Specialist response to proportion of arable land and pesticide input in agricultural landscapes

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ARTICLE INFO

ABSTRACT

Increases in farming practice intensity and landscape simplification are two well-known threats for many farmland bird species. Nevertheless, the effects of these two factors may differ strongly among species. Here, we propose to use the specialist–generalist concept to assess which bird species are most affected by these two factors. Bird density and intensity of farming practices were assessed within a sample of 58 farms across the Seine-et-Marne region in France, using point counts and a standardised farmer survey. The local abundance of 41 farmland and non-farmland species was related to farming intensity (pesticide applications) and landscape simplification (proportion of arable land), which was quantified using generalised least square models to account for spatial autocorrelation. The more specialised the farmland and non-farmland birds were, the more negatively affected they were by the intensity of farming practices, relative to the generalist bird species. Local habitat simplification had a more positive effect on abundance of the most specialist farmland bird species. This latter relationship was not significant when tested at a landscape scale, which strongly suggests that the sensitivity of farmland specialists to landscape simplification is scale-dependent. Some non-farmland species' abundance was also reduced by farming intensity and local landscape simplification suggesting that low-input agri-environmental measures could benefit both farmland and non-farmland specialists, regardless of local and landscape habitat context. However, if diversity-enhancing measures should benefit non-farmland populations, it is likely that they do not favour farmland specialist species which are more at risk. We suggest designing of priority areas for farmland specialist birds, and landscape managing accordingly.

1. Introduction

Agricultural intensification has become a major cause of species decline in Europe over the last 30 years (Krebs et al., 1999; Donald et al., 2001). In particular, there is a negative correlation between the trends of national farmland birds and indices of national agricultural intensity across Europe (Donald et al., 2001). In order to understand why species were declining in farmland habitat, many studies have investigated the demographic mechanisms likely to explain species declines, by focusing either on one or on a few farmland species (e.g. Potts, 1980; Evans, 2001; Brickle and Harper, 2002; Boatman et al., 2004; Morris et al., 2005) while others consider the fate of different species at the community level, using explicit measures for species functional traits (Siriwardena et al., 1999; Holzschuh et al., 2007; Verhulst et al., 2007). This latter approach is particularly illuminating as to the underlying mechanisms leading to population declines following disturbance

(McGill et al., 2006) and often lead to the conclusion that intensive land use affects specialists more than generalists.

A recent study found that specialists were less abundant than generalists in conservation-tillage and conventional farms compared with organic ones (Filippi-Codaccioni et al., 2009). As most of the differences found between farming systems lay in the farming intensity (i.e. different amount of inputs), it was hypothesised that such differences could be the cause of variations in specialists' and generalists' abundance.

Here we use the same dataset to investigate patterns of specialists' and generalists' abundances in relation to two main types of agricultural intensification: farming intensity and landscape simplification. Farming intensity is here assessed by quantifying agricultural inputs that aim to increase productivity (e.g. the amount of added pesticides and fertilizers) (Herzog et al., 2005). We assume that higher the amount of such inputs equals more intensive farming practices. We define landscape simplification as loss of heterogeneity in landscape features, in particular the removal of non-crop habitats.

Apart from intensive farming practices, which occur at the scale of individual fields, the simplification of landscapes with increased

proportion of arable fields is another threat that affects farmland biodiversity (Pain and Pienkowski, 1997; Kleijn and Sutherland, 2003). In farmland dominated landscapes, both heterogeneity and farming practices are identified as important factors in explaining community composition (Kleijn et al., 2001; Verhulst et al., 2004; Fuller et al., 2005; Roschewitz et al., 2005; Devictor and Jiguet, 2007; McMahon et al., 2008). Conversely, the relative effect of local and landscape-scaled managements for biodiversity in agricultural landscapes is not well-explored (Tscharntke et al., 2005). Furthermore, despite recent increases in publications incorporating scale issues in the analyses of agricultural intensification effects of on local biodiversity (Kleijn et al., 2001; Roschewitz et al., 2005; Batáry et al., 2007; Devictor and Jiguet, 2007; Holzschuh et al., 2007), none of these have investigated differential effects of these scales according to specialisation at the species level.

The specialist–generalist concept (Gregory et al., 2005) has been applied to farmland bird species indicators; however, such indicators consider species as being specialist or not despite the obvious continuum of specialisation among species (Fridley et al., 2007). Here, we refine this approach by using a continuous habitat specialisation measure (SSI) proposed by Julliard et al. (2006), which allows us to rank species along a specialisation gradient. Also, we consider all species encountered in farmland rather than simply focusing on more specialised species.

