

Hedgerows diminish the value of meadows for grassland birds: Potential conflicts for agri-environment schemes



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ABSTRACT

Hay meadows and hedgerows have been declining for decades in Western Europe. Conservation policies promote their protection but agri-environmental measures for both can be implemented in the same area, possibly generating loss of efficiency. We recorded grassland passerine abundance in 99 sampling plots distributed in hay meadow habitat. Sampling plots were located across a gradient of hedgerow density in a floodplain system. As expected, abundance and species richness increased with grassland area. More importantly, hedgerow density was negatively related to both response variables when grassland area was controlled for. This result is important for habitat management. Subsidizing agri-environmental measures for hedgerows and grassland is of interest for biodiversity, but incompatibility between measures may occur at the parcel scale if one landscape component (hedgerows) splits another component (meadows) down to the suitability threshold for grassland species. To optimize the payoffs of subsidies, it can be effective and efficient to manage agri-environmental schemes at the landscape scale.

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1. Introduction

Biodiversity in agricultural landscape has been declining for several decades in many parts of the World because of agriculture development and intensification (Donald et al., 2001). Agri-Environmental Schemes (AES) have been implemented as a response to halt this process at the national or supra-national level (CEC, 1998). Unfortunately, measures have not always yielded the expected benefits (Baker et al., 2012; Whittingham, 2007), and decline is still observed for threatened (Kleijn et al., 2006) or common birds species breeding in agricultural landscapes (Australia: Olsen, 2008; North America: U.S. Committee, 2009; Europe: Gregory and Strien, 2010). Therefore, it is urgent to assess current AES, identify their weaknesses, and propose novel implementation that more clearly yield benefits on biodiversity and increase economic returns of public subsidies (Sutherland et al., 2006). Because habitat loss is considered as the first cause of biodiversity decline, priority has been set to saving habitats of ecological interest. It is probably not sufficient if other factors limiting population growth are not explicitly taken into account (Baker et al.,

2012). Therefore, higher efficiency may be achieved by integrating plainly ecological processes in agri-environmental measures.

Many species are known to exhibit area sensitivity, i.e. occurrence or abundance is positively related to patch size of available habitat (Davis and Brittingham, 2004; Helzer and Jelinski, 1999; Schipper et al., 2011; Vickery et al., 1994; Winter and Faaborg, 1999). The size of the area to protect is considered in reserve planning (Arponen et al., 2007) but much less so in the implementation of environmental schemes where it has been simply advocated or entirely discarded respectively in the USA (NRCS (US Department of Agriculture), 1999; Ohterski, 2006) and in the EU. Ignoring the minimum spatial requirements of target species may have ecological and economical consequences. First, AES measures can be subscribed, within areas of high ecological interest, in patches too small to allow settlement, thus reducing the overall benefit for the population to conserve. Second, distinct AES implemented for different landscape components may be subscribed in the same parcel or patch which can potentially increase fragmentation of at least one component. Surprisingly, neither area sensitivity nor compatibility between different measures at the parcel scale – i.e. management unit – have received much attention when implementing AES. We addressed this issue in grassland passerines breeding in a large floodplain in Western Europe. Area sensitivity is a common limiting factor of grassland passerines in several areas of the World (Caplat and Fonderflick, 2009; Davis and Brittingham, 2004; Johnson and Igl, 2001; Winter and Faaborg, 1999).

