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Different policy scenarios to promote various targets of biodiversity

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ABSTRACT

Biodiversity loss in farmlands is widely documented, and agriculture intensification has been identified as a main driver of this decline. Numerous agri-environmental policies have been implemented to assess the negative impacts of agricultural intensification on biodiversity. However, most published studies focus on land-use scenarios, thus neglecting the economic dimension. We develop a bio-economic spatially explicit modelling across 620 small French agricultural areas, which couples a public decision maker under budgetary constraint, regional economic agents in a context of uncertainty and breeding bird dynamics. Using dynamic models, we analyse the direct impacts of several current economic scenarios of the Common Agriculture Policy on common bird communities through five ecological indicators, all related to breeding populations of birds in farmlands: the farmland bird index (FBI), a generalist bird index (GBI), the Shannon diversity index, a community specialization index (CSI) and a community trophic index (CTI). We consider these indicators to scan various functional traits of bird communities. Trends in the different indicators are significantly contrasted pending on economic policy scenarios. Scenarios promoting intensive crops lead to small but specialized communities with more granivorous species, hence a low trophic level for the community. By contrast, promoting extensive grasslands increases the population size, enhances high trophic level but decreases community specialization. Evaluation of agricultural policies should not rely on a single indicator per taxonomic group. In the context of potential reversal of current bird declines, bio-economic modelling, involving farmland incomes, is proposed as a relevant support for decision making about sustainable agri-environmental policies. Promoting extensive grasslands is essential for the sustainable management of bird communities and agriculture. We, however, reveal more complex economic effects and synergies between public incentives, which appear to give interesting leverage for enhancing the bio-economic effectiveness of agricultural policies.

1. Introduction

Numerous monitoring programs have reported farmland biodiversity losses across continents in recent decades with special focus on bird declines (Butchart et al., 2010; Donald et al., 2001; Flowerdew and Kirkwood, 1997; Sotherton and Self, 2000). The impacts of agricultural intensification on biodiversity are particularly strong on bird populations (Chamberlain et al., 2000; Krebs et al., 1999). Global changes in European agriculture, including intensification and land abandonment, have significantly modified farmland bird communities (Donald et al., 2001, 2006; Devictor et al., 2008). Such erosion is mainly induced by a combination of habitat loss and fragmentation and of degraded habitat quality altering the reproductive success and/or survival of individuals (Benton et al., 2003). In this context, the need to reconcile agricultural production and biodiversity is of particular concern (Jackson et al., 2005). In the European Union, since the 1990s, several public policies have been dedicated to limiting the negative impacts and externalities of agriculture on biodiversity. Typically, agri-environmental schemes have been implemented so that farmers receive financial support for adopting environment-friendly agricultural practices (Kleijn and van Zuijlen, 2004). There is an extensive and increasing literature concerning agri-environmental schemes and policies for multi-functional agriculture (Albrecht et al., 2007; Batary et al., 2007; Kleijn and van Zuijlen, 2004; Münier et al., 2004; Taylor and Morecroft, 2009). Still, fifteen years after the implementation of such instruments, whether providing habitat quality conflicts with management for agricultural production remains controversial (Butler et al., 2007; 2006; Vickery et al., 2004). One limit of the evaluations performed is the focus on land-use scenarios, ignoring the economic behaviour of farmers facing public incentives (Scholefield et al., 2009). As agricultural policies are mainly proposed in economic terms, the introduction of economic dimensions appears as essential to define sustainable management of both agriculture and biodiversity (Mouysset et al.,

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2011). As pointed out by Hughey et al. (2003) and Perrings et al. (2006), there is a need for approaches integrating economic criteria in conservation problems.

In order to analyse potential trends of different agricultural policies on biodiversity, the present paper develops a bio-economic model, which articulates a national decision maker, regional farmers and biodiversity dynamics for France. To give strong realism to the scenarios, we integrate a national budgetary constraint and calibrate the model with ecological, farming land-use and economic databases. To characterize biodiversity, we focus on breeding birds, which are largely recognized as a representative biodiversity compartment highly sensitive to agricultural practices (Gregory et al., 2009), although the metrics and the characterization of biodiversity remain an open debate (Le Roux et al., 2008; MEA, 2005). Focus on breeding birds is further justified because (i) birds lie at a high level in the trophic food chains and thus capture the variations in the chains; (ii) birds provide ecological services, such as the regulation of invertebrate and rodent populations and pest control (Sekercioglu et al., 2004); (iii) their close vicinity to humans makes them a simple and comprehensive biodiversity index for a large audience of citizens (Ormerod and Watkinson, 2000).

In direct line with these considerations, the European Union has adopted the farmland bird index (FBI (Gregory et al., 2009)) as an indicator of structural changes in biodiversity (Balmford et al., 2005). However, beyond changes in bird abundances, community traits and functions are only vaguely summarized by a single indicator (Barbault and Chevassus-au-Louis, 2004), while various indicators are available in the literature to describe and analyse bird communities. State indicators such as the Shannon–Wiener diversity index or the EU farmland bird index are widely used to quantify biological diversity and associated trends in farmlands. Doxa et al. (2010) reports the relevance of the FBI to reflect the response of farmland biodiversity to agriculture intensification. Other trait- or function-based indicators referring to community specialization (Julliard et al., 2006) or trophic level (Pauly et al., 1998) explore functional characteristics of the communities. Bird communities are more specialized in unaltered and non-fragmented habitats, including farmlands (Devictor et al., 2008), while higher trophic levels should testify to unaltered food chains and therefore communities ensuring more ecological functions. In this perspective, we used different ecological indicators to analyse the performance of our bio-economic modelling.

2. Material and methods

We developed a spatialized bio-economic model over the 620 small French metropolitan agricultural areas (PRA, "petite région agricole"). Their consistency at both agro-ecological and economic levels makes them particularly well-suited for our modelling and analysis. As in Mouysset et al. (2011), three compartments are linked: national public decision maker, regional economic agent and bird community (Fig. 1).

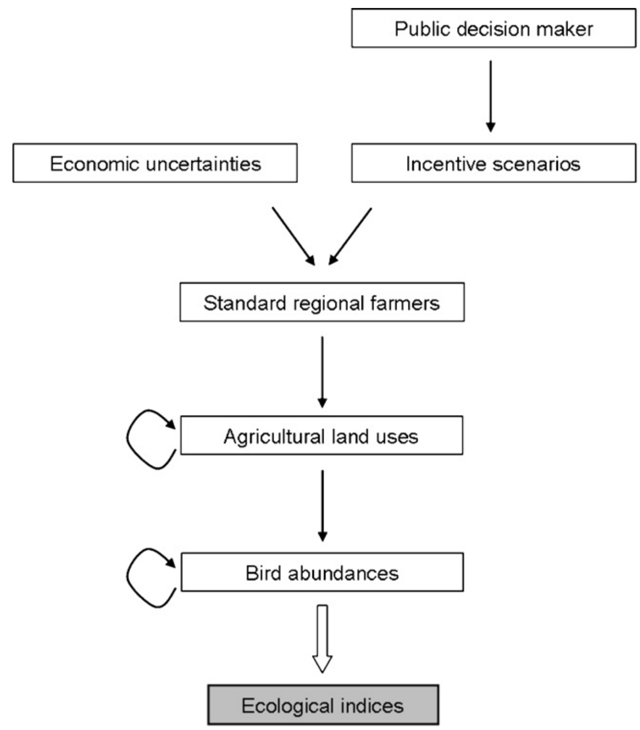

Fig. 1. Model coupling: farmers adjust their agricultural systems pending on economic uncertainty and incentives. These choices affect bird community dynamics.

2.1. The ecological model

To assess ecological performance, we focus on common farmland birds (Ormerod and Watkinson, 2000). Bird populations are driven by Beverton–Holt dynamics (Beverton and Holt, 1957) which capture intra-specific competition through the carrying capacity parameter:

$$N_{s,r}(t+1) = N_{s,r}(t)\frac{1+R_{s,r}}{1+(N_{s,r}(t))/(M_{s,r}(t))} \quad (1)$$

where $N_{s,k}(t)$ stands for the bird abundance of species s in PRA r at year t. The $R_{s,r}$ coefficient corresponds to the intrinsic growth rate specific to a given species s. The product $M_{s,k}(t) \times R_{s,r}$ represents the carrying capacity of the habitat r and the value $M_{s,k}(t)$ captures the ability of the habitat to host the species.

This habitat parameter depends on the agricultural land-uses chosen by the farmers (Eq. (2)) as follows:

$$M_{s,r}(t) = b_{s,r} + \sum_k a_{s,r,k}.A_{r,k}(t) \quad (2)$$

where surfaces $A_{r,k}(t)$ including crop or grasslands are detailed in Table 1. Consequently, the a and b coefficients, specific to each species, inform on how such species s respond to various agricultural systems k in a PRA r. The $b_{s,r}$ coefficient can be interpreted as the mean habitat coefficient for a species s in a PRA r. The ecological model is calibrated for each PRA in order to integrate regional agro-environmental features.