We predict that not all species possess the same sensitivity to farming intensity and habitat simplification. First, theory as well as empirical findings predict that specialists should suffer more from habitat disturbance (Owens and Bennet, 2000; Marvier et al., 2004; Julliard et al., 2006; Clergeau et al., 2006; Devictor and Jiguet, 2007; Schweiger et al., 2007). Since specialist species are more dependent on habitat quality, it seems likely that intensive practices like increases in pesticide spraying (either as a true disturbance or as toxic effect on their specific food), should affect them adversely and profoundly, relative to generalists, which are able to use other habitats and resources.

Secondly, since an increase in the proportion of arable area in the landscape is associated with agricultural landscape simplification (Pain and Pienkowski, 1997; Roschewitz et al., 2005), we assume that farmland specialists should be less strongly affected by this process. Indeed, generalists use various habitat types in the matrix, so they should, theoretically be more affected by the limited diversity of patches than specialists who are more dependent on one or only a few habitat types (Krauss et al., 2003). In addition, generalist species may obtain resources from various habitat patches and thus be more competitive in heterogeneous landscapes than specialists, which are restricted to certain habitat categories (Julliard et al., 2006).

We test the prediction that specialists will be affected differently from generalists by agricultural practices' intensity and habitat simplification due to agricultural practices (here studied together) by using a continuous indicator of species specialisation. Specifically, we predict that the more specialised a species is, the less abundant it will be in intensively managed fields, and highly simplified landscapes will show the inverse pattern. This prediction is tested using a bird study of 58 farms differing in their agricultural management intensity and landscape structure.

2. Methods

2.1. Study site

Our study was carried out from the 1st of April to the 15th of June over 2 years (2006 and 2007) on 58 farms within the Seine-et-Marne region (France) with 1–6 fields per farm comprising a total of 142 sampled fields (Fig. 1). The number of fields investigated depended on the farm area and on cereal crops availability. For homogeneity purposes, we only focused on cereal fields (winter wheat, and spring and winter barley). Our sample comprised different levels of farming practice intensity, with 12 organically
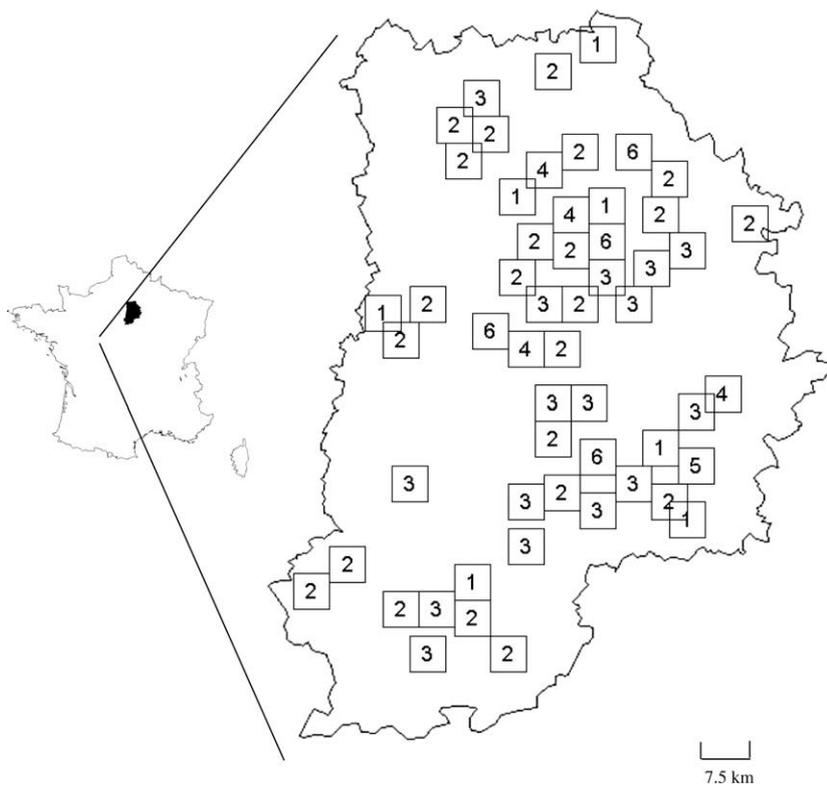

Fig. 1. Locations of farms studied in the Seine-et-Marne region (France), and numbers indicate the number of fields.

managed farms (31 fields) (with neither pesticide nor inorganic fertilizers use) and 46 conventional farms (111 fields) managed in a more or less intensive manner, using various amounts of inorganic fertilizer and pesticides.

2.2. Indicators of agricultural practice intensity

Information on the intensity of agricultural production for the 58 farms was obtained using a standardised questionnaire. Mean (±SD) farm and field size were 162.47 ± 76.49 ha and 11.49 ± 7.48 ha, respectively. The following intensity indicators were recorded: nitrogen input, rotation length, number of pesticide applications on winter wheat and yield of winter wheat. Winter wheat was always the main arable crop (covering the largest area) in the rotation and its yield at the scale of the farm was highly correlated to the yields of other major crops (r^2 = 63% and 67% with winter and spring barley, respectively), and based on these results, we assume that all the other cereal fields were grown with the same intensity.

