Agricultural Public Policy : Green or sustainable ?

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Abstract

The future of agriculture constitutes a major challenge to the achievement of sustainable development. There are new perspectives on greening(focusing on ecological objectives) and sustainability (combining both ecological and social goals). Academic papers rather study the ecological efficiency of agricultural public policies, while real public policies, such as in the European Common Agricultural Policy, examine both ecological and social considerations. The objective of this paper is to consider economic, social and ecological objectives within the design of agricultural public policies. Using a bio-economic model applied to France, we compare different optimal public strategies. We show that, when the biodiversity objectives are either very limited or very demanding, grassland subsidies are the best instruments from both green and sustainable points of view. However for medium objectives, reducing crops subsidies is the cheapest way to green the CAP, while subsidies on grasslands are the only strategy from a sustainability perspective. Our work highlights new trade-offs related to policy implementation, such as social acceptance or technical difficulties, and the spatial equity of performance among regions.

Keywords: Agriculture, Biodiversity, Public policy instrument, Poverty, Bird, Land-use, Cost-effectiveness

1. Introduction

Developing sustainable agriculture will be a major challenge in the coming decades, and particularly with regard to the impact of agriculture on the environment (Tilman *et al.* 2002). Changes in agricultural practices, including intensive of cropping, landscape homogenization, loss of semi-natural elements, mechanization and intensive use of inputs, have had several consequences such as increased water pollution (Carpenter *et al.* 2012, Volk *et al.* 2009, Tong & Chen 2002) and the loss of biodiversity (Foley *et al.* 2005, Tscharntke *et al.* 2005, Tilman *et al.* 2001, Chamberlain *et al.* 2000). In particular, consequences for biodiversity are widespread since many taxa are affected: see Flowerdew & Kirkwood (1997) for mammals, Sotherton & Self (2000) for plants and Donald *et al.* (2001) for birds. These negative effects are due mainly to degradation of habitat quality altering nesting success and survival (Benton *et al.* 2003). As a result, ecological considerations are increasingly being integrated into agricultural policy. For example, in Europe, such considerations have been introduced since the 90's through the European Common Agricultural Policy (CAP). Notably, agri-environmental schemes were developed to promote protection of biodiversity. However 15 years after their implementation, their effectiveness remains controversial (Whittingham 2007, Kleijn *et al.* 2006, Vickery *et al.* 2004). The management of biodiversity in farmlands is still an open question, especially with ongoing debates about ways to improve the use of the dedicated budget into the CAP .

In addition to these environmental considerations, the socio-economic consequences of the CAP are not ubiquitously positive, even though the agricultural incentives originally were developed for economic reasons. By subsidizing production, the goal was to ensure minimum levels of income for farmers. However, this strategy has encouraged a process of intensification, generating a two-speed agriculture (Strijker 2005). On the one hand, in the regions capable of intensification, the CAP process has stimulated enlargement of intensive farms and increased yields, generating high incomes and profits. On the other hand, in regions where intensification is impossible, incomes have remained low and a process of land abandonment has started (Mottet *et al.* 2006, MacDonald *et al.* 2000), which is deepening the gap between rich and poor farmers. More nuanced social considerations, concerning poverty for example, have now been integrated in to the CAP.

Ecological and social criteria are currently being integrated with the historical economic objectives of the CAP. More precisely, they are part of a structure referred to as the "second pillar," the first pillar being the historical support for production and incomes (Lowe *et al.* 2002). Although the public policy instruments that are part of this second pillar seem to be socially acceptable (Prager & Freese 2009), they have been strongly criticized for their ecological inefficiency (Stoate *et al.* 2009, Kleijn *et al.* 2001). The gap between the real objectives of the CAP and those studied in the literature opens interesting questions about the objectives of future CAP: do the objectives have to focus on ecological considerations or do they are in line with sustainability perspectives by considering ecological and social criteria? In other words, is the objective to green the CAP or to make it more sustainable? While the second pillar of the CAP is now defined from sustainability point of view (Sutherland 2002), the question of greening CAP remains crucial in current debates (Scherr & McNeely 2008). The originality of this paper lies in its contribution to these two viewpoints. We compare green public policies on the one hands and sustainability policies on the other.

There are various strategies to design public policies. More specifically agricultural public policies frequently are aimed at two different agricultural systems. The first concerns crops, which are characterized by high average incomes, and are associated with strong average negative impacts on biodiversity (Chamberlain *et al.* 2000, Donald *et al.* 2001, Stoate 2001). The second concerns grasslands, which are associated with lower average incomes but have a more positively effect on biodiversity by preserving the natural habitats essential of many species (Laiolo 2005). However, the impact of land-use and land-use changes on biodiversity is more complex since it has been shown that their ratio is a key element for biodiversity (Robinson *et al.* 2002). To impact these land-uses, price instruments (taxes or subsidies) are today considered by decision-makers, since they are applied to crops and grasslands respectively within the first and the second pillars of the current CAP.

In this paper, we compare green and sustainability public policies based on economic tools applied to crops or grasslands. Public policy scenarios are designed following an optimal under constraint approach in which the decision-maker maximizes a welfare criterion under social or ecological constraints. The optimal instruments are analysed and the consequences on different criteria (welfare, social, biodiversity) due to the introduction of these new constraints are compared at different scales.

The paper is organized as follows. Section 2 presents the bio-economic model and the scenarios. Section 3 describes the case study. Sections 4 and 5 are devoted to the results and the discussion.

2. The bio-economic modelling

The bio-economic model in this paper extends the model developed in Mouysset *et al.* (2013), which links the economic decisions of representative farmers to environmental quality and biodiversity at local scale. In the present paper, a third component related to national scale public policy at has been added. This multi-scale model is depicted in figure 1. A decision-maker chooses public policy scenarios, that impact on the choices of the farmers and thus, in an indirect way, affect habitat quality and biodiversity. Although the ecological equations remain similar[1], the introduction of public policy levers modifies the equations of the economic model.

2.1. The economic model of the farmer

As in Mouysset *et al.* (2013), each region r is assumed to be managed by a representative farmer who determines the areas $A_{r,k}(t)$ of each land-uses k to maximize his expected utility depending on the mean and dispersion of incomes. However the income function now includes public policy incentives in addition to agricultural rents:

$$Inc_r(t) = \sum_k A_{r,k}(t).gm_{r,k}.(1+\tau_k) \tag{1}$$

where $A_{r,k}(t)$ denotes the areas dedicated to the land uses k in the region r, $gm_{r,k}$ represents the associated expected gross margin, and τ_k the economic incentives applied to the land-use k ($\tau > 0$ for subsidies, $\tau < 0$ for taxes).

