Insectivorous birds as 'non-traditional' flagship species in vineyards: Applying a neglected conservation paradigm to agricultural systems

Giacomo Assandri[a,b,*], Giuseppe Bogliani[a], Paolo Pedrini[b], Mattia Brambilla[b,c]

[a] University of Pavia, Dept. of Earth and Environmental Sciences, Via Adolfo Ferrata 9, I-27100, Pavia, Italy
[b] MUSE Sezione Zoologia dei Vertebrati, Corso del Lavoro e della Scienza 3, I-38123, Trento, Italy
[c] Fondazione Lombardia per l'Ambiente, Settore biodiversità e aree protette, Largo 10 luglio 1976 1, 20822, Seveso (MB), Italy

ARTICLE INFO

ABSTRACT

In intensive farmland, it is often difficult to find charismatic and evocative species to serve as a flagship to inspire conservation actions and awareness. Here, 'non-traditional' flagship species may play a fundamental role in promoting more sustainable and biodiversity-friendly agriculture and achieving conservation targets in this strongly artificial habitat.

We investigated the abundance (measured in number of territories) of two insectivorous avian species, spotted flycatcher and common redstart, in response to environmental characteristics and agricultural management practices in Italian vineyards at landscape and field scales. We evaluated whether these species could be used as 'non-traditional' flagships in this agroecosystem.

We found that both species were positively affected by vineyard cover, but not all typologies of vineyards are equally suitable for them. The traditional *pergola* trellising system was favoured by both species, whereas *spalliera* vineyards exerted negative effects. The spotted flycatcher commonly occurred in intensive vineyards on valley floors where its abundance was negatively affected by young plantations, which provide limited nesting opportunities, and by the cover of organic vineyards. Conversely, the common redstart was positively affected by traditional elements interspersed in the vineyard matrix (i.e. hedges, tree rows, and isolated rural buildings), which were more abundant in the less intensively cultivated valley sides. At the field scale, common redstarts selected for grass sward shorter than 20 cm, which highlights the importance of active sward management for this ground-feeding species.

Based on our results, we proposed using the spotted flycatcher as a 'non-traditional' flagship species for intensive vineyards and the common redstart for the less intensive vineyards.

1. Introduction

Agricultural systems are often perceived as of little importance for conservation or even as species-poor environments. In fact, while traditional agroecosystems and low-intensity farmlands could host an outstanding biodiversity (Perrings et al., 2006; Tscharntke et al., 2005), modern agriculture has strong negative effects on biodiversity, largely attributable to the intensification of farming practices (Burns et al., 2016; Foley et al., 2011; Tscharntke et al., 2005) and to the abandonment of marginal and less productive areas (MacDonald et al., 2000; Rey Benayas et al., 2007).

In the European Union, the Common Agricultural Policy (CAP) has been the major driver of agriculture intensification and land abandonment (Donald et al., 2002; Young et al., 2005).

Despite increasing awareness of these processes and the claims for a "greener" CAP reform in 2013, the environmental prescriptions of the revised CAP (that will be in force until 2020) had been so weakened that they are unlikely to enhance agroecosystems and their biodiversity (Pe'er et al., 2016, 2014).

In Europe, permanent crops (e.g. vineyards and orchards) represent the third most widespread farming system (Iglesias et al., 2011). They cover about 6% of the cultivated surface in the EU, but in the Mediterranean countries, their share is much higher (17%-26%) (http://ec.europa.eu/eurostat/statistics-explained/index.php/Agri-environmental_indicator_-_cropping_patterns). They were questionably considered 'green by definition' (Hart, 2014) and were thus excluded from any 'greening' obligations, which are mandatory for farmers to be eligible for the CAP direct payments provided for other crop types.

However, permanent crops are not immune from intensification and abandonment (Caraveli, 2000; Ribeiro et al., 2014). Unfortunately, there is still limited knowledge about the impacts on biodiversity in these crop types (Balmford et al., 2012).

Among permanent crops, vineyards, due to their high economical value and in response to climate change pressure, are rapidly expanding at the expense of natural and semi-natural ecosystems in areas where they never occurred (Hannah et al., 2013; Winkler and Nicholas, 2016). Additionally, traditional vineyard agroecosystems are experiencing unprecedented intensification (e.g. increasing fertilizer and pesticides inputs, massive use of machinery, large scale irrigation) and landscape simplification (e.g. field size increasing, elimination of marginal habitats) (Martínez-Casasnovas et al., 2010), which poses new threats to biodiversity (Viers et al., 2013).

Considering organisms inhabiting agroecosystems, farmland birds have experienced a massive population depletion in the last few decades (Donald et al., 2001), and no signs of recent inverse trends have been reported (Inger et al., 2014). A decline of this magnitude in formerly abundant and widespread species has been almost unique in natural systems (Donald et al., 2002). For this reason, they have been thoroughly used as models for conservation in farmed landscapes (Brambilla et al., 2009; Gottschalk et al., 2010; Gregory et al., 2005; Santana et al., 2014). The causes of this dramatic process are still debated, but the intensification of farming practices has had a major role (Chamberlain et al., 2000; Donald et al., 2001). Among the changes related to intensification, those which probably most impacted birds were the increase in pesticides use and field size and the reduction of habitat diversity and marginal habitats occurring within the agroecosystem (Donald et al., 2002 and reference therein). The decline of farmland birds is particularly concerning when also considering the fundamental ecosystem services that birds provide; for example, insectivorous birds are efficient pest controllers (Sekercioglu et al., 2004; Whelan et al., 2015).

For these reasons, farmland bird ecological requirements and causes of their decline have been thoroughly investigated, but studies focused mainly on cereal field and grassland habitats (Bradbury and Kirby, 2006), whereas scarce attention was posed on farmland birds in permanent crops, and in particular in vineyards.

'Flagship' species are charismatic species that could serve as a symbol to inspire conservation actions and awareness (Caro et al., 2004). These are typically large mammals (e.g. elephants; Johnsingh and Joshua, 1994) and birds (e.g. eagles; Langguth et al., 2013). The protection and conservation efforts devoted to these species take advantage of their intrinsic natural appeal and fascination to the public (Brambilla et al., 2013), and often result in the conservation of their whole habitat, including a wide range of other organisms (Entwistle, 2000). Unfortunately, traditional flagship species are often missing in several areas, in particular in those impacted by human activities (Anne-Isola Nekaris et al., 2015), such as agricultural ones. In these parts of the world, there is an urgent need to identify species that could act as flagships for conservation. There, 'non-traditional' flagship species (*sensu* Entwistle, 2000) could play this role. In fact, if raising awareness and educational activities are targeted at appropriate audiences, even less-charismatic species may appear appealing for this purpose (Entwistle, 2000; Walpole and Leader-Williams, 2002). Examples of species successfully used as 'non-traditional' flagships are the snake *Alsophis antiguae* in Antigua (Daltry et al., 2001) and the bat *Pteropus voeltzkowi* in Tanzania (Trewhella et al., 2005). Unfortunately, the concept of 'non-traditional' flagship species has been rarely applied and is generally neglected in the conservation literature, despite its proved utility in conservation practice.

With this study, we focus on two insectivorous avian species: spotted flycatcher *Muscicapa striata* and common redstart *Phoenicurus phoenicurus*. These species are widely distributed in Europe, but are (or have been) declining across the continent (Sanderson et al., 2006). In our study area, they widely exploit vineyard habitats, and thus are ideal

model species to elucidate the impact of viticulture on biodiversity.

We investigate how habitat characteristics, agricultural management practices, and topographic-climatic variables affect the abundance (measured as the number of territories) of these two species (i). Additionally, we evaluated the effect of vineyard ground management practices on field-scale habitat selection by foraging common redstarts, since ground characteristics were suggested to be important for setting territories and foraging activities of ground-dwelling species in agricultural habitats (Martinez, 2012; Martinez et al., 2010; Tagmann-Ioset et al., 2012) (ii).

We finally discuss the possible role of these two species as 'non-traditional' flagship species in vineyards with different levels of intensification (iii).

2. Materials and methods

2.1. Study area

This study was performed in the Trento Province, a mountainous area in the South-eastern Alps (Italy) where vineyards are found in the valley floors and on the adjacent hillsides, up to 750 m a.s.l. Vineyard plantations cover about 10,300 ha (i.e. 2% of the total Province extent and 19% of the area below 500 m). For further details on the study area, see Assandri et al. (2016a).

2.2. Study species

We chose two species of conservation concern as models, which are relatively widespread in the study area (Assandri et al., 2016a). This allowed us to define conservation strategies with possible applications to the whole study area.

Spotted flycatcher *Muscicapa striata* is considered 'SPEC 3' in Europe (which means a species with an unfavourable conservation status but with a population not concentrated in Europe; (Birdlife International, 2004)), where it underwent a 46% population decline in the period 1980–2013. It is now considered stable (2004–2013; http://www.ebcc.info/trends2015.html), although it is decreasing in large parts of its range (Freeman and Crick, 2003). In Italy, the species has been reported to be stable by some authors (Nardelli et al., 2015), and decreasing by others (Rete Rurale Nazionale and LIPU, 2015).

