Agricultural land-use and biological conservation

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September 11, 2009

Abstract

Land use change is a main driver of biodiversity erosion, especially in agricultural landscapes. Incentive-based agri-environmental policies aim at influencing land-use pattern. In this paper, we develop a dynamic bioeconomic model of agricultural land-use and spatially explicit population dynamics. We relate incentive policies level (subsidies to grassland) to the ecological outcome (persistence probability for the species). We describe the links between increasing conservation costs and a S-shaped ecological benefit function.

Keywords: ecological-economic model, agriculture, land-use change, dynamic landscape, conservation, price volatility.

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1 Introduction

Land-use change has been identified as a main driver of changes in the abundance and geographic distribution of organisms at scales ranging from local habitats to regions, or the entire globe (Vitousek et al., 1997). In Western Europe, agriculture is the most important land use (42% of the surface area), though its importance varies among countries (12% in Sweden and Finland, 52% in France and 72% in UK according to the Eurostat references). The intensification of Western European agriculture practices after WWII has been accelerated by the adoption of the Common Agricultural Policy in 1962 and it has entailed the local, national and regional extinction of numerous species of the European flora and fauna over the last 40 years, as well as profound changes in the functioning of these agricultural ecosystems (Krebs et al., 1999; Donald et al., 2001; Benton et al., 2002). Contrary to a widespread perception, agricultural areas harbor an important part of European biodiversity: up to 20% of the British, French and German flora (Marshall et al., 2003) and 50% of bird species (Pain and Pienkowski, 1997). Many of those species have suffered steep population declines in recent years throughout Western Europe (Donald et al., 2001; Siriwardena et al., 1998; Robinson and Sutherland, 2002). The impact of agricultural intensification on biodiversity is no longer in doubt: the declines of many species of plants, insects, and more birds at both national and European levels has being repeatedly shown to be the consequence of agricultural intensification (e.g. Donald et al., 2001; Benton et al., 2002; Robinson and Sutherland, 2002).

The commonly used strategy to conserve biodiversity and habitats has been the creation of natural reserves in publicly-owned land (Armsworth et al., 2004). This approach is however inadequate in agroecosystems where land ownership is usually private. While an important part of the budget of the Common Agricultural Policy has been allocated to schemes which aim at mitigating the environmental effects of agricultural intensification (see the article by Otte et al., 2007, for an overview and references), their effectiveness in reducing the biodiversity loss in agro-ecosystems has been moderate at best (Kleijn et al., 2001, 2006). Many of these agri-environmental schemes have involved small spatial scale actions by private owners, on voluntary basis (Ohl et al., 2008). However, affecting positively the dynamics of biological species at larger scales requires to coordinate landowners decisions, for example with market-based incentives, to encourage them to convert land or retain it in the desired use (Lewis and Plantinga, 2007). Two issues arise in turn: how to assess the influence of a policy on the spatial land-use pattern, and how to evaluate the environmental outcomes that depend on this land-use pattern? The former issue is linked to the definition of efficient policies to influence the land-use pattern in a desired way. It may involve incomplete information problems, with an uncertainty on the resulting landscape. The usual way to tackle this issue is to consider different land use scenarios, resulting in potential landscapes, to assess the influence of given policies in terms of probabilities. The latter issue raises a particular problem when considering biological conservation: How to evaluate the resulting landscape with respect to the conservation objective? Most of the studies which investigate the impact of conservation policies do so in evaluating the resulting landscape, using habitat suitability scores (Polasky et al., 2005; De Koning et al., 2007; MacLeod and McIvor, 2008). In other words, the landscape is an output of the models and policies are defined or evaluated with respect to the resulting landscape (Havlik et al., 2005, 2006; Dymond et al., 2008). The habitat
suitability scores are usually defined using a static biological model for the population (for example by estimating the carrying capacity of a landscape, and thus the long-run potential population). However, such estimations may not be accurate in dynamic landscapes, as they do not account for the population actual dynamics. Ecological-Economic modeling is a good way to overcome these limits (Wätzold et al., 2006; Drechsler and Wätzold, 2007; Drechsler et al., 2007a). In particular, Drechsler and Wätzold (2001) and Drechsler et al. (2007b,c) evaluate the outcomes of incentive-based conservation schemes using biological benefit functions, but without representing explicitly ecological dynamics, either to be able to consider simultaneously several species (Drechsler et al., 2007b) or to focus on a very specific situation (e.g., butterflies in Drechsler et al., 2007c). Tichit et al. (2007) propose a viability analysis of grazing agricultural practices and their impacts on biodiversity, using explicit ecological dynamics, but without addressing the agricultural land use issue. While all these ecological-economic approaches highlight the need for coupling ecological and economic models to improve conservation in agricultural landscapes, few propose to evaluate the ecological outcome of incentive schemes on land use in a dynamic way, using an explicit dynamic ecological model of the species under consideration.