2.2. The micro-economic model

We considered 620 PRA of metropolitan France, so we have 620 regional economic standard agents with 620 representative farms. A representative farm does not really exist and represents an 'average' farm for the PRA. We compute these historical characteristics by averaging those of all the real farms of the PRA. As PRA has an agricultural and ecological homogeneity, all real farms in a PRA have similar characteristics and joining them in a 'mean' farm makes sense. The regional economic standard agents select their agricultural land-uses in order to maximise their utility under technical constraints. These choices, made in an uncertainty context, depend on expected gross margins, financial incentives specified by the public (national) decision maker and current land use areas. This approach refers to stochastic maximisation under constraints, usual in bio-economic modelling (Lien, 2002).

Farmer's income in PRA r at year t denoted by $\text{Income}_r(t)$ relies on the gross margin $\text{gm}_{r,k}(t)$ for the year t, current agricultural activities $A_{r,k}(t)$ and incentives $\tau_k(t)$ which take the form of taxes ($\tau_k < 0$) or subsidies ($\tau_k > 0$) (Eq. (3)). Gross margins $\text{gm}_{r,k}(t)$ are taken to

Table 1
Proportion of the French agricultural area dedicated to the the 14 agricultural systems named OTEX in the initial states 2008 and under the statu quo scenario in 2030 and 2050.

	Initial states	Statu quo scenario		Trend
	2008	2030	2050	
(1) Cereal, oleaginous, proteaginous	25.8%	13.3%	4.37%	↘
(2) Variegated crops	0.500%	1.88%	4.19%	↗
(3) Intensive bovine livestock breeding	17.2%	22.7%	9.79%	↘
(4) Medium bovine livestock breeding	5.75%	6.02%	3.00%	↘
(5) Extensive bovine livestock breeding	15.5%	5.13%	1.53%	↘
(6) Mixed crop-livestock farming with herbivorous direction	0.860%	3.56%	4.61%	↗
(7) Other herbivorous livestock breeding	5.07%	4.03%	3.01%	→
(8) Mixed crop-livestock farming with granivorous direction	0.01%	0.04%	0.22%	→
(9) Mixed crop-livestock farming with other direction	20.4%	11.4%	3.08%	↘
(10) Granivorous livestock breeding	2.46%	15.0%	33.0%	↗
(11) Permanent farming	1.05%	1.81%	1.36%	→
(12) Flower farming	0.971%	6.41%	21.8%	↗
(13) Viticulture	4.19%	8.10%	8.11%	↗
(14) Others associations	0.005%	0.007%	0.01%	→

be uncertain. The variability on gross margins includes both market, production and climate uncertainties. A Gaussian distribution parameterized with the mean and the covariance matrix of the historical data is chosen to capture such uncertainties.

$$\text{Income}_r(t) = \sum_k \text{gm}_{r,k}(t) \cdot A_{r,k}(t) \cdot (1 + \tau_k(t)) \qquad (3)$$

For each year t, the regional standard agents chose their agricultural systems $A_{r,k}(t)$ in order to maximise their utility in an uncertain context (Eq. (4)). This utility corresponds to the expected income, which depends on the expected gross margins $\overline{\text{gm}}_{r,k}(t)$ computed with the 7 historical gross margins (2002–2008) (Eq. (5)).

$$\max_{A_{r,k}} \text{Utility}_r(t) = \max_{A_{r,k}} \sum_k \overline{\text{gm}}_{r,k}(t) \cdot A_{r,k}(t) \cdot (1 + \tau_k(t)) \qquad (4)$$

$$\overline{\text{gm}}_{r,k} = \frac{1}{7} \sum_{t=2008}^{t=2002} \text{gm}_{r,k}(t) \qquad (5)$$

The agricultural choices are limited at every time t by capital and rigidity constraints:

$$|A_{r,k}(t) - A_{r,k}(t-1)| \leq \varepsilon \cdot A_{r,k}(t-1) \qquad (6)$$

$$\sum_k A_{r,k}(t) = A_r \qquad (7)$$

The rigidity constraint (Eq. (6)) limits the area that the farmer can change at each time for each agricultural system. It captures change costs and limits change speed quantified by the parameter ε. The constraint (Eq. (7)) ensures that the total agricultural surface A_r in a PRA is kept fixed.

2.3. The public policy model

The national decision maker selects economic incentives scenarios $\tau_k(t)$. It defines taxes and/or subsidies for different agricultural activities k according to specific objectives and a budgetary constraint. For all scenarios, incentives $\tau_k(t)$ are assumed to be linearly decreasing with time, from 2009 to achieve 0 in 2050 (Eq. (8)). Such decreasing incentives capture the current trend of common agricultural policy (CAP) perspectives.

$$\tau_k(t) = \tau_k(2009)\left(1 - \frac{t - 2009}{2050 - 2009}\right) \qquad (8)$$

We develop five scenarios to test different perspectives of CAP:

- A statu quo (SQ) scenario, with no further tax or subsidy. It prolongs the current trend and leads to defining marginal effects of the other policies compared to the current evolution.
- A crop (CR) scenario, which promotes cereal–oleaginous–proteaginous crops (COP). We here test a pattern of intensification typically associated with the development of bioenergy.
- A grassland (GL) scenario with subsidies to extensive grasslands. It corresponds to the opposite pattern of the intensification scenario and promotes extensive agricultural systems with low intensification, small fields and many linear elements.
- A double subsidies (DS) scenario with subsidies to both COP and extensive grasslands. This scenario is the closest to the current situation.
- A high quality environment (HQE) scenario with taxes on COP and subsidies to extensive grasslands. As with the Grassland scenario, the objective of this scenario is to favor extensive agriculture but with a more intensive policy.

Two scenarios make it possible to study potential synergies (HQE scenario) or antagonisms (DS scenario) between incentives.

To define the initial level of incentives $\tau_k(2009)$ for the agricultural systems underlying the scenarios, it is supposed that the public stakeholder complies with a budgetary constraint. We define the national budget as the sum of public incentives distributed over the 620 PRA (Eq. (9)).

$$\text{Budget}_{\text{Nat}}(t) = \sum_r \sum_k \text{gm}_{r,k}(t) \cdot A_{r,k}(t) \cdot \tau_k(t) \qquad (9)$$

Incentives $\tau_k(2009)$ are such that the national budget required by the decision maker has to remain lower than the current budget $\text{Budget}_{\text{Nat}}(2008)$ at each year t (Eq. (10)).

$$\text{Budget}_{\text{Nat}}(t) \leq \text{Budget}_{\text{Nat}}(2008) \qquad (10)$$

2.4. Data

To calibrate this model, several databases have been articulated: the Farm Accountancy Data Network[1] (FADN) and the Observatory of Rural Development[2] (ODR). Fourteen classes of agricultural systems, named OTEX (Orientation Technico-Economique) and displayed in Table 4 are distinguished in this manner. Each PRA is a specific combination of these OTEX. The surfaces dedicated

[1] http://ec.europa.eu/agriculture/rica/.
[2] https://esrcarto.supagro.inra.fr/intranet/.

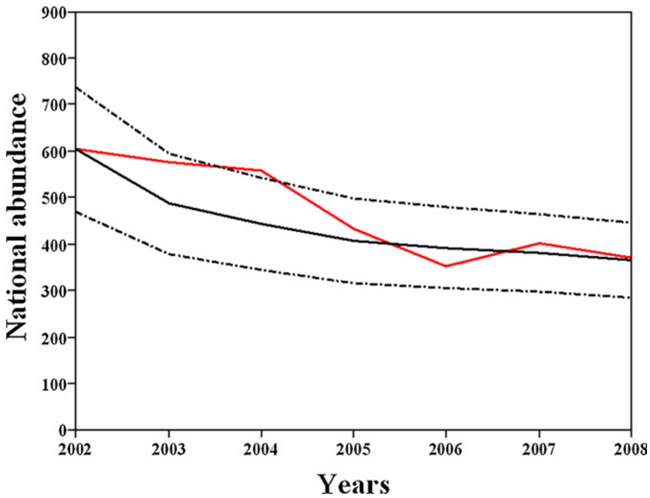

Fig. 2. Comparison between historical (red) and estimated (black) national abundances with the least square standard errors of calibration (dashed lines) for one of the species considered, the Wood Lark *Lullula arborea*. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

to each of the 14 OTEX and the associated fiscal bases which are gross margin markers, for the years 2001 to 2008 are available on the ODR website. Gross margin is an economic indicator broadly used in bio-economic modelling (ten Berge et al., 2000; Pacini et al., 2004) and agricultural economics (Lien, 2002). The budgetary constraint was calibrated with the current French CAP budget.

For the ecological part of the model, we used data provided by the national breeding bird survey (BBS) implemented in France since 2001. Among the common breeding species monitored by this scheme, we focused on those 34 species classified as farmland specialist and habitat generalist species, according to their habitat requirements (Julliard et al., 2006). The list of these species is presented in Table 2. Abundance values for each species were available for the period 2002–2008 for 1747 squares (a square is 2 km × 2 km in size) (as detailed in Jiguet, 2009; Jiguet et al., 2010). For each species, we further performed a spatial interpolation of these abundance data to obtain relative abundance values for each possible square in the country (e.g. 136 000 squares) using kriging models based on spatial autocorrelation and an exponential function. We then averaged the abundance values at the PRA scale to calibrate the ecological model. Fig. 2 illustrates the results of this calibration with one species, the Wood Lark *Lullula arborea*. Comparing the historical data with the model-generated data, we note that the model tends to smooth the variations of the observed data.

