez, 2010, Cramp, 1988). It is a long distant migrant wintering in sub-Saharan Africa (del Hoyo et al., 2005).

In the study area, it is commonly found in vineyards, with variable density depending on habitat and micro-habitat characteristics. According to this and to its peculiar natural history traits (which make it quite sensitive to environmental changes), it was proposed as a “non-traditional” flagship species to promote biodiversity-friendly agriculture in vineyard-dominated landscapes (Assandri et al. 2017b).

2.3 Aesthetic quality and cultural heritage values

We quantified two cultural ecosystem services, the aesthetic value and the cultural heritage value, identified in the M.E.A. (2005). Many people find beauty or aesthetic quality in various aspects of the ecosystem, which is routinely assessed by perception-based surveys, where quantitative measures of aesthetic quality are derived by averaging choices, ratings, or other measures across observers (Daniel et al. 2012; Van Zanten et al. 2014).

We used photographic standardized questionnaires to attribute to each sampling site an aesthetic value (see e.g. López-Santiago et al., 2014). In the questionnaires (spread by internet, see supplementary materials for details on dissemination and respondents), all the 24 sampling sites were depicted by a panoramic photograph showing the 'best representation' of the landscape. A preview of all the photographs together was firstly presented to the observers; then, photographs were presented again in a random way and observers were asked to rate each landscape from 1 to 10 considering purely aesthetic criteria. The aesthetic value of each sampling site was then calculated as the median score given by participants in the 382 questionnaires analysed.

The cultural heritage values were recognized as a cultural service because many societies place high value on the maintenance of cultural landscapes (M.E.A 2005), but its quantification is difficult (Daniel et al. 2012). Here we assessed whether the 24 landscapes selected for our study were perceived by the observers born in Trentino (N = 89) as "traditional", thus part of the cultural heritage of the region. As this index was strongly correlated with the aesthetic value ($r_s = 0.69$, $p = 0.002$), subsequent analyses were based only on the latter.

2.4 Common redstart ecological requirements and species richness assessment

2.4.1 Redstart territory mapping and species richness assessment

During the breeding season 2015 we conducted four sessions of territory mapping in each of the 24 sampling sites (census periods: 10.04-17.04; 05.05-12.05; 29.05-05.06; 27.06-05.07).

In each visit, the same observer (G.A.) followed the same route inside the site, walking at a slow pace, thoroughly surveying the entire site.

We recorded all redstart contacts within the sampling sites as precisely as possible, by using updated aerial photographs (scale 1:2500 m), starting from the first location. If a bird moved spontaneously (i.e. non-disturbed by the observer), we mapped also the subsequent location(s) until it continued the same activity for at least 10 minutes (e.g. feeding, singing from the same perch, etc.). We accurately recorded individuals' behaviours and interactions (e.g. courtship, aggressive behaviour, etc.).

Three sites per day were censused from dawn to a maximum of six hours after it (5.30 – 11.30 a.m.), when redstart song activity is highest. We changed the census order across plots from one visit to the following one, to ensure variability in the census time within the morning. We avoided bad weather conditions (e.g. strong wind, rain).

In parallel, we assessed the cumulative avian species richness based on the four visits in each sampling site. We distinguished between overall and breeding species richness. The two were highly correlated ($r_s = 0.95$, $p < 0.001$), thus subsequent analyses were based on the latter.

2.4.2. Definition of territories and control plots

We defined territories defended by redstart pairs based on the redstart locations collected within each sampling site. W established an equal number of control plots on the basis of random points scattered within the 24 sampling sites at locations where redstarts were never recorded (Fig. 1c).

In studies dealing with resource selection by animals, several methods have been adopted for the definition of territories (Manly et al. 2007). Circular buffers (e.g. around the nest) are frequently assumed as representative of the territory defended by territorial species (e.g. Coudrain et al., 2010; Jedlikowski et al., 2016).

We measured environmental variables within 1.05 ha circular buffers (radius=58 m), defined by means of a two-step procedure and representing territories defended by pairs.

We initially built the minimum convex polygon based on the locations attributed to the same redstart pair, paying attention to simultaneous locations and interactions between individuals. We discarded the records potentially attributable to migrant individuals (e.g. birds feeding in habitats unsuitable for breeding during the early part of the study season and no longer contacted in the same site).

In the first step, we calculated the centroid of each polygon and the mean surface of the polygons that were based on more than 4 points (N=37). The latter analysis suggested an average territory size equal to 0.37 ha, corresponding to a circular plot with radius 37 m. When we found the nest of a pair, we considered it as the final centroid of the pair territory.

As a second step, we calculated the distance between the nearest neighbouring centroids (in the same sample site) and divided it by two, obtaining a value of 82 m (N=78).

We finally averaged the two values obtained by the two-step procedure (radius of the circular plot corresponding to the average polygon surface and half of the average distance between neighbouring centroids), assuming the result (58 m) as the final radius of a hypothetical mean common redstart territory. We used this distance to buffer all the available centroids (N=80) and considered the resulting plots as territories in the analyses.

