ORIGINAL ARTICLE

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 to 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, recreation, and water quality. 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 payment for pastures slightly mitigates these impacts on biodiversity and water pollution. However, this turns out to be detrimental for other ecosystem services. This confirms that the design of sustainable policies cannot 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

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 years), these two drivers need to be treated differently 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, and most of the climate change (CC) 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 and supra-national strategies such as the Common Agricultural Policy (CAP) and on regional (or even more local) authorities which can indeed organize land planning and regulate human activities in order to preserve biodiversity through natural area protection, greens corridors, conservation, and valorization of species or habitats.[1]

However, some of these present and future LUCs are likely to be influenced by CC. Local opportunities and constraints indeed appear with CCs as humans adapt their land

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[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.

use 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 (Lobell et al. 2011). Consequently, an efficient conservation or climate 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) including provisioning services related to farming and forestry and 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 major productive ecosystems including crops, grassland, vineyards, and forests, as well as major urban land use in particular with the city of Bordeaux and its surrounding areas. These contrasting 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 ecosystems in the NA are facing 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 modeling 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 CC, 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 2007), TERUTI survey,[2] land prices from the French Ministry of Agriculture, and the French Breeding Bird Survey (FBBS) (Jiguet et al. 2012)

This allows us to address three main questions:

(i) What is the likely effect of CC 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 CC 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, and urban indexes), and 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 CC 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.

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

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. Secondly, we find that climate-dependent LUC amplifies the negative direct effect of CC on birds and several ESs. Thirdly, we highlight that, although a spatially uniform policy to promote pastures can counteract the negative effect of CC on biodiversity, such greening scenario turns out to be detrimental for some ESs. Thus, the design of sustainable policy for land use cannot 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: the "The ecological-economic model and scenarios" section details the case study, the model, the scenarios as well as the criteria used; the "Results" section presents the results, including the scenario trajectories and spatial patterns together with a multi-criteria comparison of the scenarios. Finally, the "Discussion and conclusion" section contains a discussion of the results and concludes. An "Online" Appendix (OA) details some methods, results, and outcomes.

The ecological-economic model and scenarios

This section describes the ecological-economic and spatio-temporal model and 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 B of the Online Appendix (OA).

Case study: the New Aquitaine region

This subsection informs on the NA region that constitutes the case study of our paper. The NA, located in the South-West of France, is the largest region in France by area as illustrated by Fig. 6 in the OA. 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 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 includes 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 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 and 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 and 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.

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:

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

For the case study of NA, we focus on five categories, namely annual crops, perennial crops (including vineyards), pastures, forests, and urban areas (Section B.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

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

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 $\mathbf{c}(t, q)$, and biophysical variables $\mathbf{e}(q)$ through six vectors of unknown coefficients to estimate $[\alpha_\ell; \boldsymbol{\eta}_\ell; \boldsymbol{\beta}_{1\ell}; \boldsymbol{\beta}_{2\ell}; \boldsymbol{\beta}_{3\ell}; \boldsymbol{\beta}_{4\ell}]$. The details of this econometric model for the NA case study are provided in the OA including Sections B.1, B.2, and Fig. 7. Lagged land uses $\mathbf{h}(t - 1, q)$ relate to conversion costs. An interaction between net returns and other exogenous variables (climate, elevation, slope, and land quality) also accounts for different spatial resolutions of the data (Section B.2 in OA). Thus, expected economic returns, climate, and biophysical variables could have heterogeneous effects on the utility.

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 land uses 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 not for the regional scale of NA at our disaggregated level (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 (2019) or the European Commission.[4] Relevant models accounting for agroecology in LUC include (Padró et al. 2020).

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[4] Tools and applications on Agroecology

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 CC 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 B.3 in OA). Table 3 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 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.

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 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:

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

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 B.4 of OA). The vector $\mathbf{c}(t, q)$ again stands for climate. Including $\mathbf{z}(q)$ the spatial coordinates (here the centroid of each FBBS square) 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.

Land-use scenarios

From the calibrated ecological-economic model, we explore several scenarios that differ in the dynamics of landowners' utilities described in (3) and economic returns $\mathbf{r}(t, q)$ described in (6). Regionalized climate scenarios are based on the IPCC SRES greenhouse gas emissions scenario A1B coupled with the Météo-France Arpège climate model (Déqué 2007). 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), and biodiversity conservation (BCS), respectively, whose structures and differences are depicted in Fig. 1.