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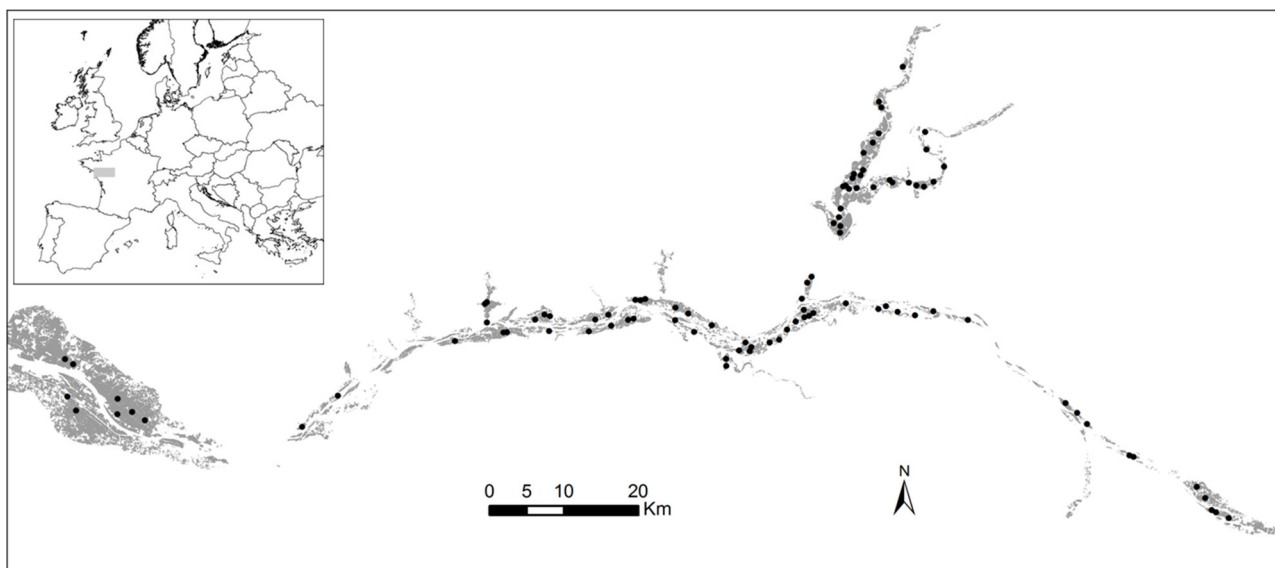


Fig. 1. Map of the study area. The grey zone corresponds to floodplain grasslands and black dots to the sample plots. The inset shows the location of the study area in Western Europe.

Environmental policies have been implemented partly to protect grassland species like the Corncrake *Crex crex* or the Whinchat *Saxicola rubetra*, which both continue to decline in western Europe (Fuller et al., 1995). It is thus striking that area sensitivity is still not implemented in such AES.

Permanent hay meadows have largely disappeared from Western Europe except from areas subjected to strong environmental constraints like frequent flooding or high elevation (Krause et al., 2011). Risks of productivity loss impair the development of intensive practices and maintain landscapes dominated by extensively managed grasslands. As a consequence, these areas still host grassland specialists (Gibbs, 2000) and shelter less specialized species vulnerable to other human activities like urbanization or intensive agriculture (Britschgi et al., 2006; Broyer, 2009; Godreau et al., 1999; Plantureux et al., 2005; Zahn et al., 2010).

In the floodplains in our study region the traditional hedgerow network still cover large areas (Forman and Baudry, 1984). Hedges are primary habitats for many species, provide refuge or complementary habitats to species vulnerable to intensive agriculture (Hinsley and Bellamy, 2000; Siriwardena et al., 2012), and offer ecosystem services like shelterbelt and shade for livestock, and wood resource (Baudry et al., 2000). For all these reasons, the maintenance and planting of hedgerows is subsidized by environmental policies (Kleijn and Sutherland, 2003). However, hedges can potentially cause habitat fragmentation for grassland species by increasing predation risk, smaller patches becoming unsuitable for breeding (Morris and Gilroy, 2008), or limiting the number of available territories in species that tend to aggregate (Ahlering et al., 2006). Furthermore, the possibility to receive subsidies for the protection of grassland birds and hedges on the same parcel may favour the densification of the hedgerow network, enhancing grassland fragmentation. In our study area, the size and patchiness of meadows is largely determined by hedgerows. Therefore, we predicted that increasing hedgerow density would reduce habitat suitability at the patch scale for grassland breeding birds. We analyzed the variation in abundance and richness of four grassland passerines breeding in hay meadows across a hedgerow density gradient. We expected a positive relationship between abundance, or richness, and the area of hay meadow, but a negative relationship with hedgerow density. We also searched the optimal spatial scale to detect area-sensitivity.

