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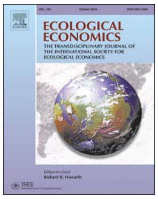

Analysis

A Spatially Explicit Choice Model to Assess the Impact of Conservation Policy on High Nature Value Farming Systems

Paulo Flores Ribeiro[a,*], Luís Catela Nunes[b], Pedro Beja[c,d], Luís Reino[c,d], Joana Santana[c,d], Francisco Moreira[c,d], José Lima Santos[a]

[a] CEF – Centro de Estudos Florestais, Instituto Superior de Agronomia, Universidade de Lisboa, Tapada da Ajuda, 1349-017 Lisboa, Portugal
[b] Nova School of Business and Economics, Universidade Nova de Lisboa, Campus de Campolide, 1099-032 Lisboa, Portugal
[c] CIBIO/InBio, Centro de Investigação em Biodiversidade e Recursos Genéticos, Universidade do Porto, Campus Agrário de Vairão, Vairão, Portugal
[d] CEABN/InBio, Centro de Ecologia Aplicada "Professor Baeta Neves", Instituto Superior de Agronomia, Universidade de Lisboa, Tapada da Ajuda, 1349-017 Lisboa, Portugal

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ABSTRACT

High Nature Value (HNV) farmland is declining in the EU, with negative consequences for biodiversity conservation. Agri-environment schemes implemented under the Common Agricultural Policy have addressed this problem, with recent proposals advocating direct support to HNV farming systems. However, research is lacking on the economics of HNV farming, which makes it difficult to set the level and type of support that ensure its sustainability. In this paper, we focused on a Special Protection Area for steppe bird conservation, analysing how economic incentives, biophysical and structural features govern the choice of farming system. We found that persistence of the traditional farming system important for steppe birds was associated with economic incentives, resistance to change, and good quality soils, whereas a shift to specialised livestock production systems was favoured by higher rainfall and less fragmented farms. A supply curve built using the choice model predicted that the proportion of traditional farming increased from 20% to 80% of the landscape, when economic incentives increased from about 100€/ha to 160€/ha. Overall, our study highlights the dependence of HNV farming systems on economic incentives, and provides a framework to assess the effects of alternative policy and market scenarios to sustain farmland landscapes promoting biodiversity conservation.

1. Introduction

The concept of High Nature Value (HNV) farmland was introduced in the early 1990s to demonstrate the dependence of European biodiversity on traditional and low-input farming systems (Beaufoy et al., 1994). Despite their importance, HNV farmland is declining due to social, economic and policy pressures for either agricultural intensification or land abandonment (Oppermann and Paracchini, 2012). This is compromising the objectives established under the European Union (EU) Biodiversity Strategy to 2020 (European Commission, 2011), and it reveals a failure of the Common Agricultural Policy (CAP) to safeguard farmland biodiversity (Henle et al., 2008; Pe'er et al., 2014).

To improve the support for HNV farmland under the CAP, a recent report for the European Commission suggested an approach based on payments to farms in HNV farmland or operating HNV farming systems (Keenleyside et al., 2014a). There are, however, major operational challenges related to lack of data or indicators to identify HNV farmland or farming systems (Keenleyside et al., 2014a), as well as limited research on economic aspects of HNV farming needed to establish the level and type of funding necessary for its sustainability (Keenleyside et al., 2014a). Indeed, most studies carried out so far aimed at estimating the costs for farmers to participate in agri-environment schemes (AES) (Oñate et al., 2007; Bamière et al., 2011; Wätzold et al., 2016), or to assess farmer's willingness to accept a compensatory payment for management options benefiting the environment (Buckley et al., 2012; Ruto and Garrod, 2009). These studies typically rely on survey data from hypothetical choice experiment designs, or use models to estimate the costs of farm management or land-use changes to comply with policy regulations. In both cases, stated-preference or *ex-ante* assessments are usually applied, confirming a lack of studies using revealed preference approaches relying on observed *ex-post* behavioural data. Other recent works advocate results-based payments, as an alternative to management-based schemes, for farmland biodiversity conservation

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* Corresponding author.
E-mail addresses: pfribeiro@isa.ulisboa.pt (P.F. Ribeiro), lcnunes@novasbe.pt (L.C. Nunes), pbeja@cibio.up.pt (P. Beja), luis.reino@cibio.up.pt (L. Reino), joanafsantana@cibio.up.pt (J. Santana), fmoreira@isa.ulisboa.pt (F. Moreira), jlsantos@isa.ulisboa.pt (J.L. Santos).

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in HNV farmland (Keenleyside et al., 2014b). However, the payment calculations are still based on the same principles set out in the EU Regulations, which provide compensations for additional costs or income foregone resulting from the commitments made, including a possible additional to cover for transaction costs (Article 28(6) of Regulation (EU) No 1305/2013).

The potential of farming systems as a basis for developing agri-environment policy has been suggested (Beaufoy and Marsden, 2011; Poux, 2013; Ribeiro et al., 2016a), supported by studies evidencing links between farming systems and landscape features or farming practices of conservation relevance (Bamière et al., 2011; Ribeiro et al., 2016a, 2016b). This farming system approach represents a significant departure from current agri-environment schemes, which are based on specific management requirements and imply significant transaction costs (Mettepenningen et al., 2011; McCann, 2013; Pannell et al., 2013). This approach could be implemented, for instance, using the concept of greening the Pillar 1 of the CAP by granting a top-up payment to farmers operating farming systems associated with HNV farmland in a specific region (Ribeiro et al., 2016a). This would require identifying these HNV farming systems for different regions across the EU, and calculating the payment level required to ensure sufficient uptake by farmers. Although the underlying idea of an agri-environment policy aimed at supporting HNV farming systems may sound interesting, however, the factors driving the farmer's decision in choosing the farming system are not well understood, nor is the role that economic incentives provided by policies play in that decision.

Here, we developed a case study on a HNV farmland of extensive cereal-steppes in southern Portugal, where previous research has shown that a range of bird species of conservation concern are associated with a traditional farming system involving rotational cereal cultivation and sheep pasturing of fallows (Delgado and Moreira, 2000; Leitão et al., 2010; Moreira, 1999; Moreira et al., 2004, 2007, 2012a, 2012b). In previous studies we have demonstrated a strong dynamics of farming systems in this area in response to the CAP reform of 2003 (Ribeiro et al., 2014), which may have affected landscape patterns (Ribeiro et al., 2016b) and agricultural practices relevant for biodiversity conservation (Ribeiro et al., 2016a). In this new study, we use the same setting to model the economic rational of farming system changes, aiming at: 1) investigating the factors that influence farmer's choice of farming system, subject to biophysical, structural, policy and economic drivers and constraints; and 2) developing a framework to simulate, on a spatially explicit basis, the effects of different policy and market scenarios on HNV farmland. Results were then used to evaluate the potential of our framework to outline empirical supply curves for conservation services (Santos et al., 2008; Lewis and Wu, 2014), relating levels of payment per hectare paid to farmers operating HNV farming systems with the amount of farmland managed under such systems.