The scenarios and trajectories are computed in a recursive way with decennial steps t from the initial land use $\mathbf{h}(2003, q)$. Thus, from the past land use $\mathbf{h}(t - 1, q)$, the environmental variables $\mathbf{c}(t, q)$ and $\mathbf{e}(q)$, and the LUC model of Eq. (2) together with economic model (6) for the identification of $\mathbf{r}(t, q)$ and the transition matrix $\mathbf{P}(t, q)$ derived from Eq. (4), we can deduce the conversion of the different land uses at time t in every location q in the following matrix sense:

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

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

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

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 $\mathbf{p}(1, q) = (80, 15, 3, 1, 1)$ (%). This means that 80 ha is 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 (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. 2017). The three scenarios differ in LUC as follows:

- **Status quo scenario** (SQS): only time t is a driver on (6) about 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 is neglected as follows:

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

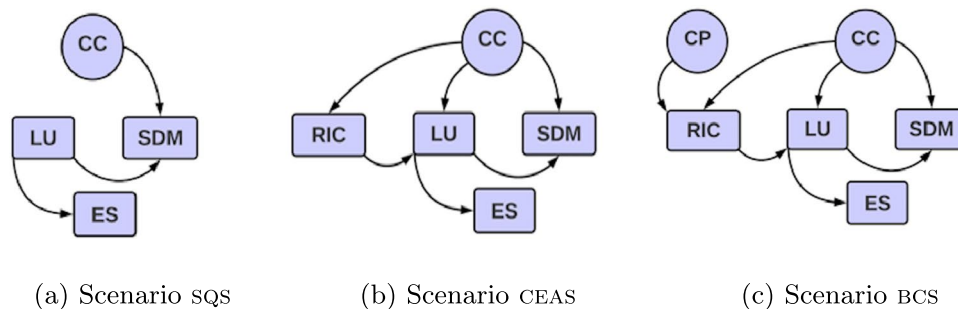(a) Scenario SQS   (b) Scenario CEAS   (c) Scenario BCS

Fig. 1 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. Scenario CEAS accounts for the effects of CC on the economic returns from land and consequently on LUC. Scenario BCS is similar to scenario CEAS but with a greening policy providing payments for pastures

where γ_ℓ captures the temporal trends of land-use returns. Similarly, estimations 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 Eq. (6) and the logistic Eq. (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 ($\ell = 3$):

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

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 6, are surprisingly close to the 200 euro scenario reported in the main text. 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 the "Discussion and conclusion" section elaborates the interest of such alternative scenarios and strategies.

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, and urban indexes), the Shannon diversity, and 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 (EFESE 2019); in line with IPBES,

the French assessment of ecosystems and ecosystem services, known as EFESE, is a platform between science and decision-making to strengthen the inclusion of ES in policies and decisions in France. The evaluation of recreational services is here based on the minimal distance to forest areas, in line with travel cost methods as in Pirikiya et al. (2016); Tardieu and Tuffery (2019).

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 the "Species distribution models" section and Eq. (7). Firstly, we compute a global community size by aggregating and averaging bird abundances across the species of the community. We also compute other abundance metrics related to different habitat type including farmland, forest, and urban birds (Balmford et al. 2005; 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 species within the community. The division of every abundance at year t by abundances $N_s(t_0,q)$ at year $t_0 = 2003$ normalizes the scores.

Habitat abundance indicators The global indicator BI is refined for several habitat classes S including farmland, forest, and urban areas as follows:

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

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 Farmland Bird Index (FBI) (Balmford et al. 2005; Devictor et al. 2007; Mouysset et al. 2012). The habitat specializations metrics rely on 17 farmland birds, 19 forest specialists, and 13 urban birds as detailed in Table 7 of the OA.

Shannon index The Shannon index denoted by H provides information on birds distribution within the community. It informs on the evenness of the community:

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[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.

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

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 trophic webs (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 7 of the Appendix. The community trophic index is here computed as the arithmetic mean of abundances weighted by the specific trophic level:

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

The exponentials are here used to have more contrasted trophic values between birds.

Ecosystem services

We here focus on three ESs, namely carbon sink intensity, forest recreation, and water quality. As captured by Fig. 1 and described in Table 8 of the Appendix, 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 CC and biodiversity and do not have their own dynamics. For water quality and carbon sequestration, such an estimation is in line with EFESE estimations (EFESE 2019). We discuss alternative and more systemic approaches for ESs as in Fezzi et al. (2015) within the conclusion (the "Discussion and conclusion" section).