2. Methods

2.1. Study area

The study area extends over the lower 200 km section of the Loire river drainage in Western France (Fig. 1). Grassland is the dominant habitat type in the main channel and several major tributaries. Such an ecological continuity is now unusual in Western Europe. It provides an ideal setup to investigate the ecological factors affecting the distribution of grassland species. We focused on the four passerines that dominate a grassland bird community of seven breeding species in this area (Noël, 2003): the Whinchat *S. rubetra*, the Yellow Wagtail *Motacilla flava*, the Corn Bunting *Emberiza calandra*, and the Reed Bunting *Emberiza schoeniclus*. The other grassland species are too scarce to analyze their habitat requirements. These species are expected to benefit from agri-environmental schemes implemented to enhance their breeding success (no or little fertilization, delayed mowing).

2.2. Patch size, hedgerow length, grassland area and topographic wetness index

Land use is simple in the study area. Grasslands represent the main land use (75%) and all other habitats (poplar, crops, water course, hedgerows) are not suitable for the target species. Therefore, identifying grassland patches is straightforward in the study area. We used ArcGIS 10 and aerial photographs (BD Ortho® IGN) to quantify environmental predictors around each count point. We determined 3 buffer zones around each sampling point with a 100 m, 250 m or 500 m radius. We measured patch size and hedgerow density. We defined a patch as a continuous polygon of hay meadow. Patch size ranged from 1.3 ha to 265.3 ha (mean = 34.7 ± 50.3 SD ha). Hence a patch can cover several parcels in the study area, since practices are very homogeneous (no fertilization, grazing after mowing) and differences mainly concern the mowing date. We defined a hedgerow as an alignment of trees or shrubs bordering parcels.

We also estimated the effect of the Topographic wetness index (TWI). This index is a proxy of water accumulation (Beven and Kirkby, 1979) that we previously showed to affect the probability of occurrence of some of the target species (Besnard et al., 2013).

We calculated TWI for the entire Loire drainage with the multiple flow algorithm FD8 (Freeman, 1991; Quinn et al., 1991) using a 250 m pixel size resolution (Besnard et al., 2013). Within each buffer we extracted the grassland area, hedgerow length, and the mean TWI value for all pixels of the buffer. A 100-m buffer approximates the territory scale whereas a 250-m buffer corresponds to the parcel scale which can host several territories. A 500-m buffer more closely matches the landscape scale.

2.3. Species sampling

We recorded bird abundance in 99 sampling plots spanning the range of patch sizes available for hay meadows. Sampling points were distributed across the entire study area. They were located at the centroid of grassland patches except for some of the largest patches where we have laid two sampling points away from each other (minimum distance: 336 m) to avoid double counting.

We surveyed all sampling plots once between 2nd and 18th of May 2011. Sampling was carried out from 30 min to 5 h after sunrise. We recorded all visual contacts within a radius of 100 m using a laser telemeter (RangeFinder, Bushnell – Elite 1500). In the 11 smallest patches visual contact was not possible in the entire 100 m counting zone because of the presence of hedges. Nevertheless, we have included these small sampling plots in our sample to span the whole range of patch size available in the study area and ensure the detection of area sensitivity. We did not record any of the study species in these patches which support their very low suitability for the target species. We recorded both male and female occurrences. We mostly observed males though. They are more detectable than females because they often perch on the vegetation canopy to sing and watch their territory. Controlled survey (low wind, no precipitation) conditions ensured both optimal bird activity and detection. Given the relative homogeneity of grassland habitat in the study area, we assumed a constant detection probability so that counts are expected to provide standardized estimates of abundance and richness.

