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Climate-dependent scenarios of land use for biodiversity and ecosystem services in the New Aquitaine region

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Abstract The synergies and trade-offs between human well-being, biodiversity, and ecosystem services are under debate for the design of more sustainable public policies. In that perspective, there is a need of quantitative methods to compare all these outcomes under alternative policy scenarios. The present paper provides scenarios at the horizon 2053 for the New-Aquitaine region in France. They rely on spatio-temporal models derived from individual land-use choices under climate change. The models are estimated at the national level from 1993-2003 fine-scale data. We focus on farming, forestry, and urban land-uses along with bird biodiversity scores and a basket of ecosystem services namely carbon sink intensity, forest recreation, and water pollution. A 'climate-economic adaptation' scenario shows that climate-induced land-use worsens the negative effects of climate change on biodiversity and several ecosystem services in the long run as compared to a 'status quo' scenario. Another scenario with an incentive policy, based on a flat payment for pastures, slightly mitigates these negative impacts on biodiversity and water pollution. However, this turns out to be detrimental for other ecosystem services. This result confirms that the design of sustainable policies can not be limited to uniform strategies and should account for the complexity of ecosystem management.

Keywords Model-based scenarios · bio-economics · climate · land-use · incentive policy · birds biodiversity · ecosystem services

1 Introduction

Balancing biodiversity conservation with food security and the preservation of a broader set of ecosystem services (ES), in a context of global change, is among the greatest challenges of the century (Godfray et al. 2010). Climate and land-use changes are the main drivers of past and future variations in terrestrial biodiversity and ecosystems (MEA 2005; Pereira et al. 2010; Willis and MacDonald 2011; Leclère et al. 2020). For medium-term analyses (ca. 50 yrs), these two drivers need to be treated differently

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in terms of scenarios and sustainable management policies in particular at regional scale. Global warming can indeed be considered as exogenous since climate is very inertial, most of the climate change over this period is already committed and depends on global greenhouse gas emissions scenarios. By contrast, Land Use Changes (LUCs) are operationalized by more local stakeholders, in particular landowners responding to changing economic incentives, and therefore can be seen as more directly observable and controllable for public policies targeting biodiversity and ecosystems. For instance, at the European scale, LUC depends both on national or supra-national strategies such as the Common Agricultural Policy (CAP) and on regional (or even more local) authorities which can indeed organise land planning and regulate human activities in order to preserve biodiversity through natural area protection, greens corridors, conservation and valorisation of species or habitats[1].

However, some of these present and future LUC are likely to be influenced by climate change. Local opportunities and constraints indeed appear with climate changes as humans adapt their use of land resources in particular with respect to provisioning services underlying farming and forestry. Thus there are already signs of negative impacts of recent climate warming on corn and wheat yields (Brisson et al. 2010; Lobell et al. 2011). In the same vein, models foresee that future climate change will result in projected northward shifts of maize area in the United States, or rice area in China (Tubiello et al. 2002; Xiong et al. 2009). Consequently, an efficient conservation or climate-economic adaptation policy has to rely both on the direct climate effects on ecosystems and the indirect effects induced by human adaptations, strategies and public policies on habitats and ecosystems (Hannah et al. 2002; Berrang-Ford et al. 2011).

Moreover, changes in land use pattern affect not only biodiversity but also ES (Bennett et al. 2009; Bullock et al. 2011; Raudsepp-Hearne et al. 2010) including not only provisioning services related to farming and forestry but also cultural and regulating services. LUC can enhance the value of one ES at the expense or for the benefit of others (Bateman et al. 2013; Leclère et al. 2020). Consequently, trade-offs as well as synergies between ES may occur which complexifies the design of sustainable land-use policies balancing these different outcomes under feasibility constraints including the compatibility with private choices.

Regarding terrestrial biodiversity, ES and land-use, the Nouvelle Aquitaine (NA) region, located in the South-West of France, represents a challenging and stimulating case study (Bretagnolle 2020). The region indeed encompasses several major productive ecosystems including crops, grassland, vineyards, forests, as well as major urban land-use in particular with the city of Bordeaux and its surrounding areas. These contrasting and interacting socio-ecological systems provide a large set of commodities and ecosystem services among which important provisioning services of high economic values such as food, timber, and wine production. They also cover different degrees of anthropogenic pressures, from ‘natural’ ecosystems with minor human impacts to intensively managed agricultural landscapes or urban areas. The NA Region is also a hotspot of bird biodiversity with numerous wetlands including the Gironde Estuary, the Arcachon Basin and the farmland birds of intensive cereal systems. All these

[1] Typical instances of strategic, prescriptive and integrating plans at regional scale in France are SRADDET (Schémas régionaux d’aménagement, de développement durable et d’égalité des territoires) https://www.ecologie.gouv.fr/sraddet-schema-strategique-prescriptif-et-integrateur-regions

ecosystems in the region are facing important threats caused by global changes, which raises various concerns about their sustainability and stresses the need to identify viable management and scenarios.

In that perspective, this paper downscales and refines the outputs of the national integrated model from Ay et al. (2014) to the NA region. The modelling framework articulates four compartments: LUC models from micro-data, econometric Ricardian models about the effects of climate on economic returns from land, Species Distribution Models (SDMs, Peterson et al. (2011)) about common birds and assessment of ESs from land-use. This ecological-economic framework allows us to explore the interplay between climate change, LUCs, biodiversity and ESs through model-based scenarios at the horizon 2053. Such ecological-economic models and scenarios track many of the guidelines listed in IPBES (2016); Doyen (2018) in particular by accounting for complex dynamics and multi-criteria analysis. More specifically, we develop and compare scenarios of LUC, under varying economic returns from land consecutively to climate or policy inputs. We choose to put LUC at the core of the modeling approach because it is the part of the system that is the most under control locally. Our framework also provides an explicit modeling on the consequences of LUCs and climate on biodiversity and of LUCs on ESs. We draw on regionalized data from Météo-France for climate (Déqué 2007a), TERUTI survey[2], land prices from the French Ministry of Agriculture and French Breeding Bird Survey (FBBS) (Jiguet et al. 2012).

This allows us to address three main questions:

(i) What is the likely effect of climate change on bird biodiversity and ESs ?
(ii) Does climate-induced LUC mitigate or amplify the raw effect of climate?
(iii) What is the contribution of a 'conservation' payment for pastures?

Consequently three scenarios are compared. The first scenario named 'Status Quo Scenario' and denoted by SQS assumes that climate affects birds dynamics but not LUC. The second scenario called 'Climate-Economic Adaptation Scenario' and denoted by CEAS integrates climate-induced LUC and a feedback of climate change on economic decisions. A third 'Biodiversity Conservation' scenario (BCS) also accounts for direct climate effect on birds and LUC, but differs from CEAS by integrating an incentive policy through a uniform payment for pastures which modifies micro-economic decisions and LUC. The focus on pastures underpinning scenario BCS and question (iii) stems from the conjecture that biodiversity (in particular birds) and ecosystems as a whole would benefit from the greening of land-use through grasslands (Mouysset et al. 2012; Bateman et al. 2013; Ay et al. 2014). To assess and compare the performances of these scenarios, we rely on both biodiversity and ESs indicators. Regarding biodiversity metrics, we consider a global bird abundance index, several bird habitat scores (farmland, forest, urban and generalist indexes), the Shannon diversity, as well as the community trophic index. In terms of ecosystem services, we here focus on carbon sink intensity, forest recreation, and water quality.

Beyond the methodological interest of the proposed ecological-economic modeling framework, the main contribution of the paper is threefold. Firstly, we find a negative effect of climate change on bird biodiversity at 2053 in line with the regional (Bretagnolle 2020), national (Ay et al. 2014) and international evidence (Gregory et al. 2009; Leclère et al. 2020). This effect is strongly related to the effect of projected LUC and a greater elevation shift of birds instead of a northern shift as expected.

[2] https://www.casd.eu/en/source/land-use-terruti-lucas/

Secondly, we find that climate-dependent LUC amplifies the negative direct effect of climate change on birds and several ESs. Thirdly, we highlight that, although a spatially-uniform policy to promote pastures can counteract the negative effect of climate change on biodiversity, such greening scenario turns out to be detrimental for some ESs. Thus the design of sustainable policy for land-use can not be limited to land-specialized strategies. In other words, a single policy instrument is not sufficient to achieve multiple objectives underlying sustainability and synergies between land use, ESs and biodiversity.

The paper is structured as follows: Section 2 details the case study, the model, the scenarios as well as the criteria used; Section 3 presents the results, including the scenario trajectories and spatial patterns together with a multi-criteria comparison of the scenarios. Finally, Section 4 contains a discussion of the results and concludes. An 'Online' Appendix (OA) details some methods, results and outcomes.

2 The ecological-economic model and scenarios

This section describes the ecological-economic and spatio-temporal model, the three contrasted scenarios as well as the different criteria relating to biodiversity and ESs. We start with a brief description of the regional case study. More details about data and estimation methods are reported in Section A of the Online Appendix (OA).

2.1 Case study: the New Aquitaine region

This subsection informs on the NA region that constitutes the case study of our paper. The NA region, located in the South-West of France, is the largest region in France by area as illustrated by Figure 1. NA economy is mainly based on agriculture and viticulture (vineyards of Bordeaux and Cognac), tourism and aerospace industry. Its largest city, Bordeaux, together with its suburbs and satellite cities, forms the seventh-largest metropolitan area of France, with 850,000 inhabitants. The growth of its population, particularly marked on the coast, shows that NA is a very attractive area in France.

NA also constitutes an interesting case study in terms of land-use, biodiversity and ESs as emphasized in Bretagnolle (2020) and is representative of what is observed in metropolitan France as a whole. Agriculture areas, including mainly permanent crops, arable land, and grasslands, indeed occupy 60% of the region while the share of forest is 2 points higher than the national share, due mainly to the presence of the Landes forest (988,000 ha). The share (4.2%) of artificialized territories include urban areas, industrial or commercial areas, communication networks and non-agricultural artificial green spaces. Beyond its state, the dynamics of NA land-use is also informative and representative. Between 2006 and 2012, the surface area of artificial territories increased by 12% while agricultural land and forests and natural environments shrank by 0.5%. Agricultural land is shrinking mainly in Gironde and Charente-Maritime departments (subregions) when it is progressing in Les Landes department. The decrease in forests and natural ecosystems (non artificial territories) is significant in the Landes.

