

Benefits and costs of biodiversity in agricultural public policies

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Abstract

This paper examines the role played by biodiversity goals in the design of agricultural policies. A bio-economic model is developed with a dynamic and multi-scale perspective. It combines biodiversity dynamics, farming land-uses selected at the micro level and public policies at the macro level based on financial incentives for land-uses. The public decision-maker identifies optimal subsidies or taxes with respect to both biodiversity and budgetary constraints. These optimal policies are then analysed through their private, public and social costs. The model is calibrated and applied to metropolitan France at the small agricultural region scale, using common birds as biodiversity metrics. First results relying on optimality curves and private costs stress the bio-economic trade-off between biodiversity and economic scores. In contrast, the analysis of public costs suggests that accounting for biodiversity can generate a second benefit in terms of public budget. Social costs defined as the sum of private and public costs also show possible bio-economic synergies.

Keywords: biodiversity, land-use, bio-economics, modelling, optimality, scenarios, birds, private cost, public cost

JEL classification: Q57

1. Introduction

In many European countries, a strong decline in biodiversity is observable in agricultural landscapes (Flowerdew and Kirkwood, 1997; Sotherton and Self, 2000; Donald, Green and Heath, 2001). Numerous studies (Chamberlain *et al.*, 2000; Wretenberg *et al.*, 2007) identify the changes in farming systems over the last decades and especially the intensification processes at play as major drivers of this erosion. Populations of breeding birds are particularly

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vulnerable to agricultural change (Krebs *et al.*, 1999; Jiguet *et al.*, 2010). Such a negative effect is due mainly to a degradation in habitat quality altering nesting success and survival (Benton *et al.*, 2003). In this context, the European Union has formally adopted the Farmland Bird Index (FBI) as an indicator of structural changes in biodiversity in response to land-use changes (Balmford, Green and Jenkins, 2003).

A challenge to reach sustainability for agricultural land-use is therefore to reconcile farming production and farmland biodiversity. The approaches usually implemented to achieve such multifunctional goals for farming rely on public policies (Pacini *et al.*, 2004; Drechsler *et al.*, 2007; Mouysset *et al.*, 2011a). For Alavalapati, Stainback and Carter (2002) and Shi and Gill (2005), financial incentives are essential for convincing farmers to adopt eco-friendly activities. These policies modify the farmer's choices and thus impact the habitat and the dynamics of biodiversity (Doherty, Marschall and Grubb, 1999; Holzkämper and Seppelt, 2007; Rashford, Dugger and Adams, 2008). In this perspective, many public policies including agri-environmental schemes have been proposed by decision-makers. However, 15 years after the initial implementation of such instruments at a large scale, their efficiency for enhancing biodiversity remains controversial (Vickery *et al.*, 2004; Kleijn *et al.*, 2006; Butler *et al.*, 2009). These policies face a variety of difficulties.

From the ecological point of view, insufficient knowledge about the agro-ecological processes at play and the focus on a few emblematic species limit the results. From the economic point of view, the low level of acceptability by farmers constitutes a major obstacle for the effectiveness of these policies. In this context, assessing and comparing the different agricultural policy scenarios through quantitative methods and bio-economic models is useful. The cost-benefit method (Boardman *et al.*, 2005) compares monetary costs and benefits of a policy. However, in view of the difficulties of quantifying biodiversity (Diamond and Hausman, 1994) in financial terms, cost-effectiveness analysis appears as a relevant alternative. Based on optimisation under constraints, it leads to defining either the least expensive policy satisfying a biodiversity goal or the policy with the best biodiversity performance under budgetary constraints (Naidoo *et al.*, 2006). Many authors (Polasky *et al.*, 2005, 2008; Drechsler *et al.*, 2007) using this approach for land-use policy have identified different possible trade-offs between ecological and economic objectives.

The objective of this paper is to contribute to accounting for biodiversity goals in the design of agricultural policies and, more specifically, to find possible synergies instead of trade-offs between biodiversity and economic performances. The paper extends the bio-economic modelling works of Mouysset *et al.* (2011a) by focusing on optimal public policies under different biodiversity constraints. More precisely, it is assumed that a public decision-maker identifies an optimal vector of land-use taxes/subsidies which maximises the present value of the national income under different biodiversity targets and a budgetary constraint. The optimal policies are then analysed through their private, public and social costs (Semaan *et al.*, 2007). By private cost is meant the loss due to biodiversity requirements of the farmers' income including transfers (taxes and subsidies)

while the public cost stands for the public monetary balances of public policy. The social cost refers to the loss of the farmer's income without transfers. The study is based on a spatio-temporal bio-economic model applied to the metropolitan France case study. The calibration relies on a French time series of the abundance of 34 birds and 14 farming land-uses over the years 2001–2009 and 620 small agricultural regions (SAR) in metropolitan France. The ecological performance is captured by two biodiversity indicators: the FBI which has been adopted by the European Union (Balmford, Green and Jenkins, 2003) and the community trophic index (CTI) which informs on the functional state of the community (Pauly *et al.*, 1998; Mouysset, Doyen and Jiguet, 2012).

The first main contribution of the paper is to confirm the known trade-off between biodiversity and economic scores in terms of optimality curve and private costs. The second major contribution is to show through public costs how the integration of biodiversity goals is not detrimental to the whole society in the sense that it can generate a benefit in terms of budget balance. The biodiversity-oriented policy thus yields a double benefit. In the same vein, it is pointed out through social (private plus public) costs how ecological-economic synergies and reconciliation may exist. In line with this, it is suggested that the redistribution of the induced earnings to the farmers could compensate their private loss and so increase their acceptance of high biodiversity objectives in the design of agricultural policies.

2. The bio-economic modelling

Illustrated by Figure 1, the bio-economic model links public decision-making at the macro scale with land-use decisions of farmers, habitats and bird population dynamics at the micro scale.

2.1. The biodiversity model

The biodiversity model is based on the population dynamics of a community of species $s = 1, \dots, n_S$ with intra-specific competition¹ depending on habitat and especially on agricultural land-use in several regions $r = 1, \dots, n_R$. A Beverton–Holt function² captures intra-specific competition through a time-dependent carrying capacity as follows:

$$N_{s,r}(t+1) = N_{s,r}(t) \cdot (1 + R_{s,r}) \left(1 + \frac{N_{s,r}(t)}{M_{s,r}(t)} \right)^{-1}, \quad (1)$$

where $N_{s,r}(t)$ stands for the abundance of species s in region r at year t . The coefficient $R_{s,r}$ corresponds to the intrinsic growth rate of species s in region r . The value $M_{s,r}(t)$ captures the ability of this habitat to host the species s in region r . The product $M_{s,r}(t) \cdot R_{s,r}$ is indeed the carrying capacity in the sense

1 The model does not integrate inter-specific competition or migration at this stage.

2 Other density-dependent population dynamics could have been used such as logistic, Ricker or Gompertz models.

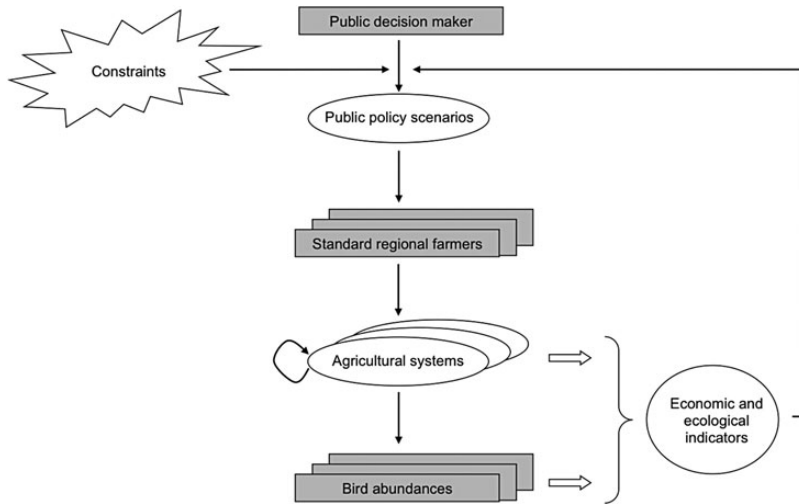


Fig. 1. Bio-economic model coupling. The decision-maker determines an incentive scenario according to a bio-economic optimisation. The farmers choose their agricultural systems by maximising their income under technical constraints. These choices affect the habitat and the bird communities.