2.4. Statistical analyses

We estimated spatial autocorrelation between sampling units using spatial eigenvector mapping (SEVM; Griffith, 2000; Dormann et al., 2007; Corkeron et al., 2011). We selected this method because we could not satisfactorily fit a theoretical variogram to our dataset which may result from both the particular distribution of the sampling units in a river network (Fig. 1). Unlike variogram, SEVM is a flexible method that does not assume a specific spatial model. Spatial eigenvectors were computed on geographical distances using Spatial Analysis in Macroecology software (SAM 4.0) (Rangel et al., 2010). We used the broken-stick method to select the eigenvectors that capture most of the spatial autocorrelation in a dataset (Diniz-filho and Bini, 2005; Diniz-Filho et al., 1998; Jackson, 1993). Three eigenvectors were selected because their observed eigenvalues exceeded the values on the expected distribution (Appendix S1 in Supporting Information). Selected eigenvectors were added as covariables in all models to control for spatial autocorrelation (Diniz-filho and Bini, 2005). We checked the spatial structure of residuals using Moran's *I* test in the SAM software (Rangel et al., 2010).

We first tested area-sensitivity on abundance (all species) and richness using linear regression and log-transformed patch size. Log transformation was used to linearize relationship between predictors and response variables. We then tested the effect of hedgerow length, grassland area and TWI on the same response variables. We also included the interaction between hedge density and grassland area. We computed GLMs with a log link and a Poisson distribution of errors for richness models and a negative binomial

distribution of errors for abundance models. We carried out multimodel inference using AICc (Akaike Information Criterion corrected) and model weights (Burnham and Anderson, 2002) with the R package “MuMIn” (Barton, 2013). Full models included eigenvectors and environmental predictors. Null models included only eigenvectors, i.e. the spatial information about the distribution of our sampling units. We computed VIF, variance inflation factors, using the R package “CAR” (Fox and Weisberg, 2011) to assess multicollinearity which appeared not to be an issue in our models (all VIFs < 1.40) (Zuur et al., 2009). We used R package “visreg” (Breheny and Burchett, 2011) to represent partial relationships in our models. Analyses were carried out using R version 2.15.2 software (R development core team, 2012).

3. Results

We recorded 356 grassland passerines: 158 Whinchats, 79 Corn Buntings, 69 Yellow Wagtails and 50 Reed Buntings. We observed no grassland birds in 46 sampling plots, 1 species in 12 plots, 2 species in 15 plots, 3 species in 17 plots and 4 species in 9 plots. Mean abundance of grassland passerines was 3.6 ± 4.6 SD birds (range 0–17 individuals) and mean richness was 1.3 ± 1.4 SD species (range 0–4 species). According to the broken stick method, we selected the first three eigenvectors that we kept in all models. We found positive relationships between the logarithm of patch size and both richness (coefficient = 1.805 ± 0.167 SE, $F_{4,94} = 32.64$, $p < 0.0001$) and abundance (coefficient = 5.328 ± 0.584 SE, $F_{4,94} = 24.98$, $p < 0.0001$) of grassland birds (Appendix S2 in Supporting Information). Thus, abundance and richness increased with patch size.

Furthermore, we analyzed the relationship between environmental predictors and response variables (abundance and richness) at the three spatial scales (100 m, 250 m and 500 m) around count points. Model fits depended on the spatial scale. For abundance and richness, AICc scores were consistently lower when predictors were measured from 250-m buffers than from 100-m or 500-m buffers. However, the best models included the same environmental predictors at all scales (Table 1). We provide in the text the detailed results for the 250-m scale but the outputs of all models are given as supplementary material (Appendix S3 in Supporting Information). The effect size (coefficient) of hedgerow density and grassland area decreased when buffer size increased (Table 1). Thus the 100-m-buffer size provides the strongest effect size.