We use this bio-economic model to assess the impact of public economic policies on bird communities. The selected timeframe runs up from 2009 to 2050, i.e a 42-year forecast. Adopting a shorter timeframe could consequently hide interesting medium-term effects due to the inertia of the model. Relevant $\tau_k(2009)$ for each scenario are described in Table 3. To clarify the impacts of policy scenarios, Table 1 presents the allocation of the agriculturally utilized area among the 13 OTEX at the national scale before the projection (initial state: 2008) and under the statu quo scenario. Table 4 illustrates the variations in the areas allocated to the various OTEX under the 4 policy scenario compared to the statu quo scenario. To study the community obtained after the different public policies, we develop various and complementary indicators to capture state, functional and pressure responses as described in Section 2.5.

2.5. Biodiversity indicators

2.5.1. Abundance indices

To analyse predicted trends in population abundances, we first focused on the national farmland bird index to study the structural changes in biodiversity (Balmford et al., 2005). Previous analyses have shown the relevance of the national FBI to reflect the response of farmland biodiversity to agriculture intensification (Doxa et al., 2010). This indicator reports the variation in the abundances of 20 habitat specialists distinctive of farmland habitats. A similar indicator is proposed here for 14 habitat generalists, namely a generalist bird index (GBI), similarly reporting the variations of abundances of these species (Julliard et al., 2004), with the aim of comparing the response of the two groups (Table 2). These multiple-species indicators are computed as the geometric mean of the yearly indices of the species considered in the group. In theses aggregated indices, the abundances variation of each species is taken into account similarly, independently from the abundance value. We first estimated a national population index for each species from the abundances values of all PRA r (Eq. (11)), then we calculated the aggregated indicators FBI_{Nat} and GBI_{Nat} (Eqs. (12) and (13).

$$N_{s,Nat}(t) = \sum_r N_{s,r}(t) \tag{11}$$

$$FBI_{Nat}(t) = \prod_{s \in \text{Specialist}} \left(\frac{N_{s,Nat}(t)}{N_{s,Nat}(2008)} \right)^{1/20} \tag{12}$$

$$GBI_{Nat}(t) = \prod_{s \in \text{Generalist}} \left(\frac{N_{s,Nat}(t)}{N_{s,Nat}(2008)} \right)^{1/14} \tag{13}$$

2.5.2. Shannon index

We computed the Shannon–Wiener diversity index for the whole community of the 34 species (including both habitat generalists and farmland specialists). This index informs about the repartition of individual birds within the different species in the community (Eqs. (14) and (15)). A larger value of this index means a more balanced repartition of individuals between species. The national Shannon index is the arithmetic mean of the regional Shannon indices (Eq. (16)).

$$N_{tot,r}(t) = \sum_s N_{s,r}(t) \tag{14}$$

$$\text{Shannon Index}_r(t) = -\sum_s \frac{N_{s,r}(t)}{N_{tot,r}(t)} \cdot \log \left(\frac{N_{s,r}(t)}{N_{tot,r}(t)} \right) \tag{15}$$

$$\text{Shannon Index}_{Nat}(t) = \frac{1}{620} \cdot \sum_r \text{Shannon Index}_r(t) \tag{16}$$

2.5.3. Community specialization index

We considered an indicator of pressure: the community specialization index (CSI). The objective of this indicator is to interpret the response of the composition of local bird communities to agricultural pressures. A habitat specialization species index (SSI) has been computed for each species, reporting the coefficient of variation of the abundance of a species across 18 habitat categories (see Julliard et al., 2006; Table 2[3]). For each square, the local CSI_r is then calculated as the arithmetic mean of the species specialization index weighted by the abundances (Eqs. (14) and (17). This index measures the average degree of habitat specialization among the

[3] Although the species specialization index for Buzzard is smaller than for some of generalist species, we kept it in farmland to conserve the European classification.

Table 2
The 34 bird species considered in this study, with reference to their habitat specialization, and values of their species specialization index and species trophic index.

Species	Habitat specialization	Specialization index	Trophic index
Buzzard *Buteo buteo*	Farmland	0.49	2.90
Cirl Bunting *Emberiza cirlus*	Farmland	0.59	1.30
Corn Bunting *Emberiza calandra*	Farmland	1.46	1.28
Grey Partridge *Perdix perdix*	Farmland	2.11	1.10
Hoopoe *Upupa epops*	Farmland	0.61	2.00
Kestrel *Falco tinnunculus*	Farmland	0.68	2.85
Lapwing *Vanellus vanellus*	Farmland	2.23	1.90
Linnet *Carduelis cannabina*	Farmland	0.70	1.05
Meadow Pipit *Anthus pratensis*	Farmland	1.37	1.75
Quail *Coturnix coturnix*	Farmland	1.52	1.22
Red-backed Shrike *Lanius collurio*	Farmland	1.14	2.15
Red-legged Partridge *Alectoris rufa*	Farmland	1.10	1.10
Rook *Corvus frugilegus*	Farmland	0.84	1.63
Skylark *Alauda arvensis*	Farmland	1,16	1.25
Stonechat *Saxicola torquatus*	Farmland	0.78	2.00
Whinchat *Saxicola rubetra*	Farmland	1.46	2.00
Whitethroat *Sylvia communis*	Farmland	0.65	1.60
Wood Lark *Lullula arborea*	Farmland	0.90	1.50
Yellowhammer *Emberiza citrinella*	Farmland	0.71	1.30
Yellow Wagtail *Motacillajlava*	Farmland	2.09	2.00
Blackbird *Turdus merula*	Generalist	0.23	1.60
Blackcap *Sylvia atricapilla*	Generalist	0.32	1.60
Blue Tit *Cyanistes caeruleus*	Generalist	0.35	1.80
Carrion *Crow Corvus corone*	Generalist	0.28	1.51
Chaffinch *Fringilla coelebs*	Generalist	0.27	1.10
Cuckoo *Cuculus canorus*	Generalist	0.43	2.00
Dunnock *Prunella modularis*	Generalist	0.50	1.50
Golden Oriole *Oriolus oriolus*	Generalist	0.47	1.95
Great Tit *Parus major*	Generalist	0.29	1.85
Green Woodpecker *Picus viridis*	Generalist	0.38	2.00
Jay *Garrulus glandarius*	Generalist	0.44	1.72
Melodious Warbler *Hippolais polyglotta*	Generalist	0.70	1.95
Nightingale *Luscinia megarhynchos*	Generalist	0.47	2.00
Wood Pigeon *Columba palumbus*	Generalist	0.30	1.01

Table 3
Initial incentives $\tau_k(2009)$ for the 4 policy scenarios.

	ST scenario	CR scenario	GL scenario	DS scenario	HQE scenario
(1) Cereal, oleaginous, proteaginous	0%	+65%	–	+30%	−30%
(4) Medium bovine livestock breeding	0%	–	+55%	+50%	+60%
(5) Extensive bovine livestock breeding	0%	–	+55%	+50%	+60%
(6) Mixed crop-livestock farming with herbivorous direction	0%	–	+55%	+50%	+60%
(7) Other herbivorous livestock breeding	0%	–	+55%	+50%	+60%

Table 4
Variations of the proportions of the French agricultural area dedicated to the 14 agricultural systems named OTEX with the 4 policy scenarios in 2030 and 2050 compared to those obtained under the statu quo scenario (Table 2).

	CR scenario			GL scenario			DS scenario			HQE scenario		
	2030	2050		2030	2050		2030	2050		2030	2050	
(1) Cereal, oleaginous, proteaginous	+**20** %	+9.3%	↗	−3.44%	−2%	↗	+**6.3** %	+2.09%	↗	−**7.55**%	−3.19%	↘
(2) Variegated crops	−0.34%	−0.5%		−0.1%	−0.22%		−0.16%	−0.31%		−0.01%	−0.21%	
(3) Intensive bovine livestock breeding	−8.7%	−3.63%		−13.43%	−5.43%		−15.9%	−6.2%		−13.27%	−5.47%	
(4) Medium bovine livestock breeding	−1.8%	−0.49%		+**11.9** %	+1.5%	↗	+**10.58** %	+5.02%	↗	+**12.28** %	+6.39%	↗
(5) Extensive bovine livestock breeding	−0.97%	−0.44%		+**8.47** %	+2.05%	↗	+**6.37** %	+1.43%	↗	+**8.97** %	+2.22%	↗
(6) Mixed crop-livestock farming with herbivorous direction	−0.24%	−0.24%		+0.93%	+1.42%		+0.79%	+1.11%		+0.97%	+1.55%	
(7) Other herbivorous livestock breeding	−0.84%	−0.79%		+2.32%	+5.89%		+1.87%	+1.18%		+2.6%	+1.67%	
(8) Mixed crop-livestock farming with granivorous direction	+0%	+0%		+0%	+0%		+0%	+0%		+0%	+0.01%	
(9) Mixed crop-livestock farming with other direction	−6.19%	−1.51%		−5.16%	−1.31%		−8.14%	−2.21%		−1.26%	−1.34%	
(10) Granivorous livestock breeding	−0.3%	−0.3%		−0.8%	−0.9%		−0.7%	−0.9%		−0.8%	−1%	
(11) Permanent farming	+0%	−0.01%		−0.05%	−0.02%		−0.04%	−0.02%		−0.07%	−0.02%	
(12) Flower farming	−0.31%	−0.8%		−0.31%	−0.6%		−0.45%	−0.7%		−0.33%	−0.7%	
(13) Viticulture	−0.81%	−0.19%		−0.33%	−0.13%		−0.39%	−0.17%		−0.2%	−0.13%	
(14) Others associations	−0.001%	+0%		+0%	+0%		+0%	+0.005%		+0%	+0%	

individuals of the community. It leads to discriminating the ordinary community of generalist species, which are more resilient to perturbation, from the specialized communities with more specialist species, which are especially sensitive to global change (Julliard et al., 2006). National CSI_{Nat} is the arithmetic mean of the 620 regional CSI_r (Eq. 18).