Table 1. List of the 14 farming land-uses (OTEX)

The 14 land-uses (OTEX) k

- (1) Cereal, Oleaginous, Proteaginous (COP)
- (2) Variegated crops
- (3) Intensive bovine livestock breeding
- (4) Medium bovine livestock breeding
- (5) Extensive bovine livestock breeding
- (6) Mixed crop-livestock farming with herbivorous management
- (7) Other herbivorous livestock breeding
- (8) Mixed crop-livestock farming with granivorous management
- (9) Mixed crop-livestock farming with other management
- (10) Granivorous livestock breeding
- (11) Permanent farming
- (12) Flower farming
- (13) Viticulture
- (14) Others associations

of the strictly positive equilibrium of the population dynamics. Moreover, the habitat value $M_{s,r}(t)$ is assumed to depend on different farming land-uses $k = 1, \dots, n_K$ (see Table 1) through the linear relation

$$M_{s,r}(t) = b_{s,r} + \sum_{k=1}^{n_K} a_{s,r,k} \cdot L_{r,k}(t), \tag{2}$$

where $L_{r,k}(t)$ stands for the area of land-use k implemented in region r at time t . Consequently, the $a_{s,r,k}$ and $b_{s,r}$ coefficients, depending on each species, inform on how such species s responds to agricultural land-use k in region r . The $b_{s,r}$ coefficient can be interpreted as the mean habitat coefficient for species s in region r and integrates other land-uses such as forests or urban areas.

The indicators used to assess ecological performance are computed from the abundances $N_{s,r}(t)$ of the species. We denote the biodiversity index by $Biod$ without specifying it at this stage. Such a formulation may include usual biodiversity metrics such as species richness, Simpson or trophic indices. In each region, it is defined as follows:

$$Biod(t) = h(N_1(t), \dots, N_{n_s}(t)). \quad (3)$$

2.2. The economic model

We consider a public decision-maker at the macro (typically national) scale interacting with regional farmers in a two stage model. In the first stage, the planner identifies the optimal incentive scheme in terms of taxes or subsidies related to different land-uses by maximising the aggregate income of all regions under global budgetary and biodiversity constraints. In the second stage, the regional farmers make optimal land-use choices by maximising their own rent. Although it is assumed that farmers do not directly account for biodiversity in their local decisions, their land-use is influenced by the public incentives which integrate the biodiversity objectives of the planner. The model is solved backward, beginning with stage two.

Each region r is assumed to be managed by a representative farmer. The aggregation of individual farmers of a given region into a representative farmer at the regional scale is justified by the assumption that the regions are characterised by an agro-economic homogeneity and consequently that the individual farmers within a region face similar environments and constraints. Both region and representative farmer are then associated with the index r . These representative farmers are heterogeneous since regions differ in terms of gross margin and initial allocation of land-uses. The income of representative regional farmer r at year t derives from the different agricultural land-uses k through the agricultural areas $L_{r,k}(t)$ implemented at time t , the expected gross margin per unit of scale denoted by $gm_{r,k}$ and public financial incentives denoted by τ_k . These public incentives for each land-use k include subsidies with positive values $\tau_k > 0$ and taxes when $\tau_k < 0$. The income of farmer r at year t denoted by $\Pi_r(t)$ reads as follows:

$$\Pi_r(t) = \sum_k (1 + \tau_k) \cdot gm_{r,k} \cdot L_{r,k}(t). \quad (4)$$

For a given public incentive scheme $\tau = (\tau_1, \tau_2, \dots, \tau_{n_K})$, it is assumed that the regional farmer selects each year t agricultural land-uses $L_r(t) = (L_{r,1}(t), L_{r,2}(t), \dots,$

$L_{r,nK}(t)$ in order to maximise income $\Pi_r(t)$ given by equation (4) according to capital and rigidity constraints. This reads as follows:

$$\max_{L_{r,1(t), \dots, L_{r,nK}(t)}} \Pi_r(t), \tag{5}$$

under the constraints

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

$$\sum_k L_{r,k}(t) = \bar{L}_r(t_0). \tag{7}$$

In this model, farmers follow a myopic behaviour as in [Péreau et al. \(2012\)](#) and [Sandal and Steinshamn \(2004\)](#). Constraint (6) restricts the area that the farmer can modify each year. Such rigidity can arise due to technological or behavioural reasons. This rigidity constraint reflects some delay in the adjustment process with respect to the optimal policy of farmers. The rate ε captures change costs or inertia. Hence, the speed of change refers to the necessary time for a farmer to adjust his land-use and converge towards an optimal allocation. In other words, farmers with small ε have reduced capability to change each year. Constraint (7) states that the total agricultural area \bar{L}_r remains fixed in each region r . This maximisation programme is solved numerically and yields an implicit reaction function for land-uses $L_{r,k}^*(t)$ depending on the values of the model, namely the incentive $\tau = (\tau_1, \tau_2, \dots, \tau_{nK})$, the gross margins $gm_r = (gm_{r,1}, gm_{r,2}, \dots, gm_{r,nK})$, the inertia parameter ε , the total area \bar{L}_r and the value of land-use $L_r(t - 1)$ in the previous period. Focusing on incentives and land-use, this implicit function can thus be written as follows:

$$L_{r,k}^*(t) = L_{r,k}(\tau, L_r(t - 1)). \tag{8}$$

In stage one, the public decision-maker acts as a leader of Stackelberg; he takes into account the reaction function (8) of each regional farmer when selecting the optimal vector of taxes and subsidies incentives τ_k associated with land-use k . By taxing a land-use, the decision-maker decreases its profitability and stimulates the reduction in the area dedicated to this land-use for the benefit of more gainful land-uses. In contrast, a subsidy on a land-use, by increasing its profitability, promotes its enlargement at the expense of the less profitable ones. The incentives τ_k designed by the decision-maker are defined at the national scale. However, it is worth noting that the absolute impact of these incentives varies with the regions because they are built as percent of gross margins [equation (4)] which are heterogeneous among the regions. The intertemporal programme of the decision-maker is to maximise the present value of aggregate net farm income $PVI(\tau)$ discounted at the rate ρ from the first year of the projection t_1 to

the final time horizon T

$$\max_{\tau_1, \tau_2, \dots, \tau_{n_K}} \text{PVI}(\tau) = \max_{\tau} \sum_{t=t_1}^T \rho^{t-t_1} \cdot \sum_r \Pi_r(t), \quad (9)$$

under the reaction function (8) of the regional farmers, a biodiversity constraint (11) and a budgetary constraint (10)

$$\text{Bal}(t) \leq \text{Budg}_{\text{lim}}, \quad (10)$$

$$\text{Biod}(T) \geq B_{\text{lim}}. \quad (11)$$

Budgetary constraint (10) is related to the amounts spent and received by the public decision-maker for farming over the regions. This national balance is based on incentives τ as follows:

$$\text{Bal}(t) = \sum_r \sum_k \tau_k \cdot \text{gm}_{r,k} \cdot L_{r,k}(t). \quad (12)$$

Budgetary constraint (10) ensures that the budget balance (spent subsidies – perceived taxes) for farming does not exceed the yearly public envelope Budg_{lim} each year. In other words, the public decision-maker has a financial envelope³ which he can share between the different land-uses according to its objectives, constraints and instruments. Because of this positive amount, a policy only based on subsidies is potentially feasible. If the decision-maker decides to implement taxes, tax products are added to the balance and can potentially be used to pay extra subsidies when compared with a case only with the initial envelope.

The ecological requirement [equation (11)] is based on a conservation limit B_{lim} for the biodiversity score introduced previously in equation (3) and depending on abundances of species. It is imposed only at the temporal horizon T . Different values of B_{lim} can be tested and compared between the lowest value $B_{\text{lim}} = 0$, where the constraint is not active and the maximal feasible biodiversity⁴ denoted by B^* .

The solution of the maximisation problem [equation (9)] yields optimal incentives denoted by $\tau^*(B_{\text{lim}})$ which depend especially on biodiversity threshold B_{lim} .