Number of pesticide applications (herbicides, insecticides and fungicides) on winter wheat was retained as a surrogate variable for agricultural practice intensity. Indeed, this variable reflected both the amount of inputs in the field (to increase productivity) and disturbance caused by each spray session by farming machines. This index was correlated to other farming intensity measures (Table 1) so that we only kept the number of pesticides as a proxy of agricultural practice intensity. It is important to note that this index was also the most reliable variable given by farmers because of its easy calculation and relevance. Finally, since negative effects of pesticides on birds have already been demonstrated by Morris et al. (2005), this surrogate also has a potential direct impact on communities. The average number of pesticide applications was 3.85 ± 2.70 (min = 0; max = 9). Field size (ha) and culture type information at the time of the study was obtained directly from farmers.

2.3. Landscape simplification indicator

Each field that we surveyed was digitized using the geographical information package ArcView 3.2 (ESRI, 2000) from aerial photographs. Landscape features were obtained from the MOS (IAURIF, 2003) which is a detailed regional geo-referenced database that includes the main habitats for the region in continuous polygons classified according to 83 categories. The spatial resolution of this database is 1 × 1 m, and the categories describe the various different habitats such as natural, arable or built-up areas. A simple classification comprising nine habitat types was created by us, according to their ecological relevance for birds. These categories are: arable, forest, vertical feature (hedgerow or isolated tree), natural grassland, garden, built-up area, orchard, cut or clear forest, and finally, water. Habitat structure and composition were calcu-

lated within a 200 m radius buffer area around the field margin and from 200 m to 1000 m around each field using Patch analyst extension on ArcView 3.2. (Elkie et al., 1999). We specifically selected these two scales since we expect birds to be influenced by their surrounding habitats (Devictor and Jiguet, 2007). The 200 m scale represents the size of most passerine territories when breeding, while the 1000 m scale is the typical unit for landscape studies. These scales assisted us in determining whether or not local populations were influenced by wider landscape contexts, specifically to assess conservation management at larger scales.

We selected the proportion of arable land (labelled ARABLE) as a measure of landscape simplification around fields due to its high negative correlation with other landscape metrics like natural non-crop habitat proportion (forest + vertical feature + natural grassland + water) or the habitat type diversity (Shannon–Wiener Diversity Index (SDI)) (non-crop habitat, local scale (200 m): r = −0.91, P < 0.001; non-crop habitat, landscape scale (1000 m): r = −0.93, P < 0.001; SDI, local scale (200 m): r = −0.77, P < 0.001; SDI, landscape scale (1000 m): r = −0.38, P < 0.001) (Fig. 2). In order to have a measure of landscape simplification at two scales (local versus landscape scales), we calculated the variable ARABLE for the 200 m (labelled ARABLE200) and 200 to 1000 m (ARABLE1000) buffer zones around fields. ARABLE200 ranged from 28.80% to 100% with a mean of 80.72 ± 17.28% and ARABLE1000 ranged from 9.98% to 100% with of mean of 73.46 ± 16.41, respectively.

2.4. Bird counts

Bird counts were conducted twice, i.e. once before May 8th, and again after 5 weeks. Five point counts were made in all 142 fields at about 200 m from the field centre and with approximately the same distance (ca. 200 m) separating each point count (Filippi-Codaccioni et al., 2009). Every bird seen or heard was recorded during a five minute period for a given session between 6:00 and 11:00, both within, and outside of, each studied field. Observations for the first year were conducted by two different observers, but by only one observer the following year. Final bird abundance estima-
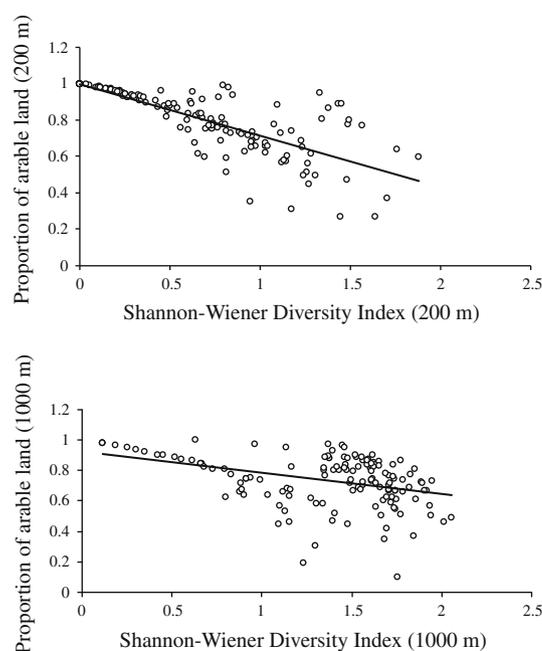
Fig. 2. Relation between proportions of arable land (proxy for landscape simplification) and the Shannon–Wiener Diversity Index (measure of habitat diversity), at the local (200 m) and landscape scale (1000 m).

