# **Sixty years of habitat decline: impact of land-cover changes in northern Italy on the decreasing ortolan bunting *Emberiza hortulana***

Mattia Brambilla[1,2,3], Marco Gustin[2], Severino Vitulano[2,4], Riccardo Falco[1], Valentina Bergero[1], Irene Negri[2], Giuseppe Bogliani[5], Claudio Celada[2]

Mattia Brambilla

Fondazione Lombardia per l'Ambiente, Settore Biodiversità e Aree protette, Largo 10 luglio 1976 1, I-20822 Seveso, MB, Italy; Lega Italiana Protezione Uccelli (LIPU/BirdLife Italia), Via Trento 49, I-43100 Parma, Italy; Museo delle Scienze, Sezione di Zoologia dei Vertebrati, Via Calepina 14, I-38122 Trento, Italy. E-mail: brambilla.mattia@gmail.com. Phone +39 02 80616123; fax +39 02 80616180 (corresponding author)

Marco Gustin

Lega Italiana Protezione Uccelli (LIPU/BirdLife Italia), Via Trento 49, I-43100 Parma, Italy. E-mail: marco.gustin@lipu.it

Severino Vitulano

Associazione FaunaViva, Viale Sarca 78, I-20125 Milano, Italy; Lega Italiana Protezione Uccelli (LIPU/BirdLife Italia), Via Trento 49, I-43100 Parma, Italy. E-mail: severino.vitulano@gmail.com

Riccardo Falco

Lombardia per l'Ambiente, Settore Biodiversità e Aree protette, Largo 10 luglio 1976 1, I-20822 Seveso, MB, Italy. E-mail: riccardo.falco@flanet.org

Valentina Bergero

Lombardia per l’Ambiente, Settore Biodiversità e Aree protette, Largo 10 luglio 1976 1, I-20822 Seveso, MB, Italy. E-mail: valentina.bergero@flanet.org

Irene Negri
Lega Italiana Protezione Uccelli (LIPU/BirdLife Italia), Via Trento 49, I-43100 Parma, Italy. E-mail: irene.negri@coldiretti.it

Giuseppe Bogliani
Department of Earth and Environmental Sciences, University of Pavia, Via Adolfo Ferrata, 9, I-27100 Pavia, Italy. E-mail: giuseppe.bogliani@unipv.it

Claudio Celada
Lega Italiana Protezione Uccelli (LIPU/BirdLife Italia), Via Trento 49, I-43100 Parma, Italy. E-mail: claudio.celada@lipu.it

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**Abstract**

Habitat loss and degradation are main global threats to biodiversity, and land-use changes in agriculture-dominated landscapes are crucial for an important portion of biodiversity, especially in Europe. We evaluated the effects of land-use changes (1954-2012) on a threatened species, the ortolan bunting, in an agricultural area crucial for its conservation in Italy. We built a distribution model for ortolan bunting in current landscapes, and then re-projected it to past scenarios (1954 and 1999-2000). We evaluated the most important land-use changes occurred and estimated their effects on habitat suitability. Bunting occurrence was mostly affected by the extent of grassland (positively; used as foraging/breeding ground), shrubland (quadratic effect; perches/shelter), forest and urbanized land (negatively), and by solar radiation (positively) and slope (quadratic), consistently with other studies carried out especially in southern Europe. The potential distribution of the species was much larger in the past: the estimated decline in suitable habitat is 44%-72% (since 1999-2000/1954), coherently with historical data suggesting strong decline and contraction. Changes in suitability (1954-2012) were mostly associated with changes in the cover of forest, vineyards and abandoned areas (negatively), and shrubland (positively). Land-use/land-cover changes are the main drivers of species occurrence and of habitat decline. The heterogeneous landscape of hilly/low-mountain sites in this area, characterized by a mix of habitats offering complementary resources to ortolan buntings and other species of conservation concern, is currently threatened by abandonment and intensification, but its maintenance may be promoted by a correct definition of Rural Development Programme measures.