The contrasting and interacting socio-ecological systems of NA provide a large set of commodities and ESs among which important provisioning services of high economic values such as food, timber, and wine production. The recreational services induced

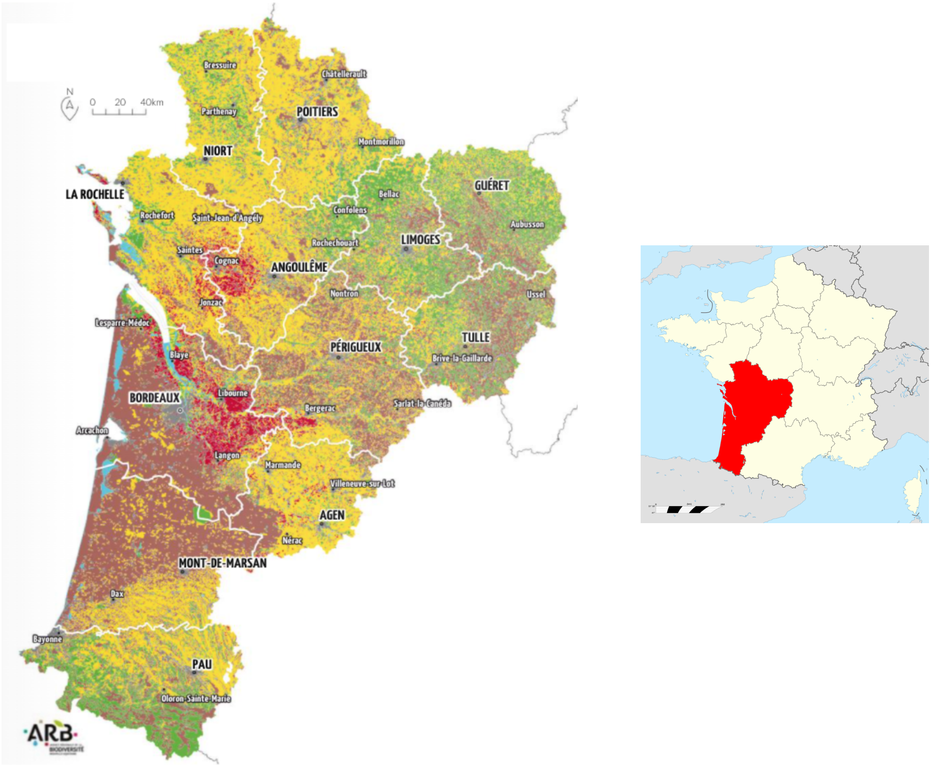


Fig. 1: Maps of the NA region: (left) Land-use 2018 (together with the administrative delineations of departments and the main cities); in red, vineyards; in yellow, croplands; in brown, forests; in green, grasslands; and in grey, urban areas; (Source : © Agence Régionale Biodiversité (ARB) NA - Agence Européenne de l'Environnement - BD Corine Land Cover 2018) ; (right) the NA region (red) as compared to other regions in metropolitan France.

by NA ecosystems are also important as illustrated by the important tourism activity which relates to both the coastal area and more continental areas such as the Dordogne department. The Landes forest also plays a major role in terms of Carbon sequestration as it is the biggest artificial forest in the whole Western Europe. The numerous wetlands including the Gironde Estuary, the Arcachon Basin as well as Poitevin marshes are hotspots of biodiversity for their flora or fauna and in particular birds. NA is hence a very interesting region for ornithology with a favorable environment for both sedentary or migratory birds. The identification of viable management, scenarios and policies balancing the economic development, biodiversity and ecosystem services conservation facing global changes is thus a key issue for the NA region.

2.2 Econometric Model of Land Use Change

The model of land-use assumes that in every location q at a given period t, land use $\mathbf{h}(t,q)$ is decomposed into L mutually exclusive classes as follows:

$$\mathbf{h}(t,q) = (h_1(t,q), h_2(t,q), \ldots, h_L(t,q))\,. \tag{1}$$

For the case study of NA, we focus on five categories namely annual crops, perennial crops (including vineyards), pastures, forests and urban areas (Section A.1 in OA). In each plot q, representative landowners[3] are assumed to favor the land use l that gives the best utility and these choices are independent for each plot. For a given land use $\ell = 1,\ldots,L$ on a given plot q at a given period t, the utility derived from land use is the sum between a deterministic and a random part such that:

$$U_\ell(t,q) = \overline{U}_\ell(t,q) + \epsilon_\ell(t,q), \tag{2}$$

with

$$\begin{aligned}\overline{U}_\ell(t,q) = \;& \alpha_\ell + \mathbf{h}(t-1,q)\boldsymbol{\eta}_\ell \\ +\;& \mathbf{r}(t,q)\boldsymbol{\beta}_{1\ell} + \mathbf{c}(t,q)\boldsymbol{\beta}_{2\ell} + \mathbf{e}(q)\boldsymbol{\beta}_{3\ell} + \mathbf{r}(t,q)\big[\mathbf{c}(t,q) + \mathbf{e}(q)\big]\boldsymbol{\beta}_{4\ell}.\end{aligned} \tag{3}$$

The deterministic part $\overline{U}_\ell(t,q)$ is parametrized from variables about previous land uses $\mathbf{h}(t-1,q)$, net returns $\mathbf{r}(t,q)$, climate variables $\mathbf{c}(t,q)$, biophysical variables $\mathbf{e}(q)$ through six vectors of unknown coefficients to estimate $[\alpha_\ell; \boldsymbol{\eta}_\ell; \boldsymbol{\beta}_{1\ell}; \boldsymbol{\beta}_{2\ell}; \boldsymbol{\beta}_{3\ell}; \boldsymbol{\beta}_{4\ell}]$. We gather up to 26 climatic and biophysical variables in order to model finely the determinants of LUCs and the other outcomes variables described below (i.e., economic returns and bird distribution). These variables are highly correlated between them, which leads to multi-colinearity issues in the statistical estimation of the models. Hence, we perform two Principal Component Analysis in order to reduce the dimension of the raw variables by keeping only the two first axis for each set of variables (Figure 8 in OA shows the relationships between the climate variables and these two principal axes, which account for 87% of the total initial variance). Lagged land uses $\mathbf{h}(t-1,q)$ are included to allow for conversion costs. An interaction between net returns and other exogenous variables (climate, elevation, slope and land quality) is included to account for different spatial resolutions of the data (section A.2 in OA). The interaction means that expected economic returns, climate, biophysical variables could have heterogeneous effects on the utility, depending on the land use.

McFadden (1974) identifies three features of the random part for deriving a multinomial logit model from this framework: independence, homoscedasticity and extreme value distribution (i.e., Gumbel). Assuming these features are met, one can show that the probabilities of the different land-use ℓ in every location q at any time t have simple closed forms, which correspond to the logit transformation of the deterministic part of the utility. Thus the probability that a plot q is in use ℓ at the period t is:

$$p_\ell(t,q) = \frac{\exp\left(\overline{U}_\ell(t,q)\right)}{\sum_k \exp\left(\overline{U}_k(t,q)\right)}, \text{ for } \ell = 1,\ldots,L. \tag{4}$$

This model is estimated on observed land use data from the TERUTI survey (France, 1993–2003), which have already been used for econometric LUC models but

[3] Representative landowners or agents in every location q are potentially farmers, foresters or urban landowners depending on the landuse in each plot q. They are rather private agents.

not for the regional scale of NA at our disaggregated level (Chakir and Parent 2009; Chakir and Gallo 2009). At this stage, it is worth to mention that we prevent the appearance of a systemic change in the prevailing agro-industrial farming. This means here that we excluded so far from the set of possible land-uses the possibility of agroecology and agroecological transitions as advocated by FAO (FAO 2019a,b) or the European Commission in (Tools and applications on Agroecology). Relevant models accounting for agroecology in LUC include Padró et al. (2020).

2.3 Models of Economic Returns

According to the Ricardian framework (Mendelsohn et al. 1994), the price of land capitalizes the expected net returns from land use. Land is considered as a classical fixed asset, implying that its price $v_\ell(t)$ at time t for the use ℓ is equal to the net present value of all expected future rents for land use ℓ. Assuming flat interest rates $\tau_t = \tau$ and flat rates of capital gains $g_t = g$, this reads as follows:

$$v_\ell(t,q) = \sum_{s=1}^{\infty} \frac{r_\ell(s,q)}{(1+\tau)^s} = \frac{r_\ell(t,q)}{(\tau - g)}, \tag{5}$$

because $r_\ell(s,q) = r_\ell(t,q) \cdot (1+g)^s$. Thus, by reversing (5), the expected return $r_\ell(t) = (\tau - g) \cdot v_\ell(t)$ of a land ℓ at time t can be calculated on the basis of its current price knowing the interest rate and the rate of capital gains $(\tau - g)$. This result depends on the assumption of well-functioning markets (i.e., competitive and balanced) and so has to be considered as a theoretically-consistent first approximation.

We use a Ricardian equation to model the effect of climate change on land prices $v_\ell(t,q)$ or, equivalently, on the expected net returns $r_\ell(t,q)$ of annual crop, pasture, perennial crop and forest. The Ricardian equation relates the economic returns of land to climate, other biophysical variables and geographical coordinates as follows:

$$\log(r_\ell(t,q)) = G_\ell\big[\mathbf{c}(t,q), \mathbf{e}(q), \mathbf{z}(q)\big] + \gamma_\ell \cdot t. \tag{6}$$

In (6), function $G_\ell(\cdot)$ is a spline-based smooth function whose endogenous structure depends on the type of land use ℓ. For the case study, these functions and the γ_ℓ are estimated on the cross-sectional variations between Small Agricultural Regions and the Terruti time series 1993–2003 from the statistical services of French Ministry of Agriculture (see Section A.3 in OA). The Ricardian equations are estimated separately for annual crop, pasture, perennial crop and forest using Generalized Additive Models (GAM, Hastie and Tibshirani 1990; Wood 2006). The smoothing functions and the penalization parameter have to be estimated jointly, with a distribution from the Gaussian family and a natural logarithm link. For the dynamics of the urban returns, we use the spatialized projections of population growth by the French demographic institute. Because these projections are available at the *département* scale[4], we have downscaled them by assuming that each municipality keeps a constant proportion of the aggregate values. Table 5 in the Appendix shows the detailed results of the calibration for the Ricardian model of economic returns.

At this stage, some shortcomings of the economic model deserve to be also mentioned as some important variables affecting the returns and hence the land-use of

[4] Département is a French administrative division ranging in size from ca. 600 to 10,550 km^2

private landowners are not so far taken into account. It includes the rising costs of fossil fuels due to peak oil scenarios, which will also affect the relative prices of synthetic fertilizers, tillage costs with tractors, and particularly the transportation costs to very long distances.

2.4 Species Distribution Models

Bird abundance and distributions are modeled with an SDM that accounts for the potential impact of climate and habitat from LUCs (Pearson and Dawson 2003). The calibration of the SDM model relies on both FBBS (Jiguet et al. 2012) and TERUTI data together with historical climate again from *Météo-France*. For a general description of the method, we note $N_s(t,q)$ the abundance of species $s \in \{1,\ldots,S\}$ at time t and location q and we assume the following relationship between the outcome and its predictors:

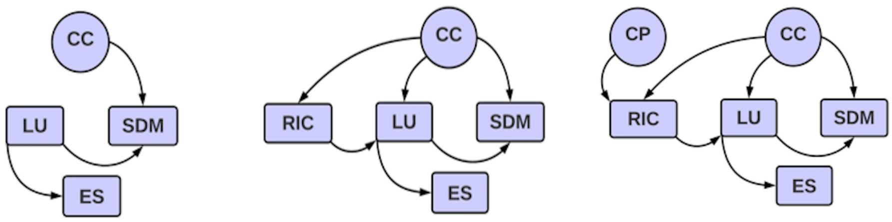
$$\log\big[N_s(t,q)\big] = F_s\big[\mathbf{c}(t,q), \mathbf{h}(t,q), \mathbf{e}(q), \mathbf{z}(q)\big] + \delta_s \cdot t. \tag{7}$$

Functions $F_s(\cdot)$ above are spline-based smoothing functions with an endogenous penalized structure common for GAM, jointly with the scalars δ_s that capture the linear growth 2003–2009 for each species s (see Section A.4 of OA). The vector $\mathbf{c}(t,q)$ again stands for climate variables (here again the two principal axes at time t and location q of a Principal Component Analysis of the climatic variables matrix to reduce colinearity problems). Figure 8 in OA shows the relationships between the climate variables and these 2 principal axes, which account for 87% of the total variance. Including $\mathbf{z}(q)$ the spatial coordinates (here the centroid of each FBBS square) in the $F_s(\cdot)$ functions allows us to separate the unobserved contextual effects (i.e., inter-species competition, spillovers from anthropogenic perturbations) from the direct topographic, climatic and habitat effects. Because bird abundances are over-dispersed positive integers, they are modeled as a distribution from the negative binomial family.