Common redstart *Phoenicurus phoenicurus* experienced a sharp decline in Europe until the end of the last century (Birdlife International, 2004). Then, it underwent a strong recovery with a 2004–2013 increase of 41% (http://www.ebcc.info/trends2015.html), which was also reported for the Italian population in the period 2000–2014 (Rete Rurale Nazionale and LIPU, 2015).

2.3. Bird counts and territory definition

We surveyed the two target species along 47 200-m long linear transects spread across the entire Trentino wine district with half in the valley floors and half on hillsides (Fig. 1). Birds were counted within 100 m from the transect (i.e. buffer zone), covering a total surface of 7.15 ha for each transects.

We mapped as precisely as possible all contact with every individual on the most updated aerial photographs available (scale 1:2500). We took into account both direct observations and songs/calls of the target species. We discarded overflying birds. To avoid double counting of the same individual, the minimum distance between neighbouring buffer zones of the transect was set at 300 m.

We surveyed each transect three times during the breeding season 2014 (10–18 April; 13–22 May; 12–22 June). Five to six transects per day were censused from dawn to a maximum of five hours later (5.30–10.30 a.m.). All counts were carried out by one of the authors (G.A.), who walked at a slow pace along each transect, on days with calm weather without precipitation and strong winds. The census of a
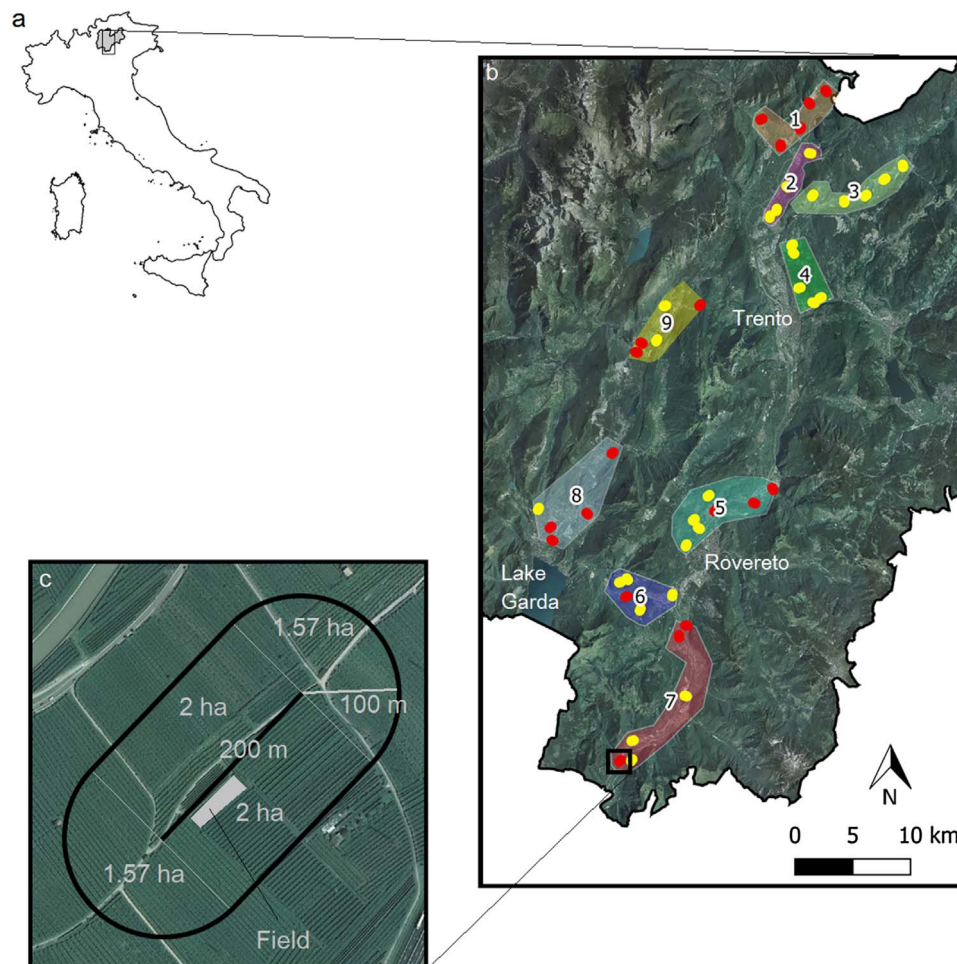
Fig. 1. a) Location of the study area in the Trento Province (in grey) and in Italy. b) Location of the 47 transects surveyed (red: valley floor transects; yellow: hillside transects), with the nine geographical areas used as random intercepts in the GLMMs. They are numbered as follow: 1-Piana Rotaliana; 2-colline di Lavis-San Michele; 3-Val di Cembra; 4-colline di Trento; 5-Alta Vallagarina; 6-Mori; 7-Bassa Vallagarina; 8-Benaco; 9-Valle dei Laghi. c) Detailed view of an example 200-transect, with its 7.15 ha-buffer zone and a sample of a vineyard considered for the field scale analysis (area shaded in light grey). Base map: Ortofoto 2011 ©AGEA − Agenzia per le Erogazioni in Agricoltura, Roma.

transect required 35–45 min, depending on both the local topography and habitat complexity. The census order across transects was changed from one visit to the next.

From these counts, we estimated the number of territories totally or partly found in the transect buffer zone, which became our response variable. We conducted three (or two, in the case of the spotted flycatcher, which appeared at the beginning of May in the study area) visits in each transect, which might be not enough for a detailed definition of territory boundaries. In any case, we are confident that this method is suited for estimating relative abundances and should be much better than using the simple mean or maximum abundance, because it considers the spatial arrangement of the individual bird locations (Brambilla et al., 2008). We also adopted an alternative approach, based on N-mixture models, to estimate population size from replicated counts (Royle, 2004). We evaluated the potential effect of census date and time of the day on detection probability through this approach, in addition to the effect of the independent (environmental) variable on the (latent) abundance of the species considered. Results showed that the effect of date and time of the day was not supported for the common redstart, whereas for the spotted flycatcher, only the effect of the date was slightly supported (over the two replicates that were useful for this species, which in April were still absent in the study area). These findings suggested that detectability was likely a minor issue in our system. Thus, we decided to continue with territory estimates based on the number and spatial location of the contacted individuals.

2.4. Environmental variables and data analysis

Data analysis was carried out at two different spatial scales: transect and field. Field scale is 'nested' within the transect.

2.4.1. Transect level

We measured in a GIS and directly in the field several landscape, management, and topographic-climatic variables at the transect level (Table 1).

Land cover variables were measured in GIS and validated or modified by means of a field survey on all the transects aimed at improving the quality of land cover estimates. We considered six habitat categories: woodlands, intensive apple orchards, marginal habitats, croplands, urban areas, and vineyards, with the last one being the dominant typology.

We measured the length of hedgerows and tree rows on aerial photographs, and counted the number of isolated trees and rural buildings. We also counted the number of habitat and vineyard patches (recognisable portions of habitat or vineyard fields with the same spatial arrangement and management) totally or partially overlapping with the buffer zone of the transect.

In Trentino vineyards, phytosanitary treatments are quite uniform since they are regulated by a central agricultural institute. The only difference lies in the use of synthetic fungicides, insecticides, herbicides, and fertilizers in conventional fields that are not allowed in organic ones. We consequently quantified the cover of vineyards under

Table 1

List of the independent variables used in the transect-level analysis. Mean ± SD is calculated over the 47 transects on the starting variable (i.e. without transformations or standardizations).

Variable	Description	Mean ± SD
Landscape variables		
Woods	% cover of woodlands (mostly broadleaved woodlands). Log + 1 transformed in the models (see text).	6.2 ± 8.9%
Crops	% cover of croplands (mainly small family-run fields)	2.3 ± 5.7%
Marginal	% cover of marginal habitats (hedgerows and tree rows, abandoned areas with scattered shrubs, field margins, roads)	14.2 ± 6.1%
Apple	% cover of intensive apple orchards. Log + 1 transformed in the models (see text).	4.6 ± 9.4%
Urban	% cover of urban areas	2.7 ± 4.3%
Patches	Number of patches partially or totally overlapping with the buffer area of the transect. We tested its quadratic effect on the common redstart (see text).	29 ± 11
Management variables		
Vineyards	% cover of vineyards	64 ± 18.7%
Hedgerows	Length of hedgerows and tree rows in the buffer area of the transect as defined in Assandri et al. (2016a)	318 ± 285.7 m
Isolated trees	Number of isolated trees. Log + 1 transformed in the models (see text).	10 ± 2
Isolated buildings	Number of isolated rural buildings	2 ± 2
Organic	% of organic vineyards (the remaining part is conventional)	13.9 ± 26.7%
Spalliera	% of *spalliera* vineyards (the remaining part is *pergola* vineyards)	18.3 ± 29.7%
Vineyard patches	Number of vineyard patches totally or partially overlapping with the buffer area of the transect	20 ± 9
Young plantations	% of *young* vineyard plantations (< 15 years). Log + 1 transformed in the models (see text).	30.2 ± 21.1%
Topographic-climatic variables		
Slope	Mean slope stated in degrees of inclination from the horizontal	8.9 ± 7.8°
Elevation	Mean altitude as derived from a DEM. This variable was excluded from the analysis due to high collinearity (see text).	307 ± 152 m
Solar radiation	Mean solar radiation on 21th June	1774 ± 460 W/m^2; 8610 ± 240 W/m^2
BIO1	Mean annual temperature derived from Hijmans et al., 2005	11.6 ± 1.5 °C
BIO12	Annual precipitation derived from Hijmans et al., 2005. This variable was excluded from the analysis due to high collinearity (see text).	865.20 ± 22.47 mm

these two different management systems for each transect. Organic viticulture in our study area is however limited, accounting for only about 3% of the entire area covered by vineyards. We were thus able to include only some (mean ± sd: 13.9 ± 26.7%; range: 0–100%) organic vineyards in one third of our sampling plots.