2.3. Private, public and social biodiversity costs

The public policies induce public and private costs both at micro and macro levels as proposed by [Semaan et al. \(2007\)](#). Analysing such costs is helpful

3 Typically based on the current CAP budget as developed in Section 3.

4 This maximum B^* is defined by a biodiversity maximization with respect to the vector of fiscal incentives and under the budgetary constraint

$$B^* = \max \text{Biod}(\tau) \text{ under constr. } \text{Bal}(t) \leq \text{Budg}_{\text{lim}}. \quad (13)$$

for evaluating the economic performances of the different policies for the entire society, including public and private agents.

The aggregated private cost is based on the loss of farmers' income (including public transfers) due to biodiversity requirements. The private cost, denoted by $\text{PrC}(B_{\text{lim}})$, is computed as the difference between the maximum feasible present value of aggregate net farm income $\text{PVI}(\tau^*(0))$ without any biodiversity target and the optimal present value $\text{PVI}(\tau^*(B_{\text{lim}}))$ under biodiversity goal B_{lim}

$$\text{PrC}(B_{\text{lim}}) = \text{PVI}(\tau^*(0)) - \text{PVI}(\tau^*(B_{\text{lim}})). \quad (14)$$

In contrast, the public cost denoted by $\text{PuC}(B_{\text{lim}})$ evaluates the impact in terms of public balance of achieving particular biodiversity targets B_{lim} . It is the difference of the discounted sums of public monetary balances without and with biodiversity target. It depends on the biodiversity target B_{lim} as the net public revenue is itself a function of the optimal incentives $\tau^*(B_{\text{lim}})$. The public aggregate cost reads as follows:

$$\text{PuC}(B_{\text{lim}}) = \sum_{t=t_1}^T \rho^{t-t_1} (\text{Bal}^*(t, 0) - \text{Bal}^*(t, B_{\text{lim}})), \quad (15)$$

where $\text{Bal}^*(t, B_{\text{lim}})$ stands for the budget evaluated for the optimal policy $\tau^*(B_{\text{lim}})$ in the following sense

$$\text{Bal}^*(t, B_{\text{lim}}) = \sum_r \sum_k \tau_k^*(B_{\text{lim}}) \cdot \text{gm}_{r,k} \cdot L_{r,k}^*(t, B_{\text{lim}}). \quad (16)$$

$L_{r,k}^*(t, B_{\text{lim}})$ means the optimal land-uses determined by the farmers according to equation (8) under the optimal policy $\tau^*(B_{\text{lim}})$.

Defining as the sum of the private and public costs, the social cost $\text{SoC}(B_{\text{lim}})$ refers to the loss entailed by the biodiversity constraint of the farmer's income without transfers (taxes and subsidies). It measures the difference in terms of farming income between an optimal policy without any ecological constraint and an optimal environment-friendly policy

$$\text{SoC}(B_{\text{lim}}) = \sum_{t=t_1}^T \rho^{t-t_1} \sum_r \sum_k \text{gm}_{r,k} \cdot (L_{r,k}^*(t, 0) - L_{r,k}^*(t, B_{\text{lim}})). \quad (17)$$

The question whether these costs are positive or not is decisive for the acceptability of biodiversity requirements by farmers on one hand and by the whole society on the other hand, and thus for the adoption of eco-friendly agricultural policies.

3. The French case study

3.1. Context

This bio-economic modelling framework is applied to metropolitan France. France is split into $n_R = 620$ SAR. A SAR is part of a department (a major French administrative entity) characterised by an agro-ecological homogeneity. This consistency from both the ecological and economic points of view makes the SAR a relevant regional scale for economic and biodiversity models. Ecological and economic data are available from 2001 to 2008 (t_0). The policy scenarios are tested between $t_1 = 2009$ and $T = 2050$. Selecting a shorter time frame could consequently hide interesting long-term effects due to the inertia of the model. When compared with [Mouysset *et al.* \(2011a\)](#), the precision of the model has been reinforced thanks to a refined spatial scale (from regional to SAR) for every bio-economic data and a refinement of the classes used for the land-uses. In other words, the model better accounts for local particularities through the calibration process.

3.2. Biodiversity data

As regards biodiversity, we focus on common bird populations and related indicators ([Gregory *et al.*, 2004](#)). Although the metric and the characterisation of biodiversity remain an open debate ([MEA, 2005](#)), such a choice is justified for several reasons ([Ormerod and Watkinson, 2000](#)): (i) Birds lie at a high level in the trophic food chains and thus capture variations in the chains. (ii) Birds provide many ecological services, such as the regulation of rodent populations and pest control, thus justifying our interest in their conservation ([Sekercioglu, Daily and Ehrlich, 2004](#)). (iii) Their close vicinity to humans makes them a simple and comprehensive example of biodiversity for a large audience of citizens.

The French Bird Breeding Survey (STOC) database⁵ provides information related to the bird abundances across the whole country. Abundance values for each species are available⁶ for the period 2001–2008. Among the species monitored by this large-scale long-term survey, we selected $n_S = 34$ species which have been classified according to their habitat requirements at an European scale ([European Bird Census Council, 2007](#)). Table 2 lists the 14 habitat generalist species and the 20 farmland specialist species used as a reference by the European Union ([Gregory *et al.*, 2004](#)).

3.3. Economic data

Agro-economic data arise from the French agro-economic classification orientation technico-economique (OTEX) developed by the French Farm

5 See the Vigie-Nature website <http://www2.mnhn.fr/vigie-nature/>. Standardized monitoring of spring-breeding birds at $1,747 \times 2 \times 2 \text{ km}^2$ plots across the whole country. Details of the monitoring method and sampling design can be found in [Jiguet \(2009\)](#).

6 For each species, a spatial interpolation of the abundance data is performed to obtain relative abundance values for each possible square in the country ([Doxa *et al.*, 2010](#)). We then average the abundance values at the SAR scale.

Table 2. List of the 20 farmland and 14 generalist bird species

20 Farmland bird species	14 Generalist bird species
(1) Buzzard <i>Buteo buteo</i>	(1) Blackbird <i>Turdus merula</i>
(2) Cirl Bunting <i>Emberiza cirlos</i>	(2) Blackcap <i>Sylvia atricapilla</i>
(3) Corn Bunting <i>Emberiza calandra</i>	(3) Blue Tit <i>Parus caeruleus</i>
(4) Grey Partridge <i>Perdix perdix</i>	(4) Carrion crow <i>Corvus corone</i>
(5) Hoopoe <i>Upupa epops</i>	(5) Chaffinch <i>Fringilla coelebs</i>
(6) Kestrel <i>Falco tinnunculus</i>	(6) Cuckoo <i>Cuculus canorus</i>
(7) Lapwing <i>Vanellus vanellus</i>	(7) Dunnock <i>Prunella modularis</i>
(8) Linnet <i>Carduelis cannabina</i>	(8) Great Tit <i>Parus major</i>
(9) Meadow Pipit <i>Anthus pratensis</i>	(9) Green Woodpecker <i>Picus viridis</i>
(10) Quail <i>Coturnix coturnix</i>	(10) Golden Oriole <i>Oriolus oriolus</i>
(11) Red-backed Shrike <i>Lanius collurio</i>	(11) Jay <i>Garrulus glandarius</i>
(12) Red-legged Partridge <i>Alectoris rufa</i>	(12) Melodius Warbler <i>Hippolais polyglotta</i>
(13) Rook <i>Corvus frugilegus</i>	(13) Nightingale <i>Luscinia megarhynchos</i>
(14) Skylark <i>Alauda arvensis</i>	(14) Wood Pigeon <i>Columba palumbus</i>
(15) Stonechat <i>Saxicola torquatus</i>	
(16) Whinchat <i>Saxicola rubetra</i>	
(17) Whitethroat <i>Sylvia communis</i>	
(18) Wood Lark <i>Lullula arborea</i>	
(19) Yellowhammer <i>Emberiza citronella</i>	
(20) Yellow Wagtail <i>Motacilla flava</i>	

Accounting Data Network⁷ and the Observatory of Rural Development (ODR).⁸ This agency distinguishes between $n_K = 14$ classes of land-use named OTEX detailed in Table 1. Each SAR is a specific combination of these OTEX. The areas dedicated to the 14 OTEX land-uses and the associated fiscal bases (tax return) used as proxies of gross margins for the years 2001–2008 are available on the ODR website upon private request. Gross margin is an economic index widely used in agricultural economics (Lien, 2002). The 14 land-uses can potentially be chosen by the representative farmers at the SAR scale and are taken into account in the computation of their regional income [equation (4)]. They also impact the habitat variables involved in the ecological model [equation (2)].