2.5 Policy scenarios

From the calibrated ecological-economic model, we can explore several scenarios that differ in the dynamics of the deterministic part of landowners' utilities described in (3) and economic returns $\mathbf{r}(t,q)$ described in (6). Regionalized climate scenarios are based on the Intergovernmental Panel of Climate Change' SRES A1B greenhouse gas emissions scenario A1B coupled with the Météo-France Arpège climate model (Déqué 2007b). Regionalized climate projections were produced with a multivariate statistical downscaling methodology, which is able to generate local time series of temperature and precipitation, and other climatic variables at different sites (Boé et al. 2009). Consequently, depending on the scenario, probabilities of LUC induced by (4) vary in time and space. We here focus on three scenarios entitled Status-Quo (SQS), climate-economic adaptation (CEAS), biodiversity conservation (BCS) respectively, whose structures and differences are depicted in Figure 2.

The scenarios and trajectories are computed in a recursive way with decennial steps t from the initial land use $\mathbf{h}(2003,q)$ at year 2003 which is common to all scenarios. Thus, from the past land use $\mathbf{h}(t-1,q)$, the environmental variables $\mathbf{c}(t,q)$ and $\mathbf{e}(q)$, the LUC model of equation (2) together with economic model (6) for the identification



(a) Scenario SQS (b) Scenario CEAS (c) Scenario BCS

Fig. 2: Structure of the different scenarios SQS (a), CEAS (b) and BCS (c) in terms of links between climate change (CC), Ricardian models of returns from land (RIC), land use (LU), conservation payments (CP), ecosystems services (ES) and species distribution models (SDM). In scenario SQS, the model of LUC is used to extrapolate the temporal trends. In scenario CEAS, the effects of climate change on the economic returns from land, and consequently on LUC are taken into account. Scenario BCS is similar to scenario CEAS but with a greening policy providing uniform payments for pastures.

of $\mathbf{r}(t,q)$ and the transition matrix $\mathbf{P}(t,q)$ derived from equation (4), we can deduce the conversion of the different land-uses at time t in every location q in the following matrix sense:

$$\mathbf{h}(t,q) = \mathbf{P}(t,q)\mathbf{h}(t-1,q). \tag{8}$$

The matrix $\mathbf{P}(t,q)$ above is defined by:

$$\mathbf{P}(t,q) = \begin{pmatrix} p_1(t,q) & \dots & p_1(t,q) \\ \vdots & \vdots & \vdots \\ p_L(t,q) & \dots & p_L(t,q) \end{pmatrix}.$$

As an example, consider a parcel q which counts for 100 ha of annual crop in period 0 and has a predicted probability vector for period 1 of $\mathbf{p}(1,q) = (0.8, 0.15, 0.03, 0.01, 0.01)$. This means that 80 ha are predicted to remain annual crops, 15 ha to be converted to pasture, 3 ha to perennial crop, 1 ha to forest and 1 ha to urban. Given the random part of the utility in equation (2), this model gives a vector of probability of finding each land use on each plot according to the different scenarios. In our simulations, the vectors of LUCs probabilities are just summed to be translated in acreages from the plot level to any aggregated scale (Ay et al. 2017a).

The three scenarios differ in LUC simulations as follows:

– **Status-quo scenario** (SQS): only time t is a driver on the equation (6) to obtain the economic returns $\mathbf{r}$ while land-use $\mathbf{h}(t,q)$ induced by (3) and (4) does not depend on climate. In other words, the function G accounting for the role of climate in the economic returns is neglected as follows:

$$\log\left(r_\ell^{\mathrm{SQS}}(t,q)\right) = \gamma_\ell \cdot t,$$

where γ_ℓ captures the temporal trends of land-use returns. Similarly, estimation of utilities are simplified as follows:

$$\overline{U}_\ell^{\mathrm{SQS}}(t,q) = \alpha_\ell + \mathbf{r}^{\mathrm{SQS}}(t,q)(\boldsymbol{\beta}_{1\ell} + \mathbf{e}(q)\boldsymbol{\beta}_{4\ell}) + \mathbf{e}(q)\boldsymbol{\beta}_{3\ell} + \mathbf{h}^{\mathrm{SQS}}(t-1,q)\boldsymbol{\eta}_\ell.$$

– **Climate-economic adaptation scenario** (CEAS): climate variables $\mathbf{c}(t,q)$ are here drivers of both the Ricardian equation (6) and the logistic equations (4).
– **Biodiversity conservation scenario** (BCS): Because LUC transition probabilities are functions of expected returns of each land use, the inclusion of an incentive-based policy is straightforward. This possibility is illustrated here through the study of a spatially-uniform payment of 200 euro.ha^{-1} for pastures[5]. This policy consists, for $t > 1$, in increasing the rents for pastures ($\ell = 3$):

$$r_3^{\text{BCS}}(t,q) = r_3^{\text{CEAS}}(t,q) + 200. \tag{9}$$

For the other land-uses, the economic returns of BCS remain the same as compared to SQS and CEAS. To improve the validity of our simulation for a rather arbitrary value of pasture subsidy of 200, we also perform the BCS scenario with 100 and 300 euros/ha. The LUCs from these alternative amounts, reported in OA Table 8, are surprisingly close to the 200 euro scenario reported in the main text. This result reinforces our choice to consider only one amount for the representativeness of the outcomes. More globally, alternative greening scenarios aiming at fostering biodiversity will be tested in our future works including agroecology innovations (Padró et al. 2020) or more normative scenarios in line with IPBES (2016); Doyen (2018). The conclusion of Section 4 elaborates later on the interest of such alternative scenarios and strategies.

2.6 Biodiversity and ESs metrics

To obtain a detailed description of birds community and biodiversity, we draw on different and complementary indicators as in Mouysset et al. 2012. Thus we consider a global bird abundance index, several bird habitat scores (farmland, forest, urban and generalist indexes), the Shannon diversity, as well as the community trophic index. In terms of ecosystem services, we here focus on carbon sink intensity, forest recreation, and water quality. The values of the basket of ESs are directly derived from land-use. Carbon sink intensity and water pollution rely in particular on EFESE estimations (Monnoyer-Smith 2019); In line with IPBES, the French assessment of ecosystems and ecosystem services, known as Efese, is a platform between science, decision-making and society to strengthen the inclusion of ES in public policies and private decisions in France. The evaluation of recreational services is here based on the minimal distance to forest areas, a method in line with travel cost methods as in Pirikiya et al. (2016); Tardieu and Tuffery (2019).

2.6.1 Bird biodiversity

The different and complementary metrics of birds biodiversity are computed in every plot q and every time t from the abundances $N_s(t,q)$ of the different bird species s obtained from the SDM depicted in Section 2.4 and equation (7). Firstly, we compute a global community size by aggregating and averaging bird abundances across the

[5] In the European Common Agricultural Policy, a significant amount of agri-environmental schemes are payments depending on land use. Since 2007, the French government has taken over an acreage payment of 76 euros by ha and by year for pastures. Our stylized payment is close to a rather ambitious version of this, over doubling the payment.

Table 1: Ecosystem scores considered in the study

Outputs	**Metrics**	**Data and method**
Biodiversity	Bird abundances index, Shannon index, community specialization index, community trophic index	Bird abundances per TERUTI grid square from Species distribution model (SDM)
Carbon sink intensity	Carbon sink intensity net metric tons per TERUTI grid square assessed for CO2, CH4, CF4,and N20 converted to CO2 equivalent	Carbon sink intensity responses to land use predictions using net tCO2eq/ha/year carbon sink input from French Assessment of Ecosystem and ESs (Monnoyer-Smith 2019)
Water quality	Inverse of Nitrate and Phosphorus levels in surface water per TERUTI grid square	Rates of Nitrate and Phosporus by land-use (Turpin et al. 1997; Dorioz 2013)
Forests recreation	Inverse of minimum travel time from each grid square to forest areas	Shortest path routing using Dijkstra algorithm with French road network as graph (National Geographic Information)

species of the community. We also compute other abundance metrics related to different habitat type including generalist, farmland, forest and urban birds (Balmford et al. 2005; Doxa et al. 2010; Devictor et al. 2007). Afterwards, we also consider a structural metric with the Shannon index as well as a functional metric with the trophic index.

Global abundance indicator: The aggregated abundance indicator consists in the geometric mean of abundances (normalized) of the whole community :

$$BI(t,q) = \prod_{s=1,\dots,N} \left(\frac{N_s(t,q)}{N_s(t_0,q)} \right)^{1/N}, \tag{10}$$

where N is the number of bird species within the community, $N_s(t,q)$ is the abundance of species s at time t and location q as defined in (7). The division of every abundance at year t by abundances $N_s(t_0,q)$ at year $t_0 = 2003$ is used to normalize the score of every species.

Habitat abundance indicators: The previous global indicator BI is refined for several habitat classes S including farmland, forest and urban areas by evaluating the geometric mean over every class S;

$$BI^S(t,q) = \prod_{s\in S} \left(\frac{N_s(t,q)}{N_s(t_0,q)} \right)^{1/|S|}, \tag{11}$$

where S is a sub-community associated with the habitat types while $|S|$ corresponds to its cardinal. Applied to farmland specialists species, this index is the well-known European Farmland Bird Index (FBI) (Balmford et al. 2005; Doxa et al. 2010; Devictor et al. 2007; Mouysset et al. 2012). The class of generalist birds is also evaluated in the same vein. The four habitat specializations metrics rely on 17 farmland birds

Table 2: Rates of ecosystem services or disservices: water pollution no_ℓ, pho_ℓ and carbon sink intensity $co2_\ell$ for each land-use type l.

Rates	**Unit** /ha/year	**Pastures**	**Forests**	**Urban**	**Annual Crops**	**Perennial Crops**
Carbon sink $co2_\ell$	tCO2eq	0.37	5.06	0	-0.06	-0.06
Nitrate no_ℓ	kg	8.21	2	13.11	28	28
Phosphorus pho_ℓ	kg	0.6	0.12	1.425	1.6	0.12

specialists, 14 generalists, 19 forest specialists and 13 urban birds as detailed in Table (9) of the appendix.

Shannon index: The Shannon index denoted here by H provides useful information on birds distribution within the aggregate community. It informs in particular on the evenness of the community as follows:

$$H(t,q) = -\sum_{s=1,\dots,N} \frac{N_s(t,q)}{N(t,q)} \cdot \log\left(\frac{N_s(t,q)}{N(t,q)}\right), \tag{12}$$

where $N(t,q) = \sum_s N_s(t,q)$ stands for the total abundance at time t and location q.

Community trophic index: Species trophic index quantifies the average trophic level of a species within the ecosystem and trophic webs (Pauly et al. 1998; Mouysset et al. 2012; Pellissier et al. 2013). It is based on the assumption that vegetables have a value 1, invertebrates 2, and vertebrates 3 respectively. Higher values indicate that species are top-consumer in the community. The individual trophic indexes sti_s for each bird species are listed in Table (9) of the appendix. The community trophic index is here computed as the arithmetic mean of abundances weighted by the exponential of the specific trophic level (Mouysset et al. 2012):

$$CTI(t,q) = \sum_s \frac{N_s(t,q)}{N(t,q)} \cdot \exp\left(sti_s\right). \tag{13}$$

The use of the exponential in this CTI metric arises from the need to have more contrasted individual trophic values between birds.