$$CSI_r(t) = \sum_s \frac{N_{s,r}(t)}{N_{tot,r}(t)} \cdot SSI_s \quad (17)$$

$$CSI_{Nat}(t) = \frac{1}{620} \cdot \sum_r CSI_r(t) \quad (18)$$

2.5.4. Community trophic index

To consider a functional dimension of bird communities, we supplemented the previous indices with a community trophic index (CTI) (Pauly et al., 1998). The position of each species within the trophic chain was computed from information on specific diets as available in BWPi (2006), defining the proportion of each species diet made of vegetables, invertebrates and vertebrates, then estimating an average species trophic index by computing a weighted mean of the 3 diet proportions (weighting coefficients being 1 for vegetables, 2 for invertebrates, 3 for vertebrates). Specific trophic indices (STI) are described in Table 2 for the 34 studied species. The CTI_r reports on the average trophic level of the community. It is computed as the weighted arithmetic mean of the exponential of the species trophic level balanced by the abundances (Eqs. (14) and (19)). An exponential function is used to better contrast communities with or without bird individuals of the higher trophic levels. This indicator discriminates the communities with more granivorous species (e.g. low trophic level) against the communities with more insectivorous and carnivorous species (e.g. high trophic level). National CTI_{Nat} is the arithmetic mean of the 620 regional CTI_r (Eq. (20)).

$$CTI_r(t) = \sum_s \frac{N_{s,r}(t)}{N_{tot,r}(t)} \cdot \exp(STI_s) \quad (19)$$

$$CTI_{Nat}(t) = \frac{1}{620} \cdot \sum_r CTI_r(t) \quad (20)$$

2.5.5. Rate index

To clarify CSI and CTI analyses, we complement these indicators with species indices. They measure the proportion of one species among the community (Eqs. (14) and (21)). We can compare the proportions of different species in the bird population according to their functional characteristics. The National species index is the arithmetic mean of the 620 regional rate index (Eq. (22)).

$$\text{Rate index}_{s,r}(t) = \frac{N_{s,r}(t)}{N_{tot,r}(t)} \quad (21)$$

$$\text{Rate index}_{s,Nat}(t) = \frac{1}{620} \cdot \sum_r \text{Species index}_{s,r}(t) \quad (22)$$

3. Results

As the farmer choices occur in uncertainty contexts, we ran 100 simulations for each scenario with different Gaussian gross margins $gm_{r,k}(t)$ to estimate the means of community outcomes and their 95% confident interval. Species indices are illustrated for one run. We present the outcomes of the statu quo scenario (Fig. 3) then the outcomes of the crop, grassland, double subsidies and high quality environmental scenarios normalized by the outcomes of the statu quo scenario (Fig. 4). We can thus analyse the marginal benefits of the different economic policies compared to current policy trends. To clarify the ecological effects of the various scenarios, Table 1 describes the land uses allocation at the initial states and under the statu quo scenario. Table 4 illustrates the changes in land uses compared to the statu quo for the four policy scenarios and Fig. 5 presents the proportion of the utilized agricultural area dedicated to the extensive activities (OTEX 4, 5, 6, 7) under the five scenarios.

3.1. Global outcomes

Fig. 3 shows that with the SQ scenario the quantities of farmland bird species decrease while those of generalist species increase. This leads to a decreasing specialization level and a loss of diversity for the community. The community trophic index follows the historical trend. Fig. 4 illustrates non linear and non monotonous trajectories among the policy scenarios. Indicators have contrasted trends in function of the scenarios for 30–40 years, then they start to return to a baseline value (of 1) around 2030–2040. Table 4 illustrates that changes in agricultural systems within the four policy scenarios compared to the statu quo are broader in 2030 than in 2050. The farmers modify their agricultural systems according to the incentives as long as they are sufficient to significantly impact the rentabilities of the OTEX. When incentives become too small with time, representative farmers face a decision problem similar to the SQ scenario although the rigidity constraint slows down such a pattern. Hence, at the end of the projections, we obtain bio-economic performances close to those obtained with the SQ scenario, which is represented in Fig. 4 with indicators converging to one. However, the year of the optimum can vary according to the indicators (around 2030 for the Shannon index versus 2040 for the FBI), or on the scenarios for a given indicator (2028 with the HQE scenario and 2035 with the CR scenario for the generalist bird index). The marginal effects of the different scenarios compared to the current trend are strong only for abundance indices (GBI and FBI) and more particularly for the FBI. For the other indicators, the maximal variations remain around 1%. However, analysing dispersion of outcomes, we can distinguish significative differences in trajectories for the other indicators and thus significative marginal effects of economic public policies.

3.2. Abundance of habitat generalist and farmland specialist species

We note clear differences between the four scenarios with regards to both abundance indicators FBI and GBI (Fig. 4(a and b)). Both indicators classify the scenarios in the same order. If evaluating the efficiency of the scenarios to enhance bird numbers, the most effective scenario is the HQE scenario, then the GL scenario, the DS scenario and finally the least effective one is the CR scenario. The FBI is very sensitive and we observe a stronger population increase, with a maximum improvement of 60% with HQE against 8% with the same scenario for the GBI. Although the farmland bird populations increase in the four considered scenarios, compared to a statu quo trend, populations of habitat generalists show more variable responses and can increase (with the HQE and GL scenarios), remain stable (with the DS scenario) or decrease (with the CR scenario).

3.3. Shannon diversity

We observe two general trends for the Shannon index on Fig. 4(c). With the CR scenario, the index decreases compared to the current trend, while it increases with the three other scenarios. As a consequence, bird communities ensuing from scenarios promoting grasslands display a better balanced composition than would be observed by maintaining the current trends. The best level is reached with the DS scenario.

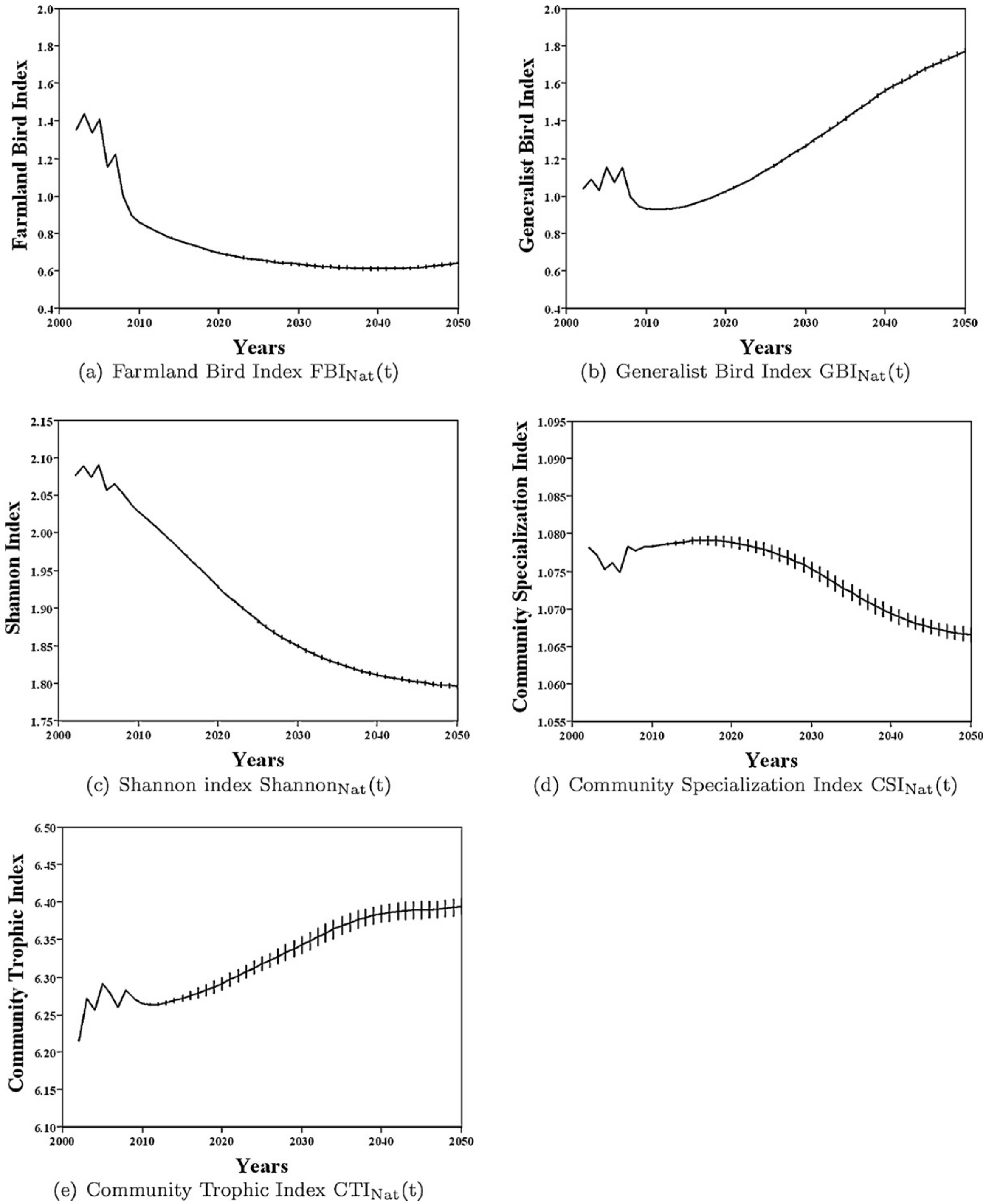

Fig. 3. Ecological indicators evolutions (mean outcomes and 95% confident interval) with the statu quo scenario.