2. Methods

2.1. Study Area

The study focused on an extensive HNV farmland area in the south of Portugal, covering ca. 180,000 ha (Fig. 1). The area is characterized by open fields, smooth relief, and typical Mediterranean climate, with hot dry summers and moderately rainy cold winters. It encompasses the Special Protection Area (SPA) of Castro Verde, classified under the EU Directive 79/409/CEE (Birds Directive) to protect several steppe bird species of conservation concern. Studies carried out during the past 20 years suggest that conservation of these steppe birds requires the maintenance of an extensive traditional farming system based on rainfed cereal crops in rotation with long-term fallows grazed by sheep, which dominated the landscape for decades (Moreira, 1999; Delgado and Moreira, 2000; Leitão et al., 2010; Reino et al., 2010; Moreira et al., 2004, 2007, 2012a, 2012b; Santana et al., 2014, 2017). To support this traditional farming system, an AES is operating since 1995, though with limited success for preventing land use changes (Ribeiro et al., 2014) and protect bird diversity (Santana et al., 2014).

Recent studies have shown a shift from the traditional to livestock-grazing specialized farming systems in the area, despite de AES, possibly resulting from the decoupling of direct payments following the 2003 CAP reform, related with the Portuguese authorities' decision to keep a direct payment on suckler cows, goats and sheep (Ribeiro et al., 2014). These changes have affected landscape patterns (Ribeiro et al., 2016b) and agricultural practices (Ribeiro et al., 2016a), but their effects on biodiversity are still poorly understood. In at least some cases, however, changes are likely to be negative, including for instance the anticipation of the cereals harvesting date under the livestock system, which is judged to increase the destruction of bird nests (Ribeiro et al., 2016a). Another problem may be the loss of the rich traditional landscape mosaic represented by cereal fields, ploughed fields, and fallows of different ages and grazing intensity, which likely reduces habitat diversity for birds (Oñate et al., 2007; Delgado and Moreira, 2000; Leitão et al., 2010; Reino et al., 2010; Moreira et al., 2012a; Santana et al., 2017).

2.2. Farming Systems Identification

The dominant farming systems in the study area and their spatial dynamics during 2000 and 2010 were assessed by a cluster analysis performed on farm-level data from the EU Integrated Administration and Control System (IACS), together with spatially explicit farm-parcel data from the EU Land Parcel Identification System (LPIS). Such data has been recommended for HNV farmland research (Beaufoy and Marsden, 2011; Beaufoy et al., 2012; Keenleyside et al., 2014a), and it was successfully tested in previous studies (Ribeiro et al., 2014, 2016b). Five farming systems were identified, including two livestock specialized systems (the Cattle and Sheep systems), two systems specialized in crop production (the Annual crops and the Permanent crops systems), and a mixed farming system (the Traditional system) (details in Annex A in Supplementary Information). Due to its land use pattern, dominated by a low-intensity cereal-fallow rotation, complemented with low-density sheep grazing, this Traditional system was acknowledged as the main farming system underpinning the HNV of the study area. Each farm in each year was assigned to one of these five farming systems, thereby providing information to assess transitions over time.

2.3. Drivers and Constraints of Farming System Choice

Each farm was characterized using biophysical (soil quality, terrain slope and average annual rainfall) and structural features (farm size, farm spatial fragmentation and oak woodlands) (details in Annex B of the Supplementary Information), expected to influence farming system choice (Keenleyside et al., 2014b; Ribeiro et al., 2014). These variables varied spatially but were largely constant over time within the study period.

To capture the effects of policy and market drivers on farmer decisions, we used the gross income ratio to compare the economic profitability of the farming systems. This indicator was used because there was no time-series on detailed farm-level production costs to compute gross margins. We believe this is acceptable, since many of these farms have their own means of production (workers and equipment), which are fixed costs largely independent of the activities in which they are used, and not subject to significant fluctuations during the 10-year time span of our study. Farm management decisions were thus expected to be mostly driven by temporal variation in gross income from sales revenues and direct subsidies. The gross income ratio for each farming system in each study year t (GIR^t) was defined as:

$$GIR^t = GI^t_{RFS}/GI^t_{AFS}$$

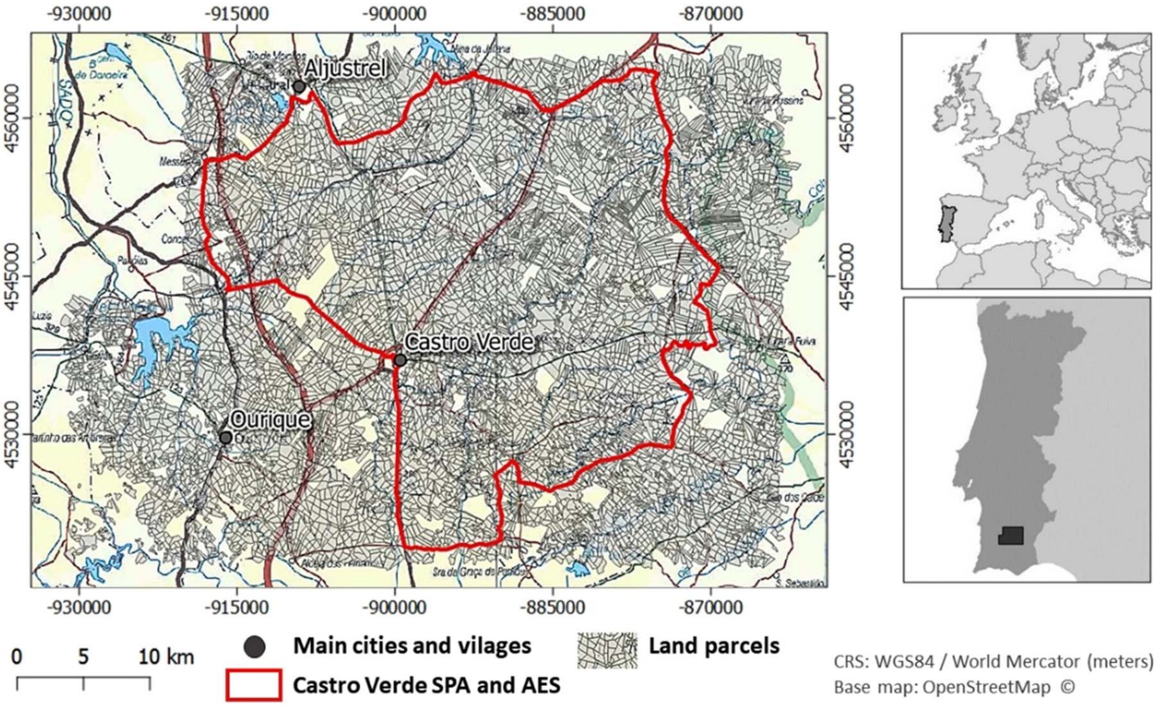


Fig. 1. Location of the study area in southern Portugal, showing the farm-parcel structure and the Special Protection Area (SPA) of Castro Verde where an agri-environment scheme (AES) is in operation since 1995.

where $GI_{RFS}{}^{t}$ is the unit gross income of a reference farming system, and $GI_{AFS}{}^{t}$ is the unit gross income of the alternative farming systems. As the reference, we considered the Traditional farming system because it is the HNV system in our study area. If the income/cost ratios of each activity do not vary significantly, then a higher GIR means a higher relative profitability of the reference farming system. However, because different productions may have very different gross income/cost ratios, we cannot interpret $GIR^t = 1$ as an indifference point between the reference and the alternative system.