Table 1
Statistic summary of the farming practices' intensity variables and their correlations with number of pesticide applications for the 58 farms studied. Pearson's correlation coefficient is presented.

	Wheat yield (100 kg/ha)	Fertilization (kg N/ha)	Rotation length
Mean ± SD	73.52 ± 18.18	135.35 ± 81.44	5.05 ± 2.28
Range	32–100	0–225	2–11
Pesticide applications	0.83[***]	0.79[***]	−0.45[***]
df	56	56	56

Fertilization = quantity of nitrogen/ha, Rotation length = number of crops in rotation.
[***] P < 0001 is a significance level.

tions were determined as the maximum value of the two visits. In order to limit contacts with birds which were too far removed from the studied zone, we only considered individuals recorded within 100 m from the observer (Filippi-Codaccioni et al., 2009).

A total of 91 species was detected over the 2 years study period. Species being present in less than 10% of the fields were removed from the study dataset, as well as the Grey Partridge *Perdix perdix* and the Common Pheasant *Phasianus colchicus*, because their local abundance is biased by local release for hunting. Aquatic species such as the Black-headed Gull *Larus ridibundus*, and the Mallard *Anas platyrhynchos* were also excluded from our analyses because of the irrelevant estimation of Species Specialisation Index (SSI) for these species. After these exclusions, 41 species out of 91 were kept for further analyses (Table 2).

2.5. Species Specialisation Index (SSI) calculation

Ideally, specialisation should be measured as ecological niche breadth of a species, considering all its dimensions (Julliard et al., 2006). However, a more practical way of quantifying specialisation of a given species is to calculate the variation in its density among various habitat classes. Julliard et al. (2006) grouped French Breed-

ing Bird Survey (FBBS) habitat categories into 18 homogenous habitat classes (Julliard and Jiguet, 2002). These authors proposed calculating a simple measure of specialisation for a given species using the coefficient of variation (SD/mean) of species population densities of a given species across habitats (Julliard et al., 2004, 2006). This measure has been calculated for 100 of the most frequently occurring terrestrial bird species using all French Breeding Bird Survey plots surveyed at least once between 2001 and 2004 ($n = 1022$, i.e., 10,220 point counts). Using this measure, we are able to rank species from specialists (occurring in few habitat classes), to generalists (occurring in many habitat classes), with lower values being attributed to generalists and higher values to specialists. Here, we use log-transformed SSI, which shows a more normal distribution (Filippi-Codaccioni et al., 2008). Previous analyses have shown that specialisation measures (SSI) were neither biased by taxonomic autocorrelation (Devictor et al., 2008a) nor by small sample sizes of species (Devictor et al., 2008b).

Additionally, we defined two groups of species (farmland and non-farmland) to investigate responses of non-farmland species to the pressure examined here (Filippi-Codaccioni et al., 2008, 2009). Farmland species are defined according to the French Breeding Bird Survey data as species more abundant in farmland habitat

Table 2
Bird species ($n = 41$) examined in this study ranked along an increasing Species Specialisation Index (SSI) gradient within farmland and non-farmland categories are presented with their responses to farming practices and landscape simplification.