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 depend directly and linearly on the local land use $h_\ell(t,q)$ on each plot q at time t as follows:

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

Here no_ℓ and pho_ℓ stand for the rates of nitrate and phosphorus (KgN/ha/an) associated with land use ℓ. These rates are derived from Turpin et al. (1997); Dorioz (2013) and are listed in Table 9 of the OA. To derive scores in terms of ES and water quality, we consider "inverse" values of NO and PHO pollution levels[6] defined in (14).

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, and a carbon equivalence of these greenhouse gas as in Bateman et al. (2013); EFESE (2019). In other words, these greenhouse gases are converted into tonnes of CO2 equivalent by assigning a "Global Warming Potential" during a given period. Thus, from the distribution $h_\ell(t,q)$ together with the carbon rates $co2_\ell$ of Table 9 in Appendix, we deduce the following carbon sink value:

$$I_{CO2}(t,q) = \sum_\ell h_\ell(t,q) \cdot co2_\ell, \tag{15}$$

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 (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 (non-adjacent) locations q and q', we used the Dijkstra algorithm:[8]

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

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[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; they do not take into account the sequestration flows emanating from changes in land use. A proposal for the future would be to add this flow as in Bateman et al. (2013) who propose a method for estimating this flow by calculating the long-term equilibrium carbon stock.

[8] The curse of dimensionality underlying the Dijkstra algorithm can be problematic and costly (in time and space) from the numerical viewpoint in particular with a dense transportation network.

Table 1 Aggregated acreage dynamics in km² of land uses $\sum_q h_l(t,q)$ for each class l and associated variation $\Delta(\%)$ from 2003 to 2053 for the scenarios SQS, CEAS, and BCS

Scenario	Year t	Perennial crops	Annual crops	Pastures	Forests	Urban areas
SQS	2003	2968	20,984	25,184	31,320	4502
	2053	1991	20,776	21,388	35,013	5790
	$\Delta(\%)$	−32	−1	−15	11	28
CEAS	2003	2968	20,984	25,184	31,320	4502
	2053	5130	26,413	13,331	32,953	7131
	$\Delta(\%)$	72	25	−47	5	58
BCS	2003	2968	20,984	25,184	31,320	4502
	2053	4209	22,236	23,940	29,417	5155
	$\Delta(\%)$	41	5	−4	−6	14

Such Dijkstra algorithm is well-suited for directed graph with a non-negative cost.[99] For the numerical implementation, we here use the scientific software `R` and in particular the `cppRouting` package. 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:

$$c^*(t,q) = \min_{\text{forest } q'(t) \text{ at time } t} c^*_{q,q'(t)} \qquad (16)$$

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[10] of $c^*(t,q)$.

Results

Hereafter, only expected (mean) results of simulations and scenarios are displayed.[11]

Land-use change

Table 1 focuses on the 2003–2053 land-use dynamics $\sum_q h_l(t,q)$ over the New Aquitaine for each land-use class l and for each scenario. Table 10 in the OA details with a decennial time step t these 2003–2053 land-use dynamics. 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% for perennial crops, −1% for annual crops, and −15% for pastures. By contrast, the CEAS scenario, accounting for the climate-economic adaptation of land use through the Ricardian model on economic returns, predicts a major increase of crops with a rise of +72% for annual crops and +25 for perennial crops until 2053. Such an increase of croplands occurs at the expense of pastures. Thus, farming intensification induced by climate warming leads to the reduction of grasslands. Moreover, there is a slight forest growth for both SQS and CEAS although it 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 pasture areas gain +47% in 2053 with BCS scenario as compared to CEAS. With such BCS policy, croplands especially annual crops are replaced by grasslands, and there is a moderate forest share loss since forest areas are projected to lose −11% 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 Fig. 2 for the three scenarios. The maps show that under the status quo scenario SQS, i.e., without effect of CC on LU, land pattern changes only slightly. However, we notice that without the effects of CC, perennial crops such as vineyards and orchards in the Gironde (center 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 returns, 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.