In Switzerland, Martinez et al. (2010) used a radius of 50 m (corresponding to an area of 0.78 ha), based on literature reference values. Our method seemed to be a good trade-off between accuracy and the inclusion of a wider area. Overlapping between different territories was limited to 4.88%.

The plots defined in this way became the sampling units of the territory scale habitat selection analysis (2.6.2).

2.4.3. Environmental variable collection

Environmental variables were measured at each territory/control plot, as defined in 2.4.2 (Fig. 1d). Land-cover variables were obtained from an aerial photograph validated and updated in the field. We calculated the relative cover of eight habitat categories for each territory/control plot and derived the land-cover H' Shannon diversity index (Laiolo, 2005).

We collected several management variables related to vineyard; specifically, we attributed each vineyard field to one of the two vineyard trellising systems occurring in the study area: spalliera and pergola. Spalliera (espalier) is the globally widespread vineyard arrangement, whereas pergola is the traditional and predominant form in the region, accounting for about 80% of the overall vineyard surface in Trentino (see Assandri et al. 2017e for details).

In our study area, vineyard and apple orchard ground is extensively (mean 91.5%, our unpub. data) covered by a dense grass sward, except at vine/tree base (a strip of about 1 m), where herbicides or mechanical grass removal are applied. However, we distinguished between vineyards and apple orchards chemically weeded/ploughed and with a full grass cover, by evaluating each single parcel in the field.

For each plot, we measured the average area of the vineyard parcels included (totally or partially) within the buffer, which is a proxy for intensive (larger fields) or extensive (smaller fields) agriculture. As redstarts are secondary cavity breeders, requiring holes for nesting, we quantified the availability of structures potentially hosting nesting sites within each plot, by counting isolated rural buildings and measuring the total length of the two types of stone walls (dry or cemented; Fig. 2) within each plot. We additionally measured from a 1-m resolution digital elevation model (DEM) the mean slope and mean direct solar radiation (for 21st June, using r.sun function in GRASS 7.0.2, considering the shadowing effect of the topography; Neteler et al. 2012) for each plot. We used QGIS 2.14.2 (QGIS Development Team 2016) for all the spatial analyses.

2.5 Statistical analysis

2.5.1 Relationships between avian species richness, cultural ecosystem services, and common redstart density

We tested whether breeding species richness and aesthetic value were associated with redstart density (redstart territories/site area).

The correlation between breeding species richness and redstart density was tested by means of a GLM with a Poisson error distribution and a log-link function, as the response variable was a count (Zuur et al. 2013). To account for the potential effect of the sampling site area on this variable we included the area of the sites in the model.

The correlation between redstart density and aesthetic value was tested by means of a cumulative link model (CLM) with a logit-link function (Agresti 2012), as the response variable is an ordered categorical variable. This analysis was conducted with the R package *ordinal* (Christensen 2015).

2.5.2 Redstart habitat selection analysis at territory scale

Explanatory variables were subdivided into two groups, separately considered when building models: land-cover/topographic and management predictors (see Table 1). We carried out an accurate data exploration for each group of predictors in order to avoid common statistical problems (e.g. collinearity), following the approach suggested by Zuur et al. (2010).

Vineyard cover was included in the management group (instead of land-cover) to correct for the amount of vineyard within each plot when evaluating the effect of the vineyard management variables, as well as to reduce collinearity in the land-cover group (Assandri et al. 2017a).

All predictors were standardized before building the models to allow comparisons of the relative effects (Schielzeth 2010), and for a better control of multicollinearity in model averaging (Cade 2015).

We used GLMMs with a binomial error distribution and a logit-link function to evaluate the effect of explanatory variables on redstart occurrence. Mixed models with sampling site as random intercept (run under R package glmmADMB; Skaug et al. 2015), were used to correct for the potential non-independence of plots within the same sampling sites.

We worked within an information-theoretic framework (Burnham & Anderson 2002) using the dredge function in the R package ‘MuMIn’ (Barton 2015) (see Supporting Information). All possible model combinations for each set of predictors were ranked based on their AICc and only the most parsimonious models (i.e. ΔAICc < 2) were selected, after discarding ‘uninformative parameters’ (Arnold 2010; Richards et al. 2011). We then averaged across the most supported models within each group of predictors, obtaining model-averaged coefficients and relative standard errors, and the relative variable importance for each predictor (Johnson & Omland 2004).

We eventually built a synthetic model, starting with the variables selected according to the above described procedure for each individual group, and adopting the same AICc-based ranking procedure (Koleček et al. 2014).

Land-cover diversity index was strongly collinear with land-cover variables (gVIF= 4.81), thus was tested in a separate single-variable model.

All the analyses were performed with R version 3.2.0 (R Core Team 2016).