We additionally distinguished young and old vine plantations (with a threshold set at an age of 15 years) and the two vineyard trellising systems occurring in the study area, *pergola* and *spalliera*. *Pergola* is the traditional system in Trentino (about 80% of vineyards of the Province; Chemolli et al. (2007)) and consists of tall (up to > 2 m) and spaced vines (up to 5 m between rows) supported by poles and beams. *Spalliera* is the standard global system; it is characterized by lower vines supported by wires held between poles and with less spacing (< 2 m between rows). Within these two systems, management is basically the same, but some mechanical activities are impeded by the *pergola* structure (e.g. mechanical harvesting and pruning).

Topographic variables (mean elevation and mean slope for each plot) were derived from a 1-m resolution digital elevation model (DEM) using GRASS 7.0.2 (Neteler et al., 2012). From this, we also obtained mean direct solar radiation for each plot on 21 June using r.sun function and taking into account the shadowing effect of the topography. We derived mean bioclimatic variables (BIO1-annual mean temperature; BIO12-annual precipitation) from WorldClim (Hijmans et al., 2005) at a 30 arc-seconds resolution for each plot.

We did a thorough data exploration for each group of explanatory variables in order to avoid common statistical problems, following Zuur et al. (2010). This led us to apply a log + 1 transformation to wood, apple, young plantations, and isolated trees to reduce the weight of outliers. We also tested the quadratic effect of patches only in common redstart, since the preliminary graphical exploration of the data suggested a possible pattern of this kind.

Based on gVIF, topographic-climatic predictors were highly collinear among each other. Thus, according to Ieno and Zuur (2015) we dropped the ones with the highest gVIF from the analyses (elevation and BIO12).

We ended up with 17 predictors subdivided into the three above defined groups (see Table 1) that we tested on our response variable. At

first, we built three separate models (one for each group of predictors) for each species (the model formulas are reported in the supplementary materials). The choice to split our analyses in three was motivated by the fact that our sample (N = 47), and the obvious collinearity of several variables (possibly relevant from a conservation and planning point of view), does not allow us to build a single comprehensive model.

For this reason we also treated the percentage cover of vineyard as a management explanatory variable (instead of as a land-cover one). This also allowed us to correct for vineyard cover in the plot when evaluating the effect of the vineyard management variables and to remove collinearity amongst covariates in the landscape group. We also placed the hedgerow and tree row length, the number of isolated trees, and buildings into the management predictor group, because often in farmlands, and invariably in our study area, their occurrence is fully determined by the farmers' choices (Assandri et al., 2016a).

We did not include any interactions in our models. We acknowledge this is a simplification of complex ecological patterns expected to occur in artificial habitats, but due to our sample size and the fact that models were already complex *per se*, we preferred to keep it simple to avoid statistical and computational problems. We standardized all explanatory variables before entering them into the models to allow comparisons of their relative effects (Schielzeth, 2010) and because recent literature has highlighted the importance of this procedure for controlling for multicollinearity in model averaging (see below) in order to obtain reliable predictor estimates (Cade, 2015).

We expected our data to show a strong spatial structure, and spatial autocorrelation is known to severely affect the results of regression analyses (Beale et al., 2010). Therefore, we ran Poisson GLMMs using the R package glmmADMB (Skaug et al., 2015) with geographical area, a categorical variable with 9 levels grouping neighbouring plots (see Fig. 1), as a random intercept. Subsequently, we verified potential residual patterns of spatial autocorrelation by means of Moran's I test (run with R ape package; Paradis et al., 2004) on the regression's Pearson's residuals. We also computed the overdispersion statistic on final model residuals in order to evaluate whether Poisson distribution was the appropriate distribution for the data (Zuur et al., 2013).

We worked within an information-theoretic framework (Burnham and Anderson, 2002) using the dredge function in the R package 'MuMIn' (Barton, 2015) to build all the possible models for each species, separately considering each of the three groups of explanatory variables. We subsequently kept models with ΔAICc < 2, saving the most supported models (i.e. considered to be essentially as good as the best model; Symonds and Moussalli, 2011). We discarded the 'uninformative parameters' (Arnold, 2010), i.e. the variables included only in models comprising more parsimonious nested models (Richards, 2008; Richards et al., 2011).

Then, following the analytical method of Koleček et al. (2014), we built a set of candidate 'synthetic' models for each of the two species by selecting the predictors from the three individual groups which were selected amongst the most supported models (as defined in the previous step). After checking again for collinearity, we built these multi-level models using the same AICc-based procedure as described above (dredge, only models with ΔAICc < 2 kept, uninformative parameters removed). We finally conducted model-averaging by obtaining model-averaged coefficients, their relative standard errors, and the relative variable importance (Johnson and Omland, 2004) for all supported explanatory variables. In Fig. S1 is presented a graphical visualization of the whole modelling process.

2.4.2. Field level

This analysis was conducted only for common redstart, which, differently from spotted flycatcher (an aerial predator), usually feeds on the ground and could be affected by ground management practices in agricultural systems (Schaub et al., 2010).

We considered exclusively the observed bird locations, exactly attributed to a vineyard field (used field), excluding all the locations falling in other habitats. We randomly selected an equal number of unused vineyard fields for each combination of transect and month, selecting for each observation the field closest to the one used by the species.

Given the high territoriality of the species and the rather limited number of territories occurring per transect, we assumed that, at a given time, if a bird was in a given field, the nearest unused field could be considered available to the individual, but unused at that precise time.

In our study area, vineyard grounds are extensively covered by a grass sward with the exception of the vine base, where herbicides or plowing are usually applied.

In each used or unused field, at the end of each morning mapping session, we measured three ground management variables by visual estimation at the field scales (according to Mermod et al., 2009): i) the percentage sward cover in each vineyard; ii) the grass height grouped into three categories (low: ≤5 cm; medium: 6–20 cm; high: > 20: high) iii) the management of the vine row (three categories: herbicide, plowing, or only mowing). Patches are generally very homogeneous according to those three variables.

At the analytical stage, we simplified sward cover variable according to three cover categories: 0–75%; 76–99%; 100%.

We modelled used and unused fields as a function of the three field-level variables using binomial GLMs. The model was defined as follows:

redstart presence/absence ~ sward cover + grass height + row management

We worked again within an information-theoretic approach by building all the possible models as in the procedure described for the transect level.

3. Results

Overall, we obtained 132 precise locations (62 in May and 70 in June) for spotted flycatcher and 146 for common redstart (48 in April, 51 in May, and 47 in June).

On the basis of those records, we defined 66 territories for spotted flycatcher (mean ± sd/transect: 1.40 ± 1.24) and 56 for common redstart (mean ± sd/transect: 1.19 ± 0.95) in the period investigated.

3.1. Transect level

3.1.1. Spotted flycatcher

The landscape model selection procedure allowed us to retain only two supported models (see supplementary materials, Table S1), which highlighted a negative effect of both crop and marginal habitat cover on the number of spotted flycatcher territories (Table S2).

Similarly, the model selection procedure for management variables only retained two supported models (Table S3), which showed a negative effect of organic management, *spalliera* trellising system, and young plantation cover, and a positive effect of vineyard cover (Table S4).

Additionally, there was only one topographic-climatic model selected (Table S5), that showed a negative effect of slope (Table S6).

The synthetic model initially tested in the final analysis was thus defined in the following way:

number of spotted flycatcher territories ~ crops + marginal + organic + young plantations + vineyards + spalliera + slope + (1|geographical area)

The synthetic model selection procedure retained six models in which all the starting variables were included at least once, except for crop cover and slope; therefore, no topographic-climatic predictors were retained in these models (Table 2, Fig. 2).

The sign of the predictor effects on the response variable was the same as in the individual groups (Table 3).

Overall, the most parsimonious model considering the three groups of predictors, but also the synthetic models, was the most supported management model (AICc = 136.1). The most parsimonious landscape model came after (AICc = 140.4), and was closely followed by the topographic-climatic model (AICc = 141.6).