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

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

[11] Given the uncertainties $\epsilon(t,q)$ underlying the utility model of Eq. (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 Eq. (8).
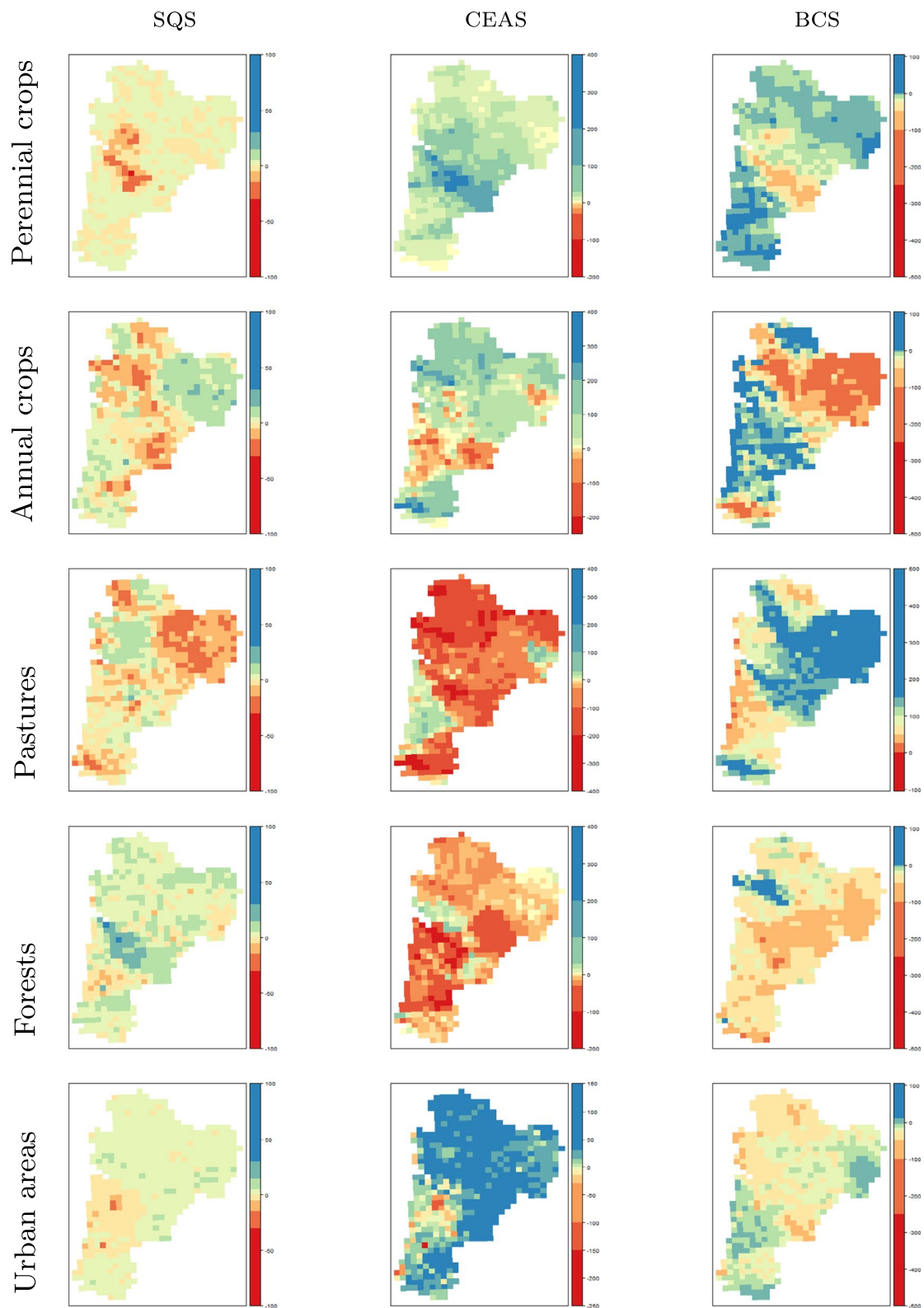
Fig. 2 Land-use variation 2003–2053 (%) for the three scenarios in terms of perennial crops, annual crops, pastures, forests, and urban areas; column 1: SQS scenario, column 2: CEAS scenario relative to SQS, and column 3: BCS scenario relative to CEAS

Bird biodiversity change

Figure 3 displays the sum over space (*q*) of the different bird biodiversity metrics defined mathematically in the previous "Bird biodiversity" section. The figure shows that the 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 decrease under the CEAS. Thus, the economic return of land, and more specifically agricultural intensification, accentuates the negative impacts of CC and habitat disturbances on birds. The conservation policy underpinning BCS mitigates the negative effects of climate in SQS and CEAS scenarios. However, the effects of such policy BCS are not sufficient to totally offset the negative effects of CC on birds because abundances decreases are almost similar for all scenarios with −10% for SQS, −11% for CEAS, and −9% for BCS.