3 Results

3.1 Relationships between avian species richness, cultural ecosystem services, and common redstart density

Overall, 96 bird species were censused in the study area, of which 72 breeding or potentially breeding. The mean breeding species richness per sampling site was 32 (SD= 7; range: 23-44) and was associated with redstart density ($\beta = 0.56 \pm 0.21$; LR $\chi^2 = 6.63$, df = 1; p=0.01; N=24; Fig. 3a). Sampling site extent did not affect richness ($\beta = -0.009 \pm 0.012$; LR $\chi^2 = 0.53$, df = 1; p=0.46; N=24). Poisson error distribution was appropriate for the model (over-dispersion parameter = 1.06). The median aesthetic value of the 24 sampling sites was 6 (interquartile range = 1) and was significantly associated with redstart density ($\beta = 4.53 \pm 2.36$; LR $\chi^2 = 3.98$, df = 1; p=0.04; N=24; Fig. 3b). Further details are available in Supporting Information.

3.2 Common redstart ecological requirements

We obtained 348 precise records (240 first locations) of redstart by territory mapping, which were used to define 80 territories and to select 80 control plots.

From the land-cover/topographic group, two supported models were retained, showing a negative effect of apple orchard cover and a positive effect of hedges and tree rows, field margins and cover of urban area, as well as of slope, on redstart occurrence. The management model selection procedure also retained two supported models, which highlighted a negative effect of *spalliera* vineyard cover and a positive effect of the length of dry-stone walls and of vineyard cover.

The synthetic model selection, based on the above listed predictors, identified three supported models (Table 2a) including all the initial predictors, except for slope and vineyard cover. The latter was however included in the first excluded model (△AICc=2.36). The signs of the predictor effects on the response variable were the same as in the individual groups (Table 2b).

Urban cover resulted in a steep increase of occurrence probability, rapidly assuming an asymptotic trend (Fig. 4a). Large confidence intervals for field margin cover (positive effect; Fig. 4b), cover of apple orchards (negative effect; Fig. 4c), and cover of *spalliera* vineyard (negative effect; Fig. 4d) were determined by several outliers, but general patterns were consistent. Positive effects of dry-stone walls and cover of hedges and tree rows on redstart occurrence were clear (Fig. 4e and 4f). Shannon land-cover diversity also had a positive effect on common restart occurrence according to the single-variable GLMM ($\beta=0.99 \pm 0.26$; $\chi^2=14.09$; df= 1; $p<0.001$; n=180; Fig. 4g).

4. Discussion

4.1 Common redstart territory selection in a cultural landscape

Common redstart abundance in Trentino was positively affected by vineyard cover (Assandri et al. 2017b): vegetation structure in vineyards is likely reminiscent of the open forests in warm and sunny climates originally exploited by the species (Cramp 1988), and limited insecticide use and regular sward management (resulting in a mosaic of high and low grass) during the breeding season could promote vineyard suitability for this insectivorous species (Martinez et al. 2010; Schaub et al. 2010). Here, we found at a finer level, i.e. the territory scale, a minor importance of vineyard cover: other characteristics in the vineyard matrix played a crucial role for territory choice. Territory selection was positively affected by environmental attributes linked to extensive agricultural practices and traditional landscapes: hedge and tree rows, dry-stone walls, marginal habitats, land-cover diversity. These attributes still occur in our study area, but not uniformly along the intensification gradient we investigated: they are still rather common where the harsh topography had prevented widespread mechanization and intensification; where the latter occurred, as in most lowland areas, these important landscape traits were dramatically reduced. In parallel, features typical of the modern and intensive agriculture, which are impacting on the traditional landscape, such as the *spalliera* vineyards (which are replacing traditional *pergola* vineyards, Chemolli et al. 2007) and the modern apple orchards characterized by dense rows of dwarfing trees (Brambilla et al. 2015), negatively influenced territory selection.

The importance of dry-stone walls for biodiversity has been scarcely investigated, but such walls may provide breeding and roosting sites (Woodhouse et al. 2005; Manenti 2014), rich and often peculiar (i.e. xeric) plant communities (Holland 1972), which can further increase the number of niches resulting in a more diversified fauna (Baur et al. 1995), and can potentially act as ecological corridors (Dover et al. 2000; Collier 2012). Dry-stone walls favoured redstart, by providing nesting cavities and enhancing habitat heterogeneity and hence feeding opportunities, being generally associated with marginal elements in which redstart may find prey. Modern cemented walls, which are replacing traditional dry-stone walls, did not have any positive effect on the species, being poor of cavities and unsuitable for rocky vegetation and invertebrates.

Redstarts were favoured by hedges and tree rows, likely offering a higher arthropod prey availability and diversity than neighbouring vineyards, both at landscape (Assandri et al. 2017b) and territory scale (this study).