3.1.2. Common redstart

Landscape model selection retained only two supported models (see supplementary materials, Table S7), which highlighted a positive effect

Table 2

Most supported synthetic GLMM models for spotted flycatcher number of territories. 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 based on most supported models (w_i), and −2 log-likelihood values (logLik) are also given. Negative (−) or positive (+) relationships between predictors and the number of spotted flycatcher territories are also shown. For variable acronyms, see Table 1. N = 47.

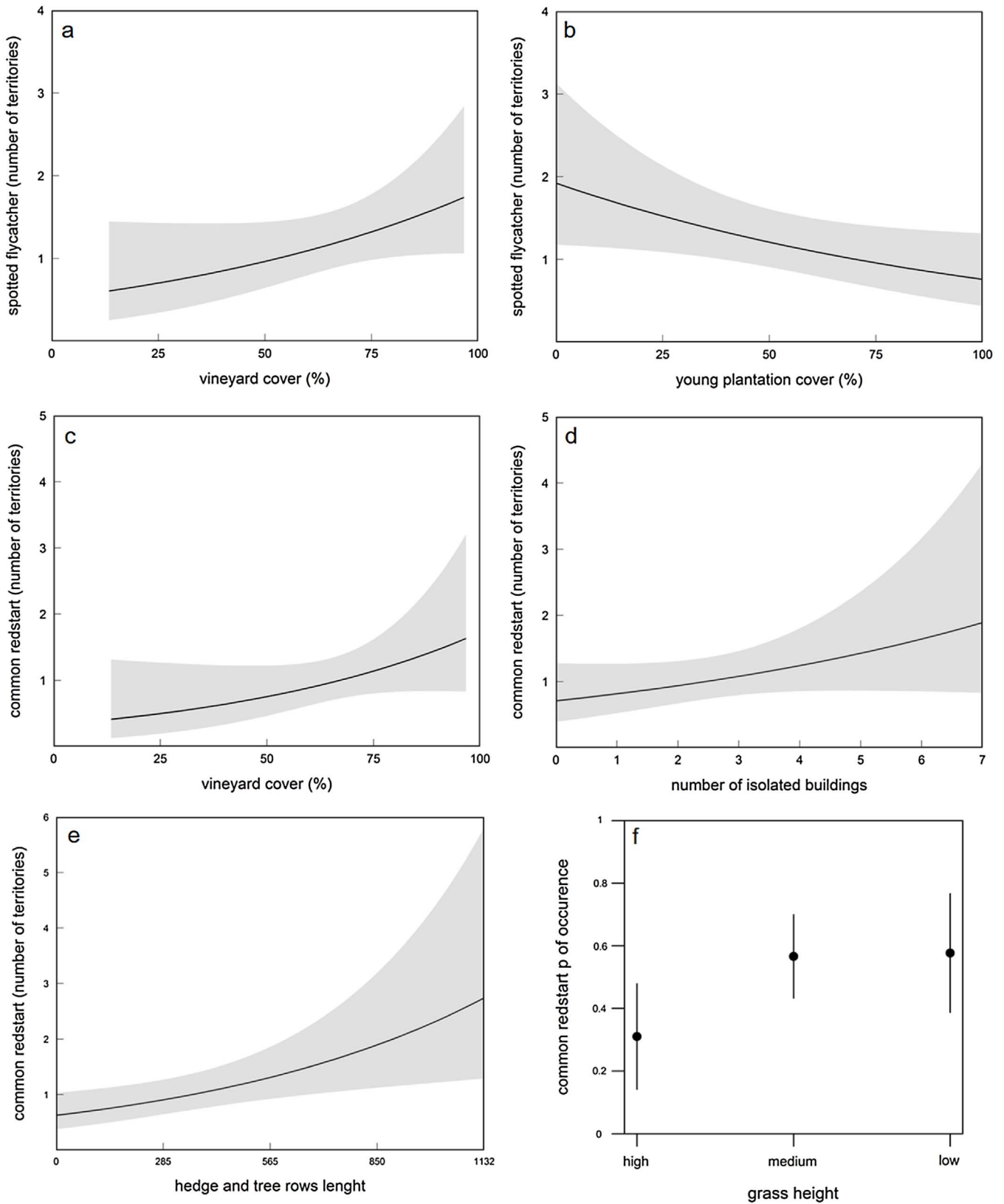| Model	df	logLik	AICc	ΔAICc	w_i
organic (−) + young plantations (−) + vineyards (+)	5	−62.30	136.1	–	0.29
spalliera (−) + young plantations (−) + vineyards (+)	5	−62.72	136.9	0.84	0.19
spalliera (−) + young plantations (−) + marginal (−)	5	−62.99	137.4	1.37	0.15
organic (−) + marginal (−) + vineyard (+)	5	−63.12	137.7	1.63	0.13
spalliera (−) + marginal (−)	4	−64.42	137.8	1.74	0.12
organic (−) + young plantations (−) + marginal (−)	5	−63.18	137.8	1.74	0.12

Fig. 2. Effect of the vineyard cover (a), young plantation cover (b) on spotted flycatcher number of territories, and vineyard cover (c), number of isolated buildings (d), and the length of hedges and tree rows (e) on common redstart number of territories. Plots are based on the predictions of the synthetic averaged GLMM models for the two species. Other predictors included in the models are kept constant at their mean value. Shaded grey polygons represent the 95% confidence intervals of the mean. N = 47.

Table 3

Model averaged standardized parameter (based on models with ∆AICc < 2) and relative variable importance (measured considering the sum of the Akaike weights over all models in which that variable appears) of predictors from synthetic models for spotted flycatcher. Covariates are ranked according to cumulative weights. For variable acronyms see Table 1. N = 47.

Variable	β	SE	Σw$_i$
intercept	0.17	0.15	–
young plantations	−0.20	0.15	0.75
vineyards	0.20	0.19	0.61
organic	−0.21	0.23	0.54
marginal	−0.15	0.18	0.52
spalliera	−0.19	0.24	0.46

Table 5

Model averaged standardized parameter (based on models with ∆AICc < 2) and relative variable importance (measured considering the sum of the Akaike weights over all models in which that variable appears) of predictors from synthetic models for the common redstart. Covariates are ranked according to cumulative weights. For variable acronyms, see Table 1. N = 47.

Variable	β	SE	Σw$_i$
intercept	0.01	0.15	–
hedgerows	0.36	0.14	1
urban	0.19	0.15	0.76
isolated buildings	0.16	0.17	0.57
spalliera	−0.18	0.24	0.46
vineyards	0.15	0.20	0.45
patches	0.02	0.09	0.08

of urban cover and a quadratic effect of the number of patches/transect on the number of territories of common redstart (Table S8).

Management model selection retained four supported models (Table S9), which showed a positive effect of hedgerow and tree row length, the number of isolated buildings, and vineyard cover, and a negative effect of *spalliera* trellising system (Table S10). Additionally, no topographic-climatic models were supported.

The synthetic model initially tested in the final analysis was thus defined in the following way:

number of common redstart territories ~ patches + patches^2 + urban + hedgerows + isolated buildings + spalliera + young plantations + (1|geographical area)

The synthetic model selection procedure retained eight models in which all the starting variables were present (Table 4, Fig. 2). The sign of the effect of predictors on the response variable was the same as in the individual groups, except for the number of patches, for which a better fit of the positive linear relationship (instead of the quadratic one) was suggested; however, the importance of this last variable was very low (Table 5).

Overall, the most parsimonious model was the synthetic model (AICc = 120.4), which was substantially comparable with the management model (AICc = 121), but clearly better than the landscape model (AICc = 125.2).

Spatial autocorrelation did not affect models (Moran's I tests on residuals always not significant). Likewise, overdispersion was always below 1.5, and thus the Poisson distribution we used was always adequate.

f) Effect of grass height on (foraging) common redstart occurrence probability as predicted from the only supported binomial GLM, as suggested by the model selection procedure. 95% confidence intervals of the mean are also shown. N = 108.

3.2. Field level

We obtained 64 common redstart precise locations in vineyards, representing the 44% of total locations of the species available over the

Table 6

Effects of grass height on the probability of foraging common redstart occurrence. The table shows the parameter estimates on a logit scale from the most (and only) supported binomial GLM fitted using treatment contrasts (i.e. the "high" level was the reference level for the comparisons). The contrast between "low" and "medium" was derived by re-fitting the model using "low" as the new baseline level. N = 108.\.

	Estimated coefficient	Standard error	z	p value
High-low	1.10	0.56	1.96	0.049
High-Medium	1.06	0.49	2.18	0.029
Low-medium	−0.04	0.48	−0.09	0.92

47 transects; of these, 54 were used for the analysis at this level (see Table S11), allowing us to define used vineyards, which we compared with an equal number of unused fields. Results showed that the only supported model (∆AICc < 2) was the grass height model (redstart occurrence ~ grass height; df = 3; logLik = −71.948; AICc = 150.12), with sward cover and row management not selected amongst the most supported models.