**Keywords**

bare ground; conservation; grassland; land abandonment; MaxEnt; Rural Development Programme

## Introduction

Habitat loss and degradation are among the main global threats to biodiversity (Newbold et al. 2015), and land use change and intensification represent a primary cause of biodiversity loss (Foley et al. 2005). Within this process, agricultural land use is particularly important, and increasing pressures on farmed landscapes are resulting in agricultural intensification with widespread loss of grasslands and other natural or semi-natural and economically less remunerative habitats (Dale et al. 2011; Tilman et al. 2001; Tilman et al. 2011; Wright and Wimberly 2013).

As an example, several farmland bird species in Europe have declined drastically due to agricultural intensification (BirdLife International 2015). Open agricultural landscapes cover large parts of Europe (Ostermann 1998). Fields and pastures attract a range of species adapted to millennia of low intensity farming in traditional farmland landscapes (Donald et al. 2002; Russo 2007). The economical growth and the demand for more food determined dramatic changes in farming practices in the second half of the past century, with a consequent widespread agricultural intensification (Robinson and Sutherland 2002). This phenomenon has been associated with an opposite trend of land abandonment in many marginal areas and in rural landscapes in mountains (MacDonald et al. 2000; Robinson and Sutherland 2002; Russo 2007). Also the latter process has detrimental effects for many species and communities tied to open and semi-open habitats (Amici et al. 2012, Chamberlain et al. 2013; Laiolo et al. 2004).

Land use change in agriculture-dominated landscapes are thus crucial for the conservation of an important portion of biodiversity, especially in Europe (see e.g. Wretenberg et al. 2007,2010). With this work, we evaluate the effects of land cover changes due to anthropogenic pressure on a threatened species: the ortolan bunting *Emberiza hortulana*. We focus on an agriculture-dominated area in northern Italy, characterized by particularly high biodiversity values, and consider an almost 60-years long period. Within this area, we evaluate the effects of changes underwent from 1954 to 2012, and of more recent variations occurred between 1999-2000 and 2012, on environmental suitability for ortolan bunting.

Ortolan bunting is one of the most severely declining species in Europe; it is vanishing through most of its range (Menz and Arlettaz 2012) and in the past 30 years it showed the most dramatic decline (-89% in Europe) among 38 widespread species which are Afro-Palearctic migrants (Vickery et al. 2014). Given that it mostly occurs in Europe (which corresponds to 50-74% of the species range) and that changes to breeding habitats are likely to have a crucial role in driving species decline (Goławski and Dombrowski 2002; Menz and Arlettaz 2012; Revaz et al. 2005; Vepsäläinen et al. 2005), adequate conservation measures are required in European countries.

This species mostly occupies open and semi-open habitats, although habitat preferences are different within the species range (Cramp and Perrins 1994). In southern Europe, where a notable portion of the species population breeds, ortolan buntings inhabit open and semi-open shrubland, heterogeneous and semi-open farmlands and steppe-like habitats (Brotons et al. 2008; Cramp and Perrins 1994; Guerrieri et al. 2006; Menz et al. 2009; Morelli 2012). They occupy areas characterized by sparse vegetation, with scattered trees (Cramp and Perrins 1994), i.e. early vegetation stages (Menz et al. 2009), including burnt areas (Brotons et al. 2008; Dale and Manceau 2003; Menz et al. 2009), while avoiding more mature vegetation stages characterized by denser and taller vegetation (Bogliani et al. 2003; Sirami et al. 2007). This kind of preferences for early stages is likely due to the foraging habits of the species, which feeds mostly on bare or sparsely vegetated ground (Menz and Arlettaz 2012). Taller elements, such as bushes, trees or rocks are used as songposts, whereas nests are made mostly on the ground (Cramp and Perrins 1994).

With this work, we aim to identify the habitat composition at the landscape scale affecting the species distribution at the regional level. This kind of information is of crucial importance for the adequate implementation of conservation measures for this dramatically declining species. We evaluate landscape traits affecting species distribution in a study area in northern Italy, where the ecology of the species has been recently investigated at the territory level (1 ha) within selected study plots, on the basis of fine-scale variables measured on the ground (Brambilla et al. 2016).