The positive effect of land-cover heterogeneity on redstart occurrence is easily explained by the fact that a high intra-territory heterogeneity is likely to better comply with the ecological requirements of the most demanding species, as insectivorous secondary-cavity nesters (Vickery & Arlettaz 2012; Barbaro et al., 2008). The positive effect of heterogeneity probably also explains the positive association with urban areas at the territory scale, which confirms previous results at the landscape scale. In fact, sparse urban areas increase heterogeneity at the landscape but also at the local scale (e.g. garden and vegetable garden patches provide a heterogeneous sward structure), with likely benefits for the species, in particular within intensive agricultural areas, in which both nesting site availability and foraging opportunities are reduced (Droz et al. 2015; Sedlacek et al. 2004; Assandri et al. 2017b, 2017d, this study).

4.2 Why beautiful cultural landscapes promote birds?

The most interesting result of our work was perhaps that landscapes with higher density of redstarts have higher aesthetic quality. The landscapes with higher aesthetic value (and higher redstart densities) were also perceived by people born in the study region as “traditional”, thus part of the local cultural heritage. This means that the landscapes with higher redstart densities are also the ones most likely to deliver aesthetic and cultural heritage ecosystem services.

What are the reasons behind this pattern? Habitat characteristics positively affecting territory occupancy, often enhance also the density of a species at a broader scale: in fact, a positive relationship between occupancy and abundance (density) is very common in animal populations (Hartley 1998). A landscape where those characteristics are more abundant, could sustain more territories of the species (Kruger et al. 2014). The breeding density of redstart at the landscape scale could thus be affected by the availability of the same habitat features which regulate territory settlement (see e.g. Brambilla et al. 2009; Broyer et al. 2014). Dry-stone walls, hedges, marginal habitats and land-cover heterogeneity favoured redstart occurrence, and in parallel were identified as the traditional traits which qualify this specific landscape as cultural (Agnoletti 2013). Traditional traits are, in fact, the footprint of the millenarian history of land domestication accomplished by humans and have also an outstanding importance in determine how the landscape is perceived by people, strongly contributing to its scenic beauty and cultural heritage value (Lindemann-Matthies et al. 2010; Daniel et al. 2012). These traits, which favour redstart occurrence and contribute to the landscape aesthetic value, have been reported to favour birds (Hinsley & Bellamy 2000; Vickery & Arlettaz 2012), also in our study area (Brambilla et al. 2015; Assandri et al. 2017c), where they promote the presence of several species of conservation concern in the cultivated matrix (Assandri et al. 2016). This explains why redstart density, as expected, was positively related with the avian (breeding) species richness. This relationship corroborates the choice of this species as a good indicator of the wider bird diversity and as a valuable non-traditional flagship species, useful to promote a biodiversity-friendly viticulture in this area (Assandri et al. 2017b), but possibly also in other viticultural districts throughout southern Europe, where the species is commonly found in vineyard (e.g. in south-western France; Barbaro et al. 2017). In the vineyard areas where the common redstart is absent or scarce (for example in Switzerland or in Central France, see Guyot et al. 2017 and Pithon et al. 2015), other species could be identified as flagships to enhance biodiversity conservation in vineyard traditional landscapes. These should share several life history traits with the redstart (e.g. to be insectivorous, to nest in cavities, to be dependent on heterogeneity at the landscape and foraging scales) and should be quite showy or well-known by local farmers. Possible candidates for this role could be the hoopoe *Upupa epops*, the wryneck *Jynx torquilla*, the woodlark *Lullula arborea*, the black redstart *Phoenicurus ochruros*, the great tit *Parus major* or the cirl bunting *Emberiza cirlus*.

4.3 Toward the integration of cultural ecosystem service delivery and biodiversity conservation in farmed cultural landscapes

Extensive agricultural management based on low-intensity, 'traditional' practices is often associated with traits promoting ecosystem resilience and productivity as well as landscape beauty (Daniel et al. 2012; Harrison et al. 2014). Considering studies carried out in the Alps, Lindemann-Matthies et al. (2010) showed that the landscapes harbouring extensively managed, species-rich grassland, isolated trees, and hedges were appreciated for their scenic beauty. Similarly, Fontana et al. (2014) reported that traditional agroforest systems host a higher plant diversity compared with abandoned or intensified systems, also providing a higher amount of two specific ecosystem services (scenic beauty and pollination). However, the former study lacks an explicit biodiversity assessment, whereas in the latter scenic beauty quantification was based only on flower colours and not on a perception-based assessment of the whole landscape.

We explicitly quantified both biodiversity and two cultural ecosystem services, linking these two facets of the cultural landscape by means of a flagship species. Our results exemplify the potential importance of the synergistic integration of multiple conservation targets - biodiversity conservation and the delivery of cultural ecosystem services. Based on them, we advocate that managing a vineyard-dominated landscape to maintain aesthetic and heritage values, which concretely means conserving and enhancing traditional attributes of that landscape such as dry-stone walls, hedges and tree rows, marginal habitats and the overall land-cover diversity, would favour also an indicator/flagship species and likely the whole bird diversity, as shown by the positive correlation between redstart density and species richness (Fig. 5). From both a political and a social point of view, it could be much easier to justify conservation efforts targeted at the delivery of ecosystem services rather than at biodiversity alone, because the former generally appear economically more valuable for the large public (Gobster et al. 2007; de Groot et al. 2010). For example, the aesthetic and environmental qualities provided by a cultural landscape correlate with its recreational/tourism potential, or with the value of the private properties found in that landscape, which then acquires a tangible and objective economic values (Daniel et al. 2012). Additionally, the conservation of birds in the agroecosystem would also support a wider bundle of ecosystem services (e.g. pest control, weed control, seed dispersal), of which they are recognized as important providers (Whelan et al. 2015), also in vineyards (Jedlicka et al. 2011; Barbaro et al., 2017).