[1] See the appendix and Mouysset *et al.* (2013) for the details of the ecological modelling

Similarly to Mouysset *et al.* (2013), a quadratic form is used for the utility function to characterize the representative agent per small agricultural region:

$$U_r(t) \quad = \quad \mathbb{E}[Inc_r(t)] - a.\mathbb{V}ar[Inc_r(t)] \tag{2}$$
$$\tag{3}$$

based on the expected income $\mathbb{E}[Inc_r(t)]$, its risky part $\mathbb{V}ar[Inc_r(t)]$ (based on the covariances between gross margins of land-uses k and k' in region r), and the risk aversion level of farmers a (see details in Mouysset *et al.* (2013)).

The maximizing program of farmer's utility in an uncertain context is defined as follows:

$$\max_{A_{r,1};\ldots;A_{r,K}} U_r(t) \tag{4}$$

Furthermore, when maximizing the utility, the standard agent must comply with three constraints at each point in time:

$$|A_{r,k}(t) - A_{r,k}(t-1)| \leq \varepsilon.A_{r,k}(t-1) \tag{5}$$

$$\sum_k A_{r,k}(t) = A_r \tag{6}$$

The first constraint (eq. 5) corresponds to a technical constraint where the coefficient ε stands for the rigidity in changes. The second one (eq. 6) is a stability constraint ensuring the total agricultural surface A_r per region constant.

2.2. The bio-economic indicators for the stakeholder

The performance of the ecological-economic model can be analysed using different indicators. These indicators are used for the design of public policies. The decisions of stakeholder are based on the classical welfare criterion. This welfare corresponds to the total wealth of society on a national scale. It includes the economic states of both private agents (farmers) and public agent (the budget available to subsidize farmers, providing from the general taxes levied society). In other words, welfare corresponds to the evolution in net income generated by the farms, excluding transfers (taxes and subsidies).

The private richness of farmers is analysed through national income $Inc(t)$ (eq. 7) computed as the sum of the product of micro incomes $Inc_r(t)$ (as defined in eq. 1):

$$Inc(t) = \sum_r Inc_r(t) = \sum_r \sum_k A_{r,k}(t).gm_{r,k}.(1+\tau_k) \tag{7}$$

The public wealth corresponds to the non-spent public budget defined as below, where Env represents the initial available envelope:

$$Budg(t) = Env - \sum_r \sum_k \tau_k.gm_{r,k}.A_{r,k}(t) \tag{8}$$

The total richness is computed as the sum of private and public wealth (eq. 9). Because of the public policies modifying rents or constraints on land-use, the $A_{r,k}(t)$ chosen by the farmers differ according to the scenario, leading to some variations in total wealth.

$$Rich(t) = Inc(t) + Budg(t) = \sum_r \sum_k gm_{r,k}.A_{r,k}(t) + Env \tag{9}$$

Finally we define inter-temporal total richness among time $Rich$. This corresponds to the sum of total richnesses discounted at the rate ρ from the first year of the projection t_1 to the final time horizon T:

$$Rich = \sum_{t=t_1}^{T} \rho^{t-t_1} \ . \ Rich(t) \tag{10}$$

Social and environmental criteria are considered as constraints by the decision-maker. First in a social perspective, the objective is to defence the poorest farmers. Following a maximin perspective (Solow 1974, Heal 1998), the criterion is then the poorest regional income among the regions:

$$Poor(t) = \min_r(Inc_r(t)) \tag{11}$$

The associated intertemporal criteriaon $Poor$ is computed as follows:

$$Poor = \sum_{t=t_1}^{T} \rho^{t-t_1} \ . \ Poor(t) \tag{12}$$

Second, in an ecological perspective, we use a criterion that ensures a minimum state of biodiversity. Large scale ecological performance is assessed using biodiversity indicators, denoted $Biod$, but not specified at this stage. This formulation includes the usual biodiversity indices such as species richness, Simpson or trophic indices. They are based on the abundance $N_{s,r}(t)$ of the species $s = 1, .., S$ and the regions $r = 1, .., R$ involved:

$$Biod(t) = f\big(N_{1,r}(t), \ldots, N_{S,R}(t)\big) \tag{13}$$

2.3. The public policy scenarios

The Laissez-Faire scenario

To analyse the impact of public policies, we first define a Laissez Faire (LF) scenario without public intervention (eq. 14), which corresponds to the situation described in Mouysset *et al.* (2013). Farmers optimize their land-use according to the economic model, win which the only drivers are the expected gross margins and the level of risk aversion. We note the ecological and economic performances of this scenario $Rich^{LF}$, $Biod^{LF}$ and $Poor^{LF}$.Then we define public policy scenarios based on the vector of economic incentives $\tau = [\tau_{k=1} \ldots \tau_{k=K}]$. Typically the LF scenario is defined as follow:

$$\tau_k^{LF} = 0 \ for \ all \ k \tag{14}$$

The Green scenario

In public policy scenarios, the decision-maker optimally determines the levels of τ using an inter-temporal maximization programme under constraints. The stakeholder makes its choice by maximizing the inter-temporal richness $Rich(\tau)$:

$$\max_{\tau} Rich(\tau) \tag{15}$$

under the budgetary (eq. 16) and the environmental (eq. 17) constraints:

$$Budg(t) \ \geq \ 0 \tag{16}$$

$$Biod(T) \ \geq \ (1+\mu) * Biod^{LF}(T) \tag{17}$$

The budgetary constraint (eq. 16) ensures that the decision maker does not spend more than the envelope (i.e. the initial envelope and the payments derived from taxes). The ecological constraint (eq. 17) is based on a conservation limit for the biodiversity goal imposed at the temporal horizon T and the performance obtained with the LF scenario. Different values of the percentages μ can be tested between the lowest value $\mu = 0$ and the maximal feasible biodiversity[2] corresponding to $b = Biod^*(T)/Biod^{LF}(T)$. These scenarios with the ecological constraint are described as green (GRE) scenarios.

The Sustainable scenario

We also tested sustainable (SUS) scenarios similarly taking account of both social and ecological constraints. In these scenarios, the decision-maker optimally determines the levels of τ by maximizing $Rich(\tau)$ (eq. 15), under the budgetary constraint (eq. 16), the environmental constraint (eq. 17) and a social constraint defined as follows:

$$Poor \ \geq \ (1+\nu) * Poor^{LF} \tag{19}$$

This social constraint (eq. 19) is computed with inter-temporal lowest incomes obtained in the LF scenario. The value $\nu = 0$ means that the criterion $Poor$ has to be at least equal to the performance obtained with the LF situation. With the value $\nu = 10\%$, the criterion increases by at least 10%. Different values of ν are tested and associated optimal public instruments designed. In this study, SUS scenarios are defined with similar account of ecological and social constraints, i.e. $\nu = \mu$. The effect of the social constraint alone will also be presented in the Result section.