Ortolan buntings have been reported to be associated at such a fine scale to the following habitat traits, listed in order of importance: bare ground (at least 5% of the potential territory), lucerne cover (around 50%), shrub cover (medium or high cover), length of hedgerows or tree rows (at least 25 m / ha) (Brambilla et al. 2016). Here, we consider a broader, landscape scale and a much higher number of territories scattered over a wider area, and use habitat variables derived from GIS (Geographical Information System) layers to describe the current and historical landscapes, and to identify the factors affecting environmental suitability for the species by means of species distribution models. We then estimate the historical variations in habitat suitability for ortolan buntings. On the basis of previous knowledge from this area and from elsewhere, at the landscape scale we consider in this work (see below), we expect i) a roughly quadratic effect of some of the features important at the territory scale, and in particular some types of land cover shaping the landscape of the study area, such as shrubland and arable land (Bogliani et al. 2003), which are oftern included in the species' territories; ii) a positive effect of the cover of semi-natural grassland grazed or used for hay-making, which characterizes low-intensity farming ecosystems in the area; iii) a negative effect of forest cover, given the association with open and semi-open habitats reported for the species in southern Europe (e.g. Guerrieri et al. 2006); iiii) a potential effect of topographical features, such as slope and solar radiation (Menz et al. 2009).

The availability of species' occurrence data and of detailed GIS habitat layers for different periods over the past 60 years offers an extraordinary opportunity to quantify the current and past distribution of suitable habitat for ortolan buntings at the landscape scale, and to relate the changes in land cover with the changes in the distribution of potential habitat for this threatened species.

## Methods

### Study area

The study was carried out in the southern belt of Lombardy, located south to the Po River and including the Apennine belt comprised within the regional boundaries. This heterogeneous area, generally referred to as Oltrepò pavese, occupies c. 1100 $km^2$ and is characterized by increasing elevation and slope from north to south. Elevation varies from c. 50 a.s.l. near the Po to 1724 m of the Mount Lesima. The lowland portion is mostly covered by arable land (mostly cereal crops), with small woodlots, poplar plantations and wetlands, whereas the foothill is largely covered by vineyards; at middle elevation, a mosaic of woodlands, non-intensive cultivations and vineyards occurs, whereas at middle-upper and upper elevations the landscape is dominated by woodlands with pastures interspersed, the latter being often abandoned (Brambilla et al. 2012).

Towns and villages occur within the entire area, according to a gradient opposite to the altitudinal one. In general, the human density is quite low if compared with lowland areas in northern Italy.

The climate of the study area is temperate, with rainfall (average amount c. 700-1500 mm/year) increasing and temperature (average mean 5°-12°C) decreasing with elevation (Bogliani et al. 2003; Abeli et al. 2012). The climate is thus largely comprised within the extremes tolerated by the species, even if the optimal rainfall amount according to the literature is on average lower (Cramp and Perrins 1994; but note that in Italy the species occurs in areas with even higher amount of rainfall, cf. Nardelli et al. 2015).

Bird census

During May-July 2010-2014 Ortolan buntings were censused at all elevations and within all major landscape types, by means of linear transects and territory mapping. 121 linear transects (fixed lenght 400m) were scattered above open and semi-open areas, and all the main landscapes types and units within the study area were included; transects had random orientation and covered all the elevational belts of the area (Brambilla et al. 2012). Territory mapping was carried out in 10 focal plots (average area 110 ha), all mainly covered by open or semi-open landscapes, with varying proportions grassland, arable land, vineyards, shrubland and forest patches (see Brambilla et al. 2016 for further details on territory mapping). Transects were censused twice in spring 2010 and the focal plots were visited four or more times in spring 2011. Some additional casual observations (obtained e.g. when moving between plots or transects, or in the course of other fieldwork activities outside the main study sites) were also included. In all cases, the exact location of ortolan buntings (considering both visual and vocal contacts) was recorded; observations of individuals just flying over the area surveyed, or recorded with an approximation larger than 100 m were discarded.