Existing policies, such as the ‘greening’ obligations of the CAP, offer some opportunities (e.g. the Ecological Focus Areas) to halt the loss of the traditional landscape traits which qualify a landscape as cultural, biodiversity-rich and provider of cultural services. Unfortunately, permanent crops such as vineyards were excluded from any greening obligations because they were considered environmentally sustainable as they are (Pe’er et al. 2014). This is particularly concerning since large parts of Southern Europe are covered with permanent crop systems (Iglesias et al. 2011), which often represent some of the most outstanding example of cultural landscapes on Earth (Kizos et al. 2012), but which are facing strong agricultural intensification, resulting in a distortion of the traditional landscape (Martínez-Casasnovas et al. 2010).

To have a chance of conserving the invaluable heritages of our rural past and the rich biodiversity therein occurring, we need to carefully think how to realize the integration of multiple conservation instances, working on synergistic strategies combining ecosystem service delivery and biodiversity conservation, with a look also on the local socio-economical distinctiveness of each landscape (Mace et al. 2012). These strategies must finally find their place into policies which adequately address the multi-faceted essence of the cultural landscape.

Supporting information

Questionnaires details and result details are available online.

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Table 1. List of the variables used in the analysis and their mean value (± standard deviation) in territories and control plots.

Variable acronym	Description	TERRITORIES (Mean ± SD)	CONTROLS (Mean ± SD)
	Land-cover/topographic variables		
woods	% cover of woodlands (large majority of broadleaved woodlands)	4.12 ± 7.83	4.06 ± 9.90
apple	% cover of intensive apple orchards	0.85 ± 2.90	6.58 ± 15.34
urban	% cover of urban areas (including isolated houses)	11.50 ± 17.31	0.27 ± 2.39
hedges	% cover of hedge and tree rows, defined as linear clusters of shrubs and/or trees, which were less than 15-m wide, isolated into the farmed landscape or originating from woodlands remains but clearly isolated from the main woodland area	3.91 ± 4.48	0.88 ± 2.18
paved	% cover of paved roads	3.77 ± 3.71	2.21 ± 2.57
open	% cover of open areas (fields, meadows, extirpated wood crops)	2.63 ± 5.85	1.39 ± 5.40
margins	% cover of field margins (also including unpaved roads and small rural buildings)	12.72 ± 5.83	9.47 ± 5.57
slope	Mean territory slope (degrees of inclination from the horizontal)	15.02 ± 6.93	9.91 ± 9.40
radiation	Mean territory solar radiation on 21th June (W/ m^2)	8589.77 ± 321.34	8648.919 ± 287.24
	Management variables		
vineyards	% cover of vineyards	57.45 ± 22.19	72.63 ± 22.75
spalliera	% cover of *spalliera* vineyards	16.99 ± 26.21	17.19 ± 30.06
weeded/ ploughed	% cover of permanent crop fields (vineyards and apple orchards) with chemically weeded or ploughed rows	48.04 ± 33.83	60.04 ± 33.28
parcel area	Mean area of vineyard patches overlapping with a territory (m^2)	2889.24 ± 2364.33	4300.79 ± 2648.79
buildings	Number of isolated rural buildings per territory	0.54 ± 0.69	0.32 ± 0.65
cemented walls	Total length of cemented walls in a territory (m). See supporting information.	56.67765 ± 80.35406	37.20 ± 90.59
stone walls	Total length of dry-stone walls in a territory (m). See supporting information.	141.54 ± 137.39	54.08 ± 110.26
	Other variables		
shannon	Shannon land-cover diversity index	1.08 ± 0.41	0.70 ± 0.44

Table 2. Synthetic model outputs for common redstart occurrence. a) Most supported synthetic GLMM models. Models are ranked according to Akaike's information criterion corrected for small sample size (AICc) and only models within an interval of ΔAICc < 2 are shown. The difference in AICc from the best supported model (ΔAICc), Akaike's weights (w_i), and -2 log-likelihood values (logLik) are also given. Negative (-) or positive (+) relationships between predictors and Common redstart occurrence are shown. b) Model averaged standardized parameter (based on models with ΔAICc < 2) and relative variable importance of predictors (measured considering the sum of the Akaike weights over the most supported models in which that variable appears) from synthetic models. Covariates are ranked according to cumulative weights.
For variable acronyms, see Table 1. N=160.

a)

