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Pesticide doses, landscape structure and their relative effects on farmland birds

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ABSTRACT

Agricultural changes related to the intensification of farming practices and the simplification of landscape elements often occur simultaneously. Their respective effects on biodiversity are thus difficult to disentangle and are poorly understood. This study assessed the relative contribution of each component of agricultural intensification on taxonomic and functional bird communities.

The bird communities studied were composed of 70 species, both farmland and non-farmland birds, found in 66 fields covering three main cereal departments of France. Herbicide dose was related to measurable negative effects on the Community Specialization Index (CSI). Overall, the proportion of habitat specialists, particularly of herbivorous species, decreased, and the proportion of generalists increased as pesticide doses increased. Pesticides also had a positive effect on total abundance and richness, whereas no influence of insecticide or fungicide doses could be detected. Landscape simplification was associated with a loss of bird species diversity and an increase in the CSI.

Our findings suggest that the intensification of agriculture in this area, as reflected by increasing pesticide doses, modified communities by homogenizing species assemblages, whereas landscape element simplification led to the selection of only a few typical farmland birds enabled to persist in a simplified arable landscape. These results highlight the importance of combining taxonomic with functional diversity indices to fully understand changes in communities that occur in response to agricultural intensification.

1. Introduction

In farmland landscapes across the world, continuous declines of animal and plant populations, as well as the simplification of communities associated with the loss of certain ecosystem services, are important conservation issues (Swift et al., 2004; Green et al., 2005). At the field level, agricultural intensification, reflected by increasing chemical inputs and field areas and decreasing crop diversity, leads to increased yield, whereas at the farm level, the spread of cropped areas results in a loss and fragmentation of natural and semi-natural habitats (Doxa et al., 2012). One of the most debated aspects of intensification is the use of pesticides, due to their potential direct and indirect consequences at the individual, population and community levels (Guerrero et al., 2011; Mitra et al., 2011). Numerous studies have addressed this issue using different methodologies. Toxic effects of chemicals can arise rapidly after treatment (Mitra et al., 2011), whereas indirect effects of pesticides on populations usually occur after one or several years of treatment. Some ecological studies have tried to assess the total effects of pesticides (i.e., both direct and indirect effects) by comparing biodiversity between differently managed agro-ecosystems represented by low and high input levels (e.g., organic versus conventional; Hole et al., 2005; Bruggisser et al., 2010) or by analyzing biodiversity along a gradient of pesticide inputs. However, these studies have not addressed the short- and long term effects of pesticides separately or the environmental risks induced by pesticide doses applied explicitly. For wheat crops, the intensity of pesticide treatment varies between farmers (Burger et al., 2012). Farmers can follow a systematic treatment plan by using the maximum recommended doses to prevent pest development. Others adapt their treatments based on their observations, which results in the use of a lower pesticide dose, adjusted to the prevailing pest

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level (Jørgensen et al., 2008). The potential toxicity of products can also vary, with relatively highly toxic products causing more severe effects on bird populations than less toxic products (Bouvier et al., 2010). Treatment schemes for a given pesticide group can also be farm-specific. For instance, the variation in insecticide use is higher compared with that of herbicide and fungicide treatments (Freier et al., 2008). Finally, farmers can treat only a portion of the field and leave the other part untreated. To better reflect the consequences of pesticide use practices on biodiversity, the present study assessed the effects of pesticide doses while accounting for all active substances used, the recommended application rate and the percentage of the treated area.