Distribution modelling

We built a distribution model for ortolan bunting in Oltrepò considering locations from independent territories: in the case of precisely mapped territories, we considered the centre of each territory; in the case of other records closer than 100 m, we randomly selected one of the records.
We used the so resulting 84 independent bunting locations to build a distribution model using MaxEnt 3.3.3 k (Phillips et al. 2006), one of the most performing and frequently used methods for modelling species distribution using presence-background methods (Elith et al. 2006, 2011), including when data have been collected under different field methods, as in our study. The background was created using 30 000 points randomly generated by MaxEnt. The model was built using linear and quadratic fitting functions (Brambilla et al. 2015) and selecting variables representing land-use and topography (Brambilla 2015; Brambilla et al. 2015; see Table 1). Climate variables were not considered as all the area is within the climate potentially suitable for the species, according to both temperature and rainfall amount (see above).

#Table 1 approximately here#

We derived land-cover variables from the regional database DUSAF 4.0 (resolution 20 m, date:

2012; Regione Lombardia 2014), a high-resolution database already used for species distribution models in other studies (e.g. Brambilla and Ficetola 2012; Brambilla et al. 2012, 2014). For each land-use type, we calculated the relative cover within a 100-m radius from each cell (20 m × 20 m cell), and used such cover value (expressed in $m^2$ x 100) for model building. Topographic variables included slope and mean solar radiation (the latter calculated as for 21st June; both potential proxies of microclimatic conditions) from a 20-m resolution digital terrain model and entered as the average value in the 100-m radius from the cell.

As a measure of the matching between the predicted suitability and the observed distribution, we calculated the area under the curve (AUC) and the relative standard deviation by means of two different approaches. In the first one, we performed 100 model runs and a bootstrap approach according to which iteratively 30% of the species records were discarded from the training data set and used for testing each model (cf. Engler et al. 2014). Secondly, we performed a 10-fold cross-validation of the distribution model. We acknowledge AUC limitations as an absolute method for the evaluation of model accuracy (e.g. Lobo et al. 2008), but we used it only to compare the same models over varying partitioning of data into training and testing data.

The continuous output (logistic) provided by the model was reclassified into a binary variable describing occurrence and absence of ortolan buntings using the Equal Training Specificity and Sensitivity threshold (Bartel and Sexton 2009, Brambilla and Ficetola 2012). The choice of this individual threshold among all the available ones was justified as such threshold led to the reclassification most closely matching the actual distribution of the species. The latter was assessed considering the actual range as reported by Nardelli et al. (2015), the distribution according to the ongoing national atlas of breeding birds (www.ornitho.it, accessed on 29th October 2015), and our own knowledge on species' distribution in the area. Notably, other frequently used thresholds, namely Maximum Training Sensitivity plus Specificity and 10th percentile, had values very close to the selected one and led to highly matching estimates of species occurrence.

Then, we re-projected the distribution model to past scenarios, obtained thanks to the availability of land cover maps describing land-use in 1954 and in 1999-2000 (available on www.geoportale.regione.lombardia.it), adopting the same categories and created following the same criteria of the DUSAF 4 map.

Land-cover changes and environmental suitability

We evaluated the most important land-use changes between 1954 and 2012, and between 1999-2000 and 2012, by estimating the relative cover of each land-use type.
To estimate what changes mostly affected the decrease in habitat suitability during the two periods considered, we randomly scattered 10 000 points throughout the entire study area and measured at each point i) the differences in environmental suitability as calculated by the MaxEnt model between current and the two past conditions, respectively, and ii) the differences in land-cover in the 100-m radius between current conditions and the two past conditions, respectively, for urbanized areas, forest, shrub, vineyard, bare ground and grassland cover. Then, we evaluated the correlation between land-cover and environmental suitability changes for each period.

## Results

Current distribution

The distribution model appeared fairly stable, showing good similar discriminatory ability over the training and the testing data and limited standard deviation (average AUC±SD over the runs: 10-fold cross-validated models: training data 0.948±0.003; test data 0.942±0.029; 100 bootstrap runs: 0.951±0.010). The most important variables affecting the environmental suitability for the ortolan buntings were related to land use: in particular, the extent of permanent grassland positively affected suitability and the cover of shrubland had a quadratical/positive effect, whereas forest cover and urbanized land negatively impacted on environmental suitability (Fig. 1).