To compute GIR^t, we first estimated the unit gross income of each farming system in each study year (GI^t), in euros per hectare, as follows:

$$GI^t = \sum_{i=1}^{n} (Q_i.\ A_i.\ P_i^t + A_i.\ S_i^{t+1})$$

where, for each activity i, Q_i is the average regional yield per hectare (e.g. wheat, sunflower, cattle, sheep), A_i is the area occupied (land shares), P_i^t is the producer price in year t, and S_i^{t+1} is the value of the direct subsidy in year $t + 1$. We considered subsidies in year $t + 1$ because farmers normally make their annual production decisions with a reasonable knowledge of the subsidies for the following year. Assuming that farmers make their management decisions based on knowledge of local average yields, these were kept constant during the study period. Decoupled CAP payments (e.g. the single payment scheme) were not included to estimate GI^t, because they do not depend on crop patterns and thus are unlikely to influence directly the management decisions on productions. Data on agricultural regional yields and producer prices were obtained from Portuguese official statistics, and CAP direct payments were provided by the Portuguese CAP paying agency (Table S4 in Annex C in Supplementary Information).

To capture the farmers' perception on the recent trend in the relative profitability of the farming systems, we use the change in GIR^t over time, defined as $GIR\text{dif}^{\,t} = GIR^t - GIR^{t\text{-}n}$, where n is the number of years used to calculate the change. Positive GIRdif indicates an upward trend in the profitability of the reference farming system, compared to alternative systems, and vice versa.

2.4. Discrete Choice Model Design

We used a discrete choice modelling approach based on logistic regression to investigate the drivers of farming systems shifts. Logistic regression can assume the binomial or multinomial form, depending on the number of farming systems specified as the categorical dependent variable. The independent variables included the biophysical, structural and policy constraints (Table S3 in Supplementary Information), and the economic variables GIR^t and GIRdif. A lagged dependent variable identifying the farming system in the previous period (FSlag) was included to account for adjustment costs and persistence effects considered by the farmer when making decisions.

To handle possible heterogeneity in farmers' preferences due to uncontrolled variables influencing motivations and attitudes towards policies, such as those describing socio-cultural profiles (de Snoo et al., 2013; Siebert et al., 2006), we used latent class models to account for heterogeneity in preferences (Greene, 2012). These models have been widely used in recent stated-preference studies using discrete choice models, including within the context of agri-environment policy evaluation (Garrod et al., 2012; Ruto and Garrod, 2009; Villanueva et al., 2014). Since data included repeated observations on the same farms over time, a panel data model was estimated, allowing for individual-specific heterogeneity in preferences that are constant over time (Greene, 2012).

A stepwise-like procedure was used in model building, starting by estimating the model with all candidate independent variables, and considering 1, 2 and 3 latent classes, and then successively removing the independent variable that showed the lower significance. The procedure was repeated until all variables were significant at the 5% level. To assess model fit and decide on the number of latent classes for the optimal model, we used the Akaike Information Criterion (AIC) and the Bayesian Information Criterion (BIC) (Boxall and Adamowicz, 2002; Gruen and Leisch, 2008; Greene, 2012; Villanueva et al., 2014). Prediction accuracy was assessed by fitting the model on a training subset containing 75% of randomly selected observations, and then applying it on a test set with the remaining 25% observations.

All statistical analyses were performed using the R software, version

3.1.1 (R Development Core Team, 2015). The latent class models were estimated using the "FlexMix" R package, version 2 (Gruen and Leisch, 2008).

2.5. Assessment of Policy and Market Scenarios

We used the choice model to assess the impacts of several policy and market scenarios on farming systems. This was carried out by evaluating how changes in GIR would affect the representation of different farming systems in the landscape. The approach was based on the idea that GIR integrates changes in economic incentives such as the complete decoupling of direct payments on suckler cows and sheep, changes in livestock market prices resulting from the review of EU border taxes on beef products, or the implementation of an AES paying a premium to HNV farming systems. Therefore, farmer's responses were assumed to respond to economic incentives inasmuch they affect GIR, rather than considering the details of policy change.

To provide a measure of uncertainty in the simulation of scenarios, we use the approach to calculate 95% confidence intervals based on Monte Carlo simulations (Krinsky and Robb, 1986). We generate 1000 random trials of the model coefficients from a multivariate normal distribution, using the estimated coefficients as the means vector and the corresponding standard errors and covariances as the variance-covariance matrix. These 1000 model replicas were successively run over a pre-set range of GIR values simulating the effects of any policy or market changes, using 2010 as a baseline, and observing the corresponding impact on the proportion of the study area covered by the farming systems. For each output simulation, we recorded the values corresponding to quantiles 0.025, 0.500 and 0.975 from the 1000 outcomes of the model, and the results were used to outline a supply curve for biodiversity conservation services, expressed as a proportion of the area covered by the Traditional system, bounded by a 95% confidence interval.

Simulation results were also used to assess the likely impact of policy and market scenarios on biodiversity, considering that the SPA of Castro Verde was created primarily to protect steppe bird assemblages (Ribeiro et al., 2014; Santana et al., 2014). Therefore, we assessed how changes in farming systems would affect habitat suitability for steppe birds, assuming that they are favoured by landscapes where livestock densities (in livestock units per hectare - LU/ha) and the proportion of cereal area early harvested for hay production (CEH) are low, and where the areas covered by the traditional farming system (P_RFS), and the mean patch area (MPA) and number of patches (NPATCH) of this farming system are large. Although these metrics provide an indirect link between farming systems and the conservation of steppe bird species, they reflect ecological information on these species collected in the study area during the last 20 years (Moreira, 1999; Delgado and Moreira, 2000; Leitão et al., 2010; Reino et al., 2010; Moreira et al., 2004, 2007, 2012a, 2012b; Santana et al., 2014, 2017). Landscape metrics were computed using the "SDMTools" R package, version 1.1–221 (Vanderwal et al., 2015).

3. Results

Significant farming system dynamics were observed between 2000 and 2010, particularly between 2003 and 2007 (Fig. 2). The main change was the transition from the Traditional farming system to livestock specialized systems (Cattle and Sheep) which, by the end of the study period, covered ca. 90% of the utilized agricultural area (UAA). The two specialized crop farming systems were poorly represented in the area, and they were nearly absent by the end of the study period. Because of this, and because there are restrictions to the expansion of these crops in the SPA, they were not considered in the development of the choice model.