The effects of scenarios on birds grouped by habitat speciality are shown on the three subfigures "Forest," "Farmland," and "Urban" specialists within Fig. 3. 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 caution 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 regards 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 Fig. 3. Three general phases emerge: first a significant increase in 2013, followed by a constant trajectory 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' 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.

Changes of ESs

Regarding water quality, we can first observe in Fig. 4 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
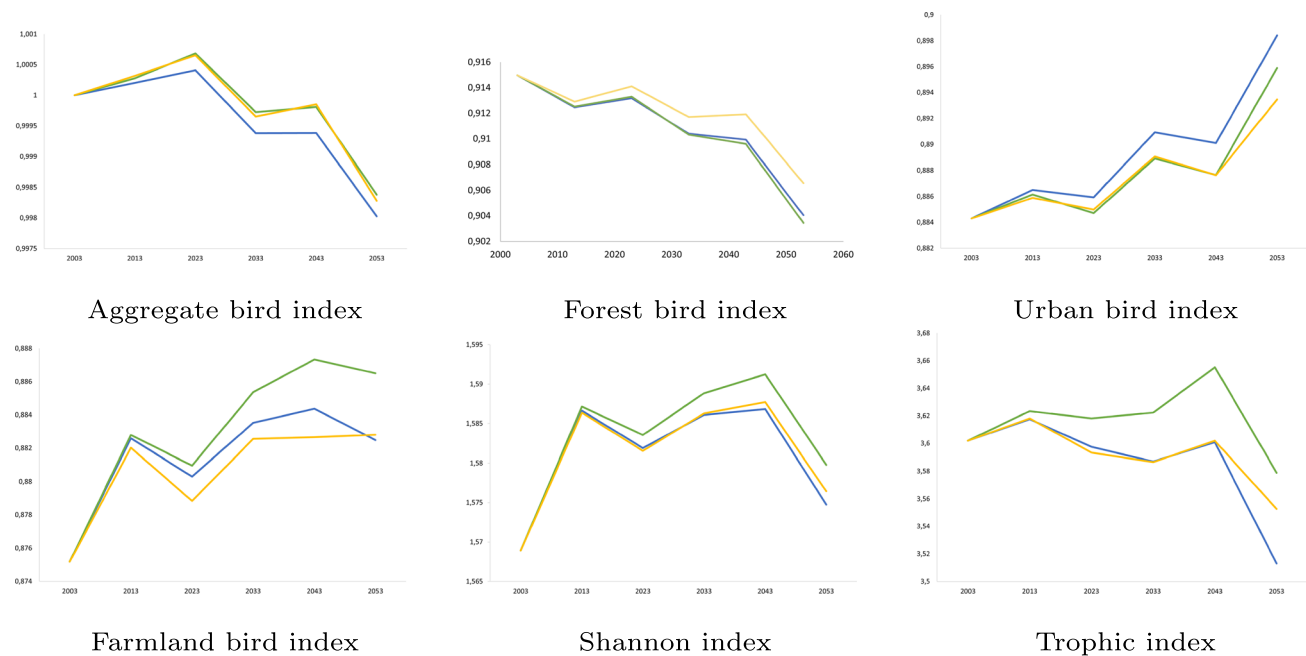

Fig. 3 Bird ecological indicators dynamics (2003–2053) over the region under the three scenarios. In yellow SQS; in blue CEAS, and in green BCS

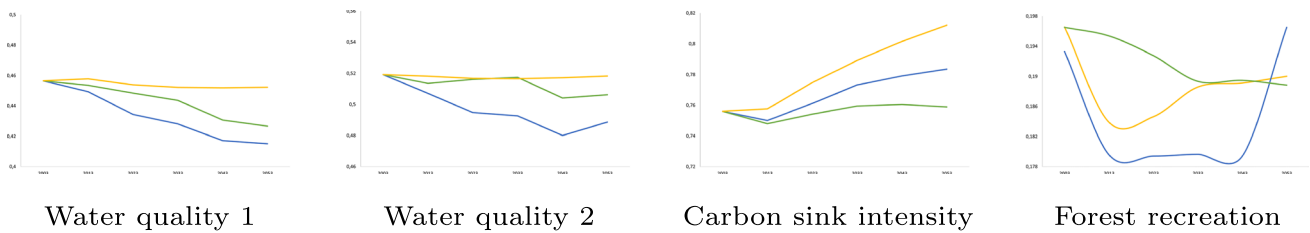| Water quality 1 | Water quality 2 | Carbon sink intensity | Forest recreation |