#Figure 1 approximately here#

Also topographical factors contributed to the definition of the suitability for the species, which is positively affected by increasing solar radiation and is associated with intermediate slope values (see Fig. 1 and Table 2). The result of the jackknife test substantially confirmed the ranking provided by the percentage contribution of the different variables, but in addition it highlighted the relative importance of permanent grassland, which resulted in highly performing single-variable models (Table 2).

#Table 2 approximately here#

Past distribution and estimated variation

The potential distribution of the species was much larger in the past. The currently suitable area is equal to 12 893 ha. The suitable area covered 46 514 ha in 1954 and 23 212 ha at the end of the last century. The estimated decline is equal to 72% since 1954, and to 44% since 1999-2000 (Fig. 2).

#Figure 2 approximately here#

Land-cover changes and habitat loss

Between 1954 and 2012, both arable land and bare soil decreased by 39%; other land-cover showed spectacular increases, and in paricular vineyards (+165%), urbanized areas (+144%), forest (+59%), shrubland (+38%). Abandoned areas (formerly used for agricultural purposes) changed from 2.3 to more than 4 000 ha. Between 1999-2000 and 2012, the pattern was substantially similar although obviously less extreme. Forests kept stable, arable land (-14%) and bare soil (-12%) decreased, whereas vineyards (+8%), urbanized areas (+9%) and shrubland (+81%) increased. The dramatic increase in abandoned areas continued (+173%).

Considering land-cover showing average variations within the periods of at least 1 $m^2$/point at the 10 000 random points, an increase from 1954 to 2012 was found for urban areas (+13.75 $m^2$/point), forest (+28.3 $m^2$/point), abandoned areas (+10.19 $m^2$/point), vineyards (+25.47 $m^2$/point), whereas a decrease was found for grassland (-1.35 $m^2$/point). From 1999-2000 to 2012, an increase was found for abandoned areas (+5.26 $m^2$/point) and vineyards (+3.07 $m^2$/point), and a decrease for grassland (-3.67 $m^2$/point).

A significant correlation was found between the changes in environmental suitability for ortolan bunting between 1954 and 2012 and the changes in the cover of forest ($r = -0.48$, $p < 0.001$), vineyards ($r = -0.33$, $p < 0.001$), abandoned areas ($r = -0.31$, $p < 0.001$) and shrubland ($r = 0.12$, $p < 0.001$). Considering changes undergone between 1999-2000 and 2012, a significant association was found with abandoned areas ($r = -0.44$, $p < 0.001$), forest ($r = -0.43$, $p < 0.001$), grassland ($r = 0.37$, $p < 0.001$), vineyards ($r = -0.15$, $p = 0.049$) and shrubland ($r = 0.11$, $p < 0.001$).

## Discussion

We evaluated the factors affecting the distribution of the ortolan bunting at the landscape level and the variations in the distribution of potential habitat for this declining species from 1954 to 2012. Land-use changes are among the main drivers of changes in species distribution and population size

(Leemans and Zuidema 1995), and the distribution of ortolan buntings in the study area was strongly affected by land use. Ortolan bunting was mainly positively associated with permanent grassland and positively/quadratically with shrubland (consistently with previous knowledge, see Cramp and Perrins 1994; Menz and Arlettaz 2012; Pruscini et al. 2013; Brambilla et al. 2016). It was also associated with an intermediate cover of bare ground (as found in other areas, Berg 2008; Menz and Arlettaz 2012; Menz et al. 2009), although the effect of the latter seemed weaker at the landscape scale (3.14 ha, this study) than at territory level (1 ha, Brambilla et al. 2016). The association with such different habitats is due to the need for perches and potential shelter, offered by shrubs, and for feeding ground, such as grassland and bare ground, the latter being particularly important for prey collection (Menz and Arlettaz 2012); ground at the basis of grass or shrubs is used as breeding site. In addition, the species was strongly and negatively affected by the cover of forest habitats and urbanized areas; this is not surprising for a species tied to open and semi-open landscapes and generally avoiding heavily 'humanized' areas (Cramp and Perrins 1994). However, elsewhere the species had been reported to be associated to small forest patches, embedded in open or complex mosaic landscapes (Kosicki and Chylarecki 2012). Considering abiotic factors, slope contributed to describe the species distribution in the area, this also mirroring previous findings (Cramp and Perrins 1994; Menz et al. 2009). This means that the ideal habitat of the species is represented by a mix of grassland and shrubland, with patches of bare ground, over gently sloping hillsides (see also Brambilla et al. 2016).