Landscape variables, i.e. hedgerow length and grassland area, were the two most informative predictors of bird richness and abundance. Richness and abundance increased with grassland area and decreased with hedgerow length (Fig. 2). The lowest AICc scores were observed when both predictors were combined in the same model (Table 2). The weight of TWI in models where it was the only predictor was low both for richness and abundance models.

4. Discussion

4.1. Area sensitivity and landscape structure

We found a positive relationship between patch size and both the abundance and richness of grassland passerines, meaning that larger meadows hosted more individuals and species of grassland passerines than smaller grassland patches. This pattern has been already observed in other bird species, especially in North America (Davis and Brittingham, 2004; Helzer and Jelinski, 1999). Thus, large continuous areas seem more effective to support grassland birds populations than several scattered small patches (Johnson and Igl, 2001). Consistently, richness and abundance increased with increasing grassland area, and decreased with increasing hedgerow density within a buffer. These two variables are not redundant though as shown by the low level of multicollinearity between our predictors. Furthermore, model fits were greatly improved when both predictors were combined than when considered singly. Thus, the two variables did contribute to explain the distribution of populations. Birds preferentially settled in landscapes hosting a large fraction of meadow habitat and avoided landscape where hedge density was too high. We evaluated whether this pattern depended on the spatial scale and found that the intermediate buffer size (250 m), which corresponds to the parcel scale, is the most relevant scale to detect the effect of landscape structure on these grassland passerines (Appendix S3 in Supporting Information). According to

Table 1
Summary of the best generalized linear models selected after the corrected Akaike information criterion (AICc) for the 3 buffer sizes (100 m, 250 m and 500 m). All models include the same 3 eigenvectors filters that control for spatial autocorrelation. The first column indicates the predictor (h: hedge density, g: grassland area and t: twi value). TWI is the mean value of the topographic wetness index in the buffer. Models were ordered according to AICc values and Akaike weights (w). For each predictor the estimated coefficients \pm the standard error were indicated. Adj R^2 is the adjusted R^2 of each model. The line in bold characters outlines the model with the lowest AICc.

Predictor	Buffer size	Hedge density	Grassland area	TWI	df	AICc	Δ AICc	Weight	Adj R^2
A. Richness									
hg	100	-0.005 ± 0.001	1.054 ± 0.420	–	6	275.9	0	0.593	0.503
hgt	100	-0.005 ± 0.001	1.069 ± 0.431	0.160 ± 0.136	7	276.8	0.9	0.381	0.51
hg	250	-0.001 ± 0.0002	0.117 ± 0.030	–	6	249.1	0	0.742	0.63
hg	500	0.0002 ± 0.0000	0.034 ± 0.006	–	6	252.6	0	0.742	0.615
B. Abundance									
hg	100	-0.007 ± 0.001	1.454 ± 0.514	–	7	421.2	0	0.509	0.837
hgt	100	-0.006 ± 0.001	1.550 ± 0.533	0.312 ± 0.193	8	421.4	0.2	0.472	0.841
hg	250	-0.001 ± 0.0002	0.150 ± 0.034	–	7	405.1	0	0.603	0.862
hgt	250	-0.001 ± 0.0002	0.152 ± 0.034	0.252 ± 0.205	8	405.9	0.8	0.397	0.864
hgt	500	-0.0003 ± 0.0000	0.046 ± 0.008	0.387 ± 0.232	8	411.6	0	0.582	0.856
hg	500	-0.0002 ± 0.0000	0.046 ± 0.008	–	7	412.3	0.7	0.418	0.851