To accelerate numerical computations, the public decision variables τ_k are restricted to only two incentives: the ‘cereal’ incentive τ_{cop} is dedicated to arable lands [Otex (1) in Table 1] and the ‘grassland’ incentive τ_{grass} is applied to non-intensive grassland systems [Otex (4), (5), (6) and (7) in Table 1]. The gross margins $gm_{r,k}$ are computed as the temporal mean of the historical gross margins:

$$gm_{r,k} = \frac{1}{8} \sum_{t=2001}^{2008} gm_{r,k}(t). \tag{18}$$

7 <http://ec.europa.eu/agriculture/rica/>

8 <https://esrcarto.supagro.inra.fr/intranet/>

The budgetary limit of constraint (10) is set at the 2008 level of French budget in the European Common Agricultural Policy (CAP) namely

$$\text{Budg}_{\text{lim}} = 1.4 \times 10^9 (\text{euros } 2008).$$

3.4. Model calibration

The agro-ecological parameters $R_{s,r}$, $a_{s,r,k}$ and $b_{s,r}$ introduced in equations (1)–(2) and the economic parameter ε of equation (6) is determined by a calibration based on a least square method. Hence, the gaps between the observed outputs and the outputs derived from the model are minimised. The considered outputs of the model are the land-use values $L_{r,k}(t)$ for the economic model and the bird abundances $N_{s,r}(t)$ for the ecological model, as detailed in [Mouysset, Doyen and Jiguet \(2013\)](#) and [Mouysset *et al.* \(2011a\)](#). The discount rate is set to $\rho = 4\%$.

3.5. Biodiversity indicators

The biodiversity indicators used in this study are the FBI and the CTI, both evaluated in final year $T = 2050$. The FBI has been adopted by the European Community as the official environmental index, especially to analyse structural changes in biodiversity ([Balmford, Green and Jenkins, 2003](#)). The relevance of the FBI to reflect the response of farmland biodiversity to agricultural intensification has been shown in [Doxa *et al.* \(2010\)](#) and [Mouysset, Doyen and Jiguet \(2012\)](#). We compute the FBI at the national scale with 20 farmland specialist species for each SAR

$$\text{FBI}(t) = \prod_{s \in \text{Specialist}} \left(\frac{N_s(t)}{N_s(2008)} \right)^{1/20}, \quad (19)$$

where $N_s(t) = \sum_{r=1}^{620} N_{s,r}(t)$ stands for the total abundance of species s over the $n_R = 620$ regions.

The CTI informs on the average trophic level of a community as in [Pauly *et al.* \(1998\)](#) and [Mouysset, Doyen and Jiguet \(2012\)](#). The CTI here integrates both the 14 generalist species and the 20 farmland specialist species (Table 2). At the regional level, it is computed as the arithmetic mean of the exponential of the species trophic level⁹ STI_s weighted by their relative abundances. National CTI is the arithmetic mean of the 620 regional CTI as follows

$$\text{CTI}(t) = \frac{1}{620} \cdot \sum_r \sum_s \frac{N_{s,r}(t)}{N_{\text{tot},r}(t)} \cdot \exp(\text{STI}_s), \quad (20)$$

where $N_{\text{tot},r}(t) = \sum_{s=1}^{34} N_{s,r}(t)$ stands for the total abundance of birds in a region r .

⁹ See [Mouysset *et al.* \(2012\)](#) for the individual trophic index of species. Typically, insectivorous and carnivorous species have higher trophic levels than granivorous species.

The exponential function is used to better contrast bird communities with high trophic levels as in [Mouysset, Doyen and Jiguet \(2012\)](#).

4. Results

4.1. Optimality curves

Figure 2 is a first illustration of the bio-economic performances. It displays the maximal present values $PVI(\tau^*(B_{lim}))$ of aggregate net farm income as computed in equation (9) under different biodiversity constraints B_{lim} (data available on row 1 of [Tables 3 and 4](#)). The biodiversity metrics is the FBI on the top, in Figure 2a, while it is the trophic index CTI on the bottom, in Figure 2b. The red diamond corresponds to the optimal policy $\tau^*(0)$ without biodiversity constraint in both cases. The green marks in Figure 2a and b represent the present values of incomes for the policies $\tau^*(FBI^*)$ and $\tau^*(CTI^*)$ with the maximal biodiversity score. We observe that the two optimal curves both decrease with respect to biodiversity target B_{lim} although the shapes differ. In particular, the curve obtained with the FBI in Figure 2a mixes convex and concave decreasing parts. In contrast, the curve obtained with the CTI constraint in Figure 2b is characterised by a global concavity, especially strong for large biodiversity level B_{lim} . Hence, the increase in the CTI target has a limited impact on the present value of farming incomes PVI for levels lower than $CTI = 6.43$. Beyond this threshold, the economic loss due to biodiversity becomes more important.

4.2. Optimal public incentives

Rows 2 and 3 of [Tables 3 and 4](#) depict the optimal incentives τ_{cop} for crops and τ_{grass} for grasslands with increasing biodiversity goals B_{lim} . When the biodiversity constraint is not active ($B_{lim} = 0$), the optimal incentive $\tau^*(0)$ consists in subsidies for both crops and grasslands. The intuition for such a result is that only economic scores matter in that case and thus subsidies, paid with the initial envelope, perform better by directly improving incomes. When the biodiversity constraint becomes active ($B_{lim} > 0$), a decrease in the cereal subsidies τ_{cop} occurs with the biodiversity objective for both ecological metrics. In particular, for the strongest biodiversity targets, the incentive on crops τ_{cop} becomes a tax. In contrast, the subsidy for extensive grasslands τ_{grass} remains at a high level globally (except for the policy with the most stringent CTI constraint, namely CTI^*). These optimal patterns to satisfy biodiversity objectives globally stress the need to promote extensive grassland at the expense of crops. According to the selected ecological indicator, this pattern is more or less emphasised.

4.3. Impact of the constraints

The rows 4 and 5 of [Table 3 and 4](#) focus on the constraints. They display the gap¹⁰ between the obtained performances on Bal and $Biod$ and the related bounds $Budg_{lim}$ and B_{lim} . It turns out that the gap for the ecological constraint

10 $\Delta Budg = \sum_{t=t_1}^T \rho^{t-t_1} ((Budg_{lim} - Bal^*(t))/Budg_{lim})$ and $\Delta Biod = (Biod(2050) - B_{lim})/B_{lim}$.