At this landscape scale, our results showed a strikingly consistency with the pattern reported for ortolan buntings in Catalonia, despite the completely different landscapes and study approach (considering both field data and statistical analyses). In burnt areas in Catalonia, the number of ortolan buntings peaked at slopes around 15°-20°, with a predicted abundance higher than one individual per transect with slopes comprised between 10° and 25° (Menz et al. 2009); in our study area, environmental suitability for the species peaked around 15° (Table 2). Moreover, in Catalonia the pattern of ortolan bunting abundance suggested an optimum level of bare ground at around 20–

30%, a figure identical to the pattern depicted by the single-variable species-habitat relationship in the MaxEnt model, which suggested maximum suitability with c. 24% of bare soil (Table 2). Similarly, also the preference for slopes well exposed to sun had been already highlighted by Menz et al. (2009).

Model validity was confirmed by both the validation statistics and by a comparison with the current knowledge on the species' distribution. A visual inspection of the predicted distribution (Fig. 2) in relation to the recently reported regional range (Fondazione Lombardia per l'Ambiente 2015) suggested that the model represents well the distribution of ortolan buntings in the study area.

Land-use changes and consequences for bunting habitat

Land-use changes occurred both historically (since 1954) and recently (since the end of the past century) resulted in a marked decline of the suitable habitat for the species, which had lost almost 75% of its potential distribution since the half of the 20$^{th}$ Century. Around half of the historical range of the species became unsuitable by the end of the century; since then, a further 45% of the remaining suitable areas has been lost. The extent suitable for the ortolan bunting decreased all over the study area, from lowlands to mountain areas. We recognize that caution is needed when applying distribution models over different periods, because of the lack of detailed information about breeding density and habitat preferences in the past on the one side, and of potentially different management applied to the same habitats, which could alter their relative suitability, on the other one.

Although precise information about the historical species distribution is lacking, the available data suggest that in the past (1966-1975) the species was much more widespread, occurring also in lowlands towards the Po river (Barbieri et al. 1980), where the species is now largely absent because of unsuitable habitats, but where it was predicted to occur in 1950s (Fig. 2). In low hills, the species has almost completely disappeared from the eastern portion (our data; see also Fig. 2), where it regularly occurred in the past (Brichetti and Fasola 1990); now most of the landscape in this portion of the study area is covered by vineyards. Therefore, even if we cannot exclude that the species currently has habitat preferences somehow different from the past, it is likley that the basic habitat-species relationships at the landscape scale remained more or less unchanged over the timeframe here considered. Coherently, in some specific areas the disappearance of the species coincided with the transition from grassland or mosaic habitats into intensive crops (e.g. poplar plantations or maize fields) or urban development in lowland, or into vineyards in hilly areas (E. Vigo pers. com.).

Land abandonment and agricultural intensification were the main causes of the observed land-use changes and of the consequent decline of suitable habitat for ortolan buntings. Land abandonment is the main issue currently threatening open and semi-open habitats in northern Apennines, especially at middle and high elevation, where shrub and tree encroachment occurs at the expense of semi-natural grassland and pastures no longer mown or grazed (Brambilla et al. 2010). Agricultural intensification in the study area could be described as a two-way process, involving on the one side the removal of marginal features (such as residual unmanaged grassland, shrub patches, tree rows or hedgerows) and the conversion of grassland into cereal crops in lowland areas, and on the other side the expansion of vineyards at the expense of grassland in foothill areas (Bogliani et al. 2003; Brambilla et al. 2007b, 2009a, 2009b). Both these intensification processes had impacted on habitat availability and suitability for ortolan buntings.