[2]This maximum $Biod^*$ is with the policy determined by the biodiversity maximisation under the budgetary constraint:

$$Biod^*(T) \ = \ \max_{\left\{\begin{array}{l}\tau \\ Budg \geq 0\end{array}\right.} Biod(T) \tag{18}$$

3. Case study

3.1. Context

As in Mouysset *et al.* (2013), the model is calibrated and applied to metropolitan France, at the Small Agricultural Region (SAR) scale using common birds as the biodiversity metrics. Metropolitan France is split into 620 Small Agricultural Regions (SAR) which exhibits agro-ecological homogeneity. The biodiversity and economic models at local scales described below are thus built and calibrated for each SAR. We then use the model to make projections from t_1=2009 to T=2050.

3.2. Biodiversity data

To measure biodiversity, we focus on common bird populations (Ormerod & Watkinson 2000, Sekercioglu *et al.* 2004). The data are from the French Bird Breeding Survey (FBBS) time series (see the Vigie-Nature website [3]). For details of the monitoring method and sample design see in Jiguet (2009). The data provides information about bird abundances across the whole country. Abundance values for each species were available for the period 2002-2008. Among the species included in the survey, we focus on the 20 farmland specialist species which are classified according to their habitat requirements at European level (European Bird Census Council 2007). Table 1 lists the 20 farmland specialist species used as a reference for the European Farmland Bird Index FBI (Gregory *et al.* 2004). Consistent with previous analyses which show the relevance of the national FBI to reflect the response of farmland biodiversity to agricultural intensification (Doxa *et al.* 2010, Mouysset *et al.* 2012), we use it as our *Biod* indicator for ecological constraint (eq. 17) in the design of the public policies. As described in equation 13, we compute a national FBI of 20 farmland specialist species for each SAR:

$$\mathrm{FBI}(t) = \prod_{s\,\in\,\mathrm{Specialist}} \left(\frac{N_{s,nat}(t)}{N_{s,nat}(2008)} \right)^{1/20} \tag{20}$$

where $N_{s,nat}(t) = \sum_{r=1}^{620} N_{s,r}(t)$ stands for the total abundance of species s across the 620 small agricultural regions r.

3.3. Agro-economic data

Our agro-economic data are from the French agro-economic classification of agricultural land-uses developed by the French Farm Accounting Data Network (FADN)[4] and the Observatory of Rural Development (ODR)[5]. This organization classifies land uses in 14 classes of agricultural systems (see tab. 2). Each SAR is a specific combination of these 14 systems. The areas dedicated to each of the

[3] http://www2.mnhn.fr/vigie-nature/
[4] http://ec.europa.eu/agriculture/rica/
[5] https://esrcarto.supagro.inra.fr/intranet/

14 agricultural systems and the associated gross margins relying on tax returns, for the years 2002 to 2008 are available at the ODR website. The budgetary constraint is calibrated with the current French CAP budget. The rigidity and aversion parameters a and ϵ have been calibrated according to a least-square minimization (see Mouysset *et al.* (2013)). As mentioned in the introduction, we focus here on two specific levers for public policy: croplands *cop* (class 1 in tab 2) and the non-intensive grasslands *grass* (classes 4-7 in tab 2).

4. Results

4.1. Welfare with green and sustainable optimal scenarios

Figure 2 presents evolutions of the welfare criteria *Rich* under optimal green GRE (black diamond) and sustainable SUS (triangle) public policies. The x-axis represents the level of constraints: in the case of the sustainability objective μ=ν the strength of both ecological and social constraints, and only μ in the case of the green target. This figure shows first that it is possible to design public policies that can increase either ecological criteria or both ecological and social objectives, but they imply a loss of richness. This decrease is due to the fact that welfare depends solely on land-use distributions and gross margins (excluding subsidies and taxes). The context of market stability (i.e. without any evolution in gross margins) implies that its evolution is due only to a modification in land-use distributions compared to the LF situation. Because the LF is the optimal land use distribution relative to current gross margins, the new distribution can only be either identical to the LF distribution or sub-optimal. In other words, the introduction of incentives due to new constraints and modifications to land use distributions necessarily implies a decrease in welfare. Finally we observe that green and sustainable objectives coincide when the objectives are rather small (between 0% and 5%) or rather high (between 18% and 30%). For medium targets, the two objectives do not imply the same optimal strategy.

4.2. Green optimal public policies

Figure 3 shows the optimal GR tools required to maximize the welfare criteria *Rich* under the budgetary and ecological constraints (tab. 3). The x-axis stands for μ, the strength of the ecological constraint. The red colour represents the levels of ecological constraints achievable with subsidies taking no account of the budgetary constraint. Incentives for grasslands or crops can be used to increase the FBI. The analysis of the instruments highlights that the cost-effective way to satisfy the FBI constraint by playing on crops is to impose a tax on them (i.d. $\tau^*_{cop} < 0$). For grasslands, the cost-effective strategies are subsidy based (i.d. $\tau^*_{grass} > 0$), yielding an increase in grassland area. Regarding maximum biodiversity performance, we observe that a tax on crops leads at best to medium objectives (up to 17 %) while subsidies on grasslands are capable of achieving the highest performances (30 % or more if the budgetary constraint is relaxed). Finally, figure 3 exhibits some structural limitations for instruments based on croplands: the impact on the FBI remains constant beyond $\tau_{cop} = -0.47$.

4.3. Socially optimal public policies

Similarly to figure 3, figure 4 depicts the optimal public policies satisfying budgetary and social constraints (tab. 4). The x-axis stands for ν, the percentage of improvement compared to the Laissez-Faire situation. We observe that, in contrast to the ecological constraint, only subsidies for grasslands are able to satisfy the social constraint. This is explained by the fact that, by enhancing the financial portfolio of farmers, subsidies imply earnings. Since the poorest regions are specialized in grasslands, the only instrument capable of increasing the social objective is a grassland subsidy. Finally, we observe use of this instrument makes it is possible to strongly increase the level of the indicator *Poor*, up to 84%. The red colour in figure 4 indicates that it is possible to design subsidies that enable an increase beyond $\nu = 84\%$. This limit is imposed by the budgetary constraint which limits the design of stronger subsidies.

4.4. Welfare under green public policies

Figure 5 compares welfare obtained under optimal instruments based on grassland (solid lines) and cropland (dashed lines) able to satisfy the ecological constraint. The x-axis stands for μ, the strength of the ecological constraint. We observe a switch between instruments able to optimally achieve ecological objectives. While subsidies on grasslands are less costly in terms of welfare for small targets ($\mu \leq 5\%$), taxes on croplands become more efficient for $\mu \geq 6\%$. However, they are not able more than $\mu \geq 17\%$. Then for $\mu \geq 18\%$, the only available strategy is to subsidize grasslands.