Many studies have focused either on one or a few species (e.g., Potts, 1986; Morris et al., 2005), while others studied the fate of the community using measures of taxonomic diversity and abundance (Verhulst et al., 2004; Geiger et al., 2010) but rarely structural and functional features of biological communities (Guerrero et al., 2011). At the species level, Filippi-Codaccioni et al. (2010) proposed the use of the specialist–generalist concept and the continuous measure of habitat specialization (Julliard et al., 2006). Farmland specialists are predicted to be more affected by intensive practices than generalists, as they are more susceptible to toxic effects and habitat degradation than generalists, which can breed and shift to other resources. This study used a measure of species and community habitat specialization as a predictor of bird responses to the overall effects of pesticides. To assess the responses of birds to pesticide treatments, we investigated the effects of insecticides, herbicides and fungicides according to species diet preferences. Pesticides can reduce food resources directly (e.g., fewer non-cultivated fruits and seeds in herbicide-treated fields) or indirectly through cascading effects on the trophic chain (e.g., fewer insects in herbicide-treated fields due to the reduction of insect-attracting plants), hence reducing bird abundance. Finally, apart from the role of field practices and pesticides, the simplification of landscapes, with increased areas of arable land and the loss of remnant or semi-natural vegetation, can have notable effects on bird populations (Pain and Pienkowski, 1997; Devictor and Jiguet, 2007; Haslem and Bennett, 2008; Kleijn et al., 2009). Our analyses thus accounted for landscape and field characteristics to assess the relative effects of pesticide input on bird community richness, abundance, and composition. Increasing doses of herbicide and non-herbicide treatments were predicted to reduce abundance, richness, habitat specialization level and the diet of bird species and communities during both the year of treatment and afterwards. The responses of the most abundant bird species to distinct pesticide groups, i.e., insecticides, fungicides and herbicides, were then evaluated. We predicted that species with more specialized diets would be less abundant in fields treated with high doses of pesticides targeting its food resources. We conclude by discussing the possible role of landscape heterogeneity in mitigating the effects of farming intensification and pesticides on bird species and communities.

2. Materials and methods

This study covered 66 cereal fields located in three French departments (27 fields in Yonne, 12 in Charente-Maritime and 27 in Seine-et-Marne) (Fig. 1). Although the crops represent a significant proportion of the studied departments (>25% of the area), they also include different habitats, such as heterogeneous arable landscapes like pastures and small fields interspaced by bushes and woody areas, and other more natural lands. The fields sampled were thus located in both heterogeneous (complex) and intensive (simple) landscapes (cf. Section 2.3). For homogeneity purposes, we selected fields that were cropped with winter wheat or spring and winter barley for two consecutive years but differed in the amount of pesticides applied and other field practices (Table 1).

2.1. Bird counts and indices

Two point counts per field were located along the field margin at least 250 m apart to avoid double counting of birds. Point counts were surveyed twice in the springs of 2008, 2009 (in Seine-et-Marne) and 2010 (in Yonne and Charente-Maritime), once before and once after the 8th of May, with 4–6 weeks between the two counting events, following the standardized French sampling method (French Breeding Bird Survey (FBBS); Jiguet et al., 2011). This sampling method allows the most common sedentary and migratory breeding birds to be detected. Bird counts were carried out in the morning, from dawn to midday, under favorable weather conditions (i.e., windy, cloudy, or rainy weather was avoided) by three experienced bird observers who covered one department each. Every bird species heard or seen around the observer during a 5-min period was recorded (except birds flying over). Birders counted the birds present inside and outside the fields (on field edges and adjacent habitats), as we were interested in both farmland specialists and other species breeding in adjacent habitats. As the detectability of birds is influenced by the distance to the observer, only those birds recorded in a radius of 100 m around the observer were retained. Among all birds observed, four species were excluded from the analysis: gray partridge *Perdix perdix*,

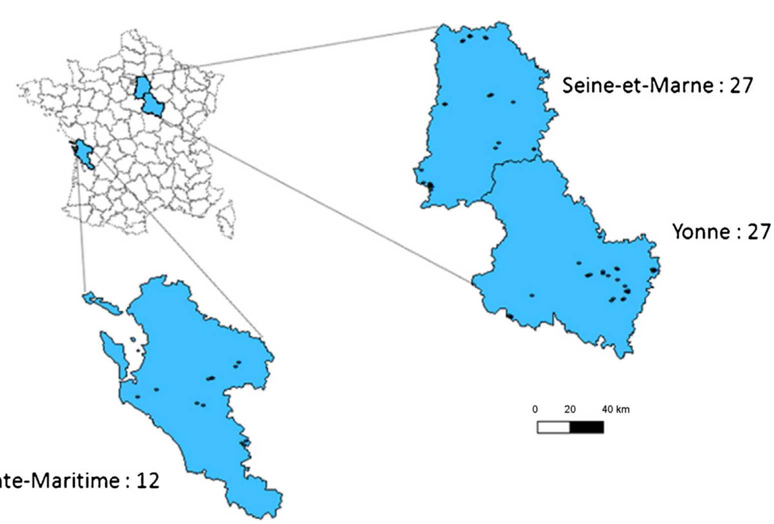

Fig. 1. Location of the 66 fields sampled within the three departments studied (Charente-Maritime, Yonne and Seine-et-Marne). Each dot represents one field.