The main changes occurred in land-cover in Oltrepò consisted indeed in a marked increase over the sixty-years period of abandoned areas, forest, vineyards, urban areas and shrubs, and of abandoned areas, shrubs and vineyards in the last 12 years. On the other side, open habitats tied to farming practices and in particular arable land declined, as already reported for the study area (Brambilla et al. 2010).

The association between changes in suitability and changes in the cover of different land-cover types suggested that the increases of forest, abandoned areas and vineyards, have been the major causes of habitat decline (in terms of extent and of suitability) for ortolan buntings in the last sixty years. Looking at the more recent variation, the expansion of abandoned areas and forests (and, secondarily, of vineyards), as well as the reduction of grassland, mostly affected variation in environmental suitability. On the other side, shrub increase slightly favoured the species over both periods.

The cover of forest and of urban area negatively affect the distribution of some other species of conservation concern in the same area, including larks (e.g. *Alauda arvensis* and *Lullula arborea*; Brambilla and Rubolini 2009; Brambilla et al. 2012), shrikes (*Lanius collurio* and *L. minor*; Brambilla et al. 2009, 2010) and other buntings (e.g. *Emberiza cirlus, E. calandra*, *E. melanocephala*; Brambilla et al. 2009b 2010; Brambilla 2015), and forest expansion over once grazed or cultivated areas is thought to be a major source of decline and loss of suitable habitat for threatened species such as the red-backed shrike (Brambilla et al. 2010). A similar pattern has been reported for Spain, where open or semi-open habitat species like red-backed shrike and tree pipit *Anthus trivialis* declined in an abandoned area in north-western Spain (Regos et al. 2016).

## Conservation implications

Preserving suitable breeding habitats is of primary importance for the conservation of ortolan bunting. Our main results could thus be used to give management recommendations for conserving or improving the species' habitat in this portion of its range and probably also abroad, given the consistency of the habitat-species relationships with studies carried out in other countries.

Coherently with recommendations given at a finer scale on the basis of territory selection in the species (Brambilla et al. 2016), a critical point is the maintenance of the low-intensity farming mosaic which still occurs in this portion of the Apennines. Hilly and low-mountain areas are characterized by a heterogeneous landscape including small fields, vineyards, grassland, hedgerows, shrublands, woodlots and calanques (eroded mountainsides with sandy/rocky soils subject to landslides). These habitat mosaics fulfil different requirements by providing breeding and foraging habitats and song-posts to ortolan buntings (Menz and Arlettaz 2012; Brambilla et al. 2016) and other species, including several species of conservation concern, such as larks (Bogliani et al. 2003; Brambilla and Rubolini 2009; Brambilla et al. 2012), shrikes (Brambilla et al. 2007a; Chiatante et al. 2014; Brambilla et al. in press) and other buntings (Brambilla et al. 2008; Brambilla 2015). This species-rich mosaic landscape is currently threatened by abandonment and by agricultural intensification (see above), mostly in terms of vineyard expansion.

Considering that ortolan buntings may tolerate some cover (and even appreciate small extent) of vineyards, it is important to limit their expansion in mosaic areas, within which their relative extent should be limited and by all means kept below 50% of the landscape (according to the modelled species-habitat relationship, which shows a drastic decline in suitability with vineyard cover exceeding half of the area of the 100-m radius circle). At the same time, in areas currently entirely covered by vineyards and thus unsuitable to buntings and other species within hilly areas otherwise potentially adequate for the species, it would be useful to restore grassland and shrubland patches with some bare ground.

Urbanization also contributed to a decline in extensively farmed habitats, and key sites especially at lower elevation (where new urbanization mostly occurs) should be preserved from further urban development.

Tree planting (often involving non-native conifers or fruit trees) over grassland (see Abeli et al. 2012) is locally causing a marked decrease of the available habitat, especially when performed on slopes partly subjected to soil erosion, particularly suitable for ortolan buntings thanks to the co-occurrence of grassland, shrubs and bare soil (Brambilla et al. 2016). Such interventions destroy or reduce highly suitable habitats characterized by the fine-scaled mosaic particularly suitable for the species.