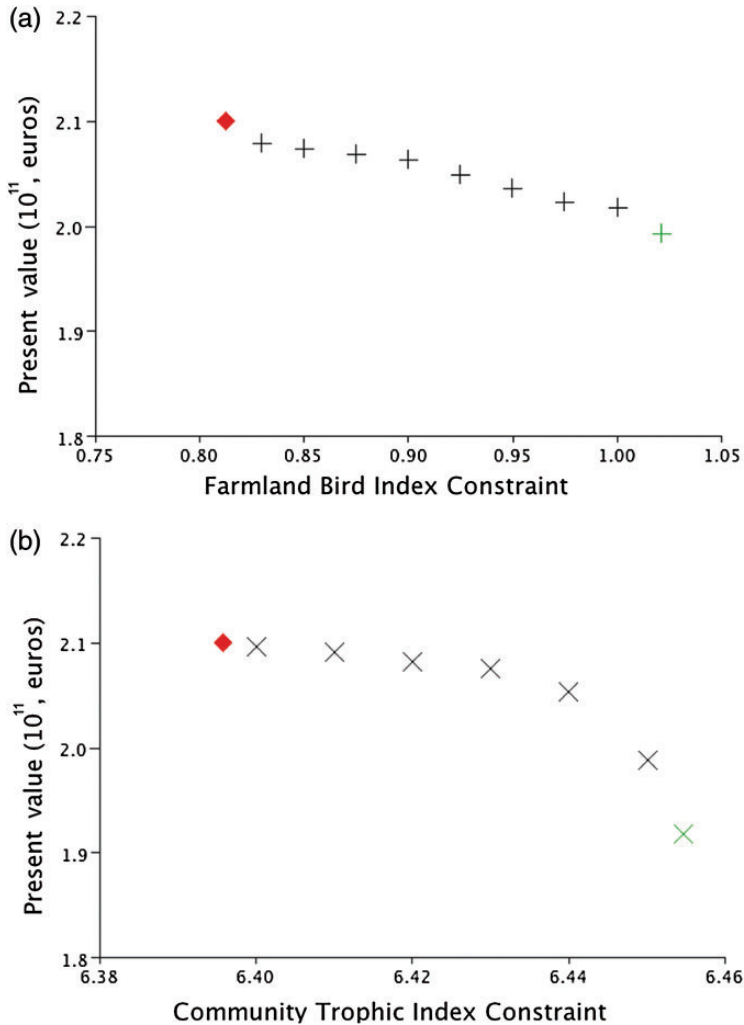


Fig. 2. Optimal present values of aggregate net farm income $PVI^*(B_{lim})$ with respect to the biodiversity constraint B_{lim} . In (a) with the FBI (2050) biodiversity indicator and in (b) with the CTI (2050) biodiversity indicator. The extreme policies $\tau^*(0)$ and $\tau^*(B^*)$ are in red (dark grey in B&W version) and green (pale grey in B&W version), respectively.

is close to 0 while the one for the financial constraint is strongly positive. This points out that the biodiversity constraint is binding while the budgetary constraint is not binding at each time t , yielding some room of manoeuvre in terms of public budget. In other words, when the biodiversity target increases, stronger changes in optimal land-uses are required, including mainly the development of grasslands at the expense of croplands as shown by the optimal incentives in Tables 3 and 4. In the context of rigidity and inertia captured by constraint (6) at the farmer scale, these optimal land-uses changes require longer time to be implemented and achieved. In that context, the annual

Table 3. Optimal results, binding levels of constraints and sensitivity analysis associated to different biodiversity targets B_{lim} based on FBI

FBI _{lim}	0	0.825	0.85	0.875	0.9	0.925	0.95	0.975	1	FBI*
PVI* (10 ¹¹ €)	2.100	2.078	2.074	2.068	2.064	2.050	2.036	2.023	2.017	1.993
τ_{cop}^*	0.47	0.27	0.23	0.23	0.14	0.02	-0.06	-0.19	-0.25	-0.54
τ_{grass}^*	0.52	0.58	0.59	0.58	0.61	0.62	0.61	0.62	0.62	0.63
Δ Budg (%)	8	15	16	18	20	25	30	32	34	41
Δ Biod (%)	/	1.7	4.3	1.2	0.5	1.5	0.1	2.9	1.7	0
t_{lim}	2026	2027	2039	2039	2040	2041	2042	2042	2043	2043
$\Delta\tau_{cop}^*$ (%)	/	-	-	-	-	-	-	-	-	-
$\Delta\tau_{grass}^*$ (%)	/	≈0	≈0	≈0	≈0	≈0	≈0	≈0	≈0	≈0

The first row presents the optimal present values of aggregate net farm income PVI*. The last two rows stand for the optimal cereal τ_{cop}^* and grassland τ_{grass}^* incentives. The rows 4 and 5 measure the binding level of the budget and biodiversity constraints, Δ Budg and Δ Biod, respectively. The row 6 presents the time t_{lim} when the budgetary constraint is saturated. Finally, the last two rows present the signs of the derivatives of τ^* related to B_{lim} ($\Delta\tau_{cop}^*$ and $\Delta\tau_{grass}^*$).

Table 4. Optimal results, binding levels of constraints and sensitivity analysis associated to different biodiversity targets B_{lim} based on CtI

CTI _{lim}	0	6.40	6.41	6.42	6.43	6.44	6.45	CTI*
PVI* (10 ¹¹ €)	2.100	2.096	2.090	2.081	2.075	2.053	1.988	1.918
τ_{cop}^*	0.47	0.42	0.37	0.33	0.23	0.20	0.34	-0.02
τ_{grass}^*	0.52	0.54	0.56	0.56	0.59	0.54	0.23	0.23
Δ Budg (%)	8	10	11	14	17	25	53	78
Δ Biod (%)	/	0.19	0.06	0.0007	0.03	0.01	0.0005	0
t_{lim}	2026	2028	2037	2038	2040	2040	2042	2043
$\Delta\tau_{cop}^*$ (%)	/	-	-	-	-	-	-	-
$\Delta\tau_{grass}^*$ (%)	/	≈0	≈0	≈0	≈0	≈0	-	-

The first row presents the optimal present values of aggregate net farm income PVI*. The last two rows stand for the optimal cereal τ_{cop}^* and grassland τ_{grass}^* incentives. The rows 4 and 5 measure the binding level of the budget and biodiversity constraints, Δ Budg and Δ Biod, respectively. The row 6 presents the time t_{lim} when the budgetary constraint is saturated. Finally, the last two rows present the signs of the derivatives of τ^* related to B_{lim} ($\Delta\tau_{cop}^*$ and $\Delta\tau_{grass}^*$).

budgetary constraint is only saturated at a specific year t_{lim} which depends on the biodiversity goal B_{lim} . The row 6 of Tables 3 and 4 highlights how the saturation for the budgetary constraint occurs later when the biodiversity goal increases. More globally, the public expenditure $Bal^*(t)$ is smaller and thus more distant from the budget limit $Budg_{lim}$ with a more demanding biodiversity target B_{lim} , as depicted by the values of Δ Budg in Tables 3 and 4.

To go further, the impact of the ecological constraint on the optimal incentives is detailed on the rows 7 and 8 of Tables 3 and 4. Thus, a sensitivity analysis is given with an approximation of the derivatives of optimal incentives τ_{cop}^* and τ_{grass}^* with respect to B_{lim} . More specifically are computed the variation

rates¹¹ of τ^* with a numerical precision at 10^{-2} . We observe that the derivatives of τ_{cop}^* are negative while the derivatives of τ_{grass}^* equal 0. This stresses that the budget distributed to grasslands remains stable, whereas the budget dedicated to crops decreases with the biodiversity target. In other words, the increase in the biodiversity constraint generates a public budget gain due to the decrease of the incentives devoted to crops.

4.4. Optimal land-uses

Figure 3 illustrates the proportions of the utilised agricultural area dedicated to the extensive grassland systems for the three extreme policies: the optimal policy $\tau^*(0)$ without a biodiversity requirement in Figure 3a, the optimal policy $\tau^*(\text{FBI}^*)$ providing the best FBI score in Figure 3b and the optimal policy $\tau^*(\text{CTI}^*)$ associated with the best CTI performance in Figure 3c. In line with the optimal incentives examined previously, the optimal strategy $\tau^*(\text{FBI}^*)$ mixing high subsidies on grassland and high taxes on crops promotes grassland through an increase in SAR with extensive grassland proportions when compared with the situation without a biodiversity constraint. As regards the optimal policy $\tau^*(\text{CTI}^*)$ combining moderate subsidies on grassland and low taxes on crops, it favours mixed land-uses with moderate grassland proportions (in blue) when compared with the $\tau^*(0)$ option.

4.5. National costs

Figure 4 plots the global social costs $\text{SoC}(B_{\text{lim}})$ by detailing the public $\text{PuC}(B_{\text{lim}})$ and the private $\text{PrC}(B_{\text{lim}})$ costs for the different optimal public strategies. Figure 4 first highlights the fact that the private cost (dashed blue lines) increases while the public cost (dotted-dashed red lines) decreases. The fact that biodiversity requirements are detrimental to private costs derives from the optimality curves mentioned in Section 4.1 where optimal present value of aggregate net farm income is reduced by biodiversity constraints. In contrast, it turns out that public costs are restricted with biodiversity objectives because the agricultural budget balance is improved accordingly. These opposite effects on private and public costs are mainly due to the reduction in subsidies or the presence of taxes on crops $\tau_{\text{cop}}^* < 0$ in the optimal policies when the biodiversity goal is more demanding, as captured by Tables 3 and 4. In other words, taxes are good for the public budget while they alter private incomes, as expected. As regards the total social cost (solid black lines), of interest is the fact that it remains rather flat. This suggests that biodiversity requirements do not necessarily penalise the total social costs. The FBI and CTI patterns are qualitatively close in this sense, although a slight decrease in social costs occurs for the highest CTI constraints.