There were significant temporal changes in gross income ratios (GIR), with a clear decline in the profitability of the Traditional system (the reference system) in relation to the Cattle and Sheep systems (Fig. S2, Annex B, in Supplementary Information). Trends in the GIRs of Sheep and Cattle systems were very highly correlated ($r = 0.98$), and so they were merged into a combined Livestock system to avoid collinearity problems in subsequent modelling. The GIR values of the Livestock system were estimated by averaging the gross income time-series of the two preceding systems (Sheep and Cattle), before recalculating the GIR. Consequently, the logistic model was specified considering a binomial choice between the Traditional (FS = 1) and the Livestock (FS = 0) systems. Given the observed temporal patterns of GIR variation (Fig. S2), we estimated choices considering the years 2001, 2004, 2007 and 2010 (1648 observations), and thus the variables GIRdif and FSlag were specified based on 3-year lags.

The independent variables retained after stepwise selection ($p < 0.050$) indicated that the choice of the Traditional system was associated with higher gross income ratio (GIR), a positive trend in GIR (GIRdif), the presence of the Traditional system three years before, and good quality soils (SOIL) (Table 1). In contrast, higher rainfall, and larger and less fragmented farms favoured the selection of the Livestock system (Table 1). We selected the model with a single latent class, as it consistently showed the lowest BIC, while AIC was nearly identical in models with either one or two classes, and much lower than in three-class models (Tables S5 and S6 in Annex D in Supplementary Information). Panel random data effects were also not included in the model structure, because they were not significant in the model with just one class (sigma $p = 0.954$). The model showed a rate of correct predictions of 86.7% in the validation estimates performed with the training and test sets.

The supply curve predicted from the choice model to assess the impact of market and policy scenarios indicates that the Traditional (reference) system is largely absent from the landscape with payments up to about 75€ per hectare (Fig. 3). The representation of this farming system then increases steadily when payments per hectare increase from about 75€/hectare to 175€/hectare, after which the landscape is almost completely occupied by this system (Fig. 3). The changes along the supply curve in the amount of the traditional farming system are predicted to affect the landscape configuration metrics used to indicate habitat suitability for steppe birds (Fig. 3). The mean patch area (MPA) of the Traditional system is expected to increase along with its proportion in the study area, while the number of patches (NPATCH) first rises as the area covered by this system also increases, and then declines as the system becomes dominant. Livestock density (LU/ha) and the proportion of cereal area early harvested for hay production (CEH) are expected to decrease with increasing area covered by the Traditional system (Fig. 3).

4. Discussion

4.1. Drivers and Constraints of Farming System Dynamics

In this study, we evaluated the factors driving major agricultural changes in the study area, which involved the decline of the Traditional farming system and its replacement by livestock specialized systems. Our results support the hypothesis raised in the previous study of Ribeiro et al. (2014) suggesting that this change was mainly driven by decoupling under the CAP reform of 2003, by showing that changes were mainly determined by variation in the gross income ratio (GIR) of the different farming systems, which in turn affected farmers' choices. In fact, the strong decrease observed in GIR between 2003 and 2007 resulted primarily from the decoupling of CAP direct payments, which affected arable crops but left mostly unchanged the direct payments on suckler cows and sheep (Table S4 in Annex C in Supplementary Information).

The role played by biophysical and structural variables was consistent with the results of previous studies. Good quality soils favoured the Traditional system, probably due to the greater relevance of crop

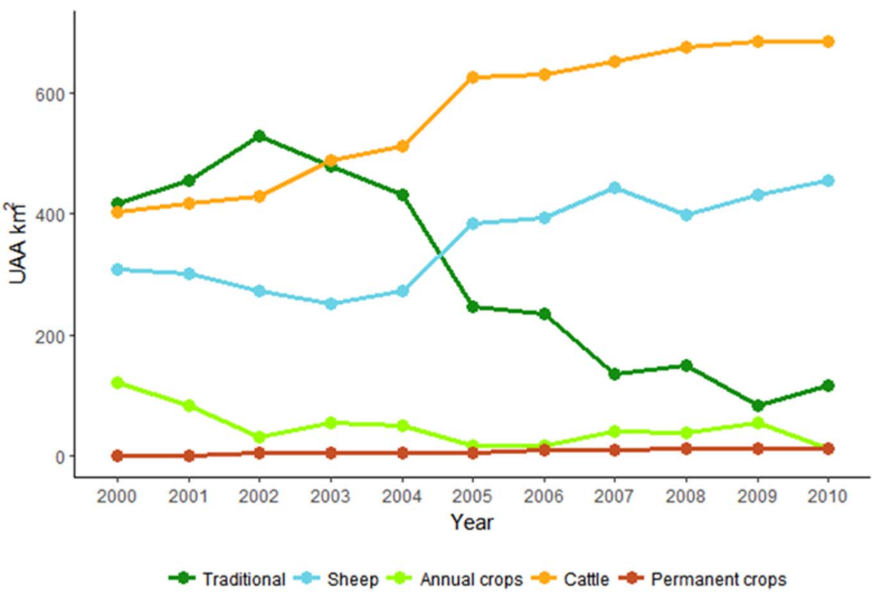


Fig. 2. Temporal variation in the percentage of the utilized agricultural area (UAA) occupied by each farming system between 2000 and 2010 in the Castro Verde region (southern Portugal).

Table 1
Binomial logistic model for the Traditional farming system choice (FS = 1).

	Coefficient (B)	Std. error	z value	Pr(> |z|)
Intercept	− 1.187	0.950	− 1.250	0.211
GIR	6.140	0.703	8.739	< 0.001***
GIRdif	4.093	1.033	3.962	< 0.001***
FSlag	2.498	0.170	14.704	< 0.001***
SOIL	1.629	0.294	5.538	< 0.001***
RAIN	− 9.525	1.530	− 6.225	< 0.001***
UAA	− 0.130	0.041	− 3.136	0.002**
JANUS	− 0.884	0.383	− 2.305	0.021*

Significance codes: 0 '***' 0.001 '**' 0.01 '*' 0.05. Model fit: log-likelihood = − 531.95 (df = 8). Number of observations = 1648.

production in this system (Ribeiro et al., 2014), while higher rainfall was associated with specialized livestock systems, possibly related to the need of higher forage yields (Howden et al., 2007). Farmers with large and less fragmented farms were more likely to choose the livestock systems, possibly to meet forage needs (Duffy, 2009), and because greater difficulty in grazing management or higher fencing (investment) costs are likely associated with more fragmented farms (Boone and Hobbs, 2004; Hobbs et al., 2008).

The choice model also evidenced resistance-to-change effects, as the farming system in any given year was positively correlated with the farming system three years before. This may be related to investment costs or risk related to changes of farming system (Pannell, 2000). For instance, shifts from the traditional farming to specialized cattle production may involve investment costs of fencing the parcels, since in this region cows usually graze in fenced parcels, while the sheep are kept by a shepherd.

The farming system dynamics observed during the study period suggest that the farmers' response to significant policy changes, such as the 2003 CAP reform, can take about 3 to 4 years to complete, since these dynamics occurred mainly between 2003 and 2007. It also suggests that many farmers anticipate the implementation of the new regulations by starting farm-management adjustments 1–2 years before (as the 2003 CAP reform only came into effect in 2005). This is arguably a potentially relevant issue in policy assessment, although nearly absent in the literature.