In this paper, we present a methodology for analyzing the links between agricultural land use and biological conservation objectives, in a dynamic framework. We develop an ecological-economic model to represent the influence of the economic context (i.e., price fluctuation and incentive agri-environmental scheme) on private owners’ land-use decisions, and their outcomes on the dynamics of a biological population at a regional level. This allows us to examine incentive policies in a dynamic framework, accounting for the inertia of population dynamics.

The remainder of the paper is organized as follows. In section 2, we present the conceptual framework used in this analysis. In section 3, we describe the ecological-economic model linking the agricultural land use to the biological population. In section 4, we show how our framework allows use to relate the ecological outcome, defined as the survival probability of the species, to the level of the incentive policy, and to its total cost. We then conclude in section 5 on the relevance of ecological-economic modeling, accounting explicitly for ecological dynamics, to address biological conservation issues in agricultural landscape.

2 Conceptual framework

In this section, we describe the conceptual framework underlying our model, we discuss the assumptions, and present the main analysis carried out.

We consider an integrated model of agricultural land use and biological population dynamics. The agricultural land use generates a dynamic landscape. This landscape is the habitat of a biological population which has a spatially explicit dynamics.

On the one hand, the economic part of the model describes farmers land-use decisions. We make the following assumptions:

- The modeled area is composed by fields. Each field can be used either as a grassland
or as a cropland.

- The yield of grassland is constant over space, but that of cropland depends on the agronomic soil quality of the considered field, which is heterogeneous over space and thus between fields, and on the level of fertilizer applied.

- Grassland generates a constant revenue which depends on an incentive policy instrument (grassland subsidy). The agricultural output of cropland is sold at market price. The market fluctuations are modeled by a dynamic system with price autocorrelation and random shocks. Farmers form anticipated prices with rational expectations. Knowing their field’s quality and anticipated prices, they can compare the certain return from grassland to the expected return from cropland.

- Farmers are assumed to be price-taker and to make their land-use decisions by maximizing intertemporal expected gross returns, with anticipated prices, without accounting for environmental externalities. Conversion costs are accounted for to represent the extra cost of changing land-use.

On the other hand, the ecological part of the model describes the evolution of the biological population over time at the regional scale, in the dynamic landscape generated by farmers land-use choices. We make the following assumptions:

- The biological population is depicted with a metapopulation model where subpopulations, which growth rates depend on the local land use, are connected by dispersal processes.

- Grasslands are favorable to the population dynamics, while croplands are not.

- The population dynamics is spatially explicit, taking into account density dependence of birth-death processes and dispersal.

The conservation objective is the survival of the species. Because survival is never certain, we use the persistence probability as a benefit function for the ecological outcome (Montgomery et al., 1994; Arponen et al., 2005).