Status	Scientific name	Response to farming intensity	Response to LS (200 m)	Response to LS (1000 m)	Specialisation	Specagri
F	*Corvus corone*	−0.027 ± 0.030	0.157 ± 0.499	−0.162 ± 0.609	−1.269	32.589
F	*Streptopelia turtur*	0.011 ± 0.021	−0.632 ± 0.387	−0.044 ± 0.452	−0.915	17.668
F	*Oriolus oriolus*	0.018 ± 0.019	−1.105 ± 0.311	0.654 ± 0.379	−0.748	41.884
F	*Buteo buteo*	0.017 ± 0.009	−0.401 ± 0.209	0.011 ± 0.152	−0.703	51.652
F	*Sturnus vulgaris*	0.003 ± 0.039	−1.482 ± 0.632	−0.579 ± 0.795	−0.561	18.341
F	*Emberiza cirlus*	−0.014 ± 0.012	−0.108 ± 0.246	−0.306 ± 0.283	−0.535	62.106
F	*Sylvia communis*	0.037 ± 0.019	−1.122 ± 0.382	0.433 ± 0.457	−0.425	61.099
F	*Falco tinnunculus*	0.001 ± 0.010	−0.598 ± 0.198	0.284 ± 0.230	−0.384	63.821
F	*Motacilla alba*	−0.022 ± 0.014	−0.171 ± 0.276	−0.083 ± 0.324	−0.366	22.016
F	*Carduelis cannabina*	−0.076 ± 0.027	0.579 ± 0.441	0.016 ± 0.552	−0.361	58.730
F	*Hippolais polyglotta*	0.003 ± 0.013	−0.626 ± 0.269	0.011 ± 0.310	−0.357	43.748
F	*Emberiza citrinella*	0.004 ± 0.025	−0.521 ± 0.471	0.103 ± 0.580	−0.341	72.699
F	*Hirundo rustica*	−0.007 ± 0.028	0.389 ± 0.576	−1.138 ± 0.663	−0.327	23.832
F	*Saxicola torquata*	−0.006 ± 0.009	−0.069 ± 0.188	0.053 ± 0.217	−0.254	79.670
F	*Corvus frugilegus*	0.010 ± 0.018	−0.313 ± 0.363	0.035 ± 0.425	−0.168	25.479
F	*Anthus trivialis*	−0.010 ± 0.013	−0.380 ± 0.255	0.059 ± 0.308	−0.094	27.561
F	*Alauda arvensis*	−0.039 ± 0.025	2.774 ± 0.458	−0.753 ± 0.547	0.144	88.929
F	*Emberiza calandra*	−0.103 ± 0.029	1.730 ± 0.449	0.964 ± 0.564	0.381	89.521
F	*Motacilla flava*	−0.007 ± 0.028	1.959 ± 0.486	−0.154 ± 0.584	0.738	80.933
NF	*Turdus merula*	0.006 ± 0.024	−0.832 ± 0.411	−0.581 ± 0.516	−1.454	−18.589
NF	*Fringilla coelebs*	0.017 ± 0.020	−2.429 ± 0.393	−0.319 ± 0.473	−1.303	−15.113
NF	*Parus caerulus*	0.014 ± 0.018	−1.166 ± 0.359	−0.024 ± 0.425	−1.221	−21.879
NF	*Sylvia atricapilla*	0.054 ± 0.024	−1.277 ± 0.453	−0.213 ± 0.529	−1.151	−14.059
NF	*Troglodytes troglodytes*	0.002 ± 0.023	−1.426 ± 0.370	0.064 ± 0.469	−0.990	−34.739
NF	*Pica pica*	0.004 ± 0.012	−0.571 ± 0.243	0.231 ± 0.279	−0.957	−3.215
NF	*Cuculus canorus*	0.017 ± 0.019	−0.515 ± 0.319	−0.803 ± 0.394	−0.836	−14.497
NF	*Garrulus glandarius*	0.018 ± 0.013	−0.488 ± 0.274	−0.003 ± 0.316	−0.813	−28.372
NF	*Turdus philomelos*	0.031 ± 0.019	−1.076 ± 0.389	0.284 ± 0.456	−0.786	−35.395
NF	*Phoenicurus ochruros*	−0.013 ± 0.013	−0.363 ± 0.253	−0.151 ± 0.298	−0.777	−27.374
NF	*Luscinia megarhynchos*	−0.013 ± 0.029	−0.944 ± 0.455	0.119 ± 0.580	−0.754	37.335
NF	*Erithacus rubecula*	−0.002 ± 0.016	−0.846 ± 0.295	−0.291 ± 0.336	−0.726	−43.599
NF	*Picus viridis*	0.052 ± 0.022	−0.633 ± 0.371	−0.633 ± 0.465	−0.703	−2.619
NF	*Streptopelia decaocto*	−0.053 ± 0.022	−1.697 ± 0.455	1.313 ± 0.524	−0.646	−40.484
NF	*Dendrocopos major*	−0.004 ± 0.012	−0.458 ± 0.202	−0.377 ± 0.240	−0.450	−49.368
NF	*Carduelis chloris*	−0.011 ± 0.018	−0.566 ± 0.365	0.107 ± 0.420	−0.420	−8.242
NF	*Phylloscopus collybita*	0.058 ± 0.021	−1.552 ± 0.430	−0.579 ± 0.495	−0.354	−24.138
NF	*Sitta europaea*	0.018 ± 0.019	−1.123 ± 0.307	0.302 ± 0.379	−0.080	−62.906
NF	*Passer domesticus*	−0.074 ± 0.035	−2.908 ± 0.730	2.195 ± 0.840	0.111	−30.057
NF	*Parus major*	0.026 ± 0.023	−2.000 ± 0.393	−0.585 ± 0.472	0.234	−20.354
NF	*Delichon urbica*	−0.018 ± 0.012	−0.006 ± 0.248	0.070 ± 0.285	0.236	−62.049
NF	*Columbia livia*	−0.017 ± 0.012	0.151 ± 0.267	−0.071 ± 0.205	0.712	−82.474