Table 1
Main characteristics of the 66 fields sampled in the three studied regions.

Variable group	Field description		Regions			Total
			Charente–Maritime	Yonne	Seine-et-Marne	
Field layout and landscape	Field area (ha ± SE)		7.54 ± 0.67	15.08 ± 1.46	16.47 ± 1.76	14.28 ± 1.53
	Land planted with arable crops – Arable1000 (% ± SE)		0.63 ± 0.03	0.71 ± 0.02	0.87 ± 0.05	0.77 ± 0.02
	Crop type diversity – Divcrop200 (Shannon index ± SE)		0.72 ± 0.07	0.38 ± 0.05	0.92 ± 0.09	0.66 ± 0.08
Farming practices	Tillage (number of fields)	Full tillage	7	18	11	36
		Conservation tillage	5	9	16	30
	Cumulative pesticide dose (STI ± SE)	Fungicides – STI_{2yFung}	3.02 ± 0.29	3.42 ± 0.13	3.22 ± 0.21	3.27 ± 0.19
		Herbicides – STI_{2yHerb}	3.36 ± 0.20	4.44 ± 0.19	2.81 ± 0.16	3.58 ± 0.20
		Insecticides – STI_{2yInse}	0.13 ± 0.05	1.29 ± 0.15	0.56 ± 0.09	0.78 ± 0.12
		All – $STI_{2y\ all\ pesticides}$	6.51 ± 0.43	9.16 ± 0.24	6.60 ± 0.33	7.63 ± 0.35
Response variables	Total abundance (mean ± SE)		19.56 ± 1.22	21.85 ± 1.00	15.00 ± 1.04	18.39 ± 1.06
	Richness (mean ± SE)		9.83 ± 0.60	9.48 ± 0.42	7.85 ± 0.54	8.87 ± 0.54
	CSI (mean ± SE)		0.79 ± 0.04	0.85 ± 0.03	1.03 ± 0.04	0.94 ± 0.04
	Proportion of invertebrate eaters – CPI (% ± SE)		0.55 ± 0.01	0.47 ± 0.01	0.51 ± 0.01	0.50 ± 0.01
	Proportion of plant eaters – CPP (% ± SE)		0.42 ± 0.01	0.52 ± 0.01	0.49 ± 0.01	0.49 ± 0.01
		Total number of fields	12	27	27	66

red-legged partridge, *Alectoris rufa*, common pheasant *Phasianus colchicus* and mallard *Anas platyrhynchos*. The abundances of these species can be locally biased by large releases organized yearly for hunting purposes. In all, 70 species were analyzed (Tables A1 and A2).

From the number of individuals of each species counted at each point count during the two visits, the abundance at the visit with the maximum number of individuals of each species was retained. The abundance at the field level was obtained by summing the maximum abundance of the two point counts. To limit bias toward flocking species, all species abundances above 10 (less than 2% of all observations) were leveled to a maximum value of 10.

At the species level, the degree of species specialization for the habitat, i.e., the Species Specialization Index (SSI), was calculated as the coefficient of variation (SD/mean) of the species' density across habitats (using the 18 habitat classes recorded by observers of FBBS; Julliard et al., 2006). SSIs were calculated for the 190 most frequent terrestrial bird species breeding in France (Julliard et al., 2006). At the species level, we analyzed the response of individual species to landscape and field parameters (cf. § 2.2.) for the 28 more frequent species recorded during the field counts (≥10% of occurrence). Following Gregory et al. (2005), we classified these 28 species as farmland birds ($n = 10$), which usually nest within the field or along its margins, and the others as non-farmland birds that occupy other habitats ($n = 18$) (see Table S1). Most farmland birds were habitat specialists (according to their SSI value), whereas most non-farmland species were generalists.

The diet composition for each bird species was determined by computing information available in 'The Birds of the Western Palaearctic' (Snow and Perrins, 1998). We report the proportions of each species' diet made of plants (SPP; essentially seeds, fruits and leaves), invertebrates (SPI; insects, worms and spiders) and vertebrates (SPV) as three distinct variables. Species diet was defined along a continuum from 0 to 100% for insects as well as for plants and vertebrates. In this study, because only a few species eat vertebrates, only the SPP and SPI (with SPP + SPI∼1) were analyzed.