3.4. Community specialization

The four scenarios are different with regards to the Community Specialization Index (Fig. 4(d)). Compared to the ongoing trend, both scenarios with subsidies for COP (CR and DS scenarios) enhance the specialization level of bird communities. The other two scenarios (GL and HQE) have lower CSI with respect to the current trend. As with the abundance indices, the extreme situations are obtained with the extreme scenarios: the CR scenario for the highest CSI (the most specialized community) and the HQE for the lowest CSI (the least specialized community). Fig. 6(a and b) reports the modelled trends for two species of similar trophic level but contrasted habitat specialization. The specialized Lapwing *Vanellus vanellus* is more sensitive to scenarios than the generalist Great Tit *Parus major*, the numbers of which always increase within the community. Fig. 6(c–e) reports the trends for three farmland specialists, though specialized to different types of farmland. The abundances of Grey Partridge *Perdix perdix* and Whinchat *Saxicola rubetra*, being more specialized in one farmland habitat type (open field versus extensive grasslands, respectively) respond more strongly to scenarios. For the Whinchat *S. rubetra*, the progressive reduction in incentives and taxes has an obvious effect and the population size decreases substantially after an initial increase in the scenarios favoring grasslands. The Common Whitethroat *Sylvia*

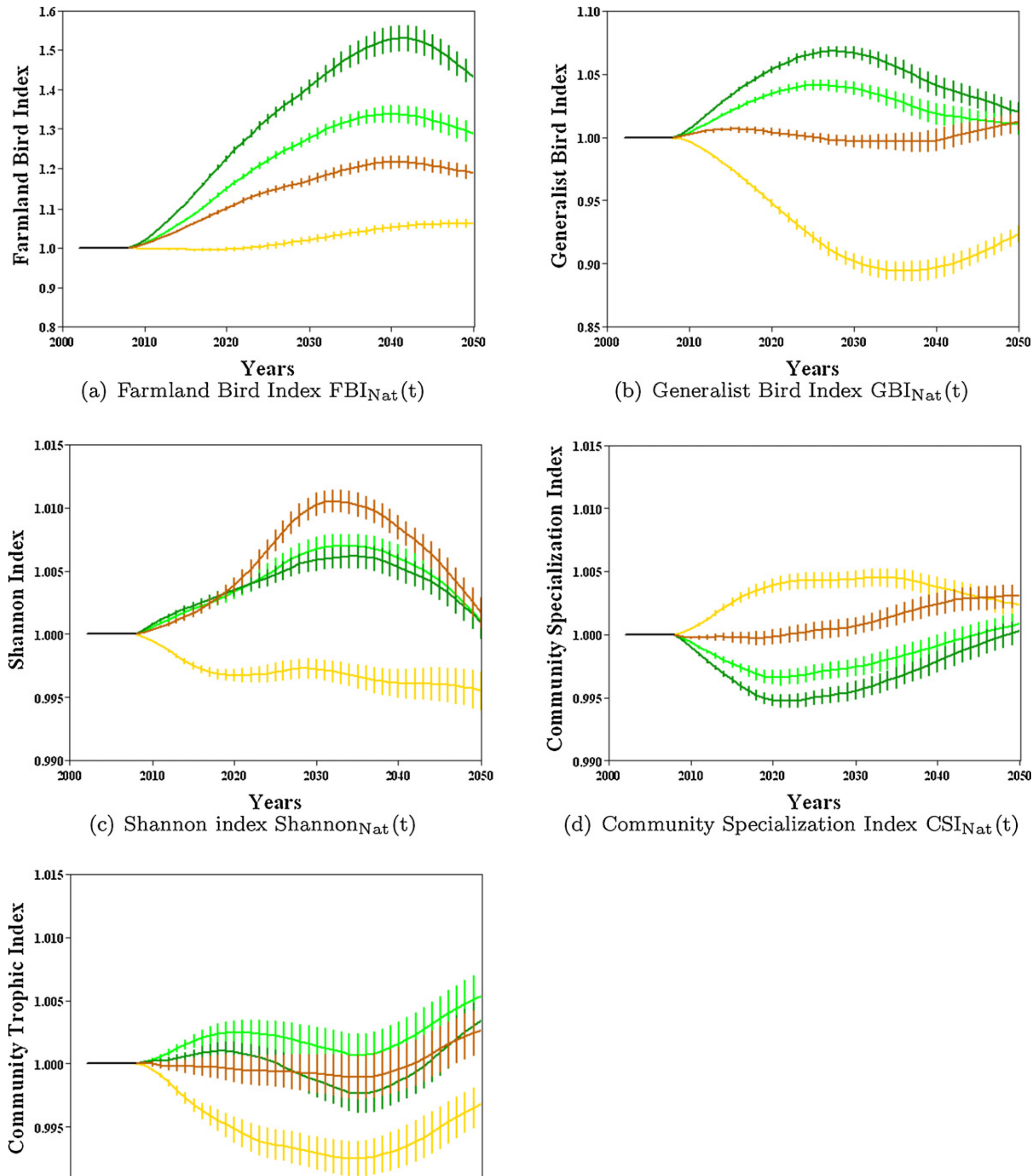

Fig. 4. Ecological indicators evolutions (mean outcomes and 95% confident interval) with the crop (yellow), grassland (light-green), high quality environmental (dark-green), and Double Subsidy (brown) scenarios, normalized by the statu quo evolutions. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

communis, favoring more mixed farmed landscapes, is less affected by variations between scenarios.

3.5. Community trophic level

As observed for the Shannon index, the CR scenario is the only one clearly discriminated compared to the other three (Fig. 4(e)): the average trophic level of the bird community is lower for the CR scenario than the current trend and the other scenarios. Differences between the three scenarios promoting grasslands are not obvious though the GL scenario seems to display slightly better trophic levels than the DS and HQE scenarios. Fig. 7 compares responses of two species with similar habitat specialization levels but contrasted trophic indices. Scenarios promoting grasslands lead to communities with larger proportions of Kestrel *Falco tinnunculus* (a bird of prey of high trophic level) and a smaller proportion of Linnet *Carduelis cannabina* (a granivorous passerine of lower trophic level).