F = farmland bird, NF = non-farmland bird; specialisation = log-transformed Species Specialisation Index; response to LS = species response estimate (±SE) to local simplification (LS) (200 m) and landscape simplification (LS) (1000 m); response to farming intensity = Species response estimate (±SE) to farming intensity (Model 1, cf. text). Specagri = abundance in farmland habitat-abundance in non-farmland habitat)/abundance in farmland habitat. This measure allowed the separation into F and NF category of status (negative values = NF, positive values = F). Specialisation is a continuum so that higher SSI values indicate the most specialised species, and the reverse.

than in non-farmland habitat (i.e. positive values of the formula: abundance in farmland habitat-abundance in non-farmland habitat)/abundance in farmland habitat), and the inverse for non-farmland species (Table 2), yielding a total of 22 non-farmland species and 19 farmland species.

2.6. Statistical analysis

To determine the effects of both management (pesticide applications) and landscape simplification (proportion of arable land) adjusted to each other, we used model-testing for differences of species abundances according to those variables. The model used was a generalised least square (GLS) model that accommodated spatial dependence between fields (Filippi-Codaccioni et al., 2009). After selection of the best fitting variogram and the corresponding parameters (range and nugget), we modelled spatial structure in a complete GLS model, with spatial structure as a random effect, species abundance as a dependent variable and the number of pesticide applications as explanatory variables accounting for differences in landscape composition and structure variables. All counts data were $\log(x + 1)$ transformed in the following analysis:

Model 1: Species abundance = Observer + Crop type + Field size + Pesticide applications + ARABLE200 + ARABLE1000,

where 'Observer' was the observer identity, 'Crop type' is the type of crop, 'Field size', represents the field size and 'Pesticide applications' the number of pesticide applications. The proportions of arable land within 200 m and 1000 m around fields are denoted by ARABLE200 and ARABLE1000, respectively. The variables ARABLE and pesticide applications were not correlated to each other (Pearson correlation coefficient: ARABLE200: $r = 0.06$, $P = 0.42$; ARABLE1000: $r = 0.08$, $P = 0.30$), while ARABLE200 and ARABLE1000 were correlated (Pearson's correlation: $r = 0.67$, $P < 0.001$).

We then modelled the various species' responses (the slope of the relationship between species abundance and the proportion of arable land or farming intensity) in Model 1 according to their SSI as follows:

Model 2: Species response = SSI + T + T.SSI,

where 'T' was the species status (farmland or non-farmland), 'SSI' the Species Specialisation Index and 'species response' is the slopes taken from the Model 1 for each species. Finally the variable 'T.SSI' represents the interaction between the two variables. Each species response estimate was weighted by the inverse of its variance. All calculations were performed using the statistical software R 2.5.1 (R Development Core Team, 2007) with function (GLS) in package nlme.

3. Results

We found a negative relationship between farmland bird species' responses (abundance) to pesticide applications and the Species Specialisation Index (SSI) (Table 3). The Species Specialisation Index, species status (farmland or non-farmland), and the interaction of both were found to be a significant explanatory variable for abundance variations with the proportion of arable land at local scale (200 m), while none of those variables had a significant effect at the landscape scale (1000 m) (Table 3). Additional negative responses – in terms of abundance- to local habitat simplification (proportion of arable land at 200 m) were found within the non-farmland community (68% and 95% of negative response for farmland and non-farmland birds, respectively; Fisher's Exact test:

Table 3
Effect of the specialisation level, species status and their interaction on abundance of species in response to farming intensity and landscape simplification ($n = 41$) ANOVA (Type II) performed on Model 2.

Variables	df	F	P
Intensity			
SSI	1, 37	8.712	0.005
T	1, 37	0.281	0.600
SSI.T	1, 37	0.663	0.421
Local landscape simplification (200 m)			
SSI	1, 37	8.952	0.005
T	1, 37	5.435	0.025
SSI.T	1, 37	4.758	0.036
Landscape simplification (1000 m)			
SSI	1, 37	0.636	0.430
T	1, 37	1.070	0.308
SSI.T	1,37	0.098	0.756

SSI = log-transformed Species Specialisation Index; T = species status (farmland or non-farmland); SSI.T = interaction between the SSI and T.