2.7 Ecosystem services

We here focus on three ESs namely carbon sink intensity, forest recreation, and water quality. As captured by Figure 2 and described in Table 1, we assume that these ESs are directly induced by land-uses $h_\ell(t,q)$. In other words, these ESs are not directly affected by climate change and biodiversity and do not have their own dynamics. For water quality and carbon sequestration, such an estimation is in line with EFESE (the French assessment of ecosystems and ecosystem service) estimations (Monnoyer-Smith 2019). We discuss alternative and more systemic approaches for ESs as in Fezzi et al. (2015) within the conclusion (Section 4).

Water Quality: We here use Nitrate and Phosphorus values of surface water as a proxy of water pollution and consequently as opposite to water quality. We consider that such water surface is produced by precipitation and water runoff mainly from nearby areas and assume that Nitrate and Phosphorus values of surface water depends directly and linearly on the local land-use $h_\ell(t,q)$ on each plot q at time t as follows

$$NO(t,q) = \sum_\ell h_\ell(t,q) \cdot no_\ell, \tag{14}$$

$$PHO(t,q) = \sum_\ell h_\ell(t,q) \cdot pho_\ell, \tag{15}$$

where no_ℓ and pho_ℓ stands for the rates of nitrate and phosphorus (KgN/ha/an) associated with land-use ℓ. The level of the rates of nitrate and phosphorus by land-use ℓ derived from Turpin et al. (1997); Dorioz (2013) respectively are listed in Table 2. To derive scores in terms of ES and water quality, we consider 'inverse' values of NO and PHO pollution levels[6] defined in (14) and (15).

Carbon sink: We use the word carbon sink' abusively here because we rely on greenhouse gas sink related to CO_2, CH_4, CF_4, N_2 to estimate a potential mitigation of climate warming through a carbon equivalence of these greenhouse gas as in Bateman et al. (2013); Monnoyer-Smith (2019). In other words, these greenhouse gas are converted into tonnes of CO2 equivalent by assigning a 'Global Warming Potential' during a given period where CO2 serves as a calculation standard. Thus, from the spatially explicit distribution $h_\ell(t,q)$ in every parcel q at time t together with the carbon rates $co2_\ell$ of Table 2, we deduce the following carbon sink value:

$$I_{CO2}(t,q) = \sum_\ell h_\ell(t,q) \cdot co2_\ell \tag{16}$$

where $co2_\ell$ stands for per tonne per hectare carbon equivalent value[7] of land-use ℓ.

Recreational service: We here assume that the recreational service is inversely related to the distance to forests; such a method is in line with travel cost methods (Pirikiya et al. 2016; Tardieu and Tuffery 2019). Therefore, we use a method of graph theory to compute the shortest path between any location q (here TERUTI points) and forests in the area. As graph, we use the French road data from IGN (*Information Géographique Nationale, ROUTE500*). Nodes are locations q while edges correspond to road sections between pairs of representative cities. A cost or distance c_{q_1,q_2} from q_1 to q_2 is associated with each edge (q_1, q_2) of the graph. To identify the shortest path between two any (non adjacent) locations q and q', we used the Dijkstra algorithm (Dijkstra 1959):

$$c^*_{q,q'} = \min_{q_1=q,q_2,\ldots,q_m=q'} \sum_i c_{q_i,q_{i+1}}$$

[6] For Nitrate and Phosphorus, we use the indicators $I_{NO}(t,q) = \exp(-NO(t,q))$ and $I_{PHO}(t,q) = \exp(-PHO(t,q))$.

[7] These values are however subject to the assumption of maintaining the land-uses; so our calculation does not take into account the sequestration flows emanating from changes in land-use. A proposal for the future would be to add this flow to the calculation as in (Bateman et al. 2013) who propose a method for estimating this flux by calculating the long-term equilibrium carbon stock which takes into account the dynamics of land use.

Such Dijkstra algorithm is well-suited for directed graph with a non-negative cost[8]. To deduce the recreational value relating to the shortest path from any location q to forest at time t, we compute the following minimal distance:

$$c^*(t,q) = \min_{\text{forest } q'(t) \text{ at time } t} c^*_{q,q'(t)} \tag{17}$$

where location $q'(t)$ is here considered as a forest at time t when $h_{\text{forest}}(t,q') > 50\%$. To derive scores in terms of ES, we again consider 'inverse' values[9] of $c^*(t,q)$.

3 Results

Hereafter, only expected (mean) results of simulations and scenarios are displayed[10] .

3.1 Land use change

Table 3 represents the 2003-2053 land-use acreages dynamics $\sum_q h_l(t,q)$ over the New Aquitaine for each land-use class l and for each scenario with a decennial time step t. First, It turns out that the projection of the prior land use trends underlying SQS scenario leads to a global decrease of agriculture in the long run with variations of -32.9% for perennial crops, -1% for annual crops and -15% for pastures. By contrast, the CEAS scenario, accounting for the climate-economic adaptation impact of climate on land-use through the Ricardian model on economic returns, predicts a major increase of crops with a rise of $+72.85\%$ for annual crops and $+25.87$ for perennial crops until 2053. Such an increase of croplands occurs at the expense of pastures. Thus agricultural intensification induced by climate warming leads to the reduction of grasslands. Moreover, there is a slight forest growth for both SQS and CEAS although this growth of forest areas is moderate for CEAS as compared to SQS. Third, BCS scenario accounting for the annual subsidy of 200 euro.ha^{-1}on pastures significantly modifies CEAS dynamics. As expected, the predicted loss of pastures induced by both SQS and CEAS is mitigated as pastures areas gain $+47.07\%$ in 2053 with BCS scenario as compared to CEAS. With such BCS policy, cropland especially annual crops are replaced by grasslands, and there is a moderate forest share loss since forest areas are projected to lose -11.28% when compared to CEAS. Finally, urban sprawl is predicted to occur for all scenarios mostly with CEAS and in a moderate way with BCS.

The variations $\frac{h_l(2053,q)-h_l(2003,q)}{h_l(2003,q)}$ of the spatial distribution of the different land use l are plotted in Figure 3 for the three scenarios. At first glance, the maps show that under the status quo scenario SQS, i.e., under the direct effect of climate change on LU, land pattern changes only slightly. However, we notice that under the effects of climate change, perennial crops such as vineyards and orchards in the Gironde (center

[8] For the numerical implementation, we here use the scientific software `R` and in particular the `cppRouting` package.

[9] We use again the exponentiel with the indicator $I_{REC}(t,q) = \exp(-c^*(t,q))$.

[10] Given the uncertainties $\epsilon(t,q)$ underlying the utility model of equation (2) or the probabilities of transition underpinning LUC (8), confidence intervals could be potentially derived for the different outcomes and figures. However, for the sake of clarity and simplicity, we choose to only show the expected values based on land-use Equation (8).

Table 3: Aggregated acreage dynamics in km^2 of land uses $\sum_q h_l(t,q)$ over the New Aquitaine for each land-use class l and associated variation $\Delta(\%)$ from 2003 to 2053 for the three scenarios SQS: status quo scenario, CEAS: climate-economic adaptation scenario, BCS: biodiversity conservation scenario

A – SQS: Status-quo scenario

Year t	Perennial crops	Annual crops	Pastures	Forests	Urban areas
2003	2968	20984	25184	31320	4502
2013	2941	21088	24268	31643	5018
2023	2560	21127	23462	32621	5187
2033	2303	21076	22685	33495	5398
2043	2123	20949	21994	34289	5602
2053	1991	20776	21388	35013	5790
$\Delta(\%)$	-32	-1	-15	11	28

B – CEAS: Climate-economic adaptation scenario

	Perennial crops	Annual crops	Pastures	Forests	Urban areas
2003	2968	20984	25184	31320	4502
2013	3324	22901	22355	31207	5170
2023	3063	25049	19800	31793	5253
2033	3481	25589	17841	32410	5637
2043	3521	27931	14690	32845	5970
2053	5130	26413	13331	32953	7131
$\Delta(\%)$	72	25	-47	5	58

C – BCS: Biodiversity conservation scenario

	Perennial crops	Annual crops	Pastures	Forests	Urban areas
2003	2968	20984	25184	31320	4502
2013	3290	21802	24068	30846	4951
2023	2959	20965	26038	30485	4510
2033	3150	20552	26667	30174	4414
2043	3117	22855	24580	30014	4392
2053	4209	22236	23940	29417	5155
$\Delta(\%)$	41	5	-4	-6	14

west of the map) will lose shares as well as polycultures in the northeast of Nouvelle Aquitaine. By contrast, under the climate-economic adaptation scenario CEAS, and so under the influence of economic return, there is an increase in farmland but a fall in pastures and forests. Thus, viticulture and polycultures in the North of the region are beneficial in this scenario. In other words, landowners choose to convert grasslands to croplands because it turns out to be economically more advantageous under climate warming.

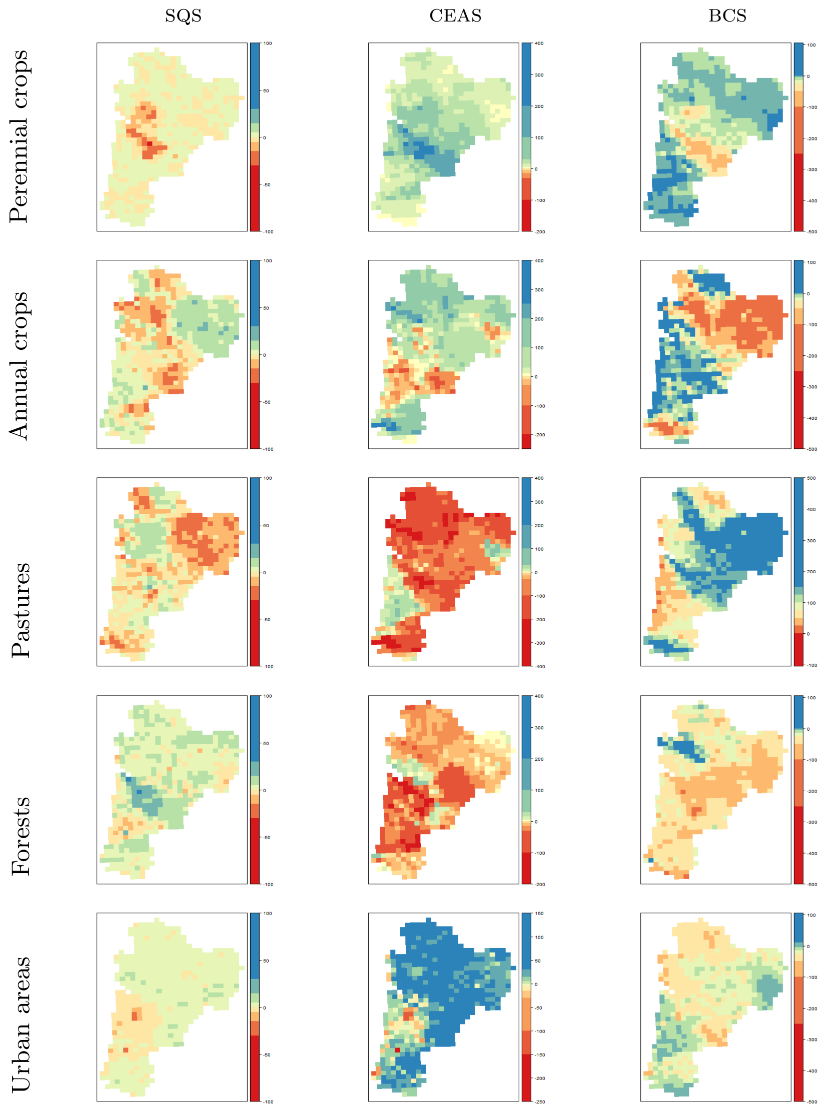
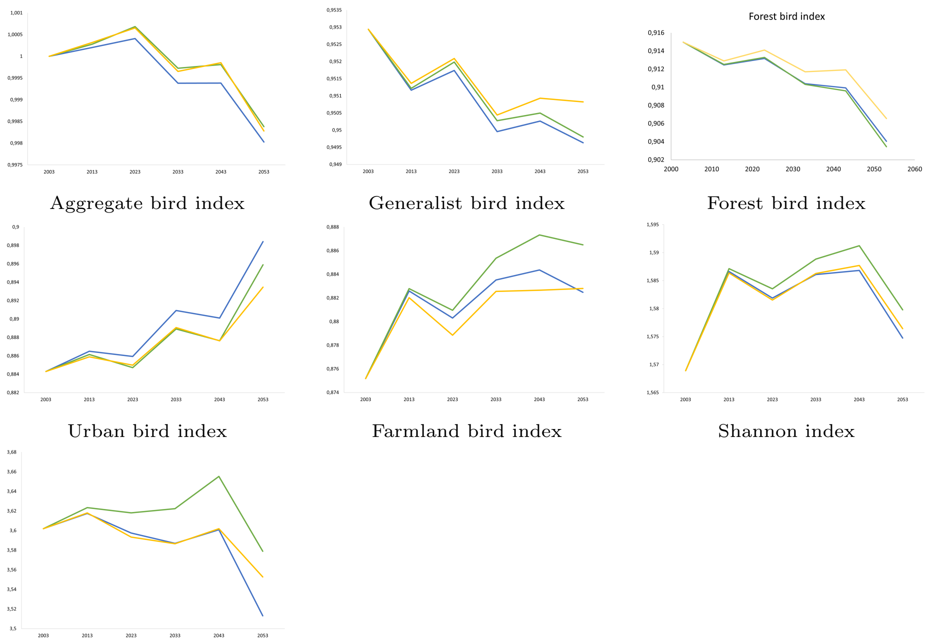