This model suggests that the species positively selected medium and low grass height (with no difference between them) against high grass (Table 6, Fig. 2e).

4. Discussion

Our results suggest that the intensive vineyards of the Trento province could harbour populations of two species of insectivorous birds, spotted flycatcher and common redstart, which have current or recent negative trends in Europe (Sanderson et al., 2006). This confirms previous findings on these and other insectivorous avian species in permanent crops, suggesting that, when food and nest sites are available, these species could maintain populations in very artificial habitats (Martinez et al., 2010; Mermod et al., 2009).

Both spotted flycatcher and common redstart were positively affected by the cover of vineyard at a landscape scale and both species

Table 4

Most supported synthetic GLMM models for common redstart. Models are ranked according to Akaike's information criterion corrected for small sample size (AICc) and only models within an interval of ∆AICc < 2 are shown. The difference in AICc from the best-supported model (∆AICc), Akaike's weights based on most supported models (w$_i$), and −2 log-likelihood values (logLik) are also given. Negative (−) or positive (+) relationships between predictors and common redstart number of territories. For variable acronyms, see Table 1. N = 47.

Model	df	logLik	AICc	∆AICc	w$_i$
isolated buildings (+) + *spalliera* (−) + hedgerows (+) + urban (+)	6	−53.13	120.4	–	0.18
isolated buildings (+) + hedgerows (+) + urban (+) + vineyards (+)	6	−53.14	120.4	0.03	0.18
spalliera (−) + hedgerows (+) + urban (+)	5	−54.47	120.4	0.05	0.17
isolated buildings (+) + hedgerows (+) + vineyards (+)	5	−54.76	121.0	0.63	0.13
spalliera (−) + hedgerows (+)	4	−56.19	121.3	0.99	0.11
isolated buildings (+) + hedgerows (+) + urban (+)	5	−55.10	121.7	1.30	0.09
patches (+) + hedgerows (+) + urban (+) + vineyard (+)	6	−53.97	122	1.69	0.08
hedgerows (+) + urban (+) + vineyards (+)	5	−55.42	122.3	1.95	0.07

seemed to be quite well adapted to this habitat, likely because they are structurally not very different from their "ancestral" habitats, such as open forests in warm and sunny climates (Cramp, 1988 ; Cramp and Perrins, 1993).

This is surprising, because the few existing studies on birds in vineyards found that, although vineyard-dominated landscape could harbour quite rich bird communities (Assandri et al., 2016a; Laiolo, 2005; Pithon et al., 2016), particularly in traditional extensively used landscapes (Verhulst et al., 2004), vineyard cover *per se* has detrimental effects on community indexes (e.g. richness and abundance) and on individual species abundance as shown by the analysis on auto-ecology at a finer scale (e.g. habitat and micro-habitat selection). Vineyards are in fact exploited only by a very few generalist species (Assandri et al., 2017a; Pithon et al., 2016) and, only in particular contexts, by a few species of conservation concern, such as woodlark *Lullula arborea* (Arlettaz et al., 2012; Pithon et al., 2016), skylark *Alauda arvensis* (Pithon et al., 2016), or lesser grey shrike *Lanius minor* (Isenmann and Debout, 2000).

The spotted flycatcher is likely favoured by vineyards due to their tree-like structure (in particular *pergola* ones), which allow the species to successfully breed on vines and vineyard supports (Assandri et al., 2017b) and to benefit from the abundance of perches (e.g. poles) used to fly-catch insects (Kirby et al., 2005). The observed negative effects of marginal habitat cover on spotted flycatcher are complex to explain, but probably are not a direct effect, since they are mediated by landscape complexity and topography. In fact, this species in the study area is mostly found in the vineyard-dominated main valley floors and transects richer in marginal habitats found on sloping valley sides have a lower cover of vineyards.

Common redstart is similarly favoured by vineyard cover, but this species is limited by the availability of suitable nesting sites, as it is a secondary cavity-nesting species. This likely explains the positive effect of urban cover on that species. In fact, sparse urban areas offer a great availability of artificial nesting sites and short-cut lawns occurring in gardens, meadows, and vegetable gardens, which confer heterogeneity at the landscape scale (Assandri et al., 2016b; Droz et al., 2015). Similarly, isolated rural buildings interspersed within the cultivated matrix (which had a positive effect on the species) are frequently used as nesting places. Additionally, hedgerows and tree rows have a clear positive effect on common redstart for two possible reasons. First, they probably offer a higher arthropod prey abundance and diversity than neighbouring vineyards. Second, they possibly provide nesting sites for the species.

4.1. Vineyard characteristics drive patterns of species abundance

Both spotted flycatcher and common restart are insectivorous, although with different specialization: common redstart is a non-specialized arthropod forager, generally hunting on the ground, whereas the spotted flycatcher is an aerial insectivore, which takes prey by fly catching (Cramp, 1985; Cramp and Perrins, 1993). As a consequence of their diet, they should find enough prey to persist in a breeding site, which is a requisite not necessarily satisfied in intensive agroecosystems (Altieri and Nicholls, 2004). Vineyards generally receive high inputs of pesticides (Aubertot et al., 2005), and these (in particular insecticides) could strongly impact arthropod prey and indirectly their insectivorous predators (Hallmann et al., 2014). Potentially, strong impacts of pesticides are thus expected on vineyard insectivores. Notwithstanding, in our study area farmers used a relevant amount of fungicides (mostly copper and sulphur); however, they use a limited amount of insecticides (normally only one treatment/year) against the leafhopper *Scaphoideus titanus*, which is the vector of the grapevine yellows *Flavescence dorée*, a serious vine phytoplasmosis (Penner et al., 2014). This limited use of selective insecticides possibly explains why some insectivorous species dwell well in this agroecosystem. In conventional fields, farmers use mainly systemic insecticides

targeted at *Scaphoideus*, whereas in organic fields, they use non-targeted insecticides (e.g. pyrethrins, *Bacillus thuringiensis*), which have been reported to affect the whole insect community (Wolfenbarger et al., 2008). This possibly explains the negative effect of organic farming that we found on spotted flycatcher abundance. Further *ad hoc* studies are however needed to confirm the effective impacts of broad-spectrum organic insecticides on the spotted flycatchers or on other insectivorous birds.

When considering vineyard characteristics, young plantations had a negative effect on spotted flycatcher. This result is likely a matter of structure, since the species can hardly find any suitable nesting sites in the poorly-structured young plantation, similarly to what happened in *spalliera* vineyards. A previous study confirmed that this is a poor nesting habitat for birds and in particular for spotted flycatcher, which totally avoids this trellising system for nesting (Assandri et al., 2017b), supporting the results of the present study (negative effect of *spalliera* on the number of spotted flycatcher territories).

The probability of foraging common redstart occurrence in vineyards at a field scale was higher when vineyard sward height was under 20 cm, and much lower when it was higher. Similar results were obtained by Martinez et al. (2010) in a field experiment in Swiss fruit orchards, where sparse vegetation and patches of bare ground emerged as key features for redstart occurrence (see also Schaub et al., 2010).

These results suggest that a minimum sward management is necessary in vineyards to guarantee the presence of this species, that is likely dependent on the "kitchen-dining room" system (Vickery and Arlettaz, 2012), according to which arthropod-prey are more abundant in tall grass, but are detectable by ground feeding birds only if patches of low or sparse grass (or of bare ground) are available.

4.2. Insectivorous birds as 'non-traditional' flagship species in vineyards

In the last few decades, vineyards have been subjected to severe intensification causing huge negative impacts on the environment including primarily loss of habitats and biodiversity. Therefore, efforts towards more sustainable viticulture should be urgently undertaken (Viers et al., 2013).

At the same time, wine consumers are becoming aware of the vulnerability of the environment and increasingly demand more sustainable wine production (Bisson et al., 2002; Tempesta et al., 2010). Landscape and biodiversity, consequently, are acquiring a new (at least economical) value, and "green" viticulture is increasingly considered as a viable way to fulfil sustainability targets (Zucca et al., 2009).

Insectivorous species are appropriate targets for conservation actions in vineyard-dominated landscapes and could be valuable flagship species, since they can be efficient pest controllers in vineyards (Barbaro et al., 2016). Bird conservation practices (e.g. nest-boxes provisioning) may enhance herbivorous arthropod removal and thus strengthen the provision of ecosystem services to wine growers (Jedlicka et al., 2017, 2011).

To be representative of a given ecosystem, flagship species have to be carefully selected according to the natural and social context of a given study area. Additionally, flagship species should be not too rare in the area/habitat considered and should have a good detectability (i.e. should be generally well visible or singing actively). In agroecosystems, it is difficult to find traditional flagship species. Thus, we propose here insectivorous birds as valuable candidates for the role of 'non-traditional' flagships in agricultural systems, in order to promote a more sustainable and biodiversity-friendly viticulture and to achieve conservation targets in this strongly artificial habitat.