11 $\Delta \tau_{\text{cop}}^* = (\tau_{\text{cop}}^*(B_{\text{lim}} + \Delta B_{\text{lim}}) - \tau_{\text{cop}}^*(B_{\text{lim}})) / \Delta B_{\text{lim}}$
 $\Delta \tau_{\text{grass}}^* = (\tau_{\text{grass}}^*(B_{\text{lim}} + \Delta B_{\text{lim}}) - \tau_{\text{grass}}^*(B_{\text{lim}})) / \Delta B_{\text{lim}}$.

and

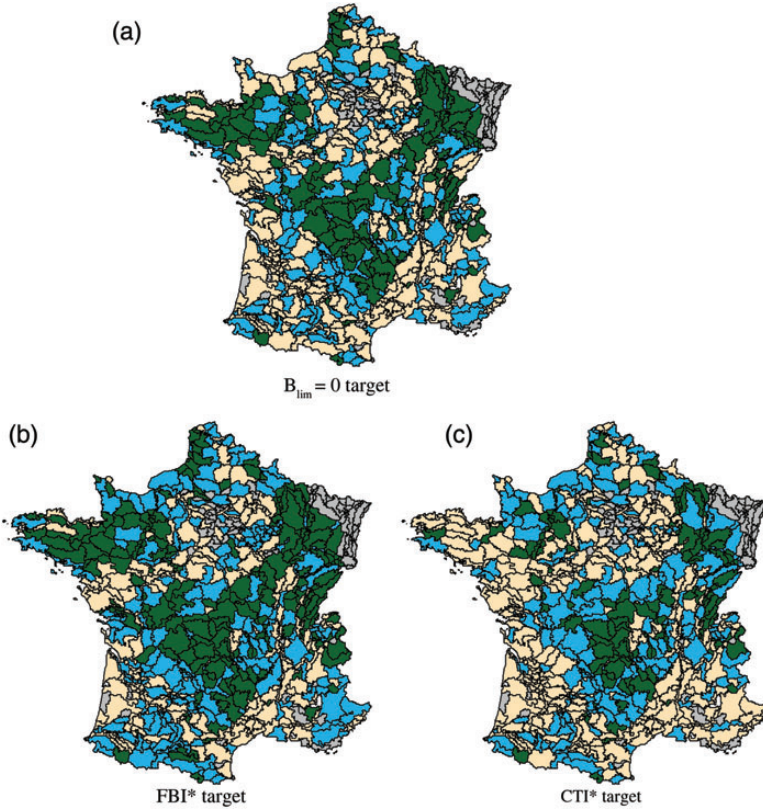


Fig. 3. Proportions of non-intensive grassland land-use $1/\bar{L}_r \sum_{k=4}^7 L_{r,k}(2050)$ at the SAR scale for optimal policies under extreme biodiversity targets B_{lim} . In green (dark grey in B&W version): 100–45 per cent, in blue (medium grey in B&W version): 45–10 per cent, in yellow (pale grey in B&W version): 10–0 per cent.

4.6. Regional costs

Figure 5 details the regional total social costs¹² $SoC_r(B_{lim})$ at the regional scale for several public policies $\tau^*(B_{lim})$. Figure 5a stands for the extreme policy $\tau^*(0)$ without a biodiversity target. Figure 5b–e represents several τ^* policies with two intermediate B_{lim} for each biodiversity indicator. Finally, Figure 5f and g depicts the $\tau^*(B^*)$ policies. It turns out that the regional social total costs remain stable among the optimal policies confirming the result obtained at the macro scale. In other words, the biodiversity constraint does not affect the social cost, even at the more micro level. However, it can be noted that this social cost differs between regions.

Figure 6 presents the distribution of the regional total social costs $SoC_r(B_{lim})$ between the regional public costs¹³ $PuC_r(B_{lim})$ (in red) and the regional private

¹² $SoC_r(B_{lim}) = \sum_{t=t_1}^T \rho^{t-t_1} \sum_k gm_{r,k} \cdot (L_{r,k}(t, 0) - L_{r,k}^*(t, B_{lim}))$.

¹³ $PuC_r(B_{lim}) = \sum_{t=t_1}^T \rho^{t-t_1} (\text{Bal}_r^*(t, 0) - \text{Bal}_r^*(t, B_{lim}))$.

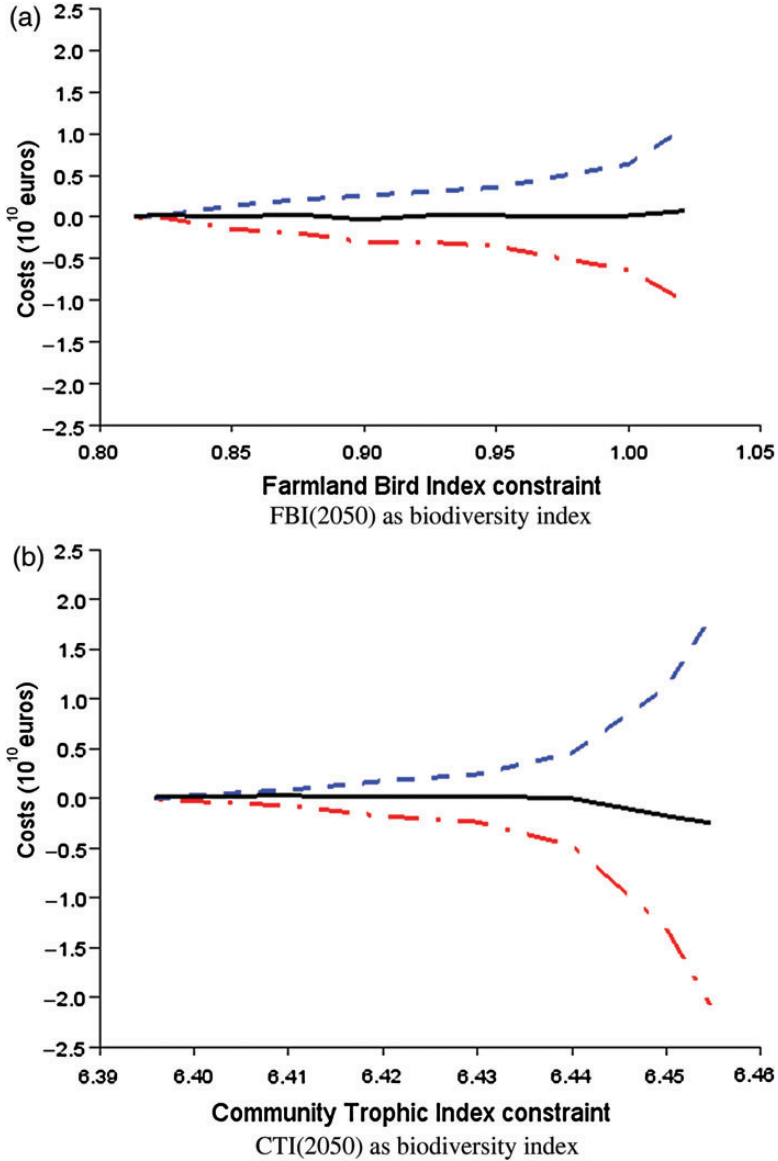


Fig. 4. Costs due to the biodiversity constraints B_{lim} : the social costs $SoC(B_{lim})$ by the solid black lines, the public costs $PuC(B_{lim})$ by the dashed-dotted red lines and the private costs $PrC(B_{lim})$ by the dashed blue lines.

costs¹⁴ $PrC_r(B_{lim})$ (in blue). From equation (14), there is no private biodiversity costs for the economic-oriented policy $\tau^*(0)$. We therefore start directly with the τ^* policies with two medium B_{lim} for each indicator. Pink represents

14 $PrC_r(B_{lim}) = \sum_{t=t_1}^T \rho^{t-t_1} (\Pi_r^*(t, 0) - \Pi_r^*(t, B_{lim}))$.