Model	df	logLik	AICc	ΔAICc	w_i
apple (-) + spalliera (-) + hedges (+) + margins (+) + stone walls (+) + urban (+)	8	-63.663	144.3	-	0.281
apple (-) + hedges (+) + margins (+) + stone walls (+) + urban (+)	7	-64.826	144.4	0.11	0.266
apple (-) + hedges (+) + stone walls (+) + urban (+)	6	-65.960	144.5	0.19	0.255

b)

Variable	β	SE	$\sum w_i$
intercept	0.70	0.47	-
apple	-0.95	0.48	1
hedges	0.97	0.34	1
stone walls	0.71	0.27	1
urban	2.79	0.99	1
margins	0.28	0.29	0.68
spalliera	-0.15	0.26	0.35

Captions

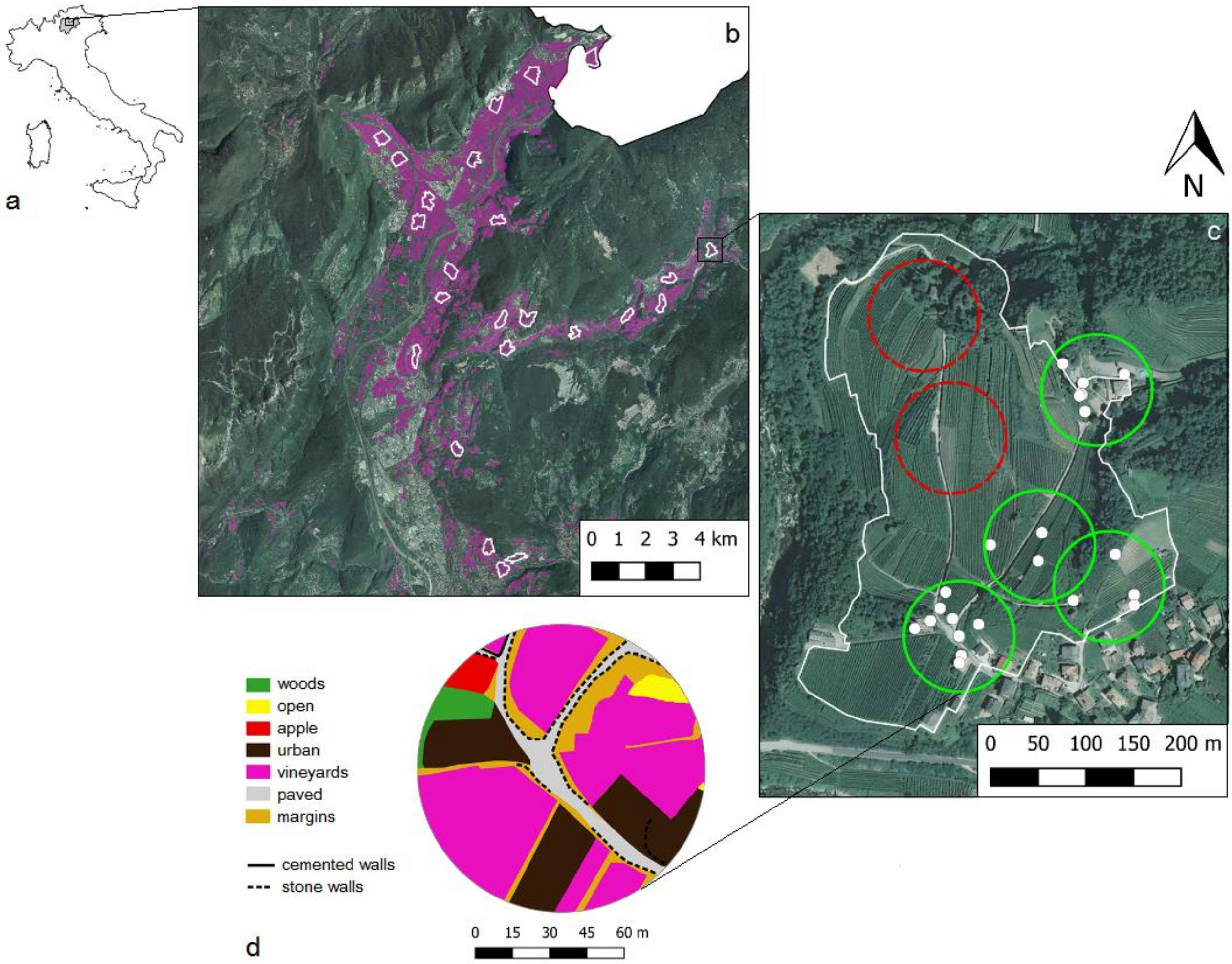
Figure 1. a) Location of the study area in Italy. Trento Province is highlighted in grey. b) The 24 sampling sites in which the present study was conducted; vineyard cover in the area is shown in violet. c) Detail on one of the sampling site (bordered in white); common redstart location obtained in the four subsequent territory mapping sessions (white dots), defined territories (solid green circles) and control plots (dashed red circles) are shown. d) Example of a land-cover-characterized plot based on photointerpretation and field validations; walls occurring in the plots are also shown. Base map: Ortofoto 2011 ©AGEA – Agenzia per le Erogazioni in Agricoltura, Roma.