the model built from our data (Fig. 2), at this scale 250-m-buffer (i.e. about 20 ha) should feature less than 1900 m of hedgerows and more than 10 ha of grassland to host at least one grassland bird. Buffers should feature less than 500 m of hedgerows and only grassland (the 19.6 ha of the sampling plot) to host more than one species. However, the effect size (coefficient) seems stronger for the 100 m buffer (Table 1). This could indicate that unfavourable conditions (i.e. a higher hedgerow density or a lower grassland surface) in the very near environment have a strong negative effect on the abundance and richness of grassland birds. The avoidance of hedgerows has been observed in other open habitat birds like Skylarks (*Alauda arvensis*) in Europe (Mason and Macdonald, 2000). Several hypotheses can explain the selection of larger patches for breeding and the avoidance of vertical boundaries like hedgerows or forest edges (Fonderflick et al., 2013). First, nest predation rate is often higher near hedgerows (Morris and Gilroy, 2008). Grassland birds could thus nest away from hedges to limit predation risk or

the perception of predation risk (Keyel et al., 2012; Laundré et al., 2010). This effect does not require a patch to be entirely circled by hedges, but more probably depends on the proximity to the nearest hedge, and thus on patch shape. Second, territorial birds can be attracted by conspecific singers (Ahlering et al., 2006) and often form territorial clusters. Hedges could thus disturb territorial and sexual interactions between individuals. Conspecific aggregation could be an important factor explaining area sensitivity in grassland birds, as individuals could only form territorial clusters in large unfragmented meadows.

In a previous study, we found that TWI was a predictor of grassland bird occurrence (Besnard et al., 2013). Sampling was carried out in large meadows only then to avoid the potential effect of area sensitivity. We did not observe a strong influence of TWI on abundance and richness here. Our sample spanned the whole range of patch sizes available in the study area, unlike in the former study, and small patches with high TWI values (high flood risk) hosted no

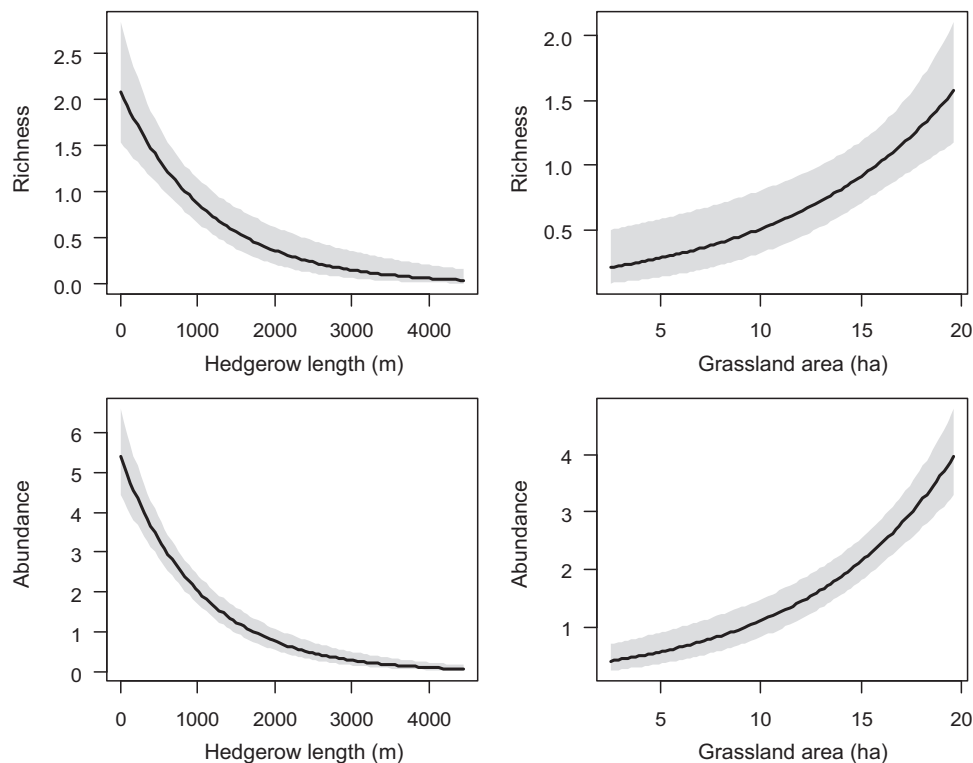


Fig. 2. Partial relationships between landscapes variables (hedgerow density, i.e. total hedgerow length, and grassland area in 250-m buffers) and richness and abundance of grassland passerines. Shaded areas represent confidence intervals.