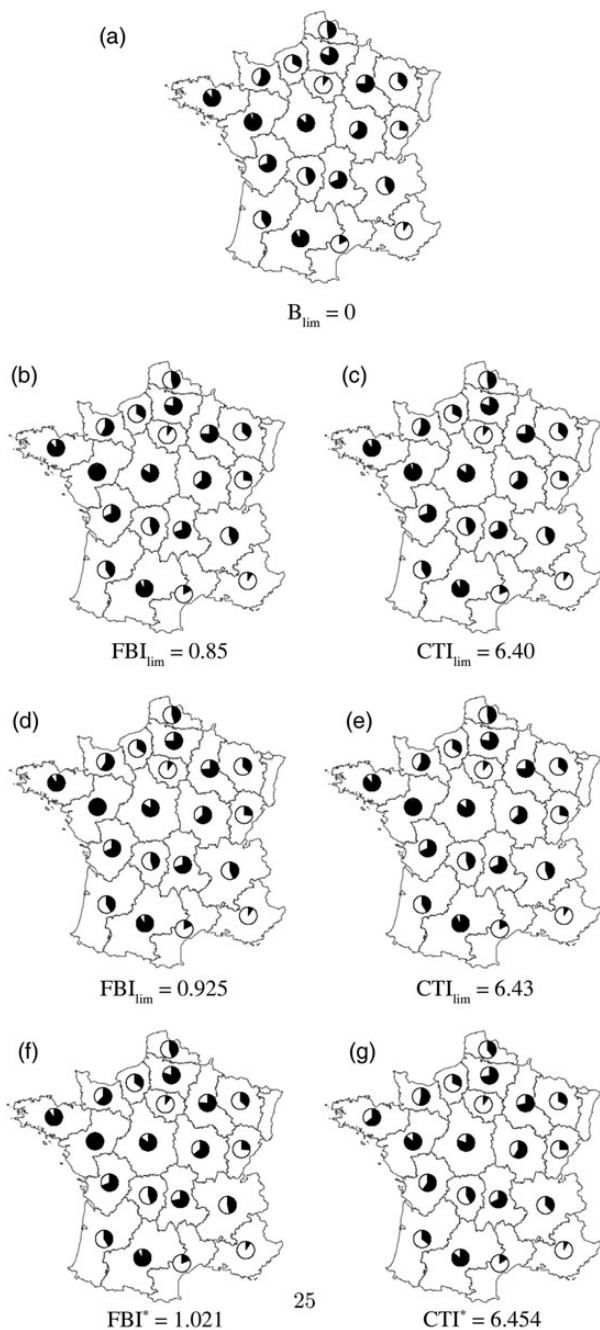


Fig. 5. Regional total social costs $SoC_r(B_{lim})$ in black under several biodiversity targets B_{lim} . The pie-charts present the position in $(-10^9; 10^9)$, quarter pie-chart stands for -5×10^8 EUR, half pie-chart stands for 0 EUR, three-quarters pie-chart stands for 5×10^8 EUR. On the left the FBI and on the right the CTI for the biodiversity index.

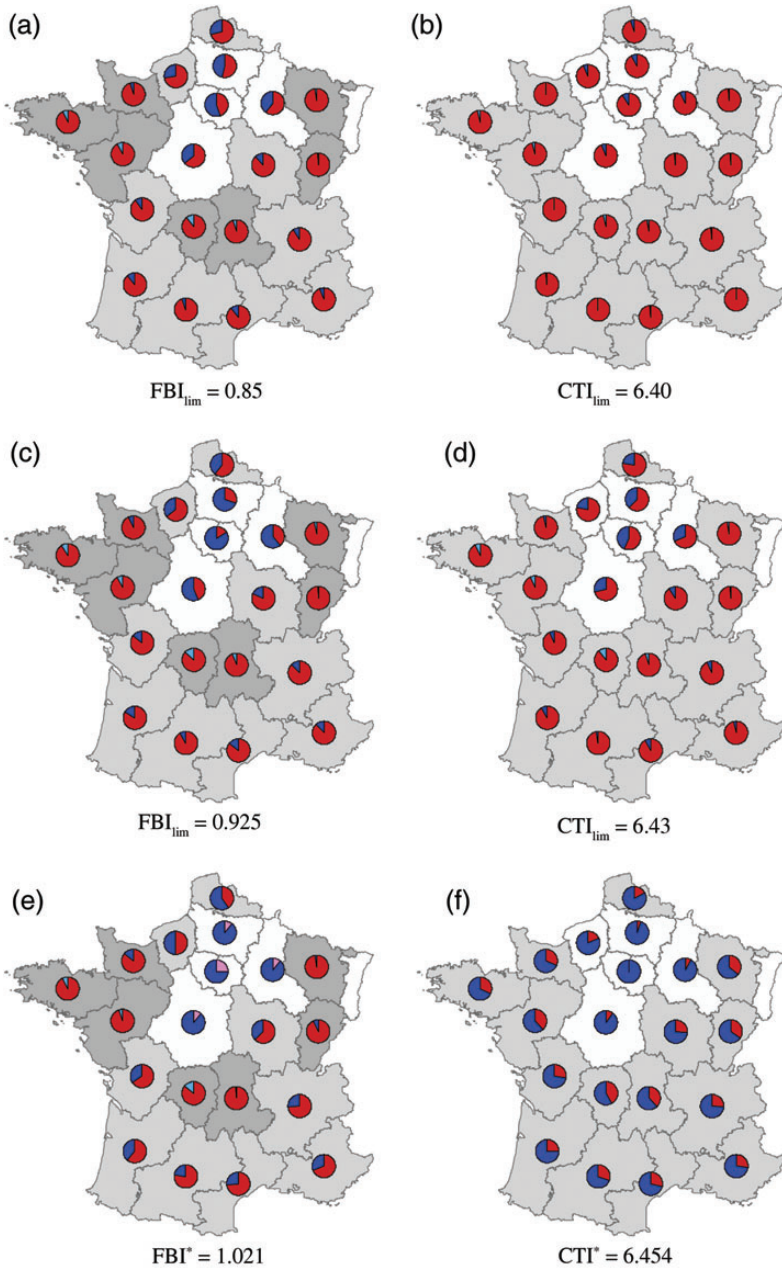


Fig. 6. Regional public $PuC_r(B_{lim})$ (in red, dark grey in B&W version) and private $PrC_r(B_{lim})$ (in blue, medium grey in B&W version) costs under several biodiversity targets B_{lim} . Pink (pale grey instead of dark grey, in the left part of the pies in B&W version) stands for negative public costs and pale blue (pale grey instead of medium grey, in the right part of the pies in B&W version) negative private costs. Grey (respectively, pale grey, white) regions present stable (respectively, intermediary stable, instable) costs.

negative public costs, where taxes exceed subsidies. Pale blue is associated with negative private costs, i.e. the regional farmer income is larger than under the $\tau^*(0)$ policy without a biodiversity requirement. Moreover, strong grey (pale grey, white respectively) regions have very stationary (intermediary stationary, unstable respectively) costs among the optimal strategies.

Although the policies affect the regions differently, the patterns are qualitatively similar in every region and for the two indicators: when the biodiversity constraint is more stringent, the public cost decreases and the private cost increases as already captured by Figure 4. As also suggested by Figure 4, there is a strong complementarity between the two costs: regions where the public cost decreases strongly are those where the private cost grows strongly. Typically, the four regions in white in Figure 6 (Ile de France, Centre, Picardie, Champagne-Ardennes), which have a historically strong specialisation in arable lands, are the most affected by 'green' policies. Hence, for the strongest biodiversity targets, they generate an important public gain with taxes on crops.