Fig. 3: Land-use variation 2003 - 2053 (%) for the three scenarios in terms of perennial crops, annual crops, pastures, forests and urban areas; column 1: SQS scenario, column 2: CEAS scenario relative to SQS and column 3: BCS scenario relative to CEAS.

3.2 *Bird biodiversity change*

Figure 4 displays the sum over space (q) of the different bird biodiversity metrics defined mathematically in previous Subsection 2.6.1. The figure shows that overall bird community (Aggregate bird index) tends to decrease up to 2053 under the three scenarios. The effects on birds are similar for all scenarios despite a higher



Aggregate bird index | Generalist bird index | Forest bird index

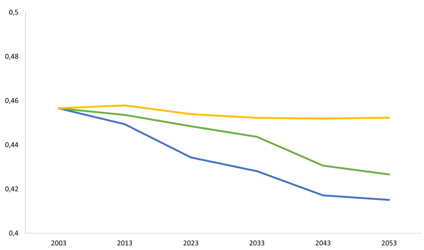
Urban bird index | Farmland bird index | Shannon index

Trophic index

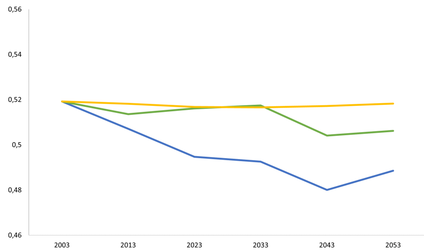
Fig. 4: Bird ecological indicators dynamics (2003-2053) over the region under the three scenarios. In yellow SQS; in blue CEAS, and in green BCS.

decrease under the CEAS. Thus, the economic return of land, and more specifically agricultural intensification, accentuates the negative impacts of climate change and habitat disturbances on birds. Figure 4 shows as well that climate-induced LUC and economic returns under CEAS amplify the effects of climate change on birds. The conservation policy underpinning BCS mitigates the negative effect of climate-induced LUC and economic returns of SQS and CEAS scenarios. However, the effects of such policy BCS are not sufficient to totally offset the negative effects of climate change on birds because abundances decreases are almost similar for all scenarios with -10.3% for SQS, -11.7% for CEAS, and -9.7% for BCS.
The effects of scenarios on birds grouped by habitat speciality are shown on the four subfigures ‘Forest’, ‘Farmland’, ‘Urban Specialists’ and ‘Generalists’ within Figure 4. Generalist and forest bird populations are the most altered by LUC, while urban specialists are beneficial. The growth of birds urban specialists is clearly explained by the increase of urban areas in every scenario. The global increase of farmland specialists over the whole period has to be analyzed with cautious as such an increase in specialization is reversed at the end of period. Said differently, the long run dynamics could also be very detrimental to farmland bird specialists. As regard forest bird specialists, the decline is due to forest losses. When focusing on the differences between the three scenarios, we can observe that only farmland specialists benefit from the BCS conservation policy as expected.

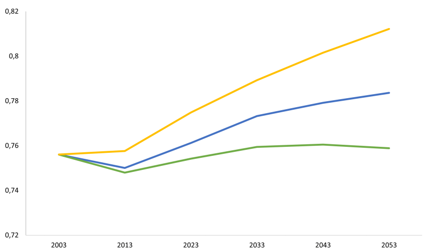
The Shannon trajectories are not linear nor constant in Figure 4. Three general phases emerge: first a significant increase in 2013, followed by a constant trajectory



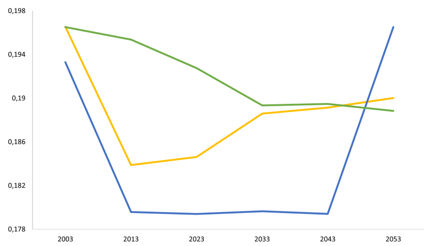
Water quality 1 (Nitrate)



Water quality 2 (Phosphorus)



Carbon sink intensity



Forest recreation

Fig. 5: ESs dynamics (2003-2053) over the NA region under the three scenarios SQS (yellow), CEAS (blue), and BCS (green). (top) Water quality (Nitrate and Phosphorus); (bottom) Carbon sink intensity, forest recreation.

until 2043, ending up with a decline towards the base value in 2053. Such a dynamics suggests a first trend of a better-balanced community ending up with a decrease and even a collapse for some species. On the other hand, bird community turns out to have more evenness with the BCS policy. Consequently, the community might have a more diversified structure with a policy promoting pastures.

Similarly to the Shannon index, the greening scenario BCS enhances the community trophic index. Thus, the birds average trophic level is higher with BCS so that birds take advantage of better functional conditions with grassland extension. Besides, the slight difference between SQS and CEAS shows that climate-induced LUC has limited impacts on bird trophic conditions.

3.3 Changes of ESs

Regarding water quality, we can first observe in Figure 5 (top) that the qualitative pattern of nitrate and phosphorus dynamics coincides for the three scenarios which put emphasis on the robustness of the results. Interestingly, the global decrease of agriculture underlying SQS scenario in NA yields a global stabilization of nitrate and phosphorus pollutions and thus a steady water quality in the long run. Of interest are also the negative trends of water quality in the scenario CEAS due mainly to the increase of annual crops and the induced increase of intrants in that case. The biodiversity conservation by promoting grassland limits the pollution of climate-economic adaptation and climate and induced scenario CEAS but not totally in the long run.

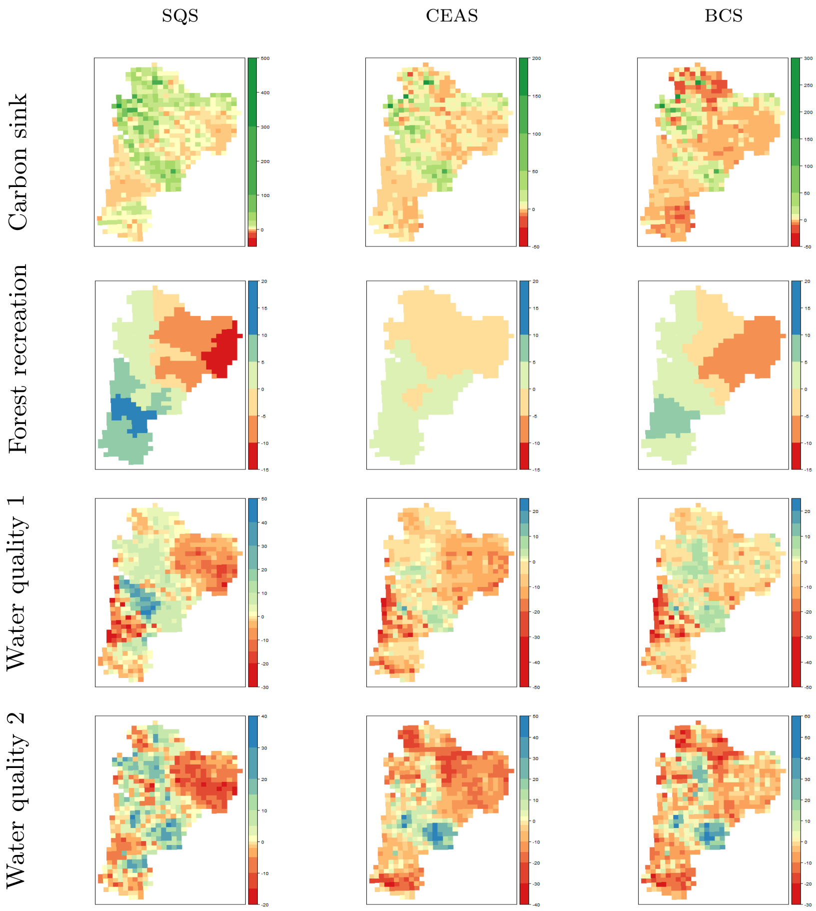


Fig. 6: Spatial distribution of ESs variation (2003-2053): Carbon sink (%), forest recreational service (%) and water quality (%).

As regards carbon sink intensity, the Figure 5 shows a striking result as the environmental policy BCS promoting grassland yields lower scores than SQS and even than CEAS scenario. This result stems from the negative performance of the policy in terms of forests. Forests indeed absorb the largest GHG quantity of GHG with 5.06 tCO_{2eq}/ha/year against 0.37 tCO_{2eq}/ha/year for pastures (Monnoyer-Smith 2019). Although BCS entails a less-extensive agricultural land use than CEAS, it is not enough to reverse forest losses.

The previous results point out the importance of LUC for ESs and the major role played by forest areas. This also holds true for recreational and cultural services. The Figure 5 shows the minimum travel time to forest according to the land use pattern and scenarios while the Figure 6 shows the spatial distribution of forest recreational services, which is the inverse of minimum travel time. Thus forests losses negatively impact recreational activities. Consequently, in the long run (2053), the BCS scenario performs badly because it is associated with a global decrease (-6%) of forests. However,

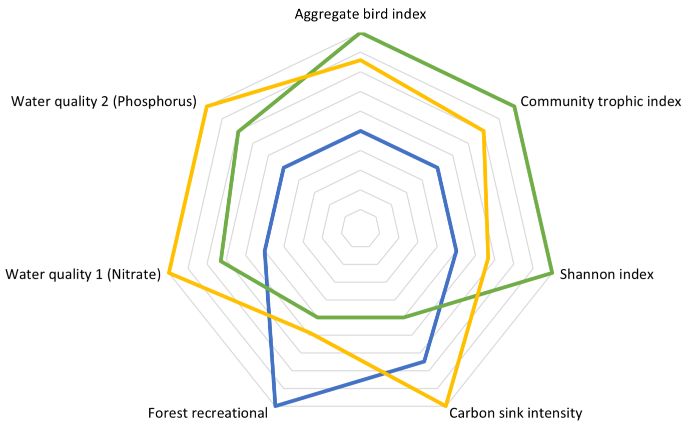


Fig. 7: Comparative radar chart with seven normalized metrics of biodiversity and ESs at year $t = 2053$ across the three scenarios SQS (yellow), CEAS (blue), and BCS (green). The metrics include three biodiversity indicators ($BI(2053)$, $H(2053)$ and $CTI(2053)$), carbon sink intensity $I_{CO2}(2053)$, recreational service $I_{REC}(2053)$ as well as water quality $I_{NO}(2053)$ and $I_{PHO}(2053)$ scores. ESs and biodiversity indicators increase when they move away from the radar center.

the transitions of the three scenarios are non linear which points out the complexity to assess and account for this recreational service.