At the community level, we quantified (1) the richness, i.e., the total number of observed species, (2) the total abundance, i.e., the observed abundance of all species combined (see above), (3) the Community Specialization Index (CSI), i.e., the average Species Specialization Index (SSI) of all species present in the community, weighted by species abundance (Julliard et al., 2006) and (4) the proportion of invertebrates (CPI) and plants (CPP) in the diet of the community, i.e., the average insect and plant proportions in the diet of all species present in the community, weighted by species abundance.

2.2. Farming practices and pesticide use

Bird observers visited farmers just after the harvesting season and asked for information about their field practices. They collected detailed information about (1) tillage, i.e., conventional tillage versus conservation tillage (also including direct sowing), (2) the crop type over the last two cropping years (wheat–barley, barley–wheat, wheat–wheat, barley–barley; wheat being mostly a winter crop and barley a spring crop) and (3) the amount of pesticides sprayed on the field. The latter information was recorded per field as a list of the following information: name of pesticide, date of treatment, applied dosage and treated area. This included all treatments with fungicides, herbicides and insecticides. Other cereal field practices likely to differ among fields included nitrate inputs (quantities in kg/ha/year). Unfortunately, the number of applications and the quantities of nitrate inputs per field were only partially collected. However, based on a subsample of 30 fields, no correlation between pesticide dose and nitrate quantity (kg/ha/year) was found (p values > 0.05, Table A4), indicating that pesticide and nitrate inputs are likely to be independent.

We used the Standardized Treatment Index (STI) as a measure of pesticide use intensity (Roßberg, 2002; Gravesen, 2003; Sattler et al., 2007). The STI reports the number of treatments in a season, proportionally corrected for partial treatments as well as below-label doses recommended by experts regarding environmental risks of active substances. The STI is easy to calculate and operational because it allows the aggregation of very different substances to measure the overall phytosanitary pressure (Butault et al., 2011). The STI was first calculated at the field level for every single pesticide treatment applied before the sampling dates, following the equation:

$$STI_{field} = \sum \frac{AD_p}{RD_p}$$

with AD_p the active dose/ha/year and AD_p the recommended dose/ha/year for a specific pesticide registered by the authorities. Next, the single values were summed up for different aggregation levels. At the field level, the STI_{field} for each pesticide group was calculated as the sum of all herbicide and non-herbicide treatments, i.e., fungicide and insecticides. STI_{field} was then calculated as a "cumulative" STI over two years, i.e., summing treatments of the sampling year and the preceding year, or as a single-year STI, including treatments applied during either the sampling year or the preceding year. The preceding year treatments allowed us to investigate indirect effects, whereas treatments applied during the sampling year helped us to assess the more direct effects of the pesticides. $STI_{2y(\text{Inse, fung, herb or no herb})}$ refers to the insecticide, fungicide, herbicide and non-herbicide treatments over two years; $STI_{y(\text{Inse, fung, herb or no herb})}$ refers to the insecticide, fungicide, herbicide and non-herbicide treatments during the sampling year; and $STI_{y-1(\text{Inse, fung, herb or no herb})}$ refers to the insecticide, fungicide, herbicide and non-herbicide treatments in the preceding year.

2.3. Landscape description

At the landscape scale, we used two landscape databases providing information on habitat and crop types present around the studied fields. The Corine Land Cover database (European Environmental Agency, 1995) describes the landscape composition around the studied fields. The CLC is an interpretation of satellite images (100 m resolution) that includes 44 habitat classes grouped into 15 larger categories (see Table A3), thus providing an informative picture of primary farmland, semi-natural, and urban habitats in France. The area covered by each of the 15 habitat categories within a radius of 1000 m centered on each field (Filippi-Codaccioni et al., 2010) was calculated, and the proportion of land cultivated with annual and permanently harvested plants was quantified, collectively labeled Arable1000. When present, non-cultivated habitats were mostly represented by forests, scrubs and transitional vegetation and very rarely by urban areas or herbaceous natural vegetation. The proportion of non-cultivated habitats was thus treated as a measure of landscape closing compared with cultivated fields, which are characterized by wide open landscapes. At a smaller scale, to identify crops planted around sampled fields, we used the map of the crop register updated yearly by farmers (ASP, 2010). This map provides information on the surface occupied by 17 different crop types for each farm in France. We determined the area of each crop, the diversity of crop types within a 200 m buffer around the field margins (calculated with Shannon's diversity index, labeled Divcrop200), and the area of each field.