Figure 2. The two types of walls occurring in the study area. a) Cemented wall. This “modern” type is made of stones taken together with cement and provides virtually no breeding sites for birds and a limited value for biodiversity. b) Dry-stone wall. This traditional type is only made of stones. The recesses between stones provide cavities used by common redstart for nesting. Debris accumulate in these spaces favour the growth of a rich vegetation hosting many invertebrates.

Figure 3. Graphical representation of the effect of common redstart density on breeding species richness (a) and aesthetic value (b), as predicted by the models. Dots represent the observed values.
In a) the other predictor included in the GLM, sampling site area, was kept constant at its mean value. 95% confidence intervals of the mean are shown in light grey. N=24.

Figure 4. Graphical representation of the effect of urban cover (a), field margin cover (b), apple orchard cover (c), *spalliera* vineyard cover (d), dry-stone wall length (e), hedge and tree row cover (f) and Shannon land-cover diversity index (g) on the common redstart’s probability of occurrence, as predicted by the averaged synthetic models. Other predictors included in the models are kept constant at their mean value. 95% confidence intervals of the mean are shown in light grey. N=160

Figure 5. Graphical visualization of the conceptual results of the study. Green arrows represent positive effect on the model species (common redstart), red arrows negative. The arrow going from biodiversity to cultural ecosystem services highlight the multiple facets of biodiversity, which could be a good, as well as a final ecosystems service itself (supporting, provisioning, regulating or cultural) (Mace et al. 2012).



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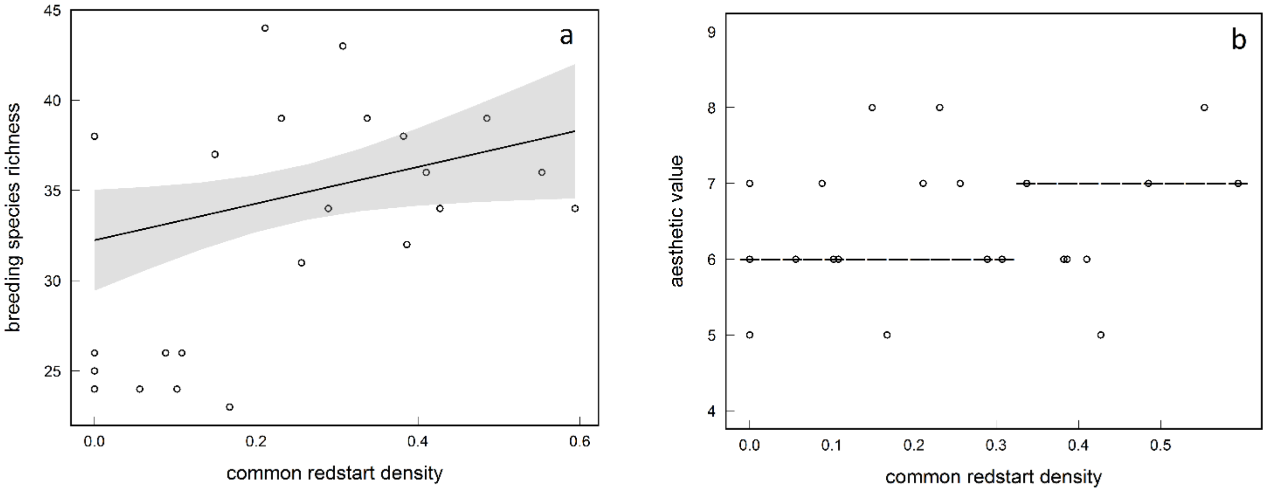