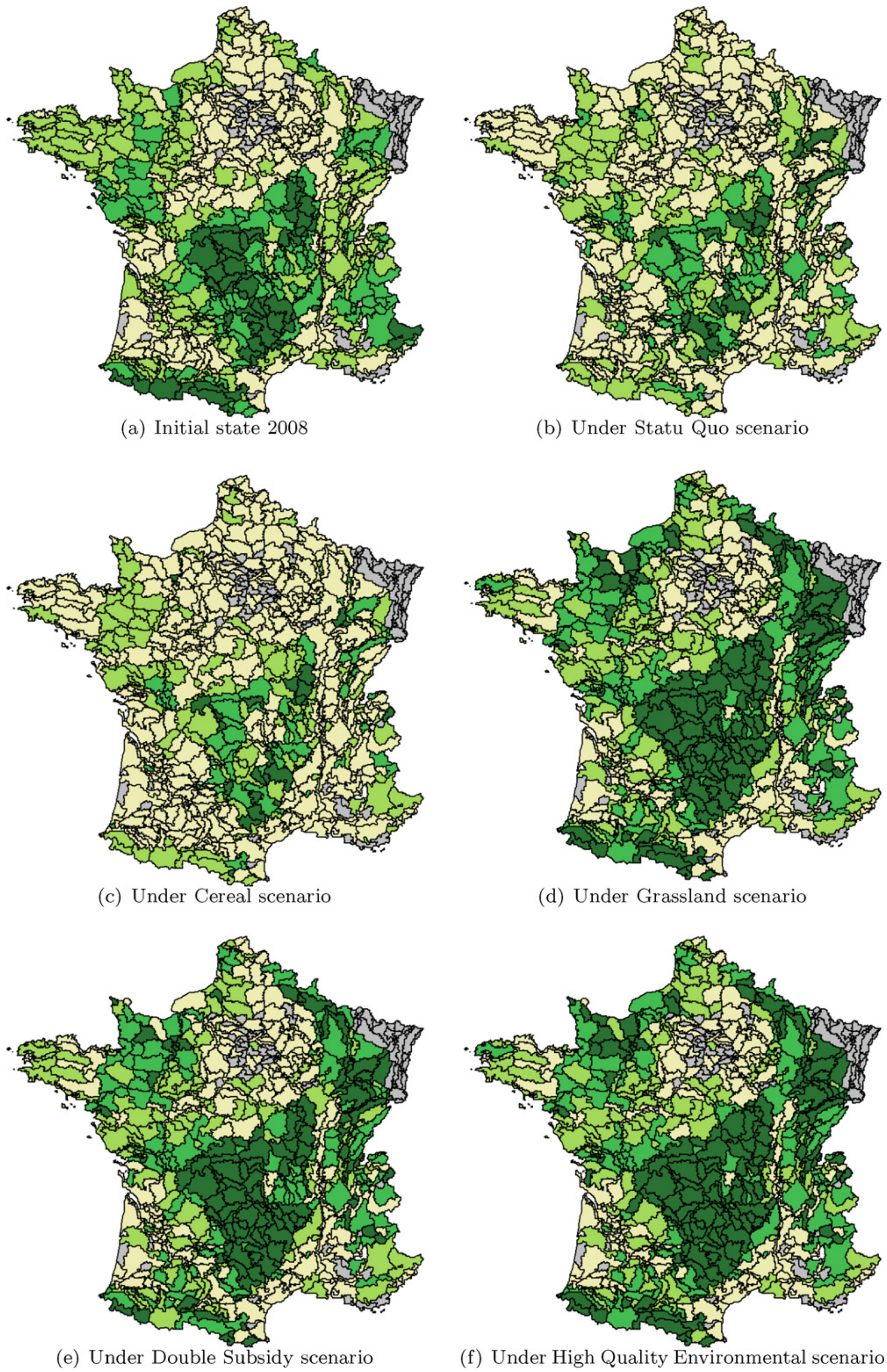

Fig. 5. Proportions of extensive grassland activities at the initial state (2008) and at 2030 under the five scenarios (yellow (resp. yellow–green, green, dark-green) when the ratio is between 0 and 0.1 (resp.0.1 and 0.35, 0.35 and 0.7, 0.7 and 1)). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

3.6. Indicators comparison

In order to compare the indicators, Fig. 8 synthesizes the different ecological performances among the scenarios at year 2040. Both population indicators FBI and GBI rank scenarios in the same way. However, the responses of the three structure indicators, the Shannon Index, the CTI and the CSI, show considerable variation. The Shannon Index and the CTI, which have similarities, are the complete opposite of the CSI.

4. Discussion

By developing dynamic models coupling economic agricultural policies and biodiversity dynamics, we intended to evaluate the

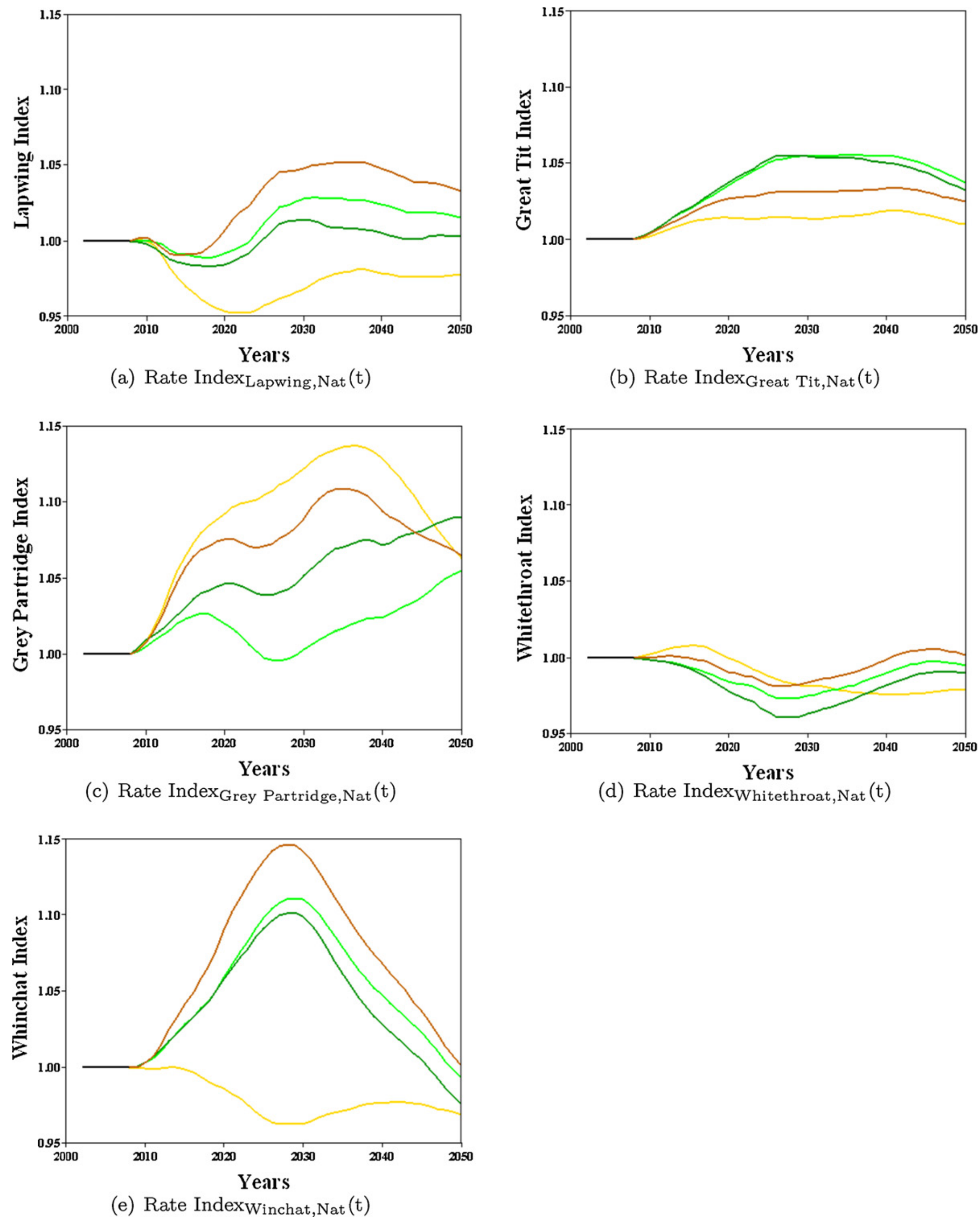

Fig. 6. Rate indices with the crop (yellow), grassland (light-green), high quality environmental (dark-green) and double subsidy (brown) scenarios normalized by the statu quo evolutions for very specialized Lapwing (SSI = 2.23, STI = 1.90) and generalist Great Tit (SSI = 0.29, STI = 1.85); specialized Grey Partridge, intermediate landscape specialized Whitethroat and Grassland specialized Winchat. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

potential impacts of incentives or taxes dedicated to crops or grasslands on various ecological indicators related to bird populations and communities. Overall, the five bird-related indicators behaved differently according to the incentive scenarios, and can be interpreted in light of their ecological meanings for bird communities (Couvet et al., 2008).

4.1. Incentives to drive ecological performance

The contrasted responses of the five indicators to the various tested economic public policies emphasize that economic incentives can be an adequate driver for bird biodiversity. The improvement of 60% in FBI with HQE scenario confirms that the

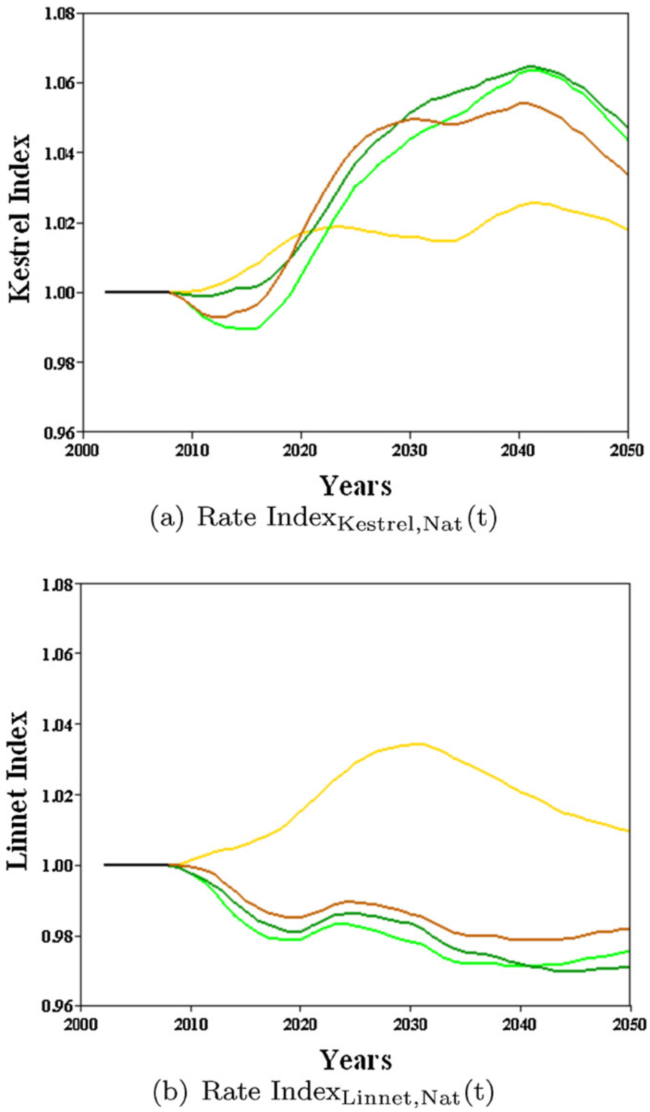

Fig. 7. Rate indices with the crop (yellow), grassland (light-green), high quality environmental (dark-green) and double subsidy brown) scenarios, normalized by the statu quo evolutions for Kestrel (SSI = 0.98, STI = 2.85) and Linnet (SSI = 0.70, STI = 1.05). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