5. Discussion

5.1. The bio-economic trade-off

The bio-economic model developed in this study leads to the design of optimal policies with respect to budgetary and biodiversity constraints. The optimal strategies maximise the national intertemporal farming income (or equivalently minimise the national private cost) under a biodiversity target with a fixed financial budget. The analysis of the optimal policies with different objectives of biodiversity provides bio-economic optimality curves. For the tested biodiversity indicators, the bio-economic trade-off is strictly negative, suggesting that integrating biodiversity goals in agricultural policies entails a loss of earnings for farmers. This result is strongly consistent with the literature as in Polasky *et al.* (2005), Drechsler *et al.* (2007), Lewis *et al.* (2011) and Barraquand and Martinet (2011), and thus constitutes another illustration of the antagonism between economic and ecological performances in agricultural public policies.

However, our study brings new insights to this debate. The first one is that the shape of the bio-economic trade-off depends on the selected indicator, and the strictly concave form cannot be generalised as the standard shape. In particular, contrary to most curves displayed in the literature (Drechsler *et al.*, 2007; Polasky *et al.*, 2005, 2008), the optimality curve for the FBI does not display a clear concave behaviour. The FBI relies on the specialist species. Because the grassland specialist species are particularly sensitive, the increase in grasslands at the expense of croplands underlying the optimal policy [as shown by the 3(b)] globally foster the FBI [as shown by the 3(b)]. However the increase is not regular because some species involved in the FBI as the skylark are specialist of crops.

The CTI does not behave in the same way. The improvement of CTI indeed requires a relative heterogeneity of farming land-uses [as highlighted by the map 3(c)] because the CTI favours high level trophic species among which many enjoy both grasslands and crops (Mouysset, Doyen and Jiguet, 2012). Typically,

the species with the highest trophic levels are the Buzzard, the Kestrel, the Red-backed Shrike and the Hoopoe. Among them, only the third is specialised in grasslands. Therefore, the optimal management with the CTI score implies more mixed land-uses or public incentives when compared with public policies involving FBI.

The diversity of optimality curves and underlying trade-offs highlights the difficulty of selecting a single policy among the optimal ones. This outcome is important for decision-makers, especially since the FBI indicator, which does not fit with the strictly concave case, is currently the one which has been adopted by the European Community as the official environmental index (Balmford, Green and Jenkins, 2003). In contrast to concave trade-off, there is here no 'balanced' choice. In the concave case, balanced choice lies at the corner of concavity, where it is possible to moderately improve the biodiversity indicator without implying strong private costs for farming (Polasky *et al.*, 2005; Barraquand and Martinet, 2011; Lewis *et al.*, 2011). Our context of multiple forms of trade-offs and the lack of a clear balanced choice suggests the adoption in future works of a multi-criteria approach to simultaneously account for the numerous facets and metrics of biodiversity.

5.2. Benefits of policies with biodiversity goals

The first benefit of policies with biodiversity goals is obviously the improvement of biodiversity performance. But public and social costs give insight into a second benefit. First, it turns out that the social cost does not rise in response to biodiversity requirements. This suggests that biodiversity is not detrimental to overall (macro) economic performance. Second, such an assertion is reinforced by the study of public cost. We observe that, for both biodiversity indicators, the increase in biodiversity objectives leads to a decrease in the agricultural public budget. In other words, the policies with demanding biodiversity goals entail a budgetary benefit. The increase in biodiversity target needs stronger changes in land-uses, in particular development of grasslands at the expense of croplands. The optimal land-uses require longer time to be achieved and yield gains in term of public expenditure. Therefore, it is possible to improve biodiversity performances while strengthening the budget balance. The stability of social costs also shows bio-economic synergies since it seems possible to improve biodiversity scores with similar social costs. This paves the way for an alternative (apart from ecosystem services) to the trade-off broadly emphasised in the literature between biodiversity and economic performance (Polasky *et al.*, 2005; Drechsler *et al.*, 2007; Barraquand and Martinet, 2011; Lewis *et al.*, 2011).

These effects are mainly implied by the taxes on crops in the optimal policies with biodiversity objectives. Understanding the role of taxes is not a straightforward affair because in the decision-making programme which maximises the present value of aggregate net farm income, subsidies should be preferred to taxes as they directly enhance the revenues. In fact, the use of taxes on crops in the optimal policies arises from the necessity to extensively develop grasslands at the expense of crops to comply with demanding ecological constraints. This

emphasises the importance of combining tools (taxes and subsidies) and levers (grasslands and croplands) to deal effectively with such a heterogeneous bio-economic system: heterogeneity in economic contexts (different profitability for land-uses among the regions), heterogeneity in biodiversity states (heterogeneity of land-use reactions of species according to the regions, heterogeneity of impacts of land-uses on population dynamics according to the initial states).

However it is well known that these taxes raise questions in terms of acceptability. At this stage, a first argument is that according to the structure of the economic data which implicitly includes current public policies, taxes on croplands in this model can be more interpreted as a reduction in the current subsidies on croplands. Second, because the objective of such public policies is to financially support farmers to fulfil the biodiversity constraint, this budgetary benefit could be redistributed to farmers in a second mechanism in order to compensate their loss of income. By reducing these private costs, their acceptability for adopting biodiversity goals in agricultural policies should be enhanced. However, this financial redistribution of the public gain questions equity between agents, or the spatial scale of redistribution. The regional analysis of the different costs provides a first insight into the second benefit redistribution. Indeed, the study shows that the stability of the total cost with respect to the biodiversity target also holds true at the regional scale. The policies do not affect all the regions with the same intensity but a gain between public and private costs is obtained for each region. As the regions with private losses are also those where the public cost decreases, a first redistribution mechanism could be tested at the regional scale. The budgetary benefit could be also distributed to other sets of land-uses regarding to other environmental issues (erosion, water pollution . . .).

5.3. Perspectives

The objective of this study is to examine the role played by biodiversity goals on agricultural policies and symmetrically to help conservation biology to take socio-economic issues into account. In this vein, ecological-economic modelling is a fruitful framework to bring together social and natural sciences in order to tackle biodiversity management issues (Cooke *et al.*, 2009) especially within an agro-ecological and terrestrial context. By stylising the agro-ecological system, this kind of modelling leads to both improvements in understanding and reinforcement of decision-making supports by fostering policy effectiveness (Mouysset *et al.*, 2011b). The integration of dynamics and spatialisation of the processes reinforces their relevance. Moreover, the relative simplicity of the bio-economic mechanisms underlying the model together with its multi-scale perspective should make it easily transferable to other case studies and other biodiversity taxa.

The results presented in this paper should, however, be viewed as suggestive rather than predictive elements. Some improvements could have a positive impact on the design of relevant policies and should be integrated into future developments. The ecological dynamics could be refined by considering interactions both between species and between regions with meta-population models

(Ferrière *et al.*, 1996). In this perspective, taking into account more explicit spatial processes within the bio-economic model should reinforce the derived assertions. For example, accounting for the level of landscape fragmentation which affects both biodiversity dynamics (Tschardt *et al.*, 2005) and agricultural land-use policies (Polasky *et al.*, 2008; Hartig and Drechsler, 2009) should be a major task. The use of other biodiversity metrics together with multi-criteria approaches should also strengthen the statements. In particular paying attention to ecosystem services (MEA, 2005) should expand biodiversity benefits and synergies between economic and ecological performances. From the economic point of view, it would be accurate to account for more macro-economic processes and price mechanisms. Typically, future rents of agricultural activities can vary according to the influence of fuel prices or technical progress. In line with this, a relevant objective would consist in expanding the myopic behaviours of farmers into optimal intertemporal strategies. The role played by transaction costs for the implementation of taxes should also be investigated. Explore the role played by some other land-uses, chosen in consistence with the new priorities of the CAP, could also constitute an interesting development. Finally, allowing for dynamic incentives instead of fixed incentives could be a relevant way to improve the effectiveness of agricultural strategies as in Hartig and Drechsler (2009). In parallel to these developments, it could be interesting to confront this work about present value of agricultural income (which consistent with the specific context of the European agricultural policy) to a more classical approach. For instance, we could study the optimal land-use distribution under an economically sound objective function excluding public incentives. This would be to maximise biodiversity subject to a constraint on social cost,¹⁵ or to minimise social cost subject to a minimum constraint on biodiversity.¹⁶

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15 max Biod_t, under the constraints Bal ≤ Budg_{lim} and SoC ≤ SoC_{lim}.

16 min SoC_t, under the constraints Bal ≤ Budg_{lim} and Biod ≥ B_{lim}.

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