3.4 Multi-criteria analysis

To compare the three scenarios SQS, CEAS and BCS, we synthetize in Figure 7 the performances of these scenarios for the different metrics using a multi-criteria approach. The radar chart thus accounts for three biodiversity indicators namely BI defined in (10), H defined in (12) and CTI defined in (13) along with four ESs including carbon sink intensity I_{CO2}, recreational service I_{REC} as well as water quality scores I_{NO} and I_{PHO}. The values for the different indicators correspond to the regional value in year $t = 2053$. Scenario performance increases as the radar surface expands. The radar is also normalized to the best scores among the 3 scenarios[11].

Such a graph gives insights into the complex linkages between ESs and biodiversity, their trade-offs and synergies. The graph here captures and confirms two general

[11] Thus, in more mathematical terms, the normalized values of the radar chart for the different scores $I_k^{scenario}(t, q)$ for each scenario (SQS, CEAS, BCS) are defined by:

$$\widetilde{I}_k^{scenario} = \frac{\sum_q I_k^{scenario}(2053, q) - I_k^{\min}}{I_k^{\max} - I_k^{\min}},$$

findings: (i) the (plausible) account of the economic feedback on land-use from climate underlying CEAS significantly worsens the biodiversity scores and the majority of ESs (except the recreational service) as compared to the SQS scenario; in other words the blue shape shrinks the yellow shape; (ii) a greening policy promoting grassland is beneficial to biodiversity and ESs but at the expense of the carbon sink and recreational services. Therefore, bringing ESs valuation into decision shows that a policy relying on a specific land use (here grassland) is not enough to promote a global ecosystem quality because of the different environmental outcomes induced by land-use and the complex and nonlinear mechanisms at play between land-use, biodiversity and ESs.

4 Discussion and conclusion

The present paper compares 3 different scenarios of biodiversity and ESs driven by IPCC climate projections at the scale of the Nouvelle Aquitaine (NA) region in France from 2013 to the next 4 decades. The scenarios differ in the account for economic returns, public policies, and climate on LU.

The scenarios first confirm that climate change is a key driver of biodiversity, ecosystems and ESs (Bennett et al. 2009; Bullock et al. 2011; Raudsepp-Hearne et al. 2010; Ay et al. 2014, 2017b; IPBES 2016; Leclère et al. 2020). This impact at the regional scale of NA is globally negative for biodiversity and water quality at least in the long run (2053) because many grasslands are replaced by croplands entailing a more intensive farming. However regarding the other ESs taken into account here namely carbon sink intensity and recreational service, the future seems less catastrophic as the forests persists in a significant way, mainly because the Landes forest plays a main economic role in NA. However, the robustness of such results in particular for biodiversity can be questioned for several reasons. First, even if we use models calibrated at the national scale for prediction at a regional level, some new species could potentially migrate and emerge in NA, which could limit the risk of local decline or extinction. In that respect, the refinement and reinforcement of models by expanding the dataset in terms of environmental conditions should be useful. In particular the extension to the European scale is a key challenge. Furthermore, one can postulate that other terrestrial species and taxon will be more affected by LUC than birds. Another shortcoming of the current work and possible improvements relate to the computation of the ESs including water quality, CO2 sequestration and recreational service. In particular direct linkages for these ESs with climate and biodiversity are missing at this stage (Locatelli 2020; Malhi et al. 2020). For water quality, models of Fezzi et al. (2015) are of particular interest.

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where the different scores k refers to three biodiversity indicators (aggregate bird, trophic, Shanon) and carbon sink intensity, recreational service, nitrate and phosphorus quality respectively. Extreme values $I_k^{\min}$ and $I_k^{\max}$ are defined by

$$I_k^{\min} = \min_{scen=\text{SQS,CEAS,BCS}} \sum_q I_k^{scen}(2053, q),$$

$$I_k^{\max} = \max_{scen=\text{SQS,CEAS,BCS}} \sum_q I_k^{scen}(2053, q).$$

For sake of clarity, the minimal values $I_k^{\min}$ are not plotted at the centroid of the radar but arbitrarily correspond to level 5 of the radar.

A second contribution of the paper is to provide an economic-based model of LUC and to account for the economic effects of returns from land and market-based policies on private decisions. In particular, the results show that changing the monetary returns from land with CEAS scenario is sufficient to induce significant differences in terms of LUC, as compared to the scenario SQS with current trends. Therefore, climate turns out to be also a strong determinant of LUC by influencing economic returns. Although the climate-economic adaptation model underlying the CEAS scenario provides informative results in its current form, it could be improved in several ways. One possible improvement is to explicitly take into account spatial autocorrelation of the outcome variables. Another improvement relates to the validity of long term extrapolations through the use of econometrics especially regarding the Ricardian equation. In that respect, again, enlarging the data at the European scale should also bring key insights. Another improvement regarding the relevance of long term projections would consist in using more mechanistic and systemic models (IPBES 2016; Doyen et al. 2013; Doyen 2018).

The robustness of our results can also be discussed in terms of climate projections since we have used a single climate projection. The selection of climate projection A1B derives from the fact that it is close to the mean of the AR4 and AR5 multi-model climate projection ensembles over the period under concern. However, the account of broader range of projected climate changes should substantially yield higher uncertainty in projections of LUC, ESs and biodiversity than those explored here. More generally, it is worthwhile to keep in mind that climate impacts are highly depending on the spatial scale of climate projections, especially in mountainous areas. The use of RCP scenarios instead of ARB projections would also provide updated results.

Our results also suggest that the projections of future biodiversity and ESs distributions cannot be based on a uniform policy and incentive. For conservation policies, this stresses the key challenge that consists in complying with the heterogeneity and complexity underlying both biodiversity, ecosystem, ES and LU responses. The paper indeed shows that accounting for multiple land-uses and reconciling biodiversity and the various terrestrial ESs is extremely challenging. The scenario BCS by promoting pastures has indeed a positive influence on biodiversity and water quality but by reducing forests is detrimental to some ESs including carbon sink intensity and recreational values. Therefore, BCS exhibits tensions and trade-offs between ESs and biodiversity. In other words, multi-criteria approaches should be developed to manage biodiversity and ES (Bateman et al. 2013; IPBES 2016; Doyen 2018). More globally, Tinbergen (1952) pointed out that incentives are generally targeting a single performance and, as such, are not designed to achieve multiple policy objectives (Brett 2013). The numerous interplays between ESs, biodiversity and land-use complexify such goal especially regarding agro-ecosystem. For example, when around 1 million-hectare Tasmanian blue gum (Eucalyptus globulus) have been planted across southern Australia in 1998 thanks to taxes (under the Managed Investment Act), timber, carbon sequestration and other biodiversity benefits were derived (Brett 2013). However, such process reduced clean water. Thus, our results pointing out that a policy promoting grassland leads to outcomes that are beneficial to biodiversity at the expense to forests and GHG sequestration are not an isolated case. Said differently, a single policy instrument is not enough to achieve multiple goals underlying sustainability and synergies between land use, ESs and biodiversity. Moreover, in contrast to the regional incentive-based policy considered here, several alternative policies could be examined. A first option would consist in spatializing the policy by applying payments to landowners according to their location. A second option inspired by a command-and-control approach relies on the use of quota or constraint in terms of land-use that implies external, regular controls on LU at farm scale. The use of offsets could constitute another strategy to promote biodiversity and ES and to bring ESs valuation into decision making (Bateman et al. 2013; Simpson et al. 2021). Both options would require more economical and ecological informations to reinforce the prospective tools developed in the present paper. Another key alternative are policies based on agroecology as advocated by (FAO 2019a,b) or the European Commission (Tools and applications on Agroecology). Agroecological transitions would indeed enlarge the range of options to react to exogenous climate changes either for private landowners, farmers, and policymakers. Furthermore, the impacts of land use changes on biodiversity and ecosystem services will also be totally different if agroecology management is adopted. For instance, changing cropland uses or even expanding them using diversified agroecology ways of farming, such as agroforestry, intercropping of grains and perennials, mixed farming of crop rotations with extensive livestock in complex landscape mosaics, etc., would entail a much more wildlife-friendly farming capable of sustaining farm-associated biodiversity. At the same time, it would change the negative signs of the impacts on water quality, carbon sequestration and recreational ecosystem services of changing cropland uses or even expanding them at the expense of grasslands and forestland. Relevant models accounting for agroecology include Padró et al. (2020) that optimize a given set of biophysical restrictions, constrains and capabilities scaling up the current farming best practices in a specific region.

More generally, our work stresses the need of ecosystem-based (and/or) nature-based models, scenarios and management (Abdelmagied and Mpheshea 2020) accounting for the different bio-economic complexities and interplays underlying land-use dynamics, sustainability and resilience facing climate change (Grafton et al. 2019).

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Supplementary Appendix (only online)

A Details about the modeling framework

A.1 Land-use changes

Data about LUC are extracted from the TERUTI survey which was carried out every year 1992-2003 by the statistical services of the French Ministry of Agriculture. TERUTI data count about 550,000 points for which we know the location in terms of French municipalities. The TERUTI survey uses a systematic area frame sampling with a two-stage sampling design. In the first stage, the total national area is divided into a 12×12 km grid. For each of these 4,700 regular meshes there are 4 aerial photographs which cover 3.5 km^2 each. In the second stage, on each photograph, a 6-by-6 grid determines the 36 points to be surveyed in June by an agent on the ground. Each point corresponds to a homogeneous unit in terms of land use and statistically represents about 100 hectares (ha) at the département scale (n = 95, median area: 5,880 km^2). On the basis of the detailed classification of land uses (81 items) we attribute to each plot a use among 5 more aggregate items: annual crop (wheat, corn, sunflowers, etc.), pasture (a rather large defintion: grassland, rangelands, productive fallows, moor), perennial crop (vineyard, orchard and greenhouses), forest (both productive and recreative, including plantation, hedgerow) or urban (cities and exurban houses but also roads, highways, airports, etc.)

The estimation of LUC models was performed using `nnet 7.3` on `R`. The unobserved factors are assumed to be uncorrelated over alternatives and periods, as well as having a constant variance. These assumptions, used to provide a convenient form for the choice probability, were found to be not restrictive (homoscedasticity cannot be rejected by a score test, p-value$= 0.283$). Moreover, these hypotheses are associated with the classical restriction of Independence of Irrelevant Alternatives (IIA) for which Hausman-McFadden specification tests are performed, with mixed evidence. The independence is not rejected for three uses: pasture, perennial crop and urban (p-values are respectively 0.001, 0.005 and 0.036) but rejected for annual crop and forest at 5%. In the land use econometric literature, use of nested multinomial logit is found not to change the results (Lubowski et al. 2008).