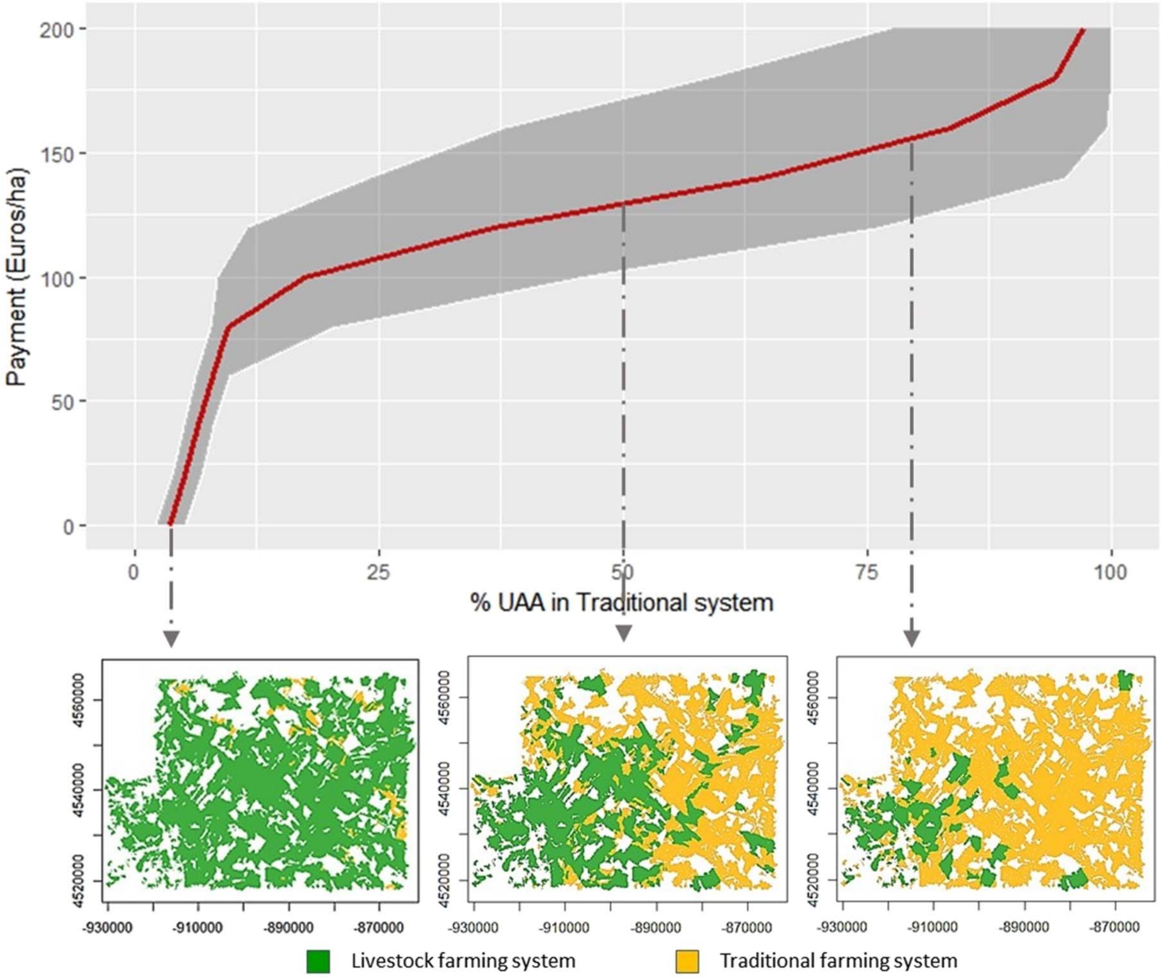
The fact that the final model had only one latent class and non-significant panel effects implies that farmers were largely homogeneous in their preferences, and thus that the independent variables capture most heterogeneity in the data. There was thus a higher homogeneity in preferences, attitudes and motivations towards economic incentives than initially expected (de Snoo et al., 2013; Siebert et al., 2006), which may be related to a focused economic rationale for profit maximization. The fact that small holdings were excluded from the analysis (Annex A) and that a large part of the remaining agricultural area is operated by business companies (ca. 50% in 2009, according to official statistics), where decisions are taken by an administration board rather than by individual farmers, could also have contributed to homogenise farm management decisions, by reducing the impact of the socioeconomic and cultural idiosyncrasies of individuals. Therefore, future studies should strive to include a higher variety of farmers, under contrasting social, economic and biophysical contexts, in order to evaluate the generality of our results.

Overall, our study provides robust information on the drivers of the replacement of the traditional by specialized livestock systems in the study area, but we could not analyse the shifts among other farming systems, and between the traditional and either sheep or cattle specialized systems. This was unavoidable given the particular characteristics of our area and the data available to undertake the study, but it implies that we could not assess the causality behind some changes that may also be relevant for biodiversity conservation (Ribeiro et al., 2016a, 2016b). Nevertheless, we think that focusing on the shifts between the traditional and the livestock systems was reasonable, because this was the main change occurring in our area during the study period, and because it entails numerous conservation management challenges (Ribeiro et al., 2016a, 2016b). In the future, our framework could be expanded to assess decisions among multiple competing farming systems, by using multinomial rather than binomial choice models.

4.2. *Policy Implications*

Results showed that local agri-environment policies within the Special Protection Area (SPA) of Castro Verde seem to have little influence on the choice of the farming system, as the variable SPA did not prove significant in the model estimations. One possible explanation is the fact that the uptake of the Castro Verde AES does not imply following a specific farming system, but rather to comply with management commitments mostly related to land use patterns, which can be met by more than one farming system (Ribeiro et al., 2016b).

The way the economic incentives were entered into the choice model, based on the ratio between the gross income of the Traditional system and the alternative system, showed high capability and efficacy



•Payment:	0 €/ha
•P_RFS:	3.6%
•GIR:	0.71
•MPA:	50.3 ha
•NPATCH:	91
•LU/ha:	0.46
•CEH:	56.9%

•Payment:	132 €/ha
•P_RFS:	50.0%
•GIR:	1.23
•MPA:	263.4 ha
•NPATCH:	235
•LU/ha:	0.34
•CEH:	43.4%

•Payment:	157 €/ha
•P_RFS:	80.0%
•GIR:	1.32
•MPA:	513,6 ha
•NPATCH:	192
•LU/ha:	0.26
•CEH:	34.8%

Fig. 3. Supply curve for biodiversity conservation services in the study area bounded by 95% confidence range (shaded area), based on model predictions from 2010 to 2013, relating increasing levels of economic incentives towards the Traditional system with the proportion of the study area managed under this farming system (axes are reversed to depict the relationship as a supply curve). The spatial arrangement of farming systems and environmental indicators are presented for three points in the curve, representing the status quo scenario of zero payment (left), an intermediate scenario where 50% of the UAA would be managed under the Traditional system requiring a payment of 132 €/ha (central), and a high payment scenario of 157€/ha where P_RFS raises to 80% (right). Indicators include livestock density (LU/ha), the proportion of cereal area early harvested for hay production (CEH), the proportion of the study area covered by the Traditional system (P_RFS), and the mean patch area (MPA) and the number of patches (NPATCH) of the Traditional system. The economic incentive (Payment) and the corresponding value for the GIR variable are also provided.

to simulate a wide range of scenarios of policy and market changes. These scenarios may include changes in CAP regulations, but also changes due to technological progress, consumption patterns, World Trade Organization negotiations or any other changes that would alter the relative prices of the outputs of the concerned farming systems. It should be noted, however, that being a ratio of gross incomes, the variable is insensitive to generalized price declines that may put farms' profitability below a sustainability threshold for all alternative farming systems, and thus may encourage farmland abandonment. This drawback might be overcome with information about the unit cost associated to each farming system, allowing the estimation of net profit. However, this would very significantly increase the data requirements to implement the framework, which should not be a problem per se if these data were readily available, which was not the case.