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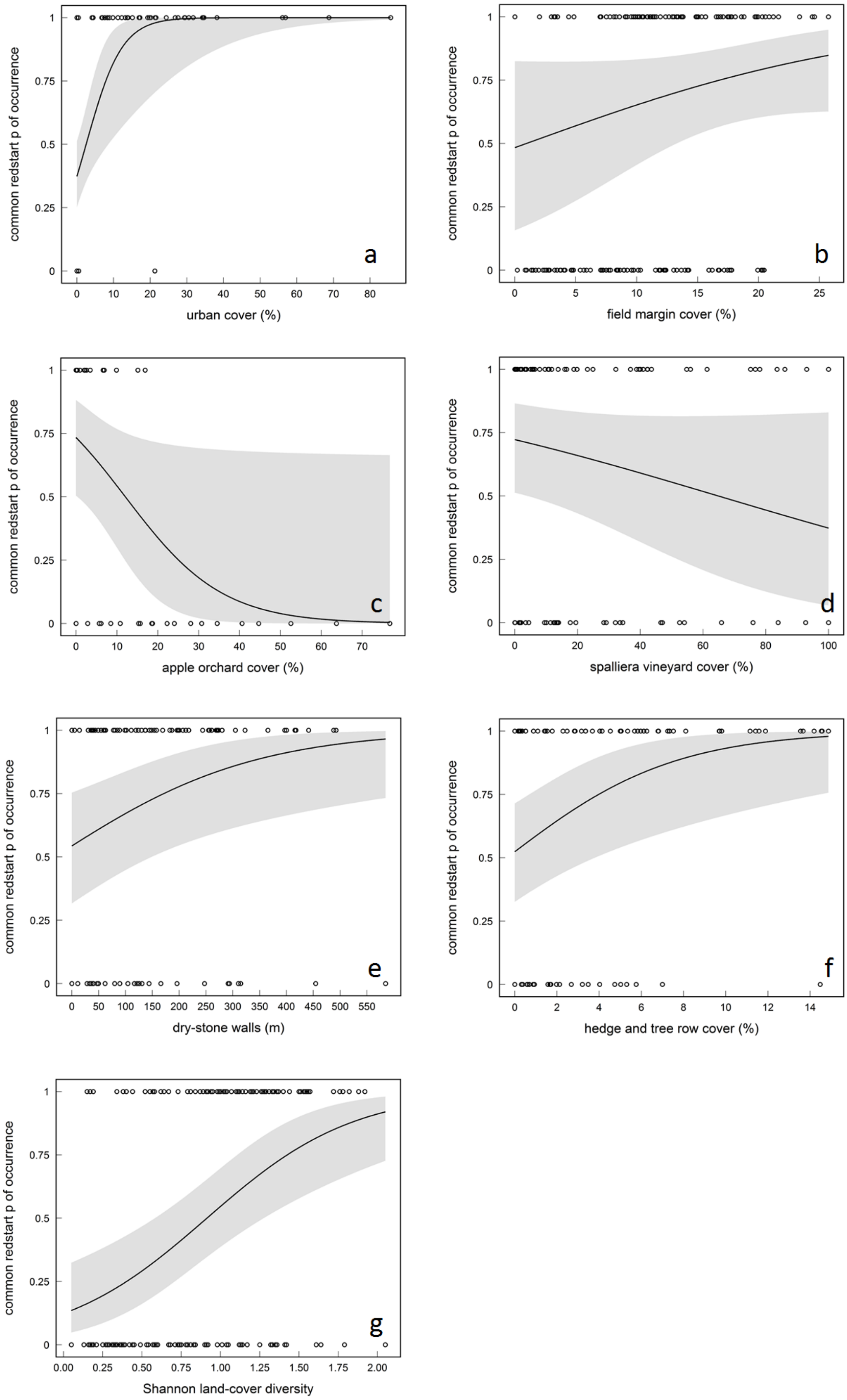

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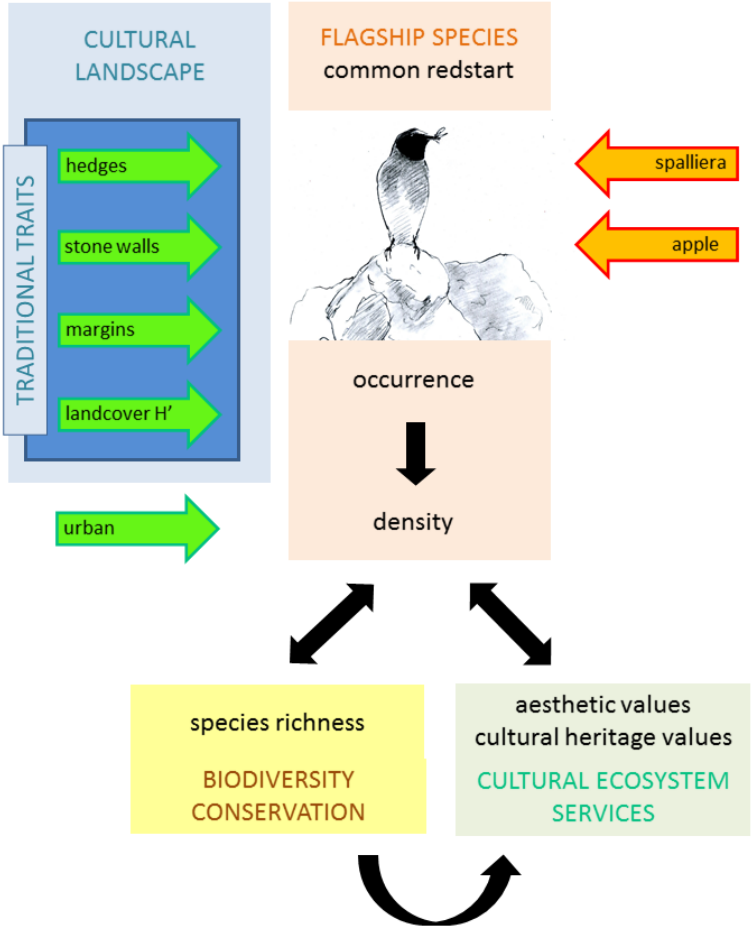

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