$P = 0.03$). In contrast, no difference in the proportion of negative responses was found in responses to landscape simplification (proportion of arable land at 1000 m) and farming intensity between farmland and non-farmland birds (Fisher's Exact test: $P = 0.35$; Fisher's Exact test: $P = 0.53$, respectively).
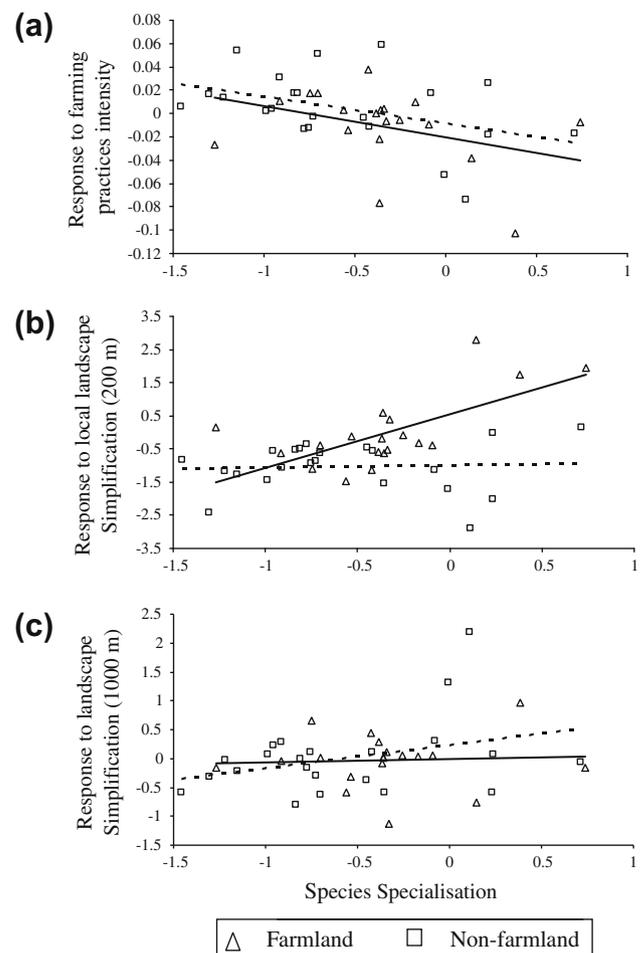

Fig. 3. Relationships between the log-transformed Species Specialisation Index and the species abundance response to indicators of (A) farming intensity (number of pesticide applications), (B) landscape simplification (proportion of arable land) in a 200 m radius and (C) landscape simplification in a 1000 m radius. Farmland bird and non-farmland bird communities are represented by solid and hatched line, respectively.

Abundance of the majority of specialist farmland species was found to decrease with farming intensity ($F_{1,17} = 4.19$, $P = 0.05$, $r^2 = 0.198$) as was the abundance of the most non-farmland specialists ($F_{1,20} = 5.17$, $P = 0.03$, $r^2 = 0.205$) (Fig. 3a).

We found a positive relationship between farmland species specialisation and their abundance response to the proportion of arable land at the local scale (200 m) ($F_{1,17} = 10.65$, $P = 0.004$, $r^2 = 0.385$), while no significant relationships was found for non-farmland species ($F_{1,20} = 2.43$, $P = 0.13$, $r^2 = 0.108$) (Fig. 3b).

No significant relationship was found between species abundance responses to the proportion of arable land (1000 m) and the SSI within the farmland and the non-farmland communities ($F_{1,17} = 0.008$, $P = 0.92$; $F_{1,20} = 0.60$, $P = 0.44$, respectively) (Fig. 3c).

4. Discussion

4.1. Farming intensity versus farming systems

The present paper tests the hypothesis proposed in Filippi-Codaccioni et al. (2009), which was that the difference of farming intensity between farming practices could determine variations of abundances of more or less specialist species between different farming systems. Although 60% of variance is shared between farming intensity (number of pesticide applications) and farming systems, higher percentages of variance in variation of species abundance are explained by farming intensity, relative to farming systems (7.07% and 1.58%, respectively, non-published results). This could suggest that management intensity could be more important than the other characteristics associated with farming systems (e.g. the use of herbicides, organic manure versus mineral fertilizer, etc.) studied here.

4.2. Specialisation and intensity of farming practices

More specialised species are more negatively affected by farming intensity, regardless of the species status, which means that the specialists' abundance was lower in more intensively managed farms. Our results agree with theoretical predictions (Owens and Bennet, 2000; Marvier et al., 2004; Julliard et al., 2006) and other authors' findings on global change effects (Clergeau et al., 2006; Devictor et al., 2008a,b; Schweiger et al., 2007). Our results also confirmed the hypothesis proposed by Filippi-Codaccioni et al. (2009) that farming intensity could be responsible for abundance variations in more or less specialist species occurring between conservation-tillage and conventional farms, compared with organic farms.

While it seems logical that pesticide applications should affect all species negatively, our results suggest that some species, mostly generalists, benefited from it. Therefore, we deduce that generalists may benefit from competition relaxation, where specialists are more affected by pesticide use.