In conclusion, a broad conservation strategy should be implemented at the landscape scale, to counteract further abandonment and intensification, thus promoting the conservation of the low-intensity farming system and of the associated mosaic landscape. In grassland subject to recent abandonment (with consequent shrub and tree encroachment) or tree plantation, the restoration of grassland habitats should be a conservation priority, for birds as well as for several reptiles, invertebrates and plant species (Bogliani et al. 2003, Abeli et al. 2012).

Within this landscape strategy, further actions could be promoted at a smaller scale to favour ortolan buntings. In particular, the creation of compact habitat mosaics including grassland, shrubs and bare ground (Brambilla et al. 2016), should be promoted within the portions of northern Apennines depicted as suitable by the distribution model (Fig. 2).

The Rural Development Programme (RDP), which in Italy is defined at the regional level, could be a potentially important tool for the species conservation. The landscape strategy could be implemented by a targeted definition of the standard measure included in the RDP, whereas the further actions at a finer, more local scale would fit well into a dedicated agri-environmental scheme (Brambilla et al. 2016).

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**Table 1** Habitat variables used to model distribution (regional scale) and fine-scaled habitat selection in ortolan buntings in northern Italy. Cover were measured (within the 100-m radius) in $m^2$ x 100.

| **Variable** | **Description** |
|---|---|
| abandoned areas | cover of recently abandoned grassland or fields progressively invaded by spontaneous vegetation (herbs or low shrubs) |
| arable land | cover of crops cultivated on arable land |
| bare soil | cover of sandy, rocky and other unvegetated (plant cover <20%) areas |
| forest cover | total cover of forest habitats |
| open water | total cover of open water |
| orchards | cover of fruit orchards |
| permanent grassland | cover of permanent grassland |
| shrubland with trees | cover of shrubland with high shrubs and/or scattered trees (cover of trees <10%) |
| urbanized | cover of urbanized areas |
| vineyards | cover of vineyards |
| slope | mean slope (°) |
| mean solar radiation | mean total solar radiation (calculated as for 21$^{st}$ June) |

**Table 2** Summary of the MaxEnt model for ortolan bunting in Oltrepò. Values for percentage contribution, permutation importance and model gains are average ones from the 10-fold cross-validated (first value) and the 100-bootstrap runs model (second value).

| Variable | Percentage contribution | Permutation importance | train gain without variable | train gain with only variable |
|---|---|---|---|---|
| abandoned areas | 0.46/1.14 | 0.32/0.61 | 2.26/2.42 | 0.04/0.05 |
| arable land | 2.76/3.66 | 0.16/1.35 | 2.26/2.43 | 0.07/0.08 |
| bare soil | 0.01/0.48 | 0.02/0.07 | 2.26/2.42 | 0.04/0.06 |
| forest cover | 27.10/25.87 | 47.70/45.63 | 1.83/2.04 | 0.40/0.40 |
| open water | 0.53/0.49 | 0.11/0.07 | 2.25/2.43 | 0.01/0.01 |
| orchards | 2.18/1.96 | 2.41/3.00 | 2.17/2.36 | 0.03/0.03 |
| permanent grassland | 17.47/16.59 | 0.28/0.19 | 2.19/2.35 | 0.37/0.40 |
| shrubland with trees | 12.74/11.88 | 0.14/0.27 | 2.23/2.39 | 0.32/0.35 |
| urbanized | 18.12/17.16 | 27.93/27.28 | 1.96/2.14 | 0.34/0.34 |
| vineyards | 1.95/2.13 | 4.72/2.68 | 2.19/2.38 | 0.03/0.06 |
| slope | 16.78/18.39 | 16.17/18.59 | 1.80/1.94 | 0.25/0.29 |
| mean solar radiation | 0.01/0.24 | 0.04/0.25 | 2.26/2.43 | 0.04/0.04 |

**Fig. 1** Species-habitat relationships for the most important variables (see previous Table 1) and bare soil. Values on the x-axis represent degrees (slope) or the relative cover (in $m^2$ * 100; all other variables). The curves shown are those of single-species models worked out according to the 100-bootstrap procedure. The area shaded grey around the curve represents the relative standard deviation.

**Fig. 2** The predicted distribution of ortolan buntings within the study area during the three study periods (see text for details).