Both species we considered largely inhabit vineyard-dominated landscapes, but not all kinds of vineyards are equally suitable for them. Moreover, they are widespread in the study area, consistently exploit vineyards for feeding and nesting, and are not much influenced by the factors that usually reduce the detectability of birds. These elements

qualify the spotted flycatcher and common redstart as valuable candidates for the role of 'non-traditional' flagships in vineyards. We showed, in fact, that the spotted flycatcher is positively affected by maintaining old vine plantations and traditional *pergola* structure in intensive vineyards. Conversely, in less intensive systems, the common redstart is favoured by a variety of marginal habitats and traditional elements (i.e. hedge and tree rows, isolated rural buildings) and < 20 cm grass height at a field scale.

Relevantly, in previous studies these elements emerged as fundamental also for the whole avian community (Assandri et al., 2016a) and for the majority of the commonest and most widespread bird species in the area (Assandri et al., 2017a). Thus, it is not too speculative to assume that by conserving them, it is possible to enhance the overall avian community and the whole agroecosystem.

In conclusion, spotted flycatcher could be an appropriate flagship for the intensively cultivated vineyard landscapes, whereas common redstart, which favours traditional and extensive systems, could be a flagship for less intensive vineyard systems.

Identifying 'non-traditional' flagship species in agricultural systems lacking more traditional ones, a goal which we developed by presenting a case study on insectivorous avian species living in vineyards, could have potentially many applications to other environmental contexts.

Our present contribution provides a first insight into the ecology of these two species in vineyards. Deeper knowledge is however needed, specifically with a focus on territory-scale preferences and breeding performance, to define specific conservation strategies. In parallel, a dissemination plan addressed to farmers, agricultural technicians, and citizens should be activated to translate the results of this work into conservation actions.

Acknowledgements

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolind.2017.05.012.

References

Altieri, M.A., Nicholls, C.I., 2004. Biodiversity and Pest Management in Agroecosystems, II. ed. Food Products Press, Bimghanton, NY.

Anne-Isola Nekaris, K., Arnell, A.P., Svensson, M.S., 2015. Selecting a conservation surrogate species for small fragmented habitats using ecological niche modelling. Animals 5, 27–40. http://dx.doi.org/10.3390/ani5010027.

Arlettaz, R., Maurer, M.L., Mosimann-Kampe, P., Nusslé, S., Abadi, F., Braunisch, V., Schaub, M., 2012. New vineyard cultivation practices create patchy ground vegetation, favouring Woodlarks. J. Ornithol. 153, 229–238. http://dx.doi.org/10.1007/s10336-011-0737-7.

Arnold, T.W., 2010. Uninformative parameters and model selection using Akaike's information criterion. J. Wildl. Manage. 74, 1175–1178. http://dx.doi.org/10.2193/2009-367.

Assandri, G., Bogliani, G., Pedrini, P., Brambilla, M., 2016a. Diversity in the monotony? Habitat traits and management practices shape avian communities in intensive vineyards. Agric. Ecosyst. Environ. 223, 250–260. http://dx.doi.org/10.1016/j.agee.2016.03.014.

Assandri, G., Bogliani, G., Pedrini, P., Brambilla, M., 2016b. Land-use and bird occurence at the urban margin in the Italian Alps: implication for planning and conservation. North West. J. Zool. e161601 (in press).

Assandri, G., Bogliani, G., Pedrini, P., Brambilla, M., 2017a. Assessing common birds' ecological requirements to address nature conservation in permanent crops: lessons from Italian vineyards. J. Environ. Manage. 191, 145–154. http://dx.doi.org/10.1016/j.jenvman.2016.12.071.

Assandri, G., Giacomazzo, M., Brambilla, M., Griggio, M., Pedrini, P., 2017b. Nest density,

nest-site selection, and breeding success of birds in vineyards: management implication for conservation in a highly intensive farming system. Biol. Conserv. 205, 23–33. http://dx.doi.org/10.1016/j.biocon.2016.11.020.

Aubertot, J.N., Barbier, J.M., Carpentier, A., Gril, J.J., Guichard, L., Lucas, P., Savary, S., Savini, I., Voltz, M., 2005. Pesticides, agriculture et environnement. Réduire l'utilisation des pesticides et en limiter les impacts environnementaux. Rapport d'Expertise Scientifique Collective INRA Et Cemagref.

Balmford, A., Green, R.E., Phalan, B., 2012. What conservationists need to know about farming. Proc. R. Soc. B Biol. Sci. 279, 2714–2724. http://dx.doi.org/10.1098/rspb.2012.0515.

Barbaro, L., Rusch, A., Muiruri, E.W., Gravellier, B., Thiery, D., Castagneyrol, B., 2016. Avian pest control in vineyards is driven by interactions between bird functional diversity and landscape heterogeneity. J. Appl. Ecol. http://dx.doi.org/10.1111/1365-2664.12740.

Barton, C., 2015. MuMIn: Multi-Model Inference. (R package version 1.13.4).

Beale, C.M., Lennon, J.J., Yearsley, J.M., Brewer, M.J., Elston, D.A., 2010. Regression analysis of spatial data. Ecol. Lett. 13, 246–264. http://dx.doi.org/10.1111/j. 1461-0248.2009.01422. x.

Birdlife International, 2004. Birds in Europe: Population Estimates, Trends and Conservation Status. BirdLife International, Cambridge, UK.

Bisson, L.F., Waterhouse, A.L., Ebeler, S.E., Walker, M.A., Lapsley, J.T., 2002. The present and future of the international wine industry. Nature 418, 696–699. http://dx.doi.org/10.1038/nature01018.

Bradbury, R.B., Kirby, W.B., 2006. Farmland birds and resource protection in the UK: Cross-cutting solutions for multi-functional farming? Biol. Conserv. 129, 530–542. http://dx.doi.org/10.1016/j.biocon.2005.11.020.

Brambilla, M., Guidali, F., Negri, I., 2008. The importance of an agricultural mosaic for Cirl Buntings Emberiza cirlus in Italy. Ibis (Lond. 1859) 150, 628–632.

Brambilla, M., Casale, F., Bergero, V., Matteo Crovetto, G., Falco, R., Negri, I., Siccardi, P., Bogliani, G., 2009. GIS-models work well, but are not enough: habitat preferences of Lanius collurio at multiple levels and conservation implications. Biol. Conserv. 142, 2033–2042. http://dx.doi.org/10.1016/j.biocon.2009.03.033.

Brambilla, M., Gustin, M., Celada, C., 2013. Species appeal predicts conservation status. Biol. Conserv. 160, 209–213. http://dx.doi.org/10.1016/j.biocon.2013.02.006.

Burnham, K.P., Anderson, D.R., 2002. Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach. Springer Science & Business Media.

Burns, F., Eaton, M. a., Barlow, K.E., Beckmann, B.C., Brereton, T., Brooks, D.R., Brown, P.M.J., Al Fulaij, N., Gent, T., Henderson, I., Noble, D.G., Parsons, M., Powney, G.D., Roy, H.E., Stroh, P., Walker, K., Wilkinson, J.W., Wotton, S.R., Gregory, R.D., 2016. Agricultural management and climatic change are the major drivers of biodiversity change in the UK. PLoS One 11, e0151595. http://dx.doi.org/10.1371/journal.pone.0151595.

Cade, B.S., 2015. Model averaging and muddled multimodel inferences. Ecology 96, 2370–2382. http://dx.doi.org/10.1890/14-1639.1.

Caraveli, H., 2000. A comparative analysis on intensification and extensification in mediterranean agriculture: dilemmas for LFAs policy. J. Rural Stud. 16, 231–242. http://dx.doi.org/10.1016/S0743-0167(99)00050-9.

Caro, T., Engilis, a, Fitzherbert, E., Gardner, T., 2004. Preliminary assessment of the flagship species concept at a small scale. Anim. Conserv. 7, 63–70. http://dx.doi.org/10.1017/S136794300300115X.

Chamberlain, D.E., Fuller, R.J., Bunce, R.G.H., Duckworth, J.C., Shrubb, M., 2000. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. J. Appl. Ecol. 37, 771–788. http://dx.doi.org/10.1046/j. 1365-2664.2000.00548.x.

Chemolli, M., Rizzo, M., Bona, E., Tonon, C., 2007. Vigneti e aziende viticole. Terra Trent. 4, 12–18.

Cramp, S., Perrins, C.M., 1993. The Handbook of the Birds of Europe, the Middle East and North Africa, vol. VII Oxford University Press, New York.

Cramp, S., 1988. The Handbook of Birds of Europe, the Middle East and North Africa, vol. V Oxford University Press, New York.

Daltry, J.C., Bloxam, Q., Cooper, G., Day, M.L., Hartley, J., Henry, M., Lindsay, K., Smith, B.E., 2001. Five years of conserving thèworld's rarest snake', the Antiguan racer Alsophis antiguae. Oryx 35, 119–127.

Donald, P.F., Green, R.E., Heath, M.F., 2001. Agricultural intensification and the collapse of Europe's farmland bird populations. Proc. R. Soc. B Biol. Sci. 268, 25–29. http://dx.doi.org/10.1098/rspb.2000.1325.

