UNIVERSIDADE DE LISBOA
Instituto Superior de Agronomia

MODELLING THE EFFECTS OF AGRICULTURAL POLICIES ON HIGH NATURE VALUE FARMLAND: A FARMING SYSTEMS APPROACH

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Resumo alargado

Esta tese aborda o tema da sustentabilidade das áreas agrícolas de elevado valor natural, essenciais à conservação de grande parte da biodiversidade europeia, analisando o efeito das políticas agrícolas nesta matéria, através de uma abordagem por sistemas de produção agrícola.

A conservação da biodiversidade é actualmente um objectivo assumido ao mais alto nível no seio da política da União Europeia. Parte significativa desta biodiversidade depende de habitats proporcionados por sistemas de agricultura tradicionais, pouco intensivos, fruto de uma longa adaptação que acompanhou a humanização da paisagem europeia. Nos últimos anos, estes sistemas agrícolas de elevado valor natural têm registado um forte declínio, em consequência de processos de transformação da agricultura europeia ocorridos sobretudo a partir de meados do Século XX, dominados pela hegemonia do modelo produtivista, colocando em causa o objectivo da conservação da biodiversidade europeia.

As medidas agro-ambientais da Política Agrícola Comum (PAC) constituem um dos principais instrumentos de política instituídos com o objectivo de assegurar a conservação destes sistemas de agricultura e da biodiversidade que deles depende. São normalmente medidas de âmbito local, implementadas numa base contratual voluntária com os agricultores, que recebem um pagamento por assumirem compromissos de gestão desenhados para fazer face a questões de conservação específicas. Muitas destas medidas agro-ambientais, no entanto, não têm revelado a eficácia esperada, por razões que incluem a insuficiente adesão dos agricultores, o desajustamento dos compromissos de gestão face às questões de conservação, ou a sua incapacidade para contrariar os efeitos de outras alterações políticas ou económicas cujos impactos se sobrepuseram aos seus objectivos