Table 2

Model selection using corrected Akaike information criterion (AICc) values and Akaike weights (w) for generalized linear models including 3 eigenvectors filters and environmental predictors measured with a 250 m-radius buffer. TWI is the mean value of the topographic wetness index in the buffer. The first column indicates the predictor (h: hedge density, g: grassland area, t: TWI value and i: interaction between hedge density and grassland area). For each predictor the estimated coefficients \pm the standard error were indicated. The line in bold characters outlines the model with the lowest AICc.

Predictor	Hedge density	Grassland area	TWI	Hedge x Grassland	df	AICc	Δ AICc	Weight	Adj R ²
A. Richness									
hg	-0.001 \pm 0.0002	0.117 \pm 0.030	–	–	6	249.1	0.0	0.74	0.63
hgi	-0.001 \pm 0.0008	0.094 \pm 0.041	–	3.79E-5 \pm 5.07E-5	7	250.8	1.7	0.22	0.63
hgt	-0.001 \pm 0.0002	0.116 \pm 0.030	0.074 \pm 0.135	–	7	251.2	2.1	0.26	0.63
hgti	-0.001 \pm 0.0008	0.093 \pm 0.041	0.073 \pm 0.164	3.77E-5 \pm 5.07E-5	8	253.0	3.9	0.07	0.63
h	-0.001 \pm 0.0002	–	–	–	5	265.8	16.6	0.00	0.54
ht	-0.001 \pm 0.0002	–	0.141 \pm 0.156	–	6	267.2	18.1	0.00	0.55
g	–	0.181 \pm 0.030	–	–	5	280.0	30.9	0.00	0.47
gt	–	0.181 \pm 0.030	0.043 \pm 0.154	–	6	282.2	33.1	0.00	0.47
	–	–	–	–	4	327.0	77.9	0.00	0.1
t	–	–	0.152 \pm 0.138	–	5	328.0	78.9	0.00	0.11
B. Abundance									
hg	-0.001 \pm 0.0002	0.150 \pm 0.034	–	–	7	405.1	0.0	0.60	0.86
hgi	-0.002 \pm 0.0009	0.101 \pm 0.051	–	7.81E-5 \pm 5.64 \pm -5	8	405.4	0.4	0.28	0.47
hgt	-0.001 \pm 0.0002	0.152 \pm 0.034	0.252 \pm 0.205	–	8	405.9	0.8	0.40	0.86
hgti	-0.002 \pm 0.0009	0.108 \pm 0.051	0.219 \pm 0.204	7.00E-5 \pm 5.57E-5	9	406.7	1.7	0.15	0.48
h	-0.001 \pm 0.0002	–	–	–	6	420.9	15.8	0.00	0.83
ht	-0.001 \pm 0.0002	–	0.260 \pm 0.220	–	7	422.0	16.9	0.00	0.84
gt	–	0.214 \pm 0.040	0.413 \pm 0.237	–	7	432.4	27.3	0.00	0.82
g	–	0.217 \pm 0.039	–	–	6	432.8	27.8	0.00	0.81
	–	–	–	–	5	454.5	49.4	0.00	0.76
t	–	–	0.378 \pm 0.256	–	6	455.4	50.3	0.00	0.76

birds because of high hedgerow density. Thus, the present sampling scheme was probably not optimized to detect the effect of TWI.