The main analysis carried out in this paper consist in linking the conservation incentive policy (total cost of the grassland subsidy) to the dynamic ecological outcome. The economic context appears to play a major role in the land use decision, and thus on the population dynamics. We provide a sensitivity analysis to examine the impact of two economic components:

\[\text{1}\]

In this paper, we are interested in the land-use and its impact on biodiversity. We aim at characterizing the impact of price fluctuation on the regional abundance of a species. Land-use decisions are based on price expectation, which are assumed to be based on stochastic price evolution in our model. An important source of uncertainty in agricultural decision making is the production level. We do not account for that uncertainty for two reasons. The first reason is that we are not interested in the actual economic outcome of the agricultural decisions as land-use decisions are based on expected prices. The second reason is that, if uncertainty on yield may affect land-use decision, one can consider a certain-equivalent yield and take it as the new benchmark, or modify the expected price to adapt it to the uncertain quantity to be produce. In the same way, the impact of uncertainty on fertilizer use is not accounted for as it modifies yield and costs, but not land-use decisions.
the level of Grassland revenue which can be adjusted thanks to subsidies. This level corresponds to a sure profit, to be compared with the fluctuating profit resulting from market price fluctuations for cropland product.

- the magnitude of price fluctuation around long-run mean price of cropland output.

3 The ecological-economic model

We consider an integrated model of agricultural land use and biological population dynamics. The agricultural land use generates a dynamic landscape. This landscape is the habitat of a biological population which has a spatially explicit dynamics.

3.1 Agricultural land use generates a dynamic landscape

We consider two types of land use: intensive cropland (hereafter C) and extensive grassland (hereafter G). At a regional level, the landscape results from the agricultural production decisions for all fields. At the field level, this decision depends on both expected agricultural commodity prices and exogenous soil quality which influences yields.

Regional soil quality map The modeled area is represented by a map of I identically shaped, contiguous cells (hereafter fields) arranged in a $20 \times 20$ regular lattice and whose positions are defined by their centers. Each cell $i = 1, \ldots, I$ is a field of area $1$, and each field $i$ has some inherent fertility or quality $Q_i$ reflecting the local variability of soil quality. For example, $Q$ might represent the soil deepness, or its water supply properties. This quality parameter is normalized in the range $[0, 1]$. It is assumed to influence crop yields but not grassland production.

Soil quality is heterogeneous at the regional level. This heterogeneity is represented by the cumulative distribution $F(Q)$ of the parameter $Q$. Moreover, we consider spatial autocorrelation of soil quality, i.e., closer fields are likely to have closer soil quality. Details on this spatial distribution of soil quality will be given when we describe the soil quality map generation in section 4.

Fig. 4 represents a given spatial distribution of soil qualities at the regional level.

Agricultural yield At the regional level, the minimum potential yield ($Y_{inf}$) and the maximum one ($Y_{sup}$) are known, which makes it possible to define the relationship between soil quality and crop yield by the following Mitscherlich-Baule yield function, which

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2Consider for example the rotation wheat / rapeseed / sunflower.

3Or perennial alfalfa crop with low input intensity and ecological-friendly practices.
Figure 1: An example of soil quality map. Soil quality cumulative distribution is represented by a Beta function $B(\ldots)$. The soil quality map is drawn with Beta random fields, with parameters ...(or Gaussian Random Fields if variance $<<$ mean)

depends on the exogenous soil quality $Q$ and on the level $f$ of fertilizer use:

$$Y(Q,f) = (Y_{inf} + Q(Y_{sup} - Y_{inf})) \left(1 - c_2 e^{-c_1 f}\right)$$

(1)

Parameters $c_1$ and $c_2$ describe the yield response to fertilizer: $c_2$ represents the loss fraction of yield when no Nitrogen fertilizer is used; $c_1$ represents the marginal effect of Nitrogen on yield. We assume that each farmer knows the quality of its field, but that this information is not available to the policymaker. This latter only knows the minimum and maximum potential yields in the area.

**Price volatility and price anticipations** We assume that the market price for the agricultural commodity is represented by the following stochastic process, accounting for serial correlation (Deaton and Laroque, 1992). The price level equation is

$$p_t = A + B p_{t-1} + u_t,$$

(2)

where $p_{t-1}$ is the agricultural price at previous year; $A$ is the coefficient of a constant exogenous variable; $B$ is the coefficient term of autocorrelation; $u_t$ is a normally distributed random shock, with $E(u_t) = 0$.