A.2 Background data

Biophysical attributes of sampled plots include both topographic and climate variables. Topography of each plot was generated by coupling a Digital Elevation Model of France (resolution of 250 meters) with the spatial geo-referencement of plots. Within a Geographical Information System (GIS), we calculated the elevation, the slope, the roughness and the exposition of each TERUTI sampled plot. Soil quality variables were extracted from the French soil database developed by the National Institute for Agricultural Research and matched by GIS. The initial data are available at the 1:1,000,000-scale (Jamagne et al. 1995) and they were downscaled to a 1-km grid with pedotransfert rules (Cheaib et al. 2012). They provide measures of the agricultural fertility of plots: plant available water capacity and soil depth. We use historical (1990–2010) and projected (2010–2053) climate data, both available at the same spatial resolution (8×8 km rasters) with a smooth transition between historical and future climate. Climate data include 13 variables about temperatures (annual means, maximum and minimum, bird breeding period means April–August and seasonality approximated by standard deviation), precipitations (annual means, maximum and minimum, breeding period means and seasonality), solar radiation (breeding period means), relative humidity (breeding period means) and wind (breeding period means). Regionalized climate scenarios are based on the Intergovernmental Panel of Climate Change' SRES A1B greenhouse gas emissions scenario A1B coupled with the Météo-France Arpège climate model (Déqué 2007b). Regionalized climate projections were produced with a multivariate statistical downscaling methodology, which is able to generate local time series of temperature and precipitation, and other climatic variables at different sites (Boé et al. 2009). The model is based on large-scale circulation predictors, here the mean sea-level pressure field, as well as the 2-meter temperature averaged over France. It starts from regional climate properties to establish discriminating weather types for the chosen local variable. Intra-type variations of the relevant forcing parameters are then taken into account by multivariate regression using the distances of a given day to the different weather types as predictors. The final step consists of conditional re-sampling (for further details in climate downscaling see Boé et al. 2009 and Cheaib et al. 2012).

A.3 Economic returns

The price of land is used to compute the expected net returns from different agricultural land uses. Defining land price as the net present value of expected future rents is standard in the economic theory (Ricardo 1817; Goodwin et al. 2003). This approach, detailed in the main text, uses data about land prices that also come from the statistical services of the French Ministry of Agriculture. Yearly prices 1990–2005 are available for three land uses (annual crops, pastures and perennial crops) and for the 713 Small Agricultural Regions (SAR) of France. SAR size ranges from 11 to 4,413 km^2 with an homogeneity in terms of both agro-ecological and economic levels, reducing intra-SAR heterogeneity (Mouysset et al. 2012). For the two others considered land uses – forest and urban – the approximations of economic returns are computed differently and at different geographic scales. For the expected net returns from forest, we use data about wood raw production (in m^3), total forest area (in ha) and wood prices (in current euro per ha), all available annually at the scale of the French départements. We compute the expected returns from forest by multiplying the aggregate production by its unitary price and dividing the result by the total forest area of each département. This simplification is based on the assumption of a myopic agent who makes decisions based on the hypothesis that future returns will be the same as today and neglect production costs. The urban returns are approximated by the population densities at the fine scale of the municipalities (number of people per total area) on the basis of the national census of French population. In France, the municipality is the administrative body where development planning choices (constructability, servicing) are operated.

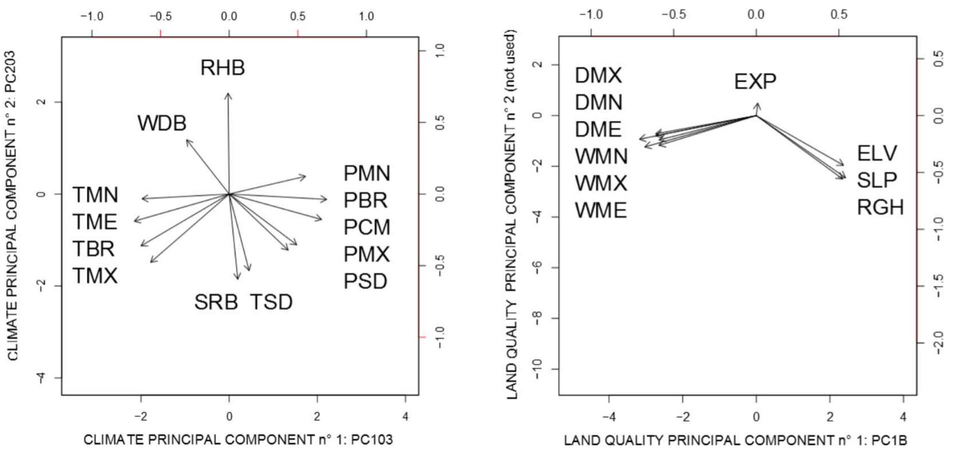
A.4 Bird abundances

We used avian data from the French Breeding Bird Survey (FBBS), a standardized monitoring scheme in which skilled volunteer ornithologists identify breeding birds by song or visual contact in spring (Jiguet et al. 2012). In FBBS, each observer provides the name of her municipality, and a 2×2 km square to be prospected is randomly selected within a 10 km radius from the gravity center of this municipality. In each square, the observer monitors 10 point counts separated by at least 300 m twice per spring (4 to 6 weeks between the sessions, 5 minutes each). Counts were repeated yearly by the same observer at the same points, on about the same date (with a maximum difference of 7 days within April to mid June) and at the same time of day (with a maximum difference of 15 minutes). FBBS data contribute to European official index of biodiversity and have been extensively used to study the effects of climate and LUC on bird populations (Barbet-Massin et al. 2011; Barnagaud et al. 2012), as well as the effects of farmers preferences (Mouysset et al. 2013)and the effects of agro-environmental policies. To simultaneously smooth annual noise and model the observed dynamics, FBBS data are used at two points of time, 2003 (the average 2002-2004, n = 1, 031) and 2009 (the average 2008–2010, n = 1, 380). For each species and each FBBS square, bird abundances are defined as the maximum number of counts. FBBS provides also a description of the habitats of the surveyed squares. The SDM are estimated with FBBS habitats description (with an equivalence of items to the other land use data to perform predictions) and each FBBS observation is weighted in the regressions according to its significance in terms of local land use. On their own, FBBS habitats description is not representative of the local land use.

To estimate SDM presented in the text, we use the `gam()` function from the `R` package `mgcv 1.7`. Because the impacts of climate change on species distributions have been shown to vary depending on choice of modeling technique (Buisson et al. 2010 and Garcia et al. 2012) and of spatial structure (Dormann et al. 2007), we have estimated other SDMs based on alternative assumptions. We also fitted negative binomial mixed models without including geographical coordinates (with the `R` package `glmmADMB`, see Table 4 in the Appendix). Including geographical coordinates increases the goodness-of-fit but have a relative limited impact on abundance variations within scenarios, we focus only on the results from the negative binomial GAMs here for the sake of clarity.

B Additional Figures and Tables

Fig. 8: Principal Component Analysis for climate and land quality variables



Notes: For climate variables (left panel), RHB: mean relative humidity during breeding, WDB: mean wind during breeding, TMN: minimal monthly temperature, TME: mean monthly temperature, TBR: mean temperature during breeding, TMX: maximal monthly temperature, PMN: minimal monthly precipitation, PBR: mean precipitation during breeding, PCM: mean monthly precipitations, PMX: maximal monthly precipitation, PSD: seasonality of precipitations, TSD: seasonality of temperatures, SRB, mean solar radiation during breeding. For land quality (right panel), DMX: maximal soil depth, DMN: minimal soil depth, DME: mean soil depth, WMN: minimal water holding capacity of soils, WMX: maximal water holding capacity, WME: mean water holding capcity, EXP: exposition, ELV: elevation, SLP: slope, RGH: roughness. The two main axis for climate variables are used, only the first for soil variables.

Table 4: Results of Species Distribution Models for the 65 common bird species in the national scale

Generalized Additive Models with Negative Binomial distributions					
N= 65	Min	Q1	Q2	Q3	Max
Adj.$-R^2$	.15	.29	.37	.43	.87
Corr(N,$\widehat{N}$)	.43	.57	.62	.67	.94
Statistical significance of explanatory variables					
VAR	CLIMATE	ANCR	PAST	FORE	URBA
p-value ¡ .01	56/65	52/65	25/65	62/65	52/65
% of species	(86)	(80)	(38)	(95)	(80)

Notes: 65 Species Distribution Models are estimated, one for each bird species of interest. The top panel of the Table presents the distribution of the adjusted R-squares and of the correlations between observed and predicted abundances. The two principal components of climate variables (left panel of Figure ORF1) are included in SDM with bivariate smoothing functions. Land use shares (ANCR for annual crops, PAST for pasture, FORE for forest and URBA for urban, perennial crop is the reference land use) have each their own additive smooth functions. The bottom panel of this Table presents the results about the 1 % statistical significance of each smoothed terms (both bivariate and univariates).

Table 5: Summary of the results from the 4 econometric Ricardian models

	$F-\mathbf{c}_{qt}$	$F-\mathbf{e}_q$	$F-POP$	$F-\mathbf{m}_q$	γ_ℓ	(n,t)	**Adj.-R^2**
ANCR	4.95**	11.6**	29.6**	14.8**	.028**	(713, 3)	.785
PAST	4.13**	11.6**	17.5**	6.11**	.012**	(713, 3)	.766
PECR	3.62**	0.43	2.90*	20.6**	.007*	(93, 2)	.914
FORE	6.46**	1.68	0.65	19.9**	.000	(93, 3)	.361

Notes: Only 4 Ricardian models (in row) are estimated because the proxy for urban returns is predicted from deterministic national projections. ** stands for 0.01% of statistical significance, * for 1%. The table reports the values of F-tests for statistical significance. Climate variables, c_{qt}, enter by their two principal components inside bivariate smoothing functions. Land quality variables, e_q, enter by their first principal component inside univariate smoothing functions. Human population (POP) is also included in these Ricardian models, as spatial coordinates m_q of the centroïds of each Small Agricultural Regions. The latters enter as two arguments of bivariate smoothing functions. The fifth column reports the coefficients for the annual trends and their significance. The table also contains the cross-sectional and temporal dimensions of the data used to estimated Ricardian models. They are principally shaped by data availability: n= 713 where the estimation is at the scale of Small Agricultural Regions and n= 93 for the French départements, see Table ST4. Three of these models are estimated on pooled data from three points of times: 1993, 1998 and 2003. Again because of data limitations, the Ricardian model for perennial crop is estimated on only two periods: 1993 and 1998. The last column contains the adjusted R-squares associated to each model.

Table 6: Robustness results of Species Distribution Models

Mixed Models with Negative Binomial distributions					
N=55	Min	Q1	Q2	Q3	Max
Corr(N,$\widehat{N}$)	.22	.40	.47	.56	.78
Statistical significance of explanatory variables					
VAR	CLIMATE	ANCR	PAST	FORE	URBA
p-value < .01 % of species	34/55 (62)	37/55 (69)	16/55 (27)	52/55 (95)	39/55 (71)

Notes: The number of bird species falls from 65 in the negative binomial GAMs to 55 here because of no convergence of estimation process for 10 species. The random effects are specified at the départements scale, see table ST4. The covariates are specified as polynomials of order 2 (the bottom panel of the Table reports joint statistical significance). The geographical coordinates are not included as covariates, to stress the differences with the other SDMs. In comparison with the results from Table ST1, the predictive abilities are smaller the climate and land use variables are less often significants.