Our findings for non-farmland birds demonstrated that some species (mostly specialists) showed negative responses to pesticide applications. Direct impact of farming intensity on those species seems unlikely, and so we propose an indirect mechanism: Firstly, as farmland specialists are affected, direct competitors like farmland and non-farmland generalists could be favoured as a result of competition relaxation; secondly, those generalists who compete with the non-farmland specialists would have an adverse effect on the specialists.

Additionally, these results suggest that other species, apart from those typically studied within farmland bird studies, are also affected by farming practices. Such species could also benefit from low-input agri-environmental measures. This study highlights the importance of enlarging the study scope to include non-farmland birds when assessing impacts of farmland intensification. We suggest that an a priori choice of species predicted to be negatively affected can lead researchers to miss certain important community ecology processes.

4.3. Specialisation and landscape simplification

We found a positive relationship between abundance responses of farmland species to local habitat simplification (proportion of arable land at 200 m) and the specialisation index of species. This relationship was non-significant for non-farmland species. This indicates that more specialised species become more abundant when their habitat area increases locally. Although such a relationship may appear obvious, studies examining this link have produced mixed results (Jonsen and Fahrig, 1997; Connor et al., 2000; Steffen-Dewenter and Tscharntke, 2000). However, when this relationship is examined at larger scale, it is not significant; meaning that farmland specialist sensitivity to arable land proportion is also scale-dependent. An explanation can be proposed: population abundance depends more on the presence of a few large suitable habitats than on a regional network of habitat patches, as stated by Connor et al. (2000). We believe it is even more accurate within a breeding context where territories have been chosen according to habitat suitability cues. In the case of open-arable species – as a majority of our specialist species are- this could mean that their local abundance depends on the availability of arable area in the vicinity rather than at larger spatial scales. This could also imply that there is little population overlap and density-dependence from 200 m to 1000 m around field territories. This finding corroborates the recent Yellow Wagtail Motacilla flava study by Gilroy (2007).

4.4. Conservation implications

Many studies have emphasised the negative impact of landscape simplification – or the positive one of landscape heterogeneity - on species diversity (Benton et al., 2003; Verhulst et al., 2004; Devictor and Jiguet, 2007; McMahon et al., 2008), or abundance and distribution (Suárez-Seoane et al., 2002; Brotons et al., 2005; Giralt et al., 2008). In contrast, our study shows that landscape simplification could have either a positive, or zero, impact on specialist species, depending on the scale examined. This emphasises the importance of considering species traits in community ecology to avoid limited or uncertain conclusions which may further be applied to conservation policies. Indeed, if habitat simplification benefits the most specialised farmland species (which are the most endangered ones), habitat diversification which is a common management approach in current agri-environmental schemes (AES), may actually constitute a negative approach for conserving specialist species.

However, one must be careful when evaluating species traits in the highly specialised species of the studied communities. For our study, it appears that the four most specialised species are also ground-nesters and open-farmland birds which generally avoid vertical features – at least for the Yellow Wagtail and the Skylark Alauda arvensis. In other types of bird communities, such as in the Mediterranean community, Red-Back Shrikes Lanius collurio or Lesser Grey Shrikes Lanius minor may be highly specialised species. In this case, the removal or disturbance of natural (shrublands) and semi-natural (fallows) habitats in the agricultural matrix could have negative effects as those habitats are linked to breeding parameters such as fledging success (Giralt et al., 2008). We therefore recommend fulfilling the most specialised species requirements in each type of agricultural region in order to protect the most vulnerable species.

These results lead us to ask questions about which kinds of species we want to promote in which landscape types, a question which arose recently from studies on AES (Vickery et al., 2004; Kleijn et al., 2006). Why not designate a certain region or area as a priority region for farmland (arable in this case) specialist birds and manage the landscape accordingly? It is impossible to design measures which will benefit all species, so that inevitably, choices are necessary. In the same manner, as it would seem inconceivable to a conservationist to retrieve hedges from a bocage region in order to benefit open-landscape species, it should be unthinkable to plant hedges in open-landscapes agricultural areas in order to promote bush species.

5. Conclusion

Our results emphasise the importance of specialisation in explaining birds' vulnerability to farming practices and landscape simplification. We showed that within bird communities encountered in farmland, the most highly specialist species showed the most negative abundance response to the farming practice intensity. Farmland bird specialist abundance was enhanced by local habitat simplification through the increased proportion of arable area. Based on the high proportion of negative responses to local scale habitat simplification and farming practice intensity found in non-farmland species, we conclude that those pressures could threaten non-farmland population levels. As another consequence, non-farmland communities could also benefit from policies targeting local diversity and less intensive practices. Importantly, the abundance of endangered farmland specialist populations appears to be negatively affected by local diversity enhancement, but reducing farming intensity should improve the survival of these endangered specialists.

Acknowledgements

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