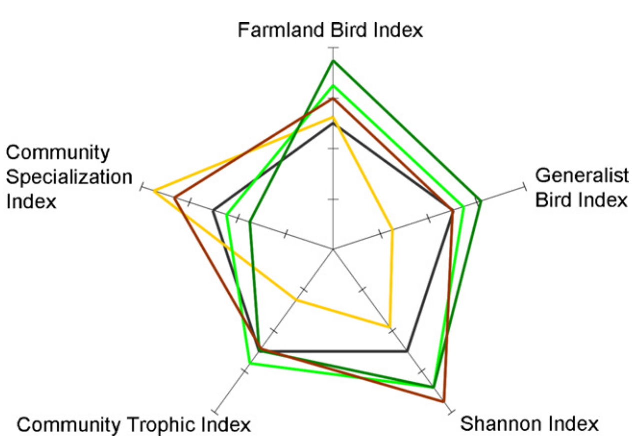

Fig. 8. Comparison of the 5 ecological indicators in 2040 with the crop scenario (yellow), grassland scenario (light-green), high quality environmental scenario (dark-green) and double subsidy scenario (brown), compared to the statu quo evolutions (represented by the black pentagon). The indicator value enhances when it gets further from the center. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

current decline of farmland biodiversity is potentially reversible (Mouysset et al., 2011), which is consistent with the recent increase of the FBI in French high nature value farmlands (Doxa et al., 2010). The impacts of public incentives on bird communities justify their use by the common agricultural policy.

However, indicators show a trend to go back to their baseline value (e.g. 1) by 2030–2040. This means that incentives become too small to influence the farmer's decisions and a return to the initial land uses occurs in the long run. This result suggests that contrary to the current CAP trend, it is important to maintain incentives to obtain sustainability for the bird community. Positive indicator evolutions for the first 20–30 years of the projections in spite of already decreasing incentives suggest that decision-makers can initially use decreasing incentives but then have to stabilize them. However, the "optimal" times differ between the indicators as well as between scenarios for a same indicator. Determining the incentive stabilization requires a specific study and will depend on ecological indicators chosen to evaluate the bird communities. However, reducing current incentives, while keeping beneficial effects on bird communities, opens many possibilities for a budget re-allocation to other environmental options. Moreover, all public policies presented here are compatible with the current decision-maker budget and in this sense are sustainable.

4.2. Contrasted populations among the scenarios

With the CR scenario, we illustrate that public policies in favor of crops, for example dedicated to bioenergy developments, could be catastrophic for bird communities. With this kind of scenario, we obtain very limited improvements of species abundances, more specialized communities (Doxa et al., 2010) and a strong decrease in the average trophic level of communities, with more granivorous species such as the partridges and the linnet. In contrast, promoting extensive grasslands appears beneficial for breeding birds when compared to the current situation. Communities are larger in size, are more diversified (with farmland and generalist species), and of higher average trophic level (with granivorous, insectivorous and carnivorous species, a more complete and balanced food chain). Species with higher positions in the food chain, such as the Red-backed Shrike, the Buzzard and the Krestel, are more abundant when extensive grasslands are economically promoted. Larger communities composed of more diverse species and spanning the complete food chain, certainly present advantages in terms of ecological services and sustainability. The sensitivity of such communities to disturbances is lower and the sustainability of the whole community improves (Ives and Carpenter, 2007; MacCann, 2000). This kind of community has a strong interest for agricultural activities through ecosystem services. Large and diverse communities are more resilient to global changes (Keesing et al., 2010), and provide more diversified and sustainable ecosystem services, such as pest control, pollination and decomposition processes (Altieri, 1999; Schlapfer et al., 1999; Tilman et al., 2002; Widly and Thomas, 2002).

However, none of the three scenarios promoting extensive grasslands (GL, DS and HQE) are the most effective for all studied ecological dimensions. The DS scenario is particularly interesting for the balance between the different species as well as for the stability of the community. The GL scenario performs better at enhancing the average trophic level of bird communities, potentially maximising the associated provision of ecosystem services. Finally, the HQE scenario induces the largest population increases, ensuring a larger bird biomass but less specialized community. Promoting extensive grasslands appears essential for the management of the bird communities, and of agriculture, even if it induces more complex economic effects. Adding other incentives to subsidies for extensive grasslands seems to favor some functional features of bird communities. Before selecting a policy scenario, the decision-maker has to prioritize the biodiversity metrics to be targeted.

4.3. Indicator relevance

In order to describe the bird communities, we developed two kinds of indicators: population size indicators (FBI and GBI), which aggregate annual indices of species population trends at the national scale, and community structure indicators (Shannon index, CSI and CTI) averaging local diversity/functional indices. These two kinds of indicators are complementary and combining them globally informs on the community dynamics. Concerning population sizes, FBI and GBI behave similarly but with a higher sensibility for the FBI. FBI therefore appears more relevant to discriminate scenarios if considering population size indices (Mouysset et al., 2011). Concerning community indicators, the Shannon diversity, CSI and CTI discriminate scenarios differently. CSI and CTI are built up from functional traits of species and are thus more informative about the community functioning than Shannon diversity. Moreover, the relevance of CSI and CTI depends on the agricultural farming system. CSI discriminates very well communities in mixed farmland and grassland landscapes, where more diverse communities will have higher CSI. However, in open field landscapes, e.g. in more intensive cropping systems, poorly diverse communities also have a high CSI. Indeed, such communities contain few species and those they have are almost all crop-specialists along with very low numbers of habitat generalists. In this context, CTI turns out to be a better functional indicator as it classifies open field communities within those of low trophic level. However, in grassland landscapes, CTI cannot clearly rank communities. A sustainable landscape will lead to both high CSI and high CTI for bird communities. Combining FBI, CSI and CTI makes it possible to describe communities with relevance and thus specify sustainable scenarios. We advocate their use as an appropriate support for policy decisions and adaptive management of farmland biodiversity.

4.4. Bio-economic model for decision support

We here argue that several characteristics of the proposed bio-economic model make it relevant for decision support in agriculture and biodiversity management. First, in complement to studies focusing on farmland use scenarios (Albrecht et al., 2007; Batary et al., 2007; Kleijn and van Zuijlen, 2004; Münier et al., 2004; Taylor and Morecroft, 2009), the bio-economic approach relies on economic scenarios, financial incentives and policies, which constitute major inputs taken into account by stakeholders in reality. Similarly, the compatibility of the policies with the current CAP budget also gives realism to such a study. Moreover, the national scale leads us to think more about global policies and general directions than about local management, the effectiveness of which is controversial (Le Roux et al., 2008). Second, another important characteristic of this bio-economic modelling regarding decision support is its the prospective dimension. With the description of different future scenarios beyond the statu quo scenario, this work is complementary with the numerous studies which focus on the impact of current policies Taylor and Morecroft, 2009; Vickery et al., 2004. These simulations are useful for public policies in agriculture and biodiversity conservation where experimental schemes are complicated to establish. In particular, our approach stresses some non-linearities between incentives and ecological indicators according to the scenarios. If outcomes with the DS scenario are often intermediate between those issued from the CR scenario and the GL/HQE scenarios, such is not the case with the Shannon index where the DS scenario leads to an extreme trajectory. As previously mentioned, synergies between incentives do not affect bird communities similarly, depending on the same functional traits considered, and can represent an interesting leverage to enhance the effectiveness of agricultural policies on different criteria. Finally focusing on one general taxon (birds) rather than on one or two emblematic species makes it possible to adopt a broad viewpoint for biodiversity and potential associated ecosystem services. Consequently, the genericity of the results is reinforced as regard biodiversity management and agri-environmental policies.

4.5. Perspectives

The present work advocates the use of bio-economic models for public farming policies and terrestrial biodiversity conservation. A major trend of the French CAP which has not been tested in this study is the regionalisation of incentives. With the national scale decomposed into economic-ecological homogeneous areas (PRA), our conceptual framework could be a fruitful instrument to test several policies of incentive regionalisation in consistency with the national budget.

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References

Albrecht, M., Duelli, P., Muller, C., Kleijn, D., Schmid, B., 2007. The Swiss agri-environmental scheme enhances pollinator divrsity and plant reproductiove success in nearby intensively managed farmland. J. Appl. Ecol. 44, 813–822.

Altieri, M.A., 1999. The ecological role of biodiversity in agroecosystems. Agric. Ecosyst. Environ. 74, 19–31.

Balmford, A., Bennun, L., Brink, B.ten., Cooper, D., Cote, I.M., Crane, P., Dobson, A., Dudley, N., Dutton, I., Green, R.E., Gregory, R.D., Harrison, J., Kennedy, E.T., Kremen, C., Leader-Williams, N., Lovejoy, T.E., Mace, G., May, R., Mayaux, P., Morling, P., Philips, J., Redford, K., Ricketts, T.H., Rodriguez, J.P., Sanjayan, M., Schei, P.J., Jaarsveld, A.S.van., Walther, B.A., 2005. The convention on biological diversity's 2010 target. Science 307, 212–213.

Barbault, R., Chevassus-au-Louis, B., 2004. Biodiversity and Global Change. Ministère des Affaires étrangères, ADPF, p. 237.

Batary, P., Baldi, A., Erdos, S., 2007. Grassland versus non-grassland bird abundance and diversity in managed grasslands: local, landscape and regional scale effects. Biodivers. Conserv. 16, 871–881.