The model predictions showed that if the baseline (2010) political and economic situation was kept unchanged for 2013 (*status quo* scenario), the HNV Traditional farming system would continue losing area for livestock systems, reducing from the ca. 10% of total study area in 2010 to < 4% in the next time-period (2013). This transition will likely have negative impacts on steppe birds, as it would lead to a significant increase in stocking density and early-harvested cereals, and thus to higher rates of trampling and nest-destruction (Ribeiro et al., 2016a). To prevent this, an economic incentive equivalent of ca. 80Euros/ha promoting the Traditional system would have to occur, whether provided by changes in market or policy conditions, or through the implementation of an equivalent agri-environment payment assigned to the Traditional system. This figure would have to rise to 132Euros/ha for the Traditional system to take up to ca. 50% of the study area, and ca. 157Euros/ha if the target was raised to ca. 80% of the study area.

Using the same simulation procedure, we concluded that fully decoupling the payments for suckler cows and sheep would be equivalent to granting a payment of 90Euros/ha, which would result in ca. 8% of the study area under the Traditional system – almost the double than in the *status quo* scenario. Using 2004 as the baseline, we can estimate that if the suckler cows and sheep direct payment had been integrated into the single payment scheme during the 2003 CAP reform, instead of being kept as a coupled payment (a national policy decision in Portugal), the Traditional farming system would occupy in 2007 ca. 89% of the study area, instead of the current 12%. This clearly shows how broad-scope (national) policy decisions may conflict with local conservation policy goals. Alternatively, if an agri-environment payment to the Traditional system had been implemented in 2004, the amount required to achieve 50% of the area under the Traditional system in the next time-period would have been ca. 85Euros/ha, instead of the above mentioned 132Euros/ha needed in 2010. These results show how the cost of maintaining environmental quality can be much lower than recovering it, and how delaying the implementation of conservation measures may significantly undermine its cost-effectiveness (Berendse et al., 2004).

The framework developed in this study seems suitable to support the design of new agri-environment policy targeted on HNV farming systems (Beaufoy and Marsden, 2011; Poux, 2013; Ribeiro et al., 2016a). Building on IACS/LPIS data is a significant advantage, as these data are readily available from CAP payments agencies in Member States, which makes the approach easily replicable elsewhere. Some adaptations are required, however, to fit the pertinent conservation issues, such as identifying farming system most relevant for conservation which should be used as reference. The spatial component of the data and model-based simulations add important advantages, not only for determining the effects of the biophysical and structural features of the farms when making the simulations, but also by allowing to assess if the policy effects are operating where the specific targeted habitats patches or natural values occur – provided these conservation targets are mapped.

By leaning on quasi-automatic farm-level selection criteria, the application of this framework to policy design could substantially contribute to implement an alternative to the greening of the CAP's Pillar 1 (top-up environment payments) using a farming system approach to support HNV farmland across the EU with much lower transaction costs (Ribeiro et al., 2016a), which are often a major cause for farmers' low uptake of AES (McCann, 2013; Pannell et al., 2013). Private transaction costs (for farmers) for participating in AES have been estimated at over 40Euros/ha (Mettepenningen et al., 2009; Wätzold et al., 2016), which can offset a significant share of the environmental payment.

The proposed approach can, moreover, be a more reliable way to estimate the required incentive for an effective level of the conservation service; this represents the per hectare compensation that the last entering farmer is willing to accept to adopt a sub-optimal farming system. This marginal cost does not necessarily correspond to the amount achieved by formal calculations following the recommended procedures in the EU regulations, based on estimates of the additional costs or income forgone resulting from the commitments made to participate in AES. Its reliability comes from using a model that “learned” from previous choices made by farmers facing similar incentive changes.

The framework has the limitation of bounding the choice of future farming systems to the options (farming systems) that were available in the recent past. Although this may not be a significant problem if the HNV farming systems to be protected are well identified, it does not allow the emergence of alternative systems with equal or higher conservation value, nor of high-profit agricultural systems, potentially destructive to the natural value. In our case study, it additionally suffered from multicollinearity problems not allowing the ratio of Cattle and Sheep systems to change in the future, since they had to be combined into the Livestock system. The approach also requires a previous knowledge of the HNV farming systems to be supported and on their minimum farmland share to meet conservation objectives. An alternative approach would be to establish, on a cost-benefit basis, the optimal point in the conservation service supply curve, for which we need the value of the marginal benefit of conservation (e.g. marginal willingness to pay for conservation), and not only that of the marginal cost expressed in the supply curve (Santos, 1998). Despite their potential usefulness to conservation management, both approaches can be a more complex issue than it seems, since there is often a lack of research data to support such decisions (Keenleyside et al., 2014a).

4.3. Conclusions

Our findings provide a significant contribution to the understanding of the factors governing farmers' decision on the choice of the farming system in areas of HNV farmland, highlighting the main role played by market and policy drivers, subject to the degrees of freedom allowed by biophysical and structural constraints. The proposed framework represents a methodological contribution to increase the empirical knowledge of the economics of HNV farming systems, taking advantage of readily available information in the EU (IACS/LPIS data) to derive spatio-temporal farming systems choice models to assess the effects of alternative policy and market scenarios on HNV farmland. The framework enabled the derivation of a supply curve for biodiversity conservation services bounded by 95% confidence intervals, showing the adoption levels of HNV farming systems under different levels of economic incentive resulting from policy and market scenarios, which can be valuable information for policy makers focused on optimizing the provision of ecosystem services. Overall, our framework supported the feasibility and usefulness of a farming systems approach to address farmland biodiversity conservation issues, providing potentially useful information to inform the design of future EU policy for HNV farming, helping to meet the biodiversity targets of the EU in a more cost-effective way.

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Appendix A. Supplementary data

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References

Bamière, L., Havlík, P., Jacquet, F., Lherm, M., Millet, G., Bretagnolle, V., 2011. Farming system modelling for agri-environmental policy design: the case of a spatially non-aggregated allocation of conservation measures. Ecol. Econ. 70, 891–899. http://dx.doi.org/10.1016/j.ecolecon.2010.12.014.

Beaufoy, G., Marsden, K., 2011. CAP Reform 2013: last chance to stop the decline of Europe's High Nature Value farming. In: European Forum on Nature Conservation and Pastoralism. Birdlife International European Division, Butterfly Conservation Europe, WWF European Policy Office.