2.4. Analysis

Statistical analyses were performed using R, version 2.13.1 (R Development Core Team, 2011), and the packages nlme (Pinheiro et al., 2013), MASS (Bates et al., 2011) and hier.part (Walsh and McNally, 2008). All independent variables (Table 1) were standardized by retrieving means and dividing by standard deviations so that the magnitudes of the modeled estimates could be directly compared.

At the community level, we modeled richness, total abundance, CSI, CPP and CPI with a linear mixed model (Eq. (1)), with Arable1000, Divcrop200, field area, crop type, tillage, STI_{2yHer} and $STI_{2yNoHer}$ as fixed factors. Department was included as a random factor to control for spatial pseudo-replications (Pinheiro and Bates, 2000).

$$\begin{aligned}&\text{Richness, Total abundance, CSI, CPP or CPI} \sim \text{Arable1000}\\ &+ \text{Divcrop200} + \text{Field area} + \text{Crop type} + \text{Tillage} + STI_{2yHerb}\\ &+ STI_{2yNoHerb} + \text{random (department)} + \text{spatial autocorrelation}\\ &(X + Y) \qquad (1)\end{aligned}$$

Because community and population indices were spatially autocorrelated (results not shown), the geographical locations (X, Y) of the fields were also included in the model to account for spatial correlation structures nested within departments. This allowed the accommodation of differences between departments, assuming autocorrelation only between fields in the same department (Dormann et al., 2007). In a derived version of Eq. (1), we replaced STI_{2yHerb} and $STI_{2yNoHerb}$ by (1) STI_{yHerb} and $STI_{yNoHerb}$ and by (2) $STI_{y-1Herb}$ and $STI_{y-1NoHerb}$ to investigate the effects on bird communities of herbicide and non-herbicide doses applied during the sampling year or the year preceding the sampling, respectively.

Independence among explanatory variables was tested using Spearman's correlations for continuous variables and other non-parametric tests for categorical variables (the Wilcoxon and Kruskal–Wallis tests). As there were significant associations between independent variables (Table A4), a hierarchical partitioning procedure was applied to alleviate problems derived from multi-colinearity (Chevan and Sutherland, 1991; McNally, 2002). The independent contribution of each explanatory variable (R-value) was reported, and the statistical significance of each was calculated as a pseudo Z-score using 100 randomizations (McNally, 2002).

At the population level, we modeled species abundance with 28 separate models (Eq. (2); one per species) using a quasi-Poisson log-link function to account for over-dispersion in the species count data and the same fixed and random factors as in Eq. (1) but replacing $STI_{2yNoHerb}$ by STI_{2yInse} and STI_{2yFung} The species' responses to STI_{2ys}, i.e., the slopes between species abundance and STI_{2yInse}, STI_{2yHerb} and STI_{2yFung}, were compiled, and the relationships between species' responses and species specialization and diet preferences were tested. For that purpose, we used a linear model with species response as the dependent variable and diet preferences (SPP and SPI) and species specialization (SSI) as the independent variables (Eq. (3)). The linear model enabled us to weight each species' response by the inverse of its variance to account for uneven precision of the slope estimate among species.

$$\begin{aligned}&\text{Species abundance} \sim \text{Arable1000} + \text{Divcrop200} + \text{Field area}\\ &+ \text{Crop type} + \text{Tillage} + (STI_{2yInse},\ STI_{2yFung} \text{ or } STI_{2yHerb})\\ &+ \text{random (department)} + \text{spatial autocorrelation} (X + Y) \qquad (2)\end{aligned}$$

$$\begin{aligned}&\text{Species' responses to } STI_{2yInse} \text{ or to } STI_{2yHerb} \sim \text{SSI} + \text{Diet}\\ &+ \text{SSI : Diet} \qquad (3)\end{aligned}$$

To reduce the number of model parameters, the year effect was accounted for through the random variable 'department', as departments were sampled in different years.

3. Results

Overall, the bird observers recorded 70 species distributed among the 66 fields (Tables A1 and A2). Bird richness was highly variable, ranging from 1 to 18 species per field (each approximately 6.28 ha in area) (mean = 8.9 ± 0.5 (SE) species, Table 1). The total bird abundance ranged from 1 to 40 individuals per field (mean = 18.4 ± 1.1 (SE) individuals, Table 1). CPI and CPP in the diet also varied among fields, ranging from 26 to 73% for CPI (mean = 49%) and 28 to 73% for CPP (mean = 50%).