By rescaling eq.(2) to the average price $\bar{p}$, we have

$$p_t - \bar{p} = B (p_{t-1} - \bar{p}) + u_t.$$
Along with the simulated prices, we calculate the corresponding conditional expectations, which we use to endow farmers with rational expectations of next period prices:

$$E_{t-1}[p_t] = (1 - B) \bar{p} + B p_{t-1}. \quad (3)$$

More generally, expectations at period $t - 1$ of prices in $n + 1$ period are given by

$$E_{t-1}[p_{t+n}] = (1 - B^{n+1}) \bar{p} + B^{n+1} p_{t-1}. \quad (4)$$

**Alternative land use profit** Each field is assumed to be owned by a farmer. The land-use and land-use change decisions depend on the relative gross return of both potential land uses, i.e., grassland $G$ or cropland $C$.

If a field is used as a grassland ($G$), the annual gross return on that field is

$$\pi_G(s_G) = p_G + s_G, \quad (5)$$

where $p_G$ is the revenue of grassland, which is assumed to be stationary, and $s_G$ is the subsidy per unit area of extensive grassland ([euros/ha]). We assume here that the revenue obtained from extensive grassland does not depend on the soil quality; costs are included in the revenue.

If a field of quality $Q$ is used as a cropland ($C$), the annual gross return on that field is

$$\pi_C(p_C, Q, f) = p_C Y(Q, f) - \omega f - C_C, \quad (6)$$

where $Y(Q, f)$ is the crop yield on that field in [tons/ha] given by eq. (1), $p_C(t)$ the crop selling price per unit produced ([euros/tons]), $C_C$ the specific cost of cropland (except fertilizer) ([euros/ha]), and $\omega$ a cost of fertilizer use per hectare ([euros/ha]).

The gross return from cropland will both depend on the soil quality, which is not chosen, and on the use of fertilizers, which is a decision variable. The optimality condition on the use of fertilizers is given by the following first order condition:

$$\frac{\partial \pi_C(p_C, Q, f)}{\partial f} = 0,$$

which implies after some basic computation

$$f^*(p_C, Q) = \frac{-1}{c_1} \ln \left( \frac{\omega}{p_C c_1 c_2 (Y_{inf} + Q(Y_{sup} - Y_{inf}))} \right). \quad (7)$$

There is thus a unique optimal fertilizer use for cropland on a field, which depends on its soil quality $Q$. Having characterized input choices, we will henceforth take them as given and fixed, focusing instead on the soil heterogeneity distribution and its impact on yields and land use. In particular, using eq. (7), one can compute the optimal production level of a given crop on soil quality $Q$:

$$Y_C^*(Q, f^*(p_C, Q)) = (Y_{inf} + Q(Y_{sup} - Y_{inf})) - \frac{\omega}{p_C c_1}. \quad (8)$$
The optimal production of a crop is linearly increasing with respect to the soil quality. One can now define the profit of that crop with respect to the soil quality:

\[
\pi_C^*(p_C, Q, f^*(p_C, Q)) = p_C Y_C^*(Q, f^*(p_C, Q)) - \omega f^*(p_C, Q)
\]

\[
= p_C (Y_{inf} + Q(Y_{sup} - Y_{inf})) - \omega \left[1 - \ln \left(\frac{\omega}{p_C c_1 c_2 (Y_{inf} + Q(Y_{sup} - Y_{inf}))}\right)\right].
\]  

Eq.(8) defines the per area unit profit of cropland on a soil of quality \(Q\) with respect to a given economic price of output \(p_C\). It is of interest to note that this relationship is increasing with respect to \(Q\), i.e., the higher the soil quality, the higher the profit of the land-use.

**Agroeconomic decision process: Land use and land-use change**  We consider land-use change costs, denoted \(C\). The conversion cost from grassland to cropland \(C_{G-C}\) corresponds to the extra mechanical work to prepare cropland after a grassland use with respect to the soil preparation after a cropland use, and to the required use of herbicide to avoid grass retake. The conversion cost from cropland to grassland \(C_{C-G}\) corresponds to the initial sowing (the grassland is then lasting), and also includes a loss of revenue (with respect to \(p_G\)) the first year because grass production is initially low. We consider that conversion cost are asymmetric: the conversion of cropland to grassland is more costly than the reverse, i.e., \(C_{C-G} > C_{G-C} > 0\).