Beaufoy, G., Beopoulos, N., Bignal, E., Dubien, I., Koumas, D., Klepacki, B., Louloudis, L., Markus, F., McCracken, D., Petretti, F., Poux, X., Theoharopoulos, J., Yudelman, T., 1994. The Nature of Farming - Low Intensity Farming Systems in Nine European Countries. Institute for European Environmental Policy, London.

Beaufoy, G., Keenleyside, C., Oppermann, R., 2012. How should EU and national policies support HNV farming? In: Oppermann, R., Beaufoy, G., Jones, G. (Eds.), High Nature Value Farming in Europe. Verlag Regionalkultur, pp. 525–535.

Berendse, F., Chamberlain, D., Kleijn, D., Schekkerman, H., 2004. Declining biodiversity in agricultural landscapes and the effectiveness of agri-environment schemes. AMBIO A J. Hum. Environ. 33, 499–502. http://dx.doi.org/10.1579/0044-7447-33.8.499.

Boone, R.B., Hobbs, N.T., 2004. Lines around fragments: effects of fencing on large herbivores. African J. Range Forage Sci. 21, 147–158. http://dx.doi.org/10.2989/10220110409485847.

Boxall, P.C., Adamowicz, W., 2002. Understanding heterogenous preferences in random utility models: a latent class approach. Environ. Resour. Econ. 23, 421–446. http://dx.doi.org/10.1023/A:1021351721619.

Buckley, C., Hynes, S., Mechan, S., 2012. Supply of an ecosystem service-farmers' willingness to adopt riparian buffer zones in agricultural catchments. Environ. Sci. Pol. 24, 101–109. http://dx.doi.org/10.1016/j.envsci.2012.07.022.

de Snoo, G.R., Herzon, I., Staats, H., Burton, R.J.F., Schindler, S., van Dijk, J., Lokhorst, A.M., Bullock, J.M., Lobley, M., Wrbka, T., Schwarz, G., Musters, C.J.M., 2013. Toward effective nature conservation on farmland: making farmers matter. Conserv. Lett. 6, 66–72. http://dx.doi.org/10.1111/j.1755-263X.2012.00296.x.

Delgado, A., Moreira, F., 2000. Bird assemblages of an Iberian cereal steppe. Agric. Ecosyst. Environ. 78, 65–76. http://dx.doi.org/10.1016/S0167-8809(99)00114-0.

Duffy, M., 2009. Economies of size in production agriculture. J. Hunger Environ. Nutr. 4, 375–392. http://dx.doi.org/10.1080/19320240903321292.

European Commission, 2011. Communication from the commission to the European parliament, the council, the economic and social committee and the committee of the regions. In: Our Life Insurance, Our Natural Capital: An EU Biodiversity Strategy to 2020. European Commission, Brussels.

Garrod, G., Ruto, E., Willis, K., Powe, N., 2012. Heterogeneity of preferences for the benefits of Environmental Stewardship: a latent-class approach. Ecol. Econ. 76, 104–111. http://dx.doi.org/10.1016/j.ecolecon.2012.02.011.

Greene, W.H., 2012. Econometric Analysis, Seventh ed. Pearson.

Gruen, B., Leisch, F., 2008. FlexMix version 2: finite mixtures with concomitant variables and varying and constant parameters. J. Stat. Softw. 28, 1–35. http://dx.doi.org/10.18637/jss.v028.i04.

Henle, K., Alard, D., Clitherow, J., Cobb, P., Firbank, L., Kull, T., McCracken, D., Moritz, R.F.a.A., Niemelä, J., Rebane, M., Wascher, D., Watt, A., Young, J., 2008. Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe–a review. Agric. Ecosyst. Environ. 124, 60–71. http://dx.doi.org/10.1016/j.agee.2007.09.005.

Hobbs, N.T., Galvin, K.A., Stokes, C.J., Lackett, J.M., Ash, A.J., Boone, R.B., Reid, R.S., Thornton, P.K., 2008. Fragmentation of rangelands: implications for humans, animals, and landscapes. Glob. Environ. Chang. 18, 776–785. http://dx.doi.org/10.1016/j.gloenvcha.2008.07.011.

Howden, S.M., Soussana, J.-F., Tubiello, F.N., Chhetri, N., Dunlop, M., Meinke, H., 2007. Adapting agriculture to climate change. Proc. Natl. Acad. Sci. 104, 19691–19696. http://dx.doi.org/10.1073/pnas.0701890104.

Keenleyside, C., Beaufoy, G., Tucker, G., Jones, G., 2014a. High Nature Value farming throughout EU-27 and its financial support under the CAP. In: Report Prepared for DG Environment, Contract No ENV B.1/ETU/2012/0035. Institute for European Environmental Policy, London. http://dx.doi.org/10.2779/91086.

Keenleyside, C., Radley, G., Tucker, G., Underwood, E., Hart, K., Allen, B., Menadue, H., 2014b. Results-based payments for biodiversity guidance handbook: designing and implementing results-based agri-environment schemes 2014–20. In: Prepared for the European Commission, DG Environment, Contract No ENV.B.2/ETU/2013/0046. Institute for European Environment.

Krinsky, I., Robb, A.L., 1986. On approximating the statistical properties of elasticities. Rev. Econ. Stat. 68, 715. http://dx.doi.org/10.2307/1924536.

Leitão, P.J., Moreira, F., Osborne, P.E., 2010. Breeding habitat selection by steppe birds in Castro Verde: a remote sensing and advanced statistics approach. Ardeola 57, 93–116.

Lewis, D.J., Wu, J., 2014. Land-use patterns and spatially dependent ecosystem services: some microeconomic foundations. Int. Rev. Environ. Resour. Econ. 1–42. http://dx.doi.org/10.1561/101.00000069.

McCann, L., 2013. Transaction costs and environmental policy design. Ecol. Econ. 88, 253–262. http://dx.doi.org/10.1016/j.ecolecon.2012.12.012.

Mettepenningen, E., Verspecht, A., Van Huylenbroeck, G., 2009. Measuring private transaction costs of European agri-environmental schemes. J. Environ. Plan. Manag. 52, 649–667. http://dx.doi.org/10.1080/09640560902958206.

Mettepenningen, E., Beckmann, V., Eggers, J., 2011. Public transaction costs of agri-environmental schemes and their determinants—analysing stakeholders' involvement and perceptions. Ecol. Econ. 70, 641–650. http://dx.doi.org/10.1016/j.ecolecon.2010.10.007.

Moreira, F., 1999. Relationships between vegetation structure and breeding bird densities in fallow cereal steppes in Castro Verde, Portugal. Bird Study 46 (3), 309–318.

Moreira, F., Morgado, R., Arthur, S., Great, S., 2004. Great bustard *Otis tarda* habitat selection in relation to agricultural use in southern Portugal. Wildl. Biol. 251–260.

Moreira, F., Leitão, P.J., Morgado, R., Alcazar, R., Morgado, R., Cardoso, A., 2007. Spatial distribution patterns, habitat correlates and population estimates of steppe birds in Castro Verde. Airo 17, 5–30.