Fig. 4 ESs dynamics (2003–2053) over the NA region under the three scenarios SQS (yellow), CEAS (blue), and BCS (green)

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. As regards carbon sink intensity, Fig. 4 shows a striking result as the environmental policy BCS promoting grassland entails 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 with 5.06 against 0.37 tCO_{2eq}/ha/year for pastures. 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. Figure 4 shows the minimum travel time to forest according to the land-use pattern and scenarios while Fig. 8 in the Appendix shows the spatial distribution of forest recreational services, which is the inverse of minimum travel time. Thus, forest losses alter 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, the transitions of the three scenarios are nonlinear which points out the complexity to assess and account for this recreational service.

Multi-criteria analysis

To compare the scenarios SQS, CEAS and BCS, we synthetize in Fig. 5 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, H, and CTI respectively defined in (10), (12), and (13) along with four ESs including carbon sink intensity I_{CO2} and 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 three scenarios.[12] 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 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 ES 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.

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

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

where the different scores k refer 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

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

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

For the 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.

Fig. 5 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 $I_{CO2}(2053)$; and recreational service $I_{REC}(2053)$ as well as water quality $I_{NO}(2053)$ and $I_{PHO}(2053)$ scores
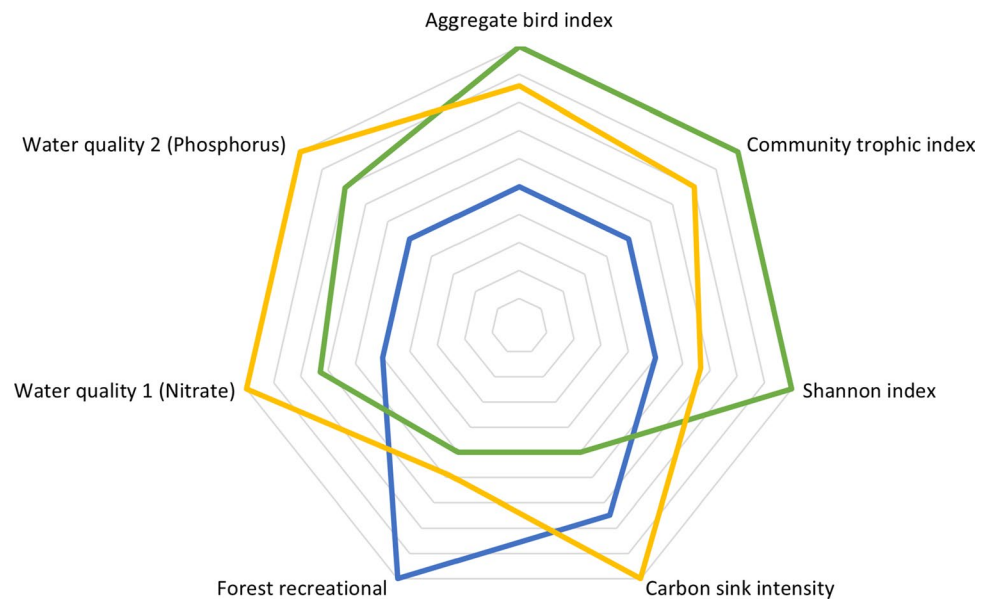
Discussion and conclusion

The present paper contrasts three different scenarios of LU, biodiversity, and ESs driven by CC at the scale of the NA region in France from 2013 to 2053. The scenarios differ in the account for economic returns, public policies, and climate on LU.

The scenarios first confirm that CC 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, 2017; 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 persist in a significant way, mainly because the Landes forest plays a main economic role in NA. Nevertheless, 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 biodiversity erosion. In that respect, the 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.

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, 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. 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 CCs should substantially yield higher uncertainty in projections. 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. The numerous interplays between ESs, biodiversity, and land use complexify such goal especially regarding agro-ecosystem. 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 ES valuation into decision-making (Bateman et al. 2013; Simpson et al. 2021). Both options would require more economical and ecological information to reinforce the prospective tools developed in the present paper. Another key alternative are policies based on agroecology as advocated by FAO (2019) or the European Commission.[13] Agroecological transitions would indeed enlarge the range of options to react to exogenous CCs either for private landowners, farmers, or 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, mixed farming of crop rotations with extensive livestock in complex landscape mosaics 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 CC.

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