As land use decisions are made before knowing the actual selling price, farmers base their decisions upon anticipated prices \(\tilde{p}\) defined by eq. (3) and (4).\(^5\) We denote the anticipated price at year \(t\) from year \(s\) with tildes, as \(\tilde{p}_s(t)\), with \(t > s\).

At the beginning of each year (from \(t_0 + 1\) in the following example), farmers decide land-use by maximizing the discounted expected profit over time, accounting for all potential land-use change and price expectations. We introduce the boolean land-use indicator \(1_{LU}(t)\),

\[
1_{LU}(t) = \begin{cases} 0 & \text{if } LU(t) = C, \\ 1 & \text{if } LU(t) = G. \end{cases}
\]

We also introduce the boolean land-use change indicator \(1_{LUC}(t)\),

\[
1_{LUC}(t) = \begin{cases} 0 & \text{if } LU(t) = LU(t-1), \\ 1 & \text{if } LU(t) \neq LU(t-1). \end{cases}
\]

Given \(1_{LU}(t_0)\), the program of the farmer is\(^6\)

\[
\max_{1_{LU}(\cdot)} \sum_{t=t_0+1}^{\infty} \frac{1}{(1+\delta)^{t-t_0}} \left( 1_{LU}(t) \pi_G + (1 - 1_{LU}(t)) \pi_C(\tilde{p}_0(t), Q) - 1_{LUC}(t)(1_{LU}(t)C_{C-G} + (1 - 1_{LU}(t))C_{G-C}) \right).
\]

\(^5\)The producers are price-takers and the local production does not influence prices.

\(^6\)To approximate the infinite time maximum discounted expected profit, which we call Net Present Value of land-use (NPV), we introduce the following benchmark \(\pi_C(Q) = \pi_C(p, Q)\), which is the profit for a field of quality \(Q\) for the mean price \(\bar{p}\). Given price expectation defined by eq.(4), the expected price converges toward the mean price \(\bar{p}\). There is thus a finite time \(T \geq t_0\) such that, for \(t > T\), \(|\tilde{p}_t(t) - \bar{p}| < \epsilon\). We compute the discounted value of constant profits of the alternative land use, i.e., \(\pi_C(Q)\) and \((p_G + s_G)\). This gives us a terminal condition \(NPV(T)\). We then define Net Present Values of both land-use over time, from \(t = t_0\) to \(T\), using Bellman principle and backward optimization.
Land use decisions at field level generate a landscape of croplands and grasslands. The landscape is thus a spatial pattern of fields endogenous to the model. Fig. 2 represents a given landscape.

Figure 2: Landscape resulting from agricultural land use. Grasslands $G$ are green (/dark) and croplands $C$ are yellow (/clear).

3.2 The landscape is an habitat for a biological population

The landscape is also an habitat for a biological population, which is depicted with a metapopulation model where subpopulations, whose growth is affected by the local land use, are connected by dispersal processes. At the field level, the population dynamics is positively influenced by grassland and negatively by cropland. This negative effect is aggravated by the fertilizer quantity. We thus assume that the population growth is positive on grassland and negative elsewhere. Population dynamics result from a local, field-scale growth process with population regulation, and a dispersal process connecting the various sub-populations by exchanging individuals. The growth process at the field scale happens during the interval $[t, t + \frac{1}{2})$ and the dispersal process connecting local populations during $[t + \frac{1}{2}, t + 1)$.