Donald, P.F., Pisano, G., Rayment, M.D., Pain, D.J., 2002. The Common Agricultural Policy, EU enlargement and the conservation of Europe's farmland birds. Agric. Ecosyst. Environ. 89, 167–182. http://dx.doi.org/10.1016/S0167-8809(01)00244-4.

Droz, B., Arnoux, R., Rey, E., Bohnenstengel, T., Laesser, J., 2015. Characterizing the habitat requirements of the Common Redstart (Phoenicurus phoenicurus) in moderately urbanized areas. Ornis Fenn. 92, 112–122.

Entwistle, A., 2000. Flagship for the future. Oryx 34, 239–240. http://dx.doi.org/10.1068/d2203ed2.

Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337–342. http://dx.doi.org/10.1038/nature10452.

Freeman, S.N., Crick, H.Q.P., 2003. The decline of the Spotted Flycatcher Muscicapa striata in the UK: An integrated population model. Ibis (Lond. 1859). 145, 400–412. http://dx.doi.org/10.1046/j.1474-919x.2003.00177.x.

Gottschalk, T.K., Dittrich, R., Diekötter, T., Sheridan, P., Wolters, V., Ekschmitt, K., 2010. Modelling land-use sustainability using farmland birds as indicators. Ecol. Indic. 10, 15–23. http://dx.doi.org/10.1016/j.ecolind.2009.05.008.

Gregory, R.D., van Strien, A., Vorisek, P., Gmelig Meyling, A.W., Noble, D.G., Foppen,

R.P.B., Gibbons, D.W., 2005. Developing indicators for European birds. Philos. Trans. R. Soc. Lond. B. Biol. Sci. 360, 269–288. http://dx.doi.org/10.1098/rstb.2004.1602.

Hallmann, C.A., Foppen, R.P.B., van Turnhout, C.A.M., de Kroon, H., Jongejans, E., 2014. Declines in insectivorous birds are associated with high neonicotinoid concentrations. Nature 511, 341–343. http://dx.doi.org/10.1038/nature13531.

Hannah, L., Roehrdanz, P.R., Ikegami, M., Shepard, A., Shaw, V., Tabor, M.R., Zhi, G., Marquet, L., a, P., Hijmans, R.J., 2013. Climate change, wine, and conservation. Proc. Natl. Acad. Sci. U. S. A. 110, 6907–6912. http://dx.doi.org/10.1073/pnas.1210127110.

Hart, K., 2014. The Fate of Green Direct Payments in the CAP Reform Negotiations: The Role of the European Parliament.

Hijmans, R.J., Cameron, S.E., Parra, J.L., Jones, P.G., Jarvis, A., 2005. Very high resolution interpolated climate surfaces for global land areas. Int. J. Climatol. 25, 1965–1978. http://dx.doi.org/10.1002/joc.1276.

Ieno, E.N., Zuur, A., 2015. A Beginner's Guide to Data Exploration and Visualisation with R. Highland Statistic Ltd.

Iglesias, A., Quiroga, S., Moneo, M., Garrote, L., 2011. From climate change impacts to the development of adaptation strategies: challenges for agriculture in Europe. Clim. Change 112, 143–168. http://dx.doi.org/10.1007/s10584-011-0344-x.

Inger, R., Gregory, R., Duffy, J.P., Stott, I., Voříšek, P., Gaston, K.J., 2014. Common European birds are declining rapidly while less abundant species' numbers are rising. Ecol. Lett. 18, 28–36. http://dx.doi.org/10.1111/ele.12387.

Isenmann, P., Debout, G., 2000. Vineyards harbour a relict population of lesser grey shrike (Lanius minor) in mediterranean France. J. Ornithol. 141, 435–440. http://dx.doi.org/10.1046/j. 1439-0361.2000.00038. x.

Jedlicka, J.A., Greenberg, R., Letourneau, D.K., 2011. Avian conservation practices strengthen ecosystem services in California vineyards. PLoS One 6, e27347. http://dx.doi.org/10.1371/journal.pone.0027347.

Jedlicka, J.A., Vo, A.-T.E., Almeida, R.P.P., 2017. Molecular scatology and high-throughput sequencing reveal predominately herbivorous insects in the diets of adult and nestling Western Bluebirds (Sialia mexicana) in California vineyards. Auk 134, 116–127. http://dx.doi.org/10.1642/auk-16-103.1.

Johnsingh, A.J.T., Joshua, J., 1994. Conserving rajaji and corbett national parks −the elephant as a flagship species. Oryx 28, 135–140. http://dx.doi.org/10.1017/S0030605300028453.

Johnson, J.B., Omland, K.S., 2004. Model selection in ecology and evolution. Trends Ecol. Evol. 19, 101–108. http://dx.doi.org/10.1016/j.tree.2003.10.013.

Kirby, W., Black, K., Pratt, S., Bradbury, R., 2005. Territory and nest-site habitat associations of Spotted Flycatchers Muscicapa striata breeding in central England. Ibis (Lond. 1859). 147, 420–424. http://dx.doi.org/10.1111/j.1474-919x.2005.00425.x.

Koleček, J., Schleuning, M., Burfield, I.J., Báldi, A., Böhning-Gaese, K., Devictor, V., Fernández-García, J.M., Hořák, D., Van Turnhout, C., a, M., Hnatyna, O., Reif, J., 2014. Birds protected by national legislation show improved population trends in Eastern Europe. Biol. Conserv. 172, 109–116. http://dx.doi.org/10.1016/j.biocon.2014.02.029.

Laiolo, P., 2005. Spatial and seasonal patterns of bird communities in Italian agroecosystems. Conserv. Biol. 19, 1547–1556. http://dx.doi.org/10.1111/j. 1523-1739.2005.00207. x.

Langguth, T., Honnen, A.C., Hailer, F., Mizera, T., Skoric, S., V??li, ??lo, Zachos, F.E., 2013. Genetic structure and phylogeography of a European flagship species, the white-tailed sea eagle Haliaeetus albicilla. J. Avian Biol. 44, 263–271. http://dx.doi.org/10.1111/j.1600-048X.2012.00075. x.

MacDonald, D., Crabtree, J., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., Gutierrez Lazpitz, Gutierrez, Gibon, A., 2000. Agricultural abandonment in mountain areas of Europe: environmental consequences and policy response. J. Environ. Manage. 59, 47–69. http://dx.doi.org/10.1006/jema.1999.0335.

Martínez-Casasnovas, J.A., Ramos, M.C., Cots-Folch, R., 2010. Influence of the EU CAP on terrain morphology and vineyard cultivation in the Priorat region of NE Spain. Land use policy 27, 11–21. http://dx.doi.org/10.1016/j.landusepol.2008.01.009.

Martinez, N., Jenni, L., Wyss, E., Zbinden, N., 2010. Habitat structure versus food abundance: the importance of sparse vegetation for the common redstart Phoenicurus phoenicurus. J. Ornithol. 151, 297–307. http://dx.doi.org/10.1007/s10336-009-0455-6.

Martinez, N., 2012. Sparse vegetation predicts clutch size in Common Redstarts Phoenicurus phoenicurus. Bird Study 59, 315–319. http://dx.doi.org/10.1080/00063657.2012.672949.

Mermod, M., Reichlin, T.S., Arlettaz, R., Schaub, M., 2009. The importance of ant-rich habitats for the persistence of the Wryneck Jynx torquilla on farmland. Ibis (Lond. 1859). 151, 731–742. http://dx.doi.org/10.1111/j.1474-919x.2009.00956.x.

Nardelli, R., Andreotti, A., Bianchi, E., Brambilla, M., Brecciaroli, B., Celada, C., Dupré, E., Gustin, M., Longoni, V., Pirrello, S., Spina, F., Volponi, S., Serra, L., 2015. Rapporto Sull'applicazione Della Direttiva 147/2009/CE in Italia: Dimensione, Distribuzione E Trend Delle Popolazioni Di Uccelli. ISPRA.pp. 2008–2012 (Serie Rapp. ed).

Neteler, M., Bowman, M.H., Landa, M., Metz, M., 2012. GRASS GIS: a multi-purpose open source GIS. Environ. Model. Softw. 31, 124–130. http://dx.doi.org/10.1016/j.envsoft.2011.11.014.

Paradis, E., Claude, J., Strimmer, K., 2004. APE: analyses of phylogenetics and evolution in r language. Bioinformatics 20, 289–290. http://dx.doi.org/10.1093/bioinformatics/btg412.

Pe'er, G., Dicks V, L., Visconti Arlettaz, P.R., Báldi, A., Benton, T.G., Collins, S., Dieterich, M., Gregory, R.D., Hartig, F., Henle, K., Hobson, P.R., Kleijn, D., Neumann, R.K., Robijns, T., Schmidt, J., Shwartz, A., Sutherland, W.J., Turbé, A., Wulf, F., Scott, A.V., 2014. EU agricultural reform fails on biodiversity. Science 344 (80), 1090–1092.