4.2. Implications for AES implementation

The influence of landscape structure for grassland passerines is of interest for the implementation of agri-environmental schemes. Currently, several measures funded by the European Union promote agricultural practices favourable to grassland birds. These measures mainly favour extensive management, i.e. no fertilizer inputs and a mowing schedule in accordance with the breeding phenology (Broyer, 2009). Within a given area, all grassland parcels are eligible to subsidies. In our study area, eligible parcels can be included in large continuous stretches of grassland as well as in meadow patches fragmented by hedgerow and too small for birds to settle. Hedgerows are of conservation interest as they favour the occurrence of many other common or threatened species (Hinsley and Bellamy, 2000). Their network can be maintained or expanded without severe reduction of productivity in many areas whereas extensively managed grasslands often remain in particular environmental conditions (flood risk, sloping ground) and disappear elsewhere. For this reason, the conservation of these habitats appears as a special concern. However, bird-friendly AES measures miss their goal in areas where hedgerow density is too high and patch size too small for birds to settle. Subsidies eligibility based only on the presence of a specific habitat, grassland here, could be spent unnecessarily in areas where breeding does not occur. Therefore, future agri-environmental schemes would probably be more effective and efficient by taking into account as a contracting condition the minimum size of continuous habitat. One line of thinking may be to provide the opportunity for aggregating contracts from small adjacent parcels to create a larger patch even when these belong to different landowners. A cautionary approach is necessary though and a special attention should be paid to maintain heterogeneity in mowing dates in order to limit the negative impact on grassland biodiversity of synchronous mowing across large areas (Cizek et al., 2011). According our data, there is no maximum patch size value for grassland birds, the larger is the better. However, there may be an optimal value at the landscape scale when considering the whole bird community breeding in grassland

and hedgerow components. A complementary study would be necessary in order to estimate this optimum patch size of grasslands. Our data highlights that it is necessary to maintain a heterogeneous landscape with large grassland areas and without hedge interspersed with area offering dense hedgerow network.

In our study area, and many others, environmental policies subsidize agricultural practices to protect hedgerow habitats and grassland species. Our results suggest that these measures managed independently can conflict and eventually reduce their interest for conservation. First, fragmenting meadow patches by hedgerows reduce habitat suitability for grassland birds. Second, planting hedgerows in open grassland areas may improve biodiversity at the patch scale (Batáry et al., 2010) but it is likely to replace grassland specialists by more generalist species, by definition found elsewhere than in semi-natural grasslands. Biodiversity is thus expected to eventually decline at the landscape scale. Patch size is an important criterion of habitat suitability for specialist birds (Davis et al., 2006; Johnson and Igl, 2001; Vickery et al., 1994). However, managers should consider the negative effects of hedgerows if the main objective is to conserve grassland bird species. Even if each scheme may appear effective when considered singly, it seems important to assess to what extent schemes designed to protect different compartment of biodiversity affect each other's effectiveness. Because area sensitivity is a rather common process, it may be difficult to effectively protect two habitats, such as grassland and hedgerow, in the same parcel. Promoting the management of a landscape mosaic may help to achieve this goal though. More generally, the integration of the landscape as the relevant conservation scale rather than the more convenient parcel scale would be wise. Moreover, a more flexible implementation that respond to the ecological needs of the communities they are meant to protect, like contract aggregation, may improve the long term efficiency of agri-environmental schemes.

5. Conclusion

Grassland birds are highly specialized birds that need large stretches of continuous favourable habitat to breed. As highlighted in other geographical areas (Baldi and Batáry, 2011), habitat

fragmentation tends to disadvantage specialized species like grassland birds. In our study area, hedgerows greatly contribute to reduce the size of grassland patches. However, currently grassland and hedgerow habitat are subjected to the same area to agri-environmental program funded by European Union in the Natura 2000 framework. Although the protection of hedgerows is a major issue for biodiversity, the extension of hedgerow networks into grassland patches may reduce the ecological interest of the latter habitat for birds. Conflicting conservation actions from the same environmental program could therefore reduce the global gain for biodiversity. Managing habitat heterogeneity at the landscape scale rather than at the parcel scale seems an advisable way to enhance avian biodiversity and preserve specialist species that need large patches of habitat.

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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.agee.2014.03.014>.

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