Population growth The local growth in each field $i$, during $[t, t + \frac{1}{2})$, is modeled according to a Poisson-Ricker growth model with average:

$$E_t(N_{i,t+\frac{1}{2}}) = N_{i,t} \exp \left( r_{i,t}(1 - \alpha_{i,t} N_{i,t}/K) \right)$$

(10)

where the maximum growth rate $r_{i,t}$ depends on the land use of field $i$ at time $t$ (with only two possible values: $r_G > 0$, or $r_C < 0$) and $\alpha_{i,t}$ is a correction term with value of zero whenever the growth rate is negative and one otherwise. Such a correction is necessary to ensure that the population always decreases when the local abundance is above the carrying capacity. The Ricker model can generate complex dynamics (see, e.g., Kot, 2001) but only for large values of $r$ that will not be considered here, so that

7We think mainly of small mammals or birds living in fields, but it might be also insects, or even plants.
it only exhibits saturating population growth. The demographic stochasticity embedded in the Poissonian variation around the mean allows for local extinction, we have here $N_{i,t+h} \sim \text{Poisson}(\mathbb{E}(N_{i,t+h}))$.

We can introduce a dependence of the local growth rate on the level of intensification by making (negative) growth rate proportional to the fertilizer level (pesticides are supposed to follow the same trend). It writes $r_C = r_0 f$. That assumption might describe, for instance, the local decline of birds deprived of insect food/having more fragile eggs because of pesticides.

**Dispersal**  The local dynamics within the $I$ fields are linked by dispersal processes during $[t + \frac{1}{2}, t + 1)$ according to

$$
\mathbb{E}_{t+\frac{1}{2}}(N_{i,t+1}) = N_{i,t+\frac{1}{2}} + \sum_{j \neq i} D_{ji}(N_{j,t+h} - N_{i,t+h})
$$

where $D_{ji}$ is the proportion of individuals that disperses from field $j$ to field $i$ as a function of the distance between fields $d_{ji}$ (calculated w.r.t. their centroids) and that is determined by

$$
D_{ji} = \beta \frac{f(d_{ji})}{\sum_{k \neq i} f(d_{ki})}
$$

The parameter $\beta$ is the percentage of disperser individuals in a field and $f$ is a 2D Gaussian dispersal kernel (integrating to 1) reflecting the declining strength of dispersal with distance. In eq.(11), we computed the expected number of individuals after dispersal, but this expected number is not always realized because of dispersal stochasticity. Ideally we should model dispersal with a multinomial distribution (urn model), having $D_{ji}$ the probability of dispersing to patch $i$ from $j$ (with $D_{ii} = (1 - \beta)$ the probability of staying in the patch). However, this is computationally very slow, so that we use a rounding of the real-valued number of arriving individual in patch instead. Such rounding prevents irrelevant values (“nano-individuals” do not disperse) but assumes that random effects associated to dispersal are nonexistent.

$$
f(d) = \frac{1}{2\pi\sigma_N^2} \exp\left(-\frac{d^2}{2\sigma^2}\right)
$$

The parameter $\sigma_N$ is the dispersal range, which is expressed in units of inter-field distance (1 inter-field distance = $\frac{1}{p}$ where $p$ is the length of the lattice, equal to 20 here). To avoid edges effects, margins are wrapped around so that dispersal happens between fields located at opposite edges of the lattice.\(^8\)

**Behavior of the model**  When growth rates are spatially unequal but constant over time, this model has a inhomogeneous positive equilibrium (see appendix) whose spatial

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\(^8\)Edges effects might be important for the population dynamics of several species in fragmented habitats. However in our case we consider that the habitat above the edges is similar, and therefore using toroidal boundaries makes sense.
arrangement is determined by the distances between patches and the shape of the dispersal kernel. When habitat types, and thus growth rates, are constant over time, the population as a whole is a generalized source-sink system where biomass flows from fields with positive growth rates toward fields with negative growth rates. However, temporal variability of the growth rates changes the pictures. In highly dynamic landscapes, the turnover of favorable habitat patches is usually detrimental to the number of patches occupied (in occupancy models, e.g. Ovaskainen 2007) or the abundance in favorable patches \((C)\) patches here). In some cases however spatiotemporal variability has the positive effect of creating ephemeral patches (Wimberly 2006): this might happen when dispersal is extremely short-ranged (very small \(\sigma N\)) and the turnover is high.