Pe'er, G., Zinngrebe, Y., Hauck, J., Schindler, S., Dittrich, A., Zingg, S., Tscharntke, T.,

Oppermann, R., Sutcliffe, L., Sirami, C., Schmidt, J., Hoyer, C., Schleyer, C., Lakner, S., 2016. Adding some green to the greening: improving the EU 's Ecological Focus Areas for biodiversity and farmers. Conserv. Lett. http://dx.doi.org/10.1111/conl.12333.

Penner, F., Ghidoni, F., Bottura, M., 2014. Manuale Di Viticoltura Sostenibile. Fondazione Edumund Mach, Centro Trasferimento Tecnologico, San Michele all'Adige TN.

Perrings, C., Jackson, L., Bawa, K., Brussaard, L., Brush, S., Gavin, T., Papa, R., Pascual, U., De Ruiter, P., 2006. Biodiversity in agricultural landscapes: saving natural capital without losing interest. Conserv. Biol. 20, 263–264. http://dx.doi.org/10.1111/j.1523-1739.2006.00390. x.

Pithon, J.A., Beaujouan, V., Daniel, H., Pain, G., Vallet, J., 2016. Are vineyards important habitats for birds at local or landscape scales? Basic Appl. Ecol. 17, 240–251. http://dx.doi.org/10.1016/j.baae.2015.12.004.

Rete Rurale Nazionale, LIPU, 2015. Uccelli comuni in Italia. Aggiornamento Degli Andamenti Di Popolazione E Del Farmland Bird Index Per La Rete Rurale Nazionale Dal 2000 Al 2014.

Rey Benayas, J.M., Martins, A., Nicolau, J.M., Schulz, J.J., 2007. Abandonment of agricultural land: an overview of drivers and consequences. CAB Rev. Perspect. Agric. Vet. Sci. Nutr. Nat. Resour. 2, 1–14. http://dx.doi.org/10.1079/PAVSNNR20072057.

Ribeiro, P.F., Santos, J.L., Bugalho, M.N., Santana, J., Reino, L., Beja, P., Moreira, F., 2014. Modelling farming system dynamics in High Nature Value Farmland under policy change. Agric. Ecosyst. Environ. 183, 138–144. http://dx.doi.org/10.1016/j.agee.2013.11.002.

Richards, S.A., Whittingham, M.J., Stephens, P.A., 2011. Model selection and model averaging in behavioural ecology: the utility of the IT-AIC framework. Behav. Ecol. Sociobiol. 65, 77–89. http://dx.doi.org/10.1007/s00265-010-1035-8.

Richards, S.A., 2008. Dealing with overdispersed count data in applied ecology. J. Appl. Ecol. 45, 218–227. http://dx.doi.org/10.1111/j. 1365-2664.2007.01377. x.

Royle, J.A., 2004. N-Mixture models for estimating population size from spatially replicated counts. Biometrics 60, 108–115. http://dx.doi.org/10.1111/j.0006-341X.2004.00142. x.

Sanderson, F.J., Donald, P.F., Pain, D.J., Burfield, I.J., van Bommel, F.P.J., 2006. Long-term population declines in Afro-Palearctic migrant birds. Biol. Conserv. 131, 93–105. http://dx.doi.org/10.1016/j.biocon.2006.02.008.

Santana, J., Reino, L., Stoate, C., Borralho, R., Carvalho, C.R., Schindler, S., Moreira, F., Bugalho, M.N., Ribeiro, P.F., Santos, J.L., Vaz, A., Morgado, R., Porto, M., Beja, P., 2014. Mixed effects of long-term conservation investment in natura 2000 farmland. Conserv. Lett. 7, 467–477. http://dx.doi.org/10.1111/conl.12077.

Schaub, M., Martinez, N., Tagmann-Ioset, A., Weisshaupt, N., Maurer, M.L., Reichlin, T.S., Abadi, F., Zbinden, N., Jenni, L., Arlettaz, R., 2010. Patches of bare ground as a staple commodity for declining ground-foraging insectivorous farmland birds. PLoS One 5, e13115. http://dx.doi.org/10.1371/journal.pone.0013115.

Schielzeth, H., 2010. Simple means to improve the interpretability of regression coefficients. Methods Ecol. Evol. 1, 103–113. http://dx.doi.org/10.1111/j.2041-210X.2010.00012. x.

Sekercioglu, C.H., Daily, G.C., Ehrlich, P.R., 2004. Ecosystem consequences of bird declines. Proc. Natl. Acad. Sci. 101, 18042–18047. http://dx.doi.org/10.1073/pnas.0408049101.

Skaug, H., Fournier, D., Bolker, B.M., Magnusson, A., Nielsen, A., 2015. Generalized Linear Mixed Models Using AD Model Builder.

Symonds, M.R.E., Moussalli, A., 2011. A brief guide to model selection, multimodel inference and model averaging in behavioural ecology using Akaike's information criterion. Behav. Ecol. Sociobiol. 65, 13–21. http://dx.doi.org/10.1007/s00265-010-1037-6.

Tagmann-Ioset, A., Schaub, M., Reichlin, T.S., Weisshaupt, N., Arlettaz, R., 2012. Bare ground as a crucial habitat feature for a rare terrestrially foraging farmland bird of Central Europe. Acta Oecol. 39, 25–32.

Tempesta, T., Giancristofaro, R.A., Corain, L., Salmaso, L., Tomasi, D., Boatto, V., 2010. The importance of landscape in wine quality perception: an integrated approach using choice-based conjoint analysis and combination-based permutation tests. Food Qual. Prefer. 21, 827–836. http://dx.doi.org/10.1016/j.foodqual.2010.04.007.

Trewhella, W.J., Rodriguez-Clark, K.M., Corp, N., Entwistle, A., Garrett, S.R.T., Granek, E., Lengel, K.L., Raboude, M.J., Reason, P.F., Sewall, B.J., 2005. Environmental education as a component of multidisciplinary conservation programs: lessons from conservation initiatives for critically endangered fruit bats in the western Indian Ocean. Conserv. Biol. http://dx.doi.org/10.1111/j. 1523-1739.2005.00548. x.

Tscharntke, T., Klein, A.-M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005. Landscape perspectives on agricultural intensification and biodiversity − ecosystem service management. Ecol. Lett. 8, 857–874. http://dx.doi.org/10.1111/j. 1461-0248.2005.00782. x.

Verhulst, J., Báldi, A., Kleijn, D., 2004. Relationship between land-use intensity and species richness and abundance of birds in Hungary. Agric. Ecosyst. Environ. 104, 465–473. http://dx.doi.org/10.1016/j.agee.2004.01.043.

Vickery, J., Arlettaz, R., 2012. The importance of habitat heterogeneity at multiple scales for birds in European agricultural landscapes. In: Fuller, R.J. (Ed.), Birds and Habitat. Relationships in Changing Landscapest. Cambridge University Press (pp. 177-204).

Viers, J.H., Williams, J.N., Nicholas, K. a., Barbosa, O., Kotzé, I., Spence, L., Webb, L.B., Merenlender, A., Reynolds, M., 2013. Vinecology: pairing wine with nature. Conserv. Lett. 6, 287–299. http://dx.doi.org/10.1111/conl.12011.

Walpole, M.J., Leader-Williams, N., 2002. Tourism and flagship species in conservation. Biodivers. Conserv. 11, 543–547.

Whelan, C.J., Şekercioğlu, Ç.H., Wenny, D.G., 2015. Why birds matter: from economic ornithology to ecosystem services. J. Ornithol. 156, 227–238. http://dx.doi.org/10.1007/s10336-015-1229-y.

Winkler, K.J., Nicholas, K.A., 2016. More than wine: cultural ecosystem services in

vineyard landscapes in England and California. Ecol. Econ. 124, 86–98. http://dx.doi.org/10.1016/j.ecolecon.2016.01.013.

Wolfenbarger, L.L., Naranjo, S.E., Lundgren, J.G., Bitzer, R.J., Watrud, L.S., 2008. Bt crop effects on functional guilds of non-target arthropods: a meta-analysis. PLoS One 3. http://dx.doi.org/10.1371/journal.pone.0002118.

Young, J., Watt, A., Nowicki, P., Alard, D., Clitherow, J., Henle, K., Johnson, R., Laczko, E., McCracken, D., Matouch, S., Niemela, J., Richards, C., 2005. Towards sustainable land use: identifying and managing the conflicts between human activities and

biodiversity conservation in Europe. Biodivers. Conserv. 14, 1641–1661. http://dx.doi.org/10.1007/s10531-004-0536-z.

Zuur, A., Ieno, E.N., Elphick, C.S., 2010. A protocol for data exploration to avoid common statistical problems. Methods Ecol. Evol. 1, 3–14. http://dx.doi.org/10.1111/j.2041-210X.2009.00001. x.

Zuur, A., Hilbe, J.M., Ieno, E.N., 2013. A Beginner's Guide to Glm and Glmm with R: A Frequentist and Bayesian Perspective for Ecologists. Highland Statistic Ltd.