Table 7: The Ricardian effects of climate change on the economic returns from land: amounts in current euros and in variations

	2003		2053		Variations 2003–2053				
Land Use	Mean	SE	Mean	SE	Min	Q1	Q2	Q3	Max
ANCR	265.4	92.27	587.7	346.2	− 100.0	+ 72.05	+ 116.8	+ 159.4	+ 323.5
PAST	113.9	73.35	191.7	103.8	− 24.10	+ 52.62	+ 73.81	+ 98.21	+ 341.7
PECR	177.3	730.1	185.6	699.4	− 75.18	+ 4.474	+ 13.35	+ 19.01	+ 196.0
FORE	80.90	60.07	69.92	53.31	− 44.76	− 16.25	− 13.18	− 8.742	+ 45.36
URBA	81.98	291.8	103.0	386.8	− 29.10	+ 13.99	+ 28.31	+ 46.81	+ 109.4

Notes: The mean values of returns are in current euros/ha for the first 4 rows and hab/km^2 for the last. SE is for standard errors, variations are expressed in %. ANCR counts for annual crops, FORE for forests, PECR for perennial crops, PAST for pastures and URBA for urban

Table 8: Biodiversity conservation scenario BCSwith alternative values for pasture payments

A – BCS: Biodiversity conservation scenario with 100 euro/ha

	Perennial crops	Annual crops	Pastures	Forests	Urban areas
2003	2968.27	20984.52	25184.69	31320.01	4502.51
2013	3152.78	21163.72	24863.33	31155.26	4624.88
2023	3337.29	21342.93	24541.97	30990.52	4747.26
2033	3521.81	21522.14	24220.62	30825.78	4869.64
2043	3706.32	21701.35	23899.26	30661.03	4992.02
2053	3890.83	21880.55	23577.90	30496.29	5114.40
Δ(%)	31.08	4.27	-6.38	-2.63	13.59

B – BCS: Biodiversity conservation scenario with 300 euro/ha

	Perennial crops	Annual crops	Pastures	Forests	Urban areas
2003	2968.27	20984.52	25184.69	31320.01	4502.51
2013	3219.61	21282.08	25005.37	30795.71	4657.21
2023	3470.96	21579.64	24826.06	30271.41	4811.92
2033	3722.30	21877.20	24646.74	29747.11	4966.62
2043	3973.65	22174.76	24467.43	29222.82	5121.33
2053	4224.99	22472.32	24288.11	28698.52	5276.04
Δ(%)	42.33	7.09	-3.56	-8.37	17.18

Table 9: Bird species with their habitat specialization and their specific trophic index

Name	HABITAT	Species	STI
Sky Lark	AGR	*Alauda arvensis*	1,25
Red-legged Partridge	AGR	*Alectoris rufa*	1,1
Common Swift	URB	*Apus apus*	1,75
Common Buzzard	AGR	*Buteo buteo*	2,9
Linnet	AGR	*Carduelis cannabina*	1,05
Goldfinch	URB	*Carduelis carduelis*	1,05
Greenfinch	URB	*Chloris chloris*	1,05
Short-toed Treecreeper	FOR	*Certhia brachydactyla*	
Hawfinch	FOR	*Coccothraustes coccothrauste*	
Wood Pigeon	GEN	*Columba palumbus*	1,01
Carrion Crow	GEN	*Corvus corone*	1,51
Western Jackdaw	URB	*Coloeus monedula*	
Common Quail	AGR	*Coturnix coturnix*	1,22
Common Cuckoo	GEN	*Cuculus canorus*	2
House Martin	URB	*Delichon urbicum*	
Great Spotted Woodpecker	FOR	*Dendrocopos major*	1,21
Black Woodpecker	FOR	*Dryocopus martius*	
Cirl Bunting	AGR	*Emberiza cirlus*	1,3
Yellowhammer	AGR	*Emberiza citrinella*	1,3
Robin	FOR	*rubecula rubecula*	
Common Kestrel	AGR	*tinnunculus rubecula*	2,85
Common Chaffinch	GEN	*Fringilla coelebs*	1,1
Eurasian Jay	GEN	*Garrulus glandarius*	1,72
Melodious Warbler	GEN	*Hippolais polyglotta*	1,95
Barn Swallow	URB	*Hirundo rustica*	
Red-backed Shrike	AGR	*Lanius collurio*	2,15
Wood Lark	AGR	*Lullula arborea*	1,5
Rufous Nightingale	GEN	*Luscinia megarhynchos*	2
Corn Bunting	AGR	*Emberiza calandra*	1,3
Yellow Wagtail	AGR	*Motacilla flava*	2
Golden Oriole	GEN	*Oriolus oriolus*	1,95
Coal Tit	FOR	*Periparus ater*	
Blue Tit	GEN	*Cyanistes caeruleus*	
Crested Tit	FOR	*Lophophanes cristatus*	2
Great Tit	GEN	*Parus major*	1,85
Marsh Tit	FOR	*Poecile palustris*	
House Sparrow	URB	*Passer domesticus*	
Tree Sparrow	URB	*Passer montanus*	
Grey Partridge	AGR	*Perdix perdix*	1,1
Black Redstart	URB	*Phoenicurus ochruros*	
Common Redstart	URB	*Phoenicurus phoenicurus*	
Western Bonelli's Warbler	FOR	*Phylloscopus bonelli*	
Common Chiffchaff	FOR	*Phylloscopus collybita*	
Wood Warbler	FOR	*Phylloscopus sibilatrix*	
Willow Warbler	FOR	*Phylloscopus trochilus*	
Magpie	URB	*Pica pica*	
Green Woodpecker	GEN	*Picus viridis*	2
Dunnock	GEN	*Prunella modularis*	1,5
Bullfinch	FOR	*Pyrrhula pyrrhula*	
Firecrest	FOR	*Regulus ignicapilla*	
Goldcrest	FOR	*Regulus regulus*	
Whinchat	AGR	*Saxicola rubetra*	2
Common Stonechat	AGR	*Saxicola rubicola*	2
European Serin	URB	*Serinus serinus*	
Eurasian Nuthatch	FOR	*Sitta europaea*	
Collared Dove	URB	*Streptopelia decaocto*	
Blackcap	GEN	*Sylvia atricapilla*	1,6
Common Whitethroat	AGR	*Sylvia communis*	1,6
Wren	FOR	*Troglodytes troglodytes*	
Blackbird	GEN	*Turdus merula*	1,6
Song Thrush	FOR	*Turdus philomelos*	1,6
Mistle Thrush	FOR	*Turdus viscivorus*	1,6
Hoopoe	AGR	*Upupa epops*	2

Fig. 9: Effect of scenario SQ on bird index for 4 different habitat speciality. Each partially-linear curve represents a bird species.

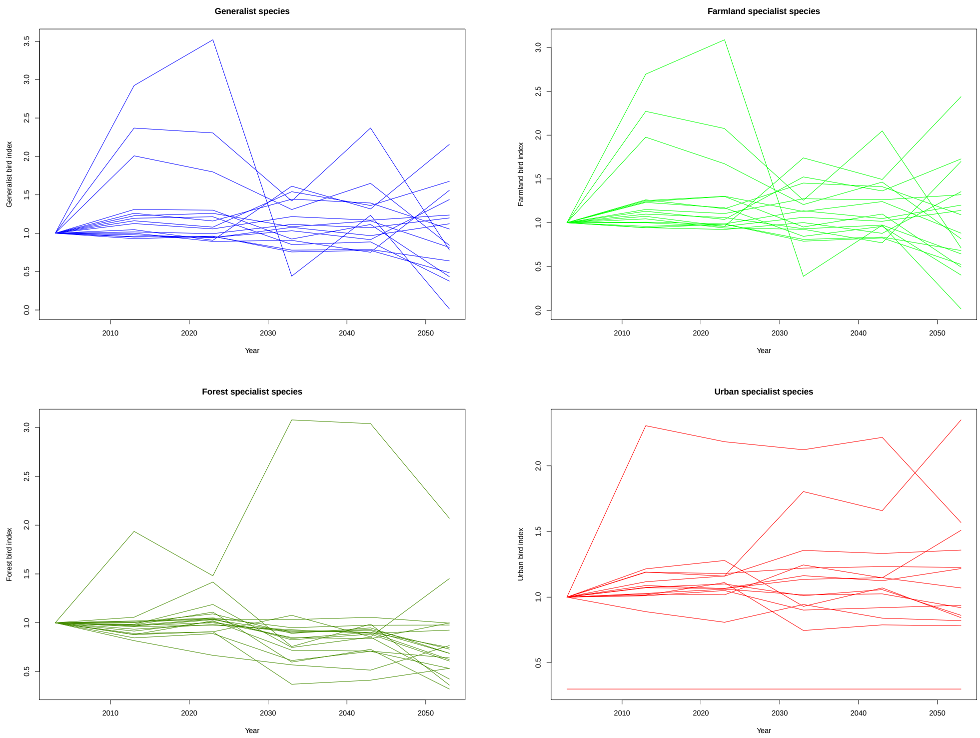


Fig. 10: Effect of scenario BES on bird index for 4 different habitat speciality.Each partially-linear curve represents a bird species.

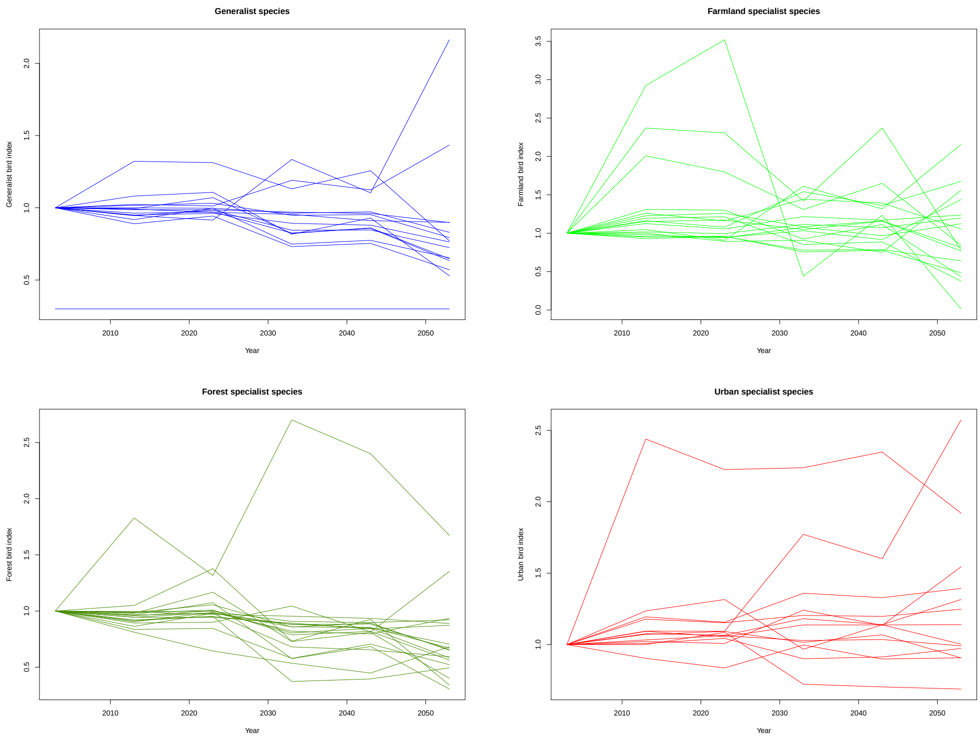


Fig. 11: Effect of scenario PI on bird index for 4 different habitat speciality. Each partially-linear curve represents a bird species.