Linear models and hierarchical partitioning indicated that bird community indices were well correlated with landscape composition (Fig. 2 and Table A5). More specifically, bird richness, total abundance and abundance of generalist species were negatively affected by simplification of the landscape (i.e., proportion of arable land).

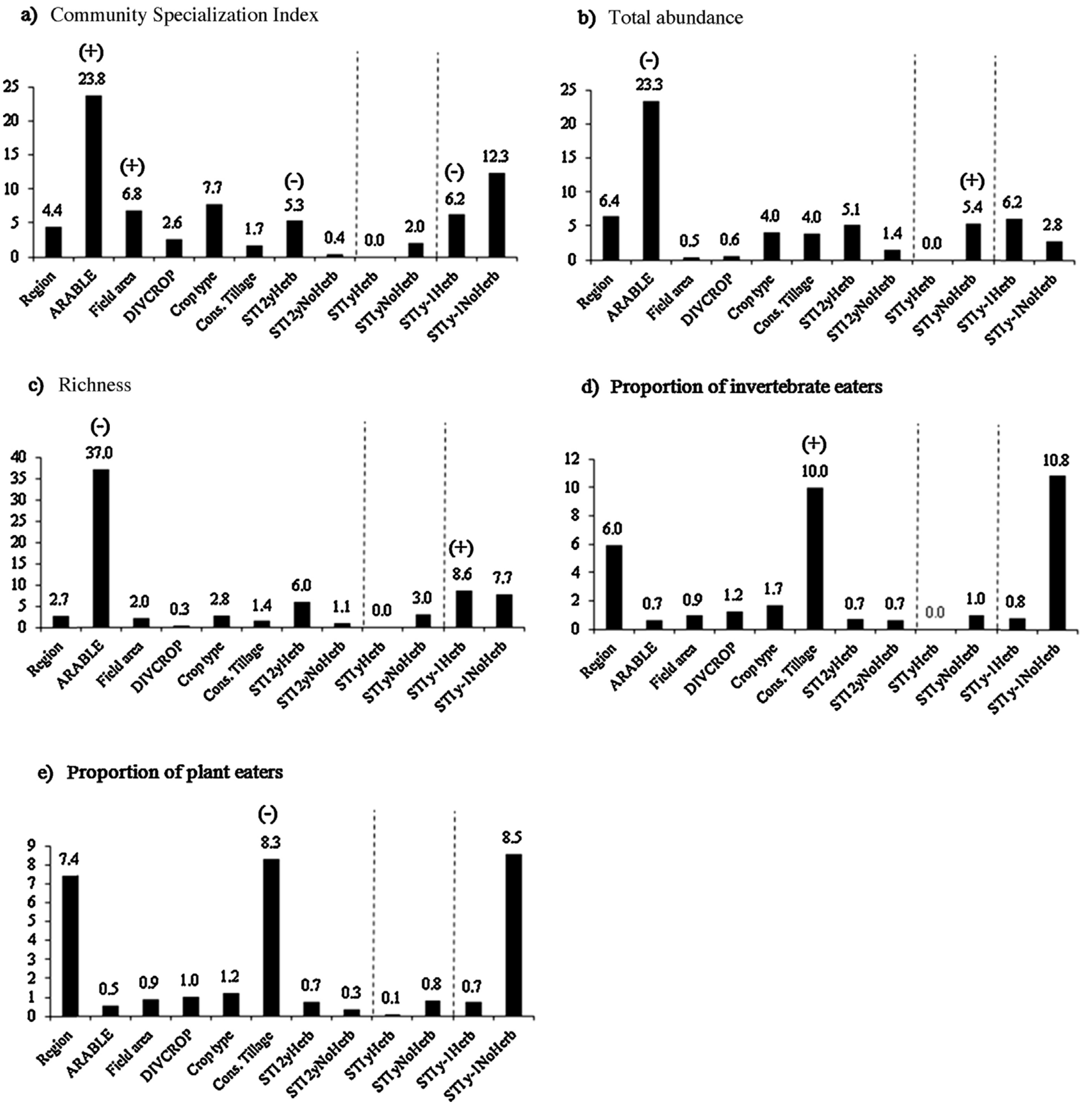

Fig. 2. Results of the hierarchical partitioning analysis for field layout, landscape and agricultural variables. The *Y*-axis indicates the independent contribution of each component to the total variance (*R*-value). STI_{2yHerb} and $STI_{2yNoHerb}$, STI_{yHerb} and $STI_{yNoHerb}$, $STI_{y-1Herb}$ and $STI_{y-1NoHerb}$ were modeled using Eq. (1) in separate models. Based on the linear model results (Table A5), the sign indicates the direction of significant relationships ($p < 0.05$) between the dependent and independent variables.