Moreira, F., Leitão, P.J., Synes, N.W., Alcazar, R., Catry, I., Carrapato, C., Delgado, A., Estanque, B., Ferreira, R., Geraldes, P., Gomes, M., Guilherme, J., Henriques, I., Lecoq, M., Leitão, D., Marques, A.T., Morgado, R., Pedroso, R., Prego, I., Reino, L., Rocha, P., Tomé, R., Zina, H., Osborne, P.E., 2012a. Population trends in the steppe birds of Castro Verde in the period 2006–2011: consequences of a drought event and land use changes? Airo 22, 79–89.

Moreira, F., Silva, J.P., Estanque, B., Palmeirim, J.M., Lecoq, M., Pinto, M., Leitão, D., Alonso, I., Pedroso, R., Santos, E., Catry, T., Silva, P., Henriques, I., Delgado, A., 2012b. Mosaic-level inference of the impact of land cover changes in agricultural landscapes on biodiversity: a case-study with a threatened grassland bird. PLoS One 7, e38876. http://dx.doi.org/10.1371/journal.pone.0038876.

Oñate, J.J.J., Atance, I., Bardají, I., Llusia, D., Bardaj'i, I., Llusia, D., 2007. Modelling the effects of alternative CAP policies for the Spanish high-nature value cereal-steppe farming systems. Agric. Syst. 94, 247–260. http://dx.doi.org/10.1016/j.agsy.2006.09.003.

Oppermann, R., Paracchini, M.L., 2012. HNV farming - central to European cultural landscapes and biodiversity. In: Oppermann, R., Beaufoy, G., Jones, G. (Eds.), High Nature Value Farming in Europe. Verlag Regionalkultur, pp. 16–22.

Pannell, D., 2000. Are we risking too much? Perspectives on risk in farm modelling. Agric. Econ. 23, 69–78. http://dx.doi.org/10.1016/S0169-5150(00)00058-X.

Pannell, D.J., Roberts, A.M., Park, G., Alexander, J., 2013. Improving environmental decisions: a transaction-costs story. Ecol. Econ. 88, 244–252. http://dx.doi.org/10.1016/j.ecolecon.2012.11.025.

Pe'er, G., Dicks, L.V., Visconti, P., Arlettaz, R., Baldi, A., Benton, T.G., Collins, S., Dieterich, M., Gregory, R.D., Hartig, F., Henle, K., Hobson, P.R., Kleijn, D., Neumann, R.K., Robijns, T., Schmidt, J., Shwartz, A., Sutherland, W.J., Turbe, A., Wulf, F., Scott, A.V., 2014. EU agricultural reform fails on biodiversity. Science 344 (80), 1090–1092. http://dx.doi.org/10.1126/science.1253425.

Poux, X., 2013. Biodiversity and Agricultural Systems in Europe: Drivers and Issues for the CAP Reform. Institut du Développement Durable et des Relations Internationales.

R Development Core Team, 2015. R: a language and environment for statistical computing. WWW Document. R Found. Stat. Comput. http://www.r-project.org, Accessed date: 26 December 2011.

Reino, L.L., Porto, M., Morgado, R., Moreira, F., Fabião, A., Santana, J., Delgado, A., Gordinho, L.L., Cal, J., Beja, P., 2010. Effects of changed grazing regimes and habitat fragmentation on Mediterranean grassland birds. Agric. Ecosyst. Environ. 138, 27–34. http://dx.doi.org/10.1016/j.agee.2010.03.013.

Ribeiro, P.F., Santos, J.L., Bugalho, M.N., Santana, J., Reino, L., Beja, P., Moreira, F., 2014. Modelling farming system dynamics in High Nature Value Farmland under policy change. Agric. Ecosyst. Environ. 183, 138–144. http://dx.doi.org/10.1016/j.agee.2013.11.002.

Ribeiro, P.F., Santos, J.L., Santana, J., Reino, L., Beja, P., 2016a. An applied farming systems approach to infer conservation-relevant agricultural practices for agri-environment policy design. Land Use Policy 58, 165–172. http://dx.doi.org/10.1016/j.landusepol.2016.07.018.

Ribeiro, P.F., Santos, J.L., Santana, J., Reino, L., Leitão, P.J., Beja, P., Moreira, F., 2016b. Landscape makers and landscape takers: links between farming systems and landscape patterns along an intensification gradient. Landsc. Ecol. 31, 791–803. http://dx.doi.org/10.1007/s10980-015-0287-0.

Ruto, E., Garrod, G., 2009. Investigating farmers' preferences for the design of agri-environment schemes: a choice experiment approach. J. Environ. Plan. Manag. 52, 631–647. http://dx.doi.org/10.1080/09640560902958172.

Santana, J., Reino, L., Stoate, C., Borralho, R., Carvalho, C.R., Schindler, S., Moreira, F., Bugalho, M.N., Ribeiro, P.F., Santos, J.L., Vaz, A., Morgado, R., Porto, M., Beja, P., 2014. Mixed effects of long-term conservation investment in Natura 2000 farmland. Conserv. Lett. 7, 467–477. http://dx.doi.org/10.1111/conl.12077.

Santana, J., Reino, L., Stoate, C., Moreira, F., Ribeiro, P.F., Santos, J.L., Rotenberry, J.T., Beja, P., 2017. Combined effects of landscape composition and heterogeneity on farmland avian diversity. Ecol. Evol. 7, 1212–1223. http://dx.doi.org/10.1002/ece3.2693.

Santos, J.L., 1998. The Economic Valuation of Landscape Change: Theory and Policies for Land Use and Conservation. Edward Elgar, Cheltenham, UK.

Santos, J.L., Carvalho, C.R., Beja, P., Gordinho, L., Reino, L., Pereira, A.J., Porto, M., Ribeiro, P.F., 2008. Medidas de Gestão Agrícola e Florestal para as Áreas Classificadas da Rede Natura 2000 Incluídas na 2a Fase de ITI/PDR - Relatório Final, Instituto. Instituto da Conservação da Natureza e da Biodiversidade, Lisbon.

Siebert, R., Toogood, M., Knierim, A., 2006. Factors affecting European farmers' participation in biodiversity policies. Sociol. Rural. 46, 318–340. http://dx.doi.org/10.1111/j.1467-9523.2006.00420.x.

Vanderwal, A.J., Falconi, L., Januchowski, S., Shoo, L., Vanderwal, M.J., 2015. Package “SDMTools”.

Villanueva, A.J., Rodríguez-Entrena, M., Gómez-Limón, J.A., Gómez-Limón, M., 2014. Agri-environmental schemes in olive growing: farmers' preferences towards collective participation and ecological focus areas. EAAE 2014 Congr. Agri-Food Rural Innov. Heal. Soc. Ljubljana (August 2014 5).

Wätzold, F., Drechsler, M., Johst, K., Mewes, M., Sturm, A., 2016. A novel, spatio-temporally explicit ecological-economic modeling procedure for the design of cost-effective agri-environment schemes to conserve biodiversity. Am. J. Agric. Econ. 98 (2), 489–512. http://dx.doi.org/10.1093/ajae/aav058. (à paraitre).