Fig. 3 represents the evolution of the population through time in a given fixed landscape. The population reaches an equilibrium quite quickly (before \(t = 40\)).

![Figure 3](image)

Figure 3: (a) Soil Quality Map (b) Population dynamics snapshots for various times and a landscape half-made of grassland. Parameters : \(r_C = -0.1, r_G = +0.1, \beta = 0.25, \sigma = 1, K = 30, N_{thresh} = 2\)

**Indicators of population persistence and variability**  Given the population model, we have to assess the biological characteristics of population: is it likely to persist in time? Is there wide fluctuations around the mean? For that purpose we compute the probability of extinction in a given time horizon \((T = 100\) in most cases), as well as the mean value and standard deviation over time of the global abundance.
Indicators of land use and land-use change  To characterize the land use and its suitability for the species, we consider the share of favorable habitat $P$. As it evolves over time, we compute the mean and standard deviation over time of the ratio of grassland over the total number of fields. It gives us an information on the evolution of the habitat.

We are also interested in the frequency of land-use change, and the way the economic components of the model make the landscape vary in time. It might be done by defining a rate of change per parcel (or patch, field) $\rho_i$. Then we can define its spatial mean $m(\rho)$ and spatial standard deviation $sd(\rho)$ (we will take, in our computations, averages over of sample of these values). $sd(\rho)$ is a measure of the spatial heterogeneity in landscape dynamics. These are rather crude measures because we do not distinguish changes $G \rightarrow C$ and $C \rightarrow G$. A cleaner approach (but more complicated) involves the matrix of transition rates

\[
\begin{pmatrix}
\rho_{GG} & \rho_{GC} \\
\rho_{CG} & \rho_{CC}
\end{pmatrix}
\]

More complex measures can take into account the correlation structure of the land-use time series (e.g. $GGCGCCGGGGG$) at higher-orders (e.g. autocorrelation functions with time lags $> 1$).

4 Evolution of population abundance in dynamic landscape

4.1 A case-study

In order to apply our methodology with some realistic parameters, we consider a simplified representation of a European agricultural landscape. We want to assess the potential impact of agricultural prices fluctuation on the long run dynamics and abundance of some Passerine bird. Our stylized model does not aim at reproducing any particular system but rather at providing some qualitative results with biologically and economically meaningful orders of magnitude. Therefore, we take the wheat production as the reference for the cereal production system, and assess the theoretical effect of price fluctuation on a (theoretical) small bird population. We thus consider two potential land-uses: grassland or cropland.

Land quality heterogeneity  We consider random soil quality maps characterized by soil quality distribution and spatial auto-correlation of qualities. Each soil quality map is generated using Beta function distribution calibrated on French soil quality data (see Fig.4).

The initial land use is defined with respect to the expected profit of each land use by comparing $\pi_G$ and $\pi_C(\bar{p}, Q)$ for each field, given its quality $Q$.

Economic parameters  This illustration is inspired from the Plaine de Niort (Deux-sèvres, France) case-study. For agronomic and economic data, we refer to Girard (2006) and Desbois and Legris (2007). The mean wheat price in the area is $\bar{p} = 113.42$ euros/t (average between 1993 and 2007). In the initial context (2008 data), the price is $p_C^{2008} = 220$ euros/t. The costs of production (excluding fertilization cost) are $c_c = 222$ euros/ha.
Nitrogen costs are $\omega = 1.15$ euros/kg. We assume that the benefits for grassland are equal to the opportunity cost of alfalfa, i.e., $p_G = 191$ euros/ha (including costs). For estimation of price fluctuation parameters, we use the Grilli and Yang (1988) commodity prices, updated by Pfaffenzeller et al. (2007). Prices are annual and extend from 1900 to 2003. We use price information on wheat. Agricultural prices have a positive first order correlation, a behavior which can be related to the effect of storage that tends to smooth shocks over several periods (Deaton and Laroque, 1992). It implies that a period of low (high) prices is most susceptible to be followed by low (high) prices. Lagged effects of one commodity over another are limited. Auto-correlation of wheat price with previous year is highly significant (at the 1% level), and the coefficient value is $B = 0.559$, with a variance residual of 0.058. Fig. 5 exhibits an example of random price time series.