Generalist species were also less numerous in areas with low crop diversity and large fields. Conservation tillage favored invertebrate eaters, whereas conventional tillage favored herbivorous birds.

Regarding pesticide use, herbicide dose (cumulative over two years and applied during the preceding year only) was negatively correlated with CSI, with specialist birds decreasing in number with higher herbicide doses. Surprisingly, herbicide dose was positively correlated with bird richness and total abundance (Fig. 2 and Table A5).

We then modeled the various species' responses according to their specialization level and diet. There was an interaction between specialization level and species diet, indicating that the abundance of herbivore specialists decreased with cumulative herbicide treatments (slope (SE) = −0.008 (0.004), $p = 0.05$, Fig. 3).

4. Discussion

4.1. *Effects of field practices and landscape simplification*

Herbicide applications decreased the specialization level of bird communities, such that communities in herbicide-treated fields contained fewer specialist than generalist species than those in low treated fields. The measurable effects of herbicides were proportional to the doses applied. As farmland specialists breed and

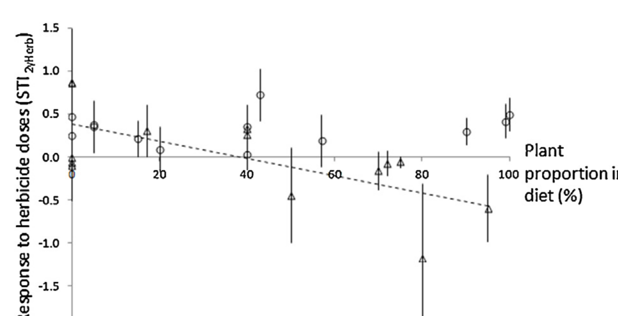

Fig. 3. Responses of specialist (triangles and dashed line, $n = 13$ species with SSI > 0.40) and generalist (dots, $n = 14$ species with SSI < 0.40) species in terms of abundance to herbicides according to the proportion of plants in their diet. We removed the white wagtail from the figure because the model did not converge for this species. The relationships follow from Eq. (3).

forage within fields and along their margins, they are more exposed than generalists to any direct or indirect disturbance caused by agricultural practices or pesticides. Habitat generalist species can use other habitats than farmland, e.g., woodland patches, and this appears to buffer them from the most negative effects of intensified herbicide treatments. This pattern is consistent with the expected higher vulnerability of specialist species with increasing habitat disturbance (Devictor et al., 2008; Pocock, 2011). This pattern also suggests that pesticides, especially herbicides, may contribute to the 'biotic homogenization' process (Clavel et al., 2011) by replacing specialists with generalists. The effect of herbicides on community specialization was mostly indirect, as we only detected effects associated with the preceding year. This indicates that the ecological effects of pesticides may last several years following their application.

Specialist herbivore species, such as the corn bunting and the yellowhammer, are more exposed to herbicides because of their diet specificity. Herbicides can reduce food, nesting and shelter availability for ground-nesting and ground-feeding bird species in cropland (Benton et al., 2002; Kleijn et al., 2009; Guerrero et al., 2011). Generalist herbivores were not affected by herbicides because they do not exclusively depend on the field habitat. Although these species, which include the woodpigeon, chaffinch and turtle dove, can feed on the seeds and leaves of plants found on field margins, they are less likely to be influenced by disturbances induced by agricultural practices and the potential toxic effects of farming products, as they more often forage in adjacent habitats.