4.5. Regional distributions

Figure 6 shows regional welfares for tools capable of satisfying the biodiversity constraint with $\mu = 10\%$. Since a subsidy on grasslands can also increase the social criteria, figure 6(b) is also related to social perspectives. The intensity of the blue shading represents the loss of welfare. We observe that the land use targeted by public policy is a strong driver of patterns of regional welfare. More precisely, a crop tool is cheaper than a grassland instrument. Because more French regions are specialized in grasslands, the regional costs of grassland instruments are more generalized.

Figure 7 shows the pattern of ecological performances at the regional scale for two optimal tools which satisfy the biodiversity constraint with $\mu = 10\%$. Again, figure 7(b) depicting subsidies on grasslands is interesting from a social point of view. The intensity of blue shading represents the decrease in FBI performance and the intensity of red shading indicates the intensity of the improvement. We observe different patterns capable of achieving the same national scores. Figure 7 shows that the crop tool (Fig. 6(a)) slightly positively affects only the half of the regions, with no consequence for the half. This contrasts with the instrument based on grassland (Fig. 6(b)) which affects all of the regions with contrasts among regions that show decreased, stable or strong increase in FBI.

5. Discussion

5.1. Designing agricultural public policies towards sustainable agriculture

The model built here offers a framework for both researchers' and decision makers' thinking about more sustainable management of agriculture in France. Due to the stylization used in the modelling, the results can not be interpreted as exact predictions of future bio-economic states. However, because the model integrates multi-scale and dynamic processes which are calibrated with historical data, it provides relevant ranges of future trends function of different public policy drivers. Its objective is thus to stimulate debates on what strategies to develop for various policy targets, and especially in the perspective of the next reform of the Common Agricultural Policy (CAP). More specifically, we have shown that it is possible to integrate and improve ecological and social criteria through public economic policies.

The first contribution of this study is that it complements knowledge on the economic drivers available to public policy. We highlighted land-use targeted by public policy as an essential driver, determinant of national performances and structuring regional patterns. The lack of attention of this target in the literature can probably be explained by the fact that most models focus only on these two land-uses such as in Barraquand & Martinet (2011). This simplification is justified because they constitute crucial levers for designing new socially (Prager & Freese 2009, Mottet *et al.* 2006) and ecologically (Laiolo 2005, Robinson & Sutherland 2002, Stoate 2001) improved public policy. However, the symmetry between a negative effect on croplands or a positive effect on grasslands in the two-land use models, raises questions in the real landscapes with multiple land uses. In the present study, we stress the importance of the target since there is no symmetry between them. The two levers do not affect the same regions, according to their specialization. Also, for regions affected by both cropland and grassland policies, the consequences differ because actions related to one land-use impact has an impact, but not necessarily in favour of or at the expense of the other. Hence, the distributions of regional ecological and economic performances depends on the public instruments.

The second contribution is more aligned to ecological perspectives. The optimal crop-based instruments have positive consequences for FBI within almost all regions, whatever the range of land-uses used to replace crops. Consistent with the literature (Chamberlain *et al.* 2000, Donald *et al.* 2001, Robinson & Sutherland 2002), we confirmed the broadly negative effect of crops on biodiversity. We also confirmed the average positive effect of grassland on biodiversity (Laiolo 2005) but we moderated it since it is characterized by contrasting regional patterns. In some regions, the development of grasslands leads to local decrease in the FBI, suggesting that land-uses replaced by grassland have a positive impact on biodiversity. Although on the whole bird communities react positively to grasslands, this land use is not optimal for all single species. If some communities are historically strongly biased towards these species, increasing grassland areas may not have such a positive effect at the local scale.

5.2. Implications for the current CAP

These previous theoretical contributions can be interpreted in the context of the current CAP. They give some insights into the design of the best public instruments and understanding the compatibility between green and sustainability objectives. We highlighted in this paper that this compatibility depends on the stringency of the objectives. Our study shows that if the ecological objectives are limited (lower than 5%) or are high (between 18% and 30%), subsidizing grassland is the best strategy for the decision-maker from both green and sustainability points of view: this strategy simultaneously satisfies the ecological and social objectives and minimizes loss of richness. These conclusions unambiguously question the current structure of the second pillar of the CAP based on subsidies for grassland. However, most of the measures in favour of the environment in the current CAP are based on medium objectives. It is because the conclusions in these situations are not trivial that the debate between green and sustainability criteria persists.

First, if the objective is only to green the CAP (i.e. cost-effectively manage biodiversity), playing on the first pillar dedicated to crops is the cheapest strategy. This conclusion questions the social acceptability of taxes. However, due to the structure of our economic dataset which includes the current public incentives, taxes are more understandable as a reduction in current crop subsidies rather than absolute taxes. Then, in contrast to what is commonly understood, reducing first pillar subsidies would be the most relevant strategy to cost-effectively manage biodiversity and greening the CAP with medium ecological objectives. This confirms the interest of the starting debate about the inconsistency of this first pillar with current environmental considerations: some of its subsidies are now subject to environmental conditions. Second, if the objective is sustainability, then the aims are broader and consist of combining ecological and socio-economic (related to poverty for instance) considerations, and subsidies for grassland consistent with the current structure of the second pillar is the only strategy. However, we should keep in mind that this strategy has a cost regarding the biodiversity criterion since loss in welfare is larger than with taxes on crops even if the difference remains small.

In other words, in the case of medium objectives, decision makers need to choose between managing biodiversity or managing it in association with improvements to social criteria and higher welfare losses. The conclusions about the best public policy instrument will depend on this political position. However, it should be noted also that, in any case, these medium objectives are not capable of achieving the classical objective of halting erosion of biodiversity in the short term. We frequently heard about goals related to halting biodiversity loss by 2010, but now the horizon has shifted to 2020. Although our study adopts a more distant and, thus easier, objective (horizon 2050), it still appears that it will be difficult to stop loss of biodiversity ($Biod(2050) \geq Biod(2009)$ will happen for $\mu = 43\%$). This raises questions about the practicality and achievability of such political objectives.

Finally, our study points to the importance of international cooperation since the introduction of ecological and social consideration constraints on a national scale necessarily results in a decrease in welfare. One way to avoid or reduce this decrease would be to re-evaluate prices on a larger scale (here world scale), independent of local public incentives (here national scale).

5.3. Perspectives

Our study highlights some interesting trade-offs which are beyond the scope of this paper, but which constitute relevant perspectives for the development of sustainable agriculture public policies.

The first perspective is related to the barriers limiting the implementation of public policy. They differ according to the instrument. Playing on crops to limit negative land uses does not exhibit any social limitations (smallest loss of welfare, equity in ecological performances)s but presents some structural limitations: it is impossible to achieve the highest biodiversity scores since there is no pressure on the land use substituting crop. In contrast, affecting grasslands to develop positive land uses does not exhibit any structural limitations (highest social and ecological scores achievable), but in associated with heavy social limitations (large loss of welfare, heterogeneity of ecological performances, public costs related to the required budget). Thus, we need to debate over choosing the more socially acceptable tool for the short-term (i.e. medium objectives) although it will be impossible later to re-assess the objectives, or choosing the tool that allows the possibility of future re-assessment of objectives although in relation to moderate objectives, it is not the most efficient instrument.

The second perspective is related to the question of spatial equity of performance. We found contrasted patterns for both welfare and biodiversity scores. For biodiversity, the consequences are tow folds: there are ecological impacts (functioning of the ecosystem, ability for dispersion among regions) and social impacts (social impacts of living in a good or bad ecological environment). In this context, taking account of equity criteria in addition to national scores for welfare and biodiversity appears an interesting perspective. It should affect the conclusions, since in our study the instruments capable of achieving the highest national scores are based on heterogeneous regional patterns, clearly highlighting the trade-off between maximizing national score versus regional equity. In this context, it would be relevant to try other schemes to see whether this result (highest national score associated with heterogeneous pattern) is reproduced using other instruments or whether it is inherent in the problematic (at least in our modelling context). Among other schemes, subsidies to diversify land uses could be tested, since the literature identifies the positive impact on biodiversity of such schemes (Schläpfer *et al.* 2002, Di Falco & Perrings 2003, Quaas *et al.* 2007, Mouysset *et al.* 2013). In this perspective, refining the economic behaviours of farmers in the model might be an interesting improvement.

Finally, green and sustainability strategies raise questions about the productive requirements and open debates at global scale. In the current context of demographic growth, some limitations of production in Europe will imply compensations somewhere else. In this sense, opportunities could arise with the introduction of Ukraine in Europe, where agriculture has dropped in the recent decades.

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References

Barraquand, F., & Martinet, V. 2011. Biological conservation in dynamic agricultural landscapes: Effectiveness of public policies and trade-offs with agricultural production. *Ecological Economics*, **70**(5), 910–920.

Benton, T.G., Vickery, J.A., & Wilson, J.D. 2003. Farmland biodiversity: Is habitat heterogeneity the key? *Trends in Ecology & Evolution*, **18**(4), 182–188.

Carpenter, S.R., Caraco, N.F., Correl, D.L., Horwarth, R.W., Sharler, A.N., & Smith, V.H. 2012. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications*, **8**(3), 559–568.

Chamberlain, D.E., Fuller, R.J., Bunce, R.G.H., Duckworth, J.C., & Shrubb, M. 2000. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. *Journal of Applied Ecology*, **37**(5), 771–788.

Di Falco, S., & Perrings, C. 2003. Crop genetic diversity, productivity and stability of agroecosystems. A theoretical and empirical investigation. *Scottish Journal of Political Economy*, **50**(2), 207–216.

Donald, P.F., Green, R.E., & Heath, M.F. 2001. Agricultural intensification and the collapse of Europe's farmland bird populations. *Proceedings of the Royal Society B-Biological Sciences*, **268**(1462), 25–30.

Doxa, A., Bas, Y., Paracchini, M-L., Pointereau, P., Terres, J-M., & Jiguet, F. 2010. Low-intensity agriculture increases farmland bird abundances in France. *Journal of Applied Ecology*, **47**(6), 1348–1356.

European Bird Census Council. 2007. *The state of Europe's common birds ?* EBCC.

Flowerdew, J.R., & Kirkwood, R.C. 1997. Mammal biodiversity in agricultural habitats. *BCPC Symposium Proceedings*, **69**, 25–40.

Foley, J.A., Defries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., & Al., Et. 2005. Global consequences of land use. *Science*, **309**(5734), 570–4.

Gregory, R.D., G. Noble, D., & Custance, J. 2004. The state of play of farmland birds: Population trends and conservation status of lowland farmland birds in the United Kingdom. *Ibis*, **146**(Suppl. 2), 1–13.

Heal, G. 1998. *Valuing the future, economic theory and sustainability*. New York: Columbia University Press.

Jiguet, F. 2009. Method-learning caused first-time observer effect in a newly-started breeding bird survey. *Bird Study*, **56**(2), 253–258.

Kleijn, D., Berendse, F., Smit, R., & Gilissen, N. 2001. Agri-environment schemes do not effectively protect biodiversity in Dutch agricultural landscapes. *Nature*, **413**(6857), 723–5.

Kleijn, D., Baquero, R.A., Clough, Y., Díaz, M., De Esteban, J., Fernández, F., Gabriel, D., Herzog, F., Holzschuh, A., Jöhl, R., Knop, E., Kruess, A., Marshall, E.J.P., Steffan-Dewenter, I., Tscharntke, T., Verhulst, J., West, T.M., & Yela, J.L. 2006. Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecology Letters*, **9**(Mar.), 243–256.

Laiolo, P. 2005. Spatial and seasonal patterns of bird communities on italian agroecosystems. *Conservation Biology*, **19**(1), 1547–1556.

Lowe, P., Buller, H., & Ward, N. 2002. Setting the next agenda ? British and French approaches to the second pillar of the Common Agricultural Policy. *Journal of Rural Studies*, **18**, 1–17.

MacDonald, D., Crabtree, J.R, Wiesinger, G., Dax, T., Stamou, N., Fleury, P., Gutierrez Lazpita, J., & Gibon, A. 2000. Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. *Journal of Environmental Management*, **59**(1), 47–69.

Mottet, A., Ladet, S., Coque, N., & Gibon, A. 2006. Agricultural land-use change and its drivers in mountain landscapes: A case study in the Pyrenees. *Agriculture, Ecosystems & Environment*, **114**(2-4), 296–310.

Mouysset, L., Doyen, L., & Jiguet, F. 2012. Different policy scenarios to promote various targets of biodiversity. *Ecological Indicators*, **14**(1), 209–221.

Mouysset, L., Doyen, L., & Jiguet, F. 2013. How does economic risk aversion affect biodiveristy ? *Ecological Applications*, In press.

Ormerod, S.J., & Watkinson, A.R. 2000. Editors' introduction: Birds and agriculture. *Journal of Applied Ecology*, **37**(5), 699–705.

Prager, K., & Freese, J. 2009. Stakeholder involvement in agri-environmental policy making-learning from a local- and a state-level approach in Germany. *Journal of environmental management*, **90**(2), 1154–67.

Quaas, M., Baumgartner, S., Becker, C., Frank, K., & Muller, B. 2007. Uncertainty and sustainability in the management of rangelands. *Ecological Economics*, **62**(2), 251–266.

Robinson, R.A., & Sutherland, W.J. 2002. Post-war changes in arable farming and biodiversity in Great Britain. *Journal of Applied Ecology*, **39**(1), 157–176.

Robinson, R.A., Wilson, J.D., & Crick, H.Q.P. 2002. The importance of arable habitat for farmland birds in grassland landscapes. *Journal of Applied Ecology*, **38**(5), 1059–1069.

Scherr, S.J., & McNeely, J.A. 2008. Biodiversity conservation and agricultural sustainability: towards a new paradigm of 'ecoagriculture' landscapes. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, **363**(1491), 477–94.

Schläpfer, F., Tucker, M., & Seidl, I. 2002. Returns from hay cultivation in fertilized low diversity and non-fertilized high diversity grassland. *Environmental and Resource Economics*, **21**, 89–100.

Sekercioglu, C.H., Daily, G.C., & Ehrlich, P.R. 2004. Ecosystem consequences of bird declines. *Proceedings of the National Academy of Sciences of the United States of America*, **101**(52), 18042–7.

Solow, R. 1974. Intergenerational equity and exhaustible resources. *Review of Economic Studies*, **41**, 29–45.

Sotherton, N.W., & Self, M.J. 2000. Changes in plant and arthropod biodiversity on lowland farmland: An overview. *Spring Conference of the British Ornithologists Union*, 26–35.

Stoate, C. 2001. Ecological impacts of arable intensification in Europe. *Journal of Environmental Management*, **63**(4), 337–365.

Stoate, C., Baldi, A., Beja, P., Boatman, N.D., Herzon, I., van Doorn, A., de Snoo, G.R., Rakosy, L., & Ramwell, C. 2009. Ecological impacts of early 21st century agricultural change in Europe-a review. *Journal of environmental management*, **91**(1), 22–46.

Strijker, D. 2005. Marginal lands in Europe - causes of decline. *Basic and Applied Ecology*, **6**(2), 99–106.

Sutherland, W.J. 2002. Restoring a sustainable countryside. *Trends in Ecology & Evolution*, **17**(3), 148–150.

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W.H., Simberloff, D., & Swackhamer, D. 2001. Forecasting agriculturally driven global environmental change. *Science*, **292**(5515), 281–4.

Tilman, D., Cassman, K.G., Matson, P.A., Naylor, R., & Polasky, S. 2002. Agricultural sustainability and intensive production practices. *Nature*, **418**(6898), 671–7.

Tong, S.T.Y., & Chen, W. 2002. Modeling the relationship between land use and surface water quality. *Journal of Environmental Management*, **66**(4), 377–393.

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., & Thies, C. 2005. Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. *Ecology Letters*, **8**(Aug.), 857–874.

Vickery, J.A., Bradbury, R.B., Henderson, I.G., Eaton, M.A., & Grice, P.V. 2004. The role of agri-environment schemes and farm management practices in reversing the decline of farmland birds in England. *Biological Conservation*, **119**(1), 19–39.

Volk, M., Liersch, S., & Schmidt, G. 2009. Towards the implementation of the European Water Framework Directive? *Land Use Policy*, **26**(3), 580–588.

Whittingham, M.J. 2007. Will agri-environment schemes deliver substantial biodiversity gain, and if not why not? *Journal of Applied Ecology*, **44**(1), 1–5.

APPENDIX

A dynamic framework has been chosen. The Beverton-Holt model accounts for the intra-specific competition and the density dependence:

$$N_{s,r}(t+1) = N_{s,r}(t).\frac{1+R_{s,r}}{1+\dfrac{N_{s,r}(t)}{M_{s,r}(t)}} \tag{21}$$

where $N_{s,r}(t)$ stands for the bird abundance of species s in a PRA r at year t. The $R_{s,r}$ coefficient corresponds to the intrinsic growth rate specific to each species s in a region r. This parameter takes into account the characteristics of each species such as clutch size, mean reproductive success, number of clutches per year. The variable $M_{s,r}$ captures the ability of the habitat to host the species and the product $M_{s,r}(t) * R_s$ represents the carrying capacity of the habitat r.

The habitat variable $M_{s,r}(t)$ is assumed to depend linearly on land-uses as follows:

$$M_{s,r}(t) = \beta_{s,r} + \sum_k \alpha_{s,r,k}.A_{r,k}(t) \tag{22}$$

where $A_{r,k}(t)$ represents the share of the PRA r dedicated to farming systems k at time t. The $\alpha_{s,r,k}$ and $\beta_{s,r}$ coefficients, specific to each species, quantify how the species s responds to the various farming systems k in a given region r. The $\beta_{s,r}$ coefficient can be interpreted as the mean habitat coefficient for a species s in a PRA r.

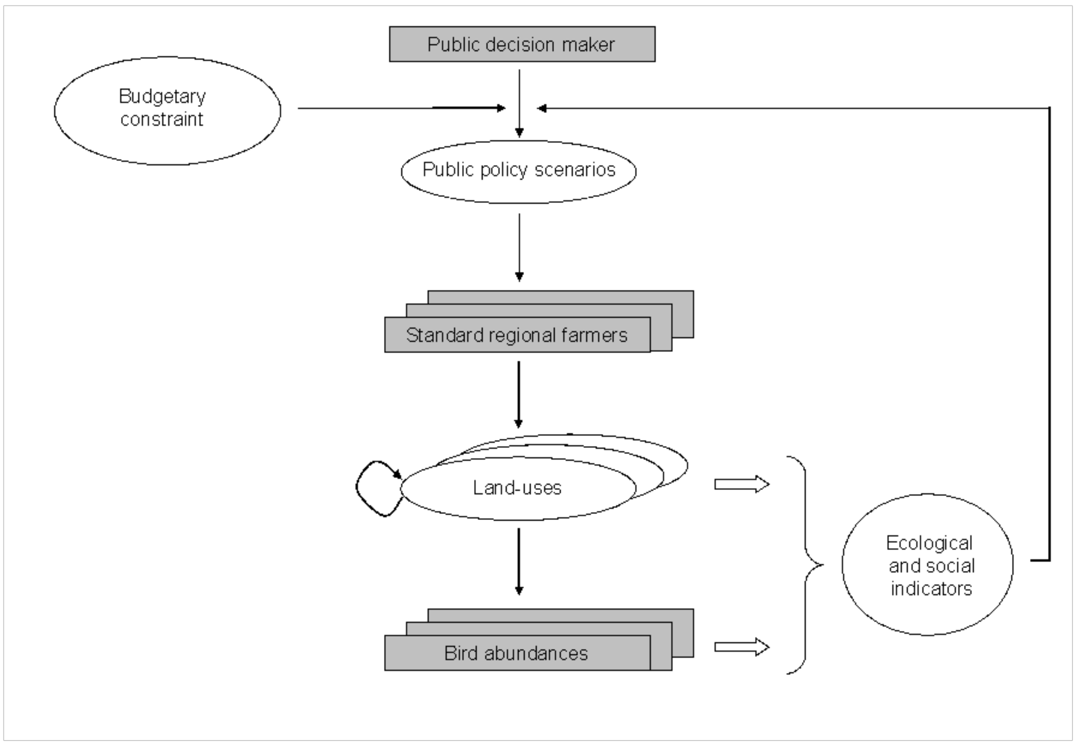


Figure 1: Bio-economic model coupling. The decision maker determines a public policy scenario according to a bio-economic optimization. The farmers choose their agricultural systems by maximizing their utility function under technical constraints. These choices affect the habitats and the bird communities.

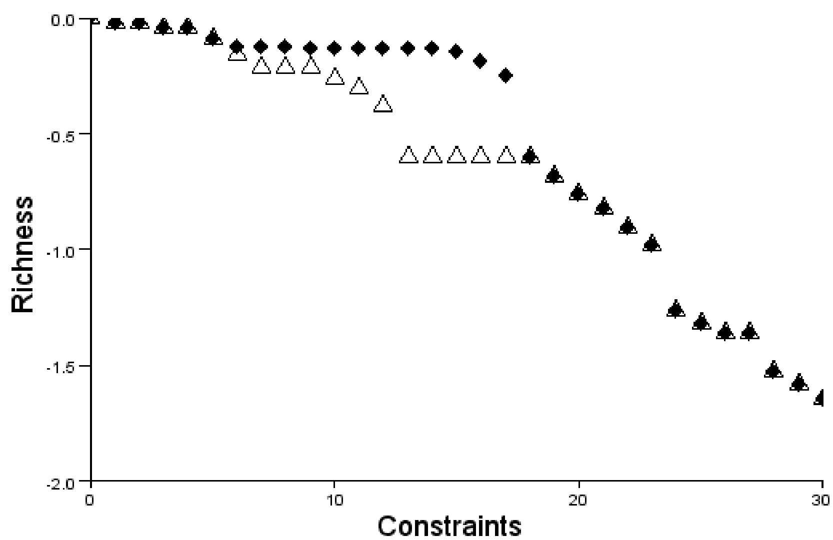


Figure 2: Comparison of the welfare criteria *Rich* under optimal green (black diamond) and sustainable (triangle) public policies with different levels of constraints.The x-axis stands for the level of constraints: in the case of sustainable objective μ=ν the strength of both ecological and social constraints, and only μ in the case of green target.

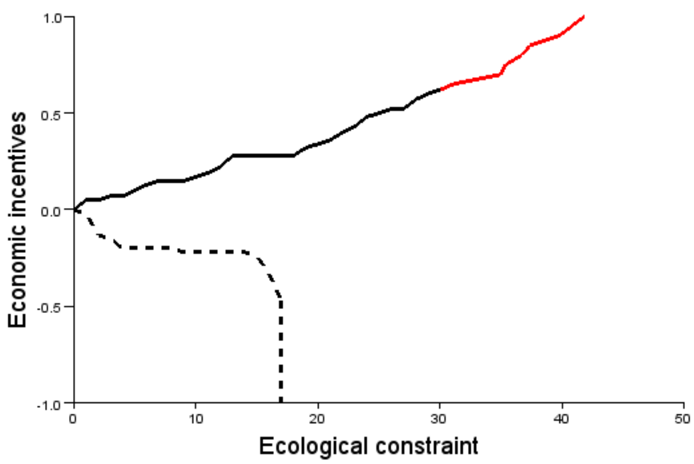


Figure 3: Optimal public policies instruments able to satisfy the budgetary and ecological constraints, based on grasslands (solid lines) or croplands (dashed lines). Red stands for out of the budgetary constraint.

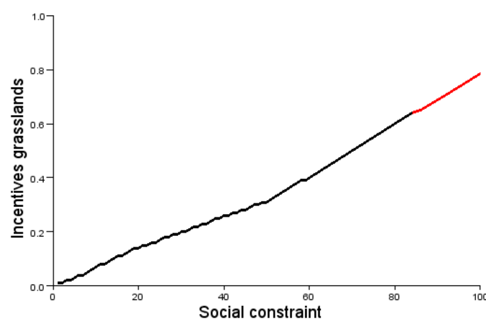
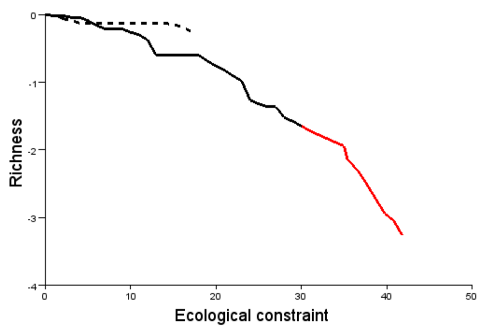
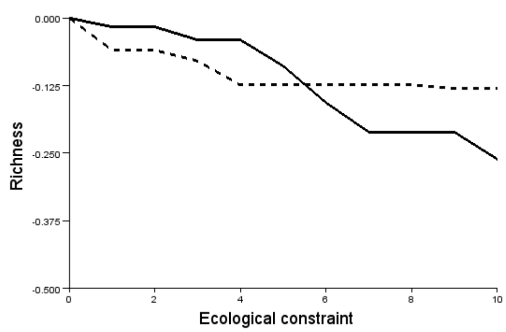


Figure 4: Optimal grasslands incentives τ_{grass} able to satisfy the budgetary and the social constraint. Red stands for out of the budgetary constraint.

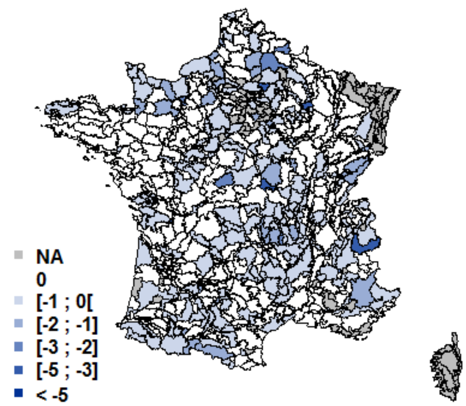


(a)

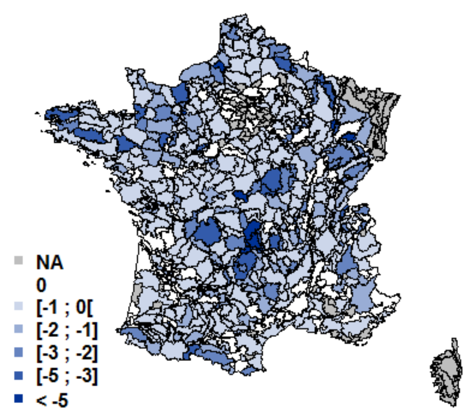


(b)

Figure 5: Evolutions of the richness *Rich* (represented as percentages of the LF performances) with the optimal instruments satisfying the budgetary constraint and the ecological constraint μ based on *Biod*. Solid lines: incentives on grasslands, dashed lines: incentives on croplands. Red stands for out of the budgetary constraint. Figure 5(b) corresponds to a zoom of figure 5(a) for the small levels of constraint.

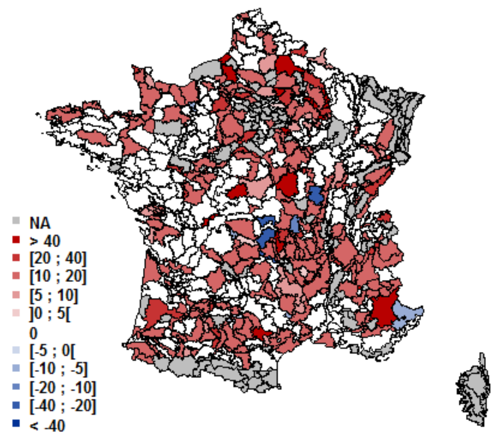


(a) Based on τ_{cop}

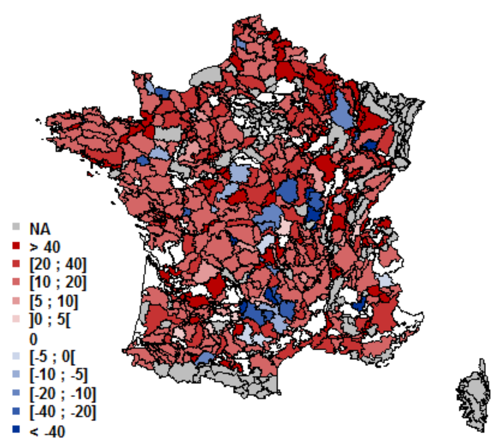


(b) Based on τ_{grass}

Figure 6: Loss of regional richness obtained with the optimal tools able to satisfy the ecological constraint with μ=10%.



(a) Based on τ_{cop}



(b) Based on τ_{grass}

Figure 7: Evolutions of regional ecological performances obtained with the optimal tools able to satisfy the ecological constraint with μ=10%.

20 farmland bird species
(1) Buzzard *Buteo buteo*
(2) Cirl Bunting *Emberiza cirlus*
(3) Corn Bunting *Emberiza calandra*
(4) Grey Partridge *Perdix perdix*
(5) Hoopoe *Upupa epops*
(6) Kestrel *Falco tinnunculus*
(7) Lapwing *Vanellus vanellus*
(8) Linnet *Carduelis cannabina*
(9) Meadow Pipit *Anthus pratensis*
(10) Quail *Coturnix coturnix*
(11) Red-backed Shrike *Lanius collurio*
(12) Red-legged Partridge *Alectoris rufa*
(13) Rook *Corvus frugilegus*
(14) Skylark *Alauda arvensis*
(15) Stonechat *Saxicola torquatus*
(16) Whinchat *Saxicola rubetra*
(17) Whitethroat *Sylvia communis*
(18) Wood Lark *Lullula arborea*
(19) Yellowhammer *Emberiza citrinella*
(20) Yellow Wagtail *Motacilla flava*

Table 1: List of the 20 farmland bird species s

The 14 agricultural systems 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

Table 2: List of the 14 farming systems

μ	τ_{cop}	τ_{grass}
1 %	-0.02	0.05
2 %	-0.13	0.05
3 %	-0.15	0.07
4 %	-0.20	0.07
5 %	-0.20	0.10
6 %	-0.20	0.13
7 %	-0.20	0.15
8 %	-0.20	0.15
9 %	-0.22	0.15
10 %	-0.22	0.17
11 %	-0.22	0.19
12 %	-0.22	0.22
13 %	-0.22	0.28
14 %	-0.22	0.28
15 %	-0.24	0.28
16 %	-0.33	0.28
17 %	-0.47	0.28
18 %		0.28
19 %		0.32
20 %		0.34
21 %		0.36
22 %		0.40
23 %		0.43
24 %		0.48
25 %		0.50
26 %		0.52
27 %		0.52
28 %		0.57
29 %		0.60
30 %		0.62

Table 3: Optimal green public policy (satisfying the ecological constraint based on μ)

ν	τ_{grass}	ν	τ_{grass}
1 %	0.01	43 %	0.27
2 %	0.01	44 %	0.28
3 %	0.02	45 %	0.28
4 %	0.02	46 %	0.29
5 %	0.03	47 %	0.30
6 %	0.04	48 %	0.30
7 %	0.04	49 %	0.31
8 %	0.05	50 %	0.31
9 %	0.06	51 %	0.32
10 %	0.07	52 %	0.33
11 %	0.08	53 %	0.34
12 %	0.08	54 %	0.35
13 %	0.09	55 %	0.36
14 %	0.10	56 %	0.37
15 %	0.11	57 %	0.38
16 %	0.11	58 %	0.39
17 %	0.12	59 %	0.39
18 %	0.13	60 %	0.40
19 %	0.14	61 %	0.41
20 %	0.14	62 %	0.42
21 %	0.15	63 %	0.43
22 %	0.15	64 %	0.44
23 %	0.16	65 %	0.45
24 %	0.16	66 %	0.46
25 %	0.17	67 %	0.47
26 %	0.18	68 %	0.48
27 %	0.18	69 %	0.49
28 %	0.19	70 %	0.50
29 %	0.19	71 %	0.51
30 %	0.20	72 %	0.52
31 %	0.20	73 %	0.53
32 %	0.21	74 %	0.54
33 %	0.22	75 %	0.55
34 %	0.22	76 %	0.56
35 %	0.23	77 %	0.57
36 %	0.23	78 %	0.58
37 %	0.24	79 %	0.59
38 %	0.25	80 %	0.60
39 %	0.25	81 %	0.61
40 %	0.26	82 %	0.62
41 %	0.26	83 %	0.63
42 %	0.27		

Table 4: Optimal social public policies (satisfying the social constraint based on ν).