Benton, T.G., Vickery, J.A., Wilson, J.D., 2003. Farmland biodiversity: is habitat heterogeneity the key? Trends Ecol. E 18, 182–188.

Berge ten, H.F.M., Ittersum van, M.K., Rossing, W.A.H., Ven van de, G.W.J., Schans, J., Sanden van de, P.A.C.M., 2000. Farming option for the Netherlands explored by multi-objective modelling. Eur. J. Agron. 13, 263–277.

Beverton, R.J.H., Holt, S.J., 1957. On the Dynamics of Exploited Fish Populations, Fishery Investigations, II. Ministry of Agriculture, Fisheries and Food.

Butchart, S.H.M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J.P.W., et al., 2010. Global biodiversity: indicators of recent declines. Science 328, 1164–1168.

Butler, S.J., Vickery, J.A., Norris, K., 2007. Farmland biodiversity and the footprint of agriculture. Science 315, 381–384.

BWPI 2006, 2006. Birds of the Western Palearctic interactive. BirdGuides Ltd., UK.

Chamberlain, D.E., Fuller, R.J., Bunges, R.G.H., Duckworth, J.C., Shrubb, M., 2000. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. J. Appl. Ecol. 37, 771–788.

Couvet, D., Jiguet, F., Julliard, R., Levrel, H., Teyssèdre, A., 2008. Enhancing citizen contributions to biodiversity science and public policy. Interdiscip. Sci. Rev. 33, 95–103.

Devictor, V., Julliard, R., Clavel, J., Jiguet, F., Lee, A., Couvet, D., 2008. Functional biotic homogenization of bird communities in disturbed landscapes. Glob. Ecol. Biogeogr. 17, 252–261.

Donald, P.F., Sanderson, F.J., Burfield, I.J., van Bommel, F.P.J., 2006. Further evidence of continent-wide impacts of agricultural intensification on European farmland birds, 1990–2000. Agric. Ecosyst. Environ. 116, 189–196.

Donald, P.F., Green, R.E., Health, M.F., 2001. Agricultural intensification and the collapse of Europe's farmland bird populations. Proc. R. Soc. Lond. B: Biol. Sci. 268, 25–29.

Doxa, A., Bas, Y., Parracchini, M.L., Pointereau, P., Terres, J-M., Jiguet, F., 2010. Low-intensity agriculture increases farmland bird abundances in France. J. Appl. Ecol., doi:10.1111/j.1365–2664.2010.01869.x.
Flowerdew, J.R., Kirkwood, R.C.,1997. Mammal biodiversity in agriculture habitats, Biodiversity and Conservation in Agriculture. In: BCPC Symposium Proceedings, vol. 69. BCPC Publications, Alton, pp. 25–40.
Gregory, R.D., Willis, S.G., Jiguet, F., Vorisek, P., Pazderová, A., van Strien, A., Huntley, B., Collingham, Y.C., Couvet, D., Green, R.E., 2009. An indicator of the impact of climate change on European bird populations. PloS ONE 4, doi:10.1371/journal.pone.0004678.
Hughey, K.F.D., Cullen, R., Moran, E., 2003. Integrating economics into priority setting and evaluation in conservation management. Conserv. Biol. 17, 93–103.
Ives, A.R., Carpenter, S.R., 2007. Stability and diversity in ecosystems. Science 317, 58–62.
Jackson, L., Bawa, K., Pascual, U., Perrings, C., 2005. Agrobiodiversity: a new science agenda for biodiversity in support of sustainable agroecosystems. DIVERSITAS Report 4, p. 40.
Jiguet, F., 2009. Method-learning caused first-time observer effect in a newly-started breeding bird survey. Bird Study 56, 253–258, doi:10.1080/00063650902791991.
Jiguet, F., Devictor, V., Ottvall, R., van Turnhout, C., van der Jeugd, H., Lindström, A., 2010. Bird population trends are linearly affected by climate change along species thermal ranges. Proc. R. Soc. Lond. B: Biol. Sci., doi:10.1098/rspb.2010.0796.
Julliard, R., Jiguet, F., Couvet, D., 2004. Common bird facing global changes: what makes a species at risk? Glob. Change Biol. 10, 148–154.
Julliard, R., Clavel, J., Devictor, V., Jiguet, F., Couvet, D., 2006. Spatial segregation of specialists and generalists in bird communities. Ecol. Lett. 9, 1237–1244.
Keesing, F., Belden, L.K., Daszak, P., Dobson, A., Harvell, C.D., Holt, R.D., Hudson, P., Jolles, K.E., Jones, K.E., Mitchell, C.E., Myers, S.S., Bogich, T., Ostfeld, R.S., 2010. Impacts of biodiversity on the emergence and transmission of infectious diseases. Nature 468, 647–652, doi:10.1038/nature09575.
Kleijn, D., van Zuijlen, G.J.C., 2004. The conservation effects of meadow bird agreements on farmland in Zeelan, The Netherlands, in the period 1989–1995. Biol. Conserv. 117, 443–451.
Kleijn, R.A., Baquero, Y., Clough, M., Daz, J., De Esteban, F., Fernndez, D., Gabriel, F., Herzog, A., Holzschuh, R., Jhl, E., Knop, A., Kruess, E.J.P., Marshall, I., Steffan-Dewenter, T., Tscharntke, J., Verhulst, T.M., West, J., Yela, L., 2006. Mixed biodiversity benefits of agri-environment schemes in five European countries. Ecol. Lett. 9, 243–254.
Krebs, J.R., Wilson, J.D., Bradbury, R.B., Siriwardena, G.M., 1999. The second silent spring? Nature 400, 611–612.
Le Roux, X., Barbault, R., Baudry, J., Burel, F., Doussan, I., Garnier, E., Herzog, F., Lavorel, S., Lifran, R., Roger-Estrade, J., Sarthou, J.P., Trometter, M. (éd) 2008. Agriculture et biodiversité, valoriser les synergies. Expertise scientifique collective, synthèse du rapport, INRA, France.
Lien, G., 2002. Non-parametric estimation of decision makers' risk aversion. Agric. Econ. 27, 75–83.
MacCann, K.S., 2000. The diversity–stability debate. Nature 405, 228–233.
Millenium Ecosystem Assessment, 2005. Cultivated systems. In: Ecosystems and Human Well Being: Current State and Trends. Island Press, Washington, pp. 745–794 (Chapter 26).
Münier, B., Birr-Pedersen, K., Schou, J.S., 2004. Combined ecological and economic modelling in agricultural land use scenarios. Ecol. Model. 174, 5–18.
Mouysset, L., Doyen, L., Jiguet, F., Allaire, G., Leger, F., 2011. Bio economic modeling for a sustainable management of biodiversity in agricultural lands. Ecol. Econ. 70, 617–626.
Ormerod, S.J., Watkinson, A.R., 2000. Editor' s introduction: birds and agriculture. J. Appl. Ecol. 37, 699–705.
Pacini, C., Wossink, A., Giesen, G., Huirne, R., 2004. Ecological-economic modelling to support multi-objective olicy making: a farming systems approach implemented for Tuscany. Agric. Ecosyst. Environ. 102, 349–364.
Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., Torres, F., 1998. Fishing down marine food webs. Science 6, 860–862.
Perrings, C.P., Jackson, L., Bawa, K.S., Brussaard, L., Brush, S., Gavin, T., Papa, R., Pascual, U., de Ruttier, P., 2006. Biodiversity in agricultural landscapes: saving natural capital without losing interest. Conserv. Biol. 20, 263–264.
Schlapfer, F., Schmid, B., Seidl, I., 1999. Expert estmitates about effects of biodiversity on ecosystem preocesses and services. Oikos 84, 346–352.
Scholefield, P., Firbank, L., Simon, B., Norris, K., Jones, L.M., Petit, S., 2009. Modelling the European farmland bird indicator in response to forecast land-use change in Europe. Ecol. Indic., doi:10.1016/j.ecolind.2009.09.008.
Sekercioglu, C., Daily, G.C., Ehrlich, P., 2004. Ecosystem consequences of bird declines. Proc. Natl. Acad. Sci. U.S.A. 101, 18042–18047.
Sotherton, N.W., Self, M.J., 2000. Changes in plant and arthropod biodiversity on lowland farmland: an overview. Ecology and Conservation of Lowland Farmland Birds. In: Aebischer, N.J., Evans, A.D., Grice, P.V., Vickery, J.A. (Eds.), Proceedings of the 1999 British Ornithologist' Union Spring Conference. BOU, Tring, pp. 26–35.
Taylor, M.E., Morecroft, M.D., 2009. Effects of agri-environmental schemes in a long-term ecological time series. Agric. Ecosyst. Environ. 130, 9–15.
Tilman, D., Cassman, K.G., Matson, P.A., Naylor, R., Polasky, S., 2002. Agricultural sustainability and intensive production practices. Nature 418, 671–677.
Vickery, J.A., Bradbury, R.B., Henderson, I.G., Eaton, M.A., Grice, P.V., 2004. The role of agri-environment schemes and farm management practices in reversing the decline of farmland birds in England. Biol. Conserv. 119, 19–39.
Widly, A., Thomas, M.B., 2002. Natural enemy diversity and pest control: patterns of pest emergence with agricultural intensification. Ecol. Lett. 5, 353–360.