The positive relationship observed between herbicide doses and bird richness was unexpected (Geiger et al., 2010; Guerrero et al., 2012). It is possible that this relationship is attributable to the ability of generalist species to arrive and establish in intensively treated fields (Chiron et al., 2010). This capacity may be enhanced because of relaxed competition with specialist species following the decrease of their populations. Furthermore, adding to the hypothesis that specialists are replaced by generalists, we can assume that the increase in richness could have resulted from the arrival of more detectable species, like the wood pigeon and corvids, within the community in fields intensively treated with herbicides, artificially leading to an overall increase in richness. These large-bodied and conspicuous species are generalists that are able to persist in managed cropped areas in the studied departments. The relationship between herbicide doses and bird richness may also result from the large number of tests carried out in our study. When multiple tests are undertaken, the probability of achieving at least one significant result by chance is greater than the accepted significance level. We did not apply any correction to lower the significance level and thereby reduce the probability of a spurious result because this would have negated several important results supported by sound hypotheses and predictions about agricultural effects on species assemblages within communities. Correction procedures are much debated in ecology for this reason (e.g., Moran, 2003).

Although we observed no effect of insecticides, reduced insect availability due to insecticide application is known to be responsible for the decline of many insectivorous bird species in cropland (Boatman et al., 2004). This may be due to the relatively low doses of insecticides (compared with herbicides) applied by farmers on cereal crops during the studied years (Table A4). Conservation tillage had a positive effect on the CSI Index and the proportion of invertebrate feeders. This tillage disturbs the surface of the soil without inverting it and incorporates, to varying degrees, the stubble of the previous crop (Cunningham et al., 2004). This technique may have benefited soil invertebrates like worms and carabid larvae and indirectly the farmland food chain and invertebrate feeders (Cunningham et al., 2004).

Landscape simplification, here defined as an increase of the arable land proportion (versus semi-natural areas) and field size, was associated with a loss of bird diversity. This pattern is consistent with the expected negative effect of reduced non-cropped habitat heterogeneity on species diversity (Benton et al., 2003; Henderson et al., 2009; Fahrig et al., 2011). However, while bird richness declined in simplified arable landscapes, the CSI increased. This result suggests the selection of a few typical farmland birds capable of persisting in simplified arable landscapes. Specialist species such as the skylark, corn bunting and yellow wagtail are ground-nesting species that initially lived in steppe habitats but now occupy cultivated landscapes, here largely dominated by cereals.

4.2. Improving our understanding of pesticide effects on birds

Our study indicates that information about pesticide doses is key to understanding the effects of pesticides on bird species and communities. Ecological studies dealing with pesticides should thus incorporate accurate and detailed information about the treatments used. In the case of our study, the STI was convenient for calculating and comparing pesticide doses between fields as well as assessing potential impacts on birds. Accordingly, dose adjustment would reflect the target efficacy expected to control pest levels by a farmer when deciding on an application as well as the optimum dose, which may still result in an acceptable yield for the farmer while leaving enough food resources for the birds in the field. Compared with the commonly used index 'total amount of pesticides', the STI provides relevant information about environmental risks due to pesticide usage, as the doses applied are standardized to the recommended application rate, and the index accounts for all active substances. However, the STI does not account for the chemical and toxic properties of certain specific substances in pesticides, which may influence the pesticide's effect on the environment.

5. Conclusion and management implications

Our findings stress the importance of combining taxonomic with functional diversity indices to fully understand the changes occurring in bird communities, as they shifted both in composition and in size in response to field and landscape characteristics. To this end, the community specialization and diet indices are suitable for monitoring the impact of field practices and landscapes. Not considering functional and compositional community indices may lead to erroneous conclusions regarding the 'health' of the community and what species and communities are at risk under disturbance intensity gradients in farmland ecosystems.

To improve the environmental friendliness of agricultural practices, the industry, consumers and politicians have encouraged the widespread development of organic systems (Paull, 2011). However, although there is positive support for the idea that organic systems can enhance the diversity of multiple taxa (Hole et al., 2005; Paull, 2011), integrated management systems have been poorly studied, though they may also contribute to biodiversity conservation. Integrated management farms engage in the reduction of agrochemical treatments (pesticides and fertilizers) but do not aim to realize a complete conversion to an organic system. Our study demonstrates that such systems can still benefit biodiversity. Increasing habitat heterogeneity in the landscape and reducing pesticide doses should benefit taxonomic and/or functional diversity in integrated management systems. This finding has important implications for biodiversity conservation at the wider agricultural landscape scale in France and elsewhere, as long as cereal systems are comparable among countries because, compared with organic systems, integrated management fields comprise a large proportion of the total land area managed for cereal production (Hendrickson et al., 2008).

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2013.12.013.

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