The Mitscherlich nitrogen response function for wheat is calibrated with the following parameters (Monod et al., 2002): $c_1 = 0.015$ and $c_2 = 0.61$. In order to calibrate nitrogen response function to the actual yield level in the considered area, the minimum expected potential yield of wheat in the area is set at $Y_{inf} = 4.8$ t/ha (tons per hectare) and the maximum potential yield is $Y_{max} = 10.8$ t/ha.
Land-use conversion cost are \( C_{C\rightarrow G} = 200 \) euros and \( C_{G\rightarrow C} = 50 \) euros.

4.2 The effect of land-use change due to price fluctuation

Using the model presented in section 3, we first describe the evolution of the biological population when agricultural price fluctuates.

Using Monte Carlo simulations, we generate a large number of price scenarios according to eq. (2).

The benchmark model uses the following set of parameters. The standard deviation of the normal distribution of price shock is set to \( \sigma_u = 20 \) euros. The discount rate is equal to \( \delta = 0.05 \). We also set aside the effect of intensification in this benchmark (\( r_C = 0.1 \)).

Fig. 6 presents some of the outputs of the model for a given random price scenario. The top panel presents the price evolution over time (blue dashed line), and the proportion of Grassland in the landscape (green plain line). One can see that when the price is higher (resp. lower) for some period, grassland proportion increases (resp. decreases). The second panel presents the global population abundance. In this example, population collapses after some time. The third panel presents land use for 100 of the fields, with respect to time (black is for cropland and white for grassland). Some fields have a stationary land use, while other exhibit land-use change. The last panel presents the local abundance in a couple of patches. Time fluctuations are due to stochastic demography. One can also observe local extinction, and also some colonization.

![Figure 6: Dynamics of the bioeconomic system for a given random price scenario.](image-url)
Doing Monte Carlo simulation over price scenarios, we can analyze the statistical effect of the various parameters on the biological conservation objective.

**The grassland subsidy effect** The first economic parameter we study is the effect of the sure revenue from grassland. We parametrize the probability of persistence with the grassland subsidy level. Results are presented in Fig. 7. The dashed blue line is the persistence probability, which is our biological outcome, as a function of the grassland subsidy. The green plain line is the mean total cost of the incentive policy with respect to the subsidy level.

![Figure 7: Effect of subsidy on conservation objective, and associated costs](image)

**Sensitivity to the price fluctuation intensity** We consider three values for $\sigma_u$: 10, 20 and 30. These results are compared to the benchmark defined in previous section with a standard deviation of 20 euros (medium fluctuation).

One can see that the larger the fluctuation, the lower the persistence probability for a given subsidy level.

5 Conclusion

Land-use change has been identified as a main driver of biodiversity erosion, especially in agricultural landscapes. The usual strategy of creating reserves cannot be applied when landowners are private agents. In this case, incentive-based measures must be considered.

Land use in agricultural landscape are the product of farmers choices, depending on the interaction between economic context and agronomic factors. Fluctuating agricultural prices may result in dynamic landscape, with change between land-uses. It may then be necessary to evaluate the outcomes of agri-environmental schemes for biological conservation using bioeconomic dynamic models.

In this paper, we develop a theoretical framework to analyze the links between agricultural land use and biological conservation objectives, in a dynamic framework. Using an ecological-economic model, we built a biological benefit function defining the survival
probability of the species as a function of the incentive conservation policy level (grassland subsidy in our stylized model). We also related that survival probability to the total cost of the conservation policy. We examine the influence of the economic context, in particular the magnitude of agricultural price fluctuation, on the biological objective. We show that the benefit function is a S-shaped curve: The higher the subsidy level, the higher the survival probability. The higher the price fluctuations, the less efficient a given subsidy level.

Future research will consider the influence of spatial heterogeneity on conservation objectives, and the design of incentive-based policies to preserve biodiversity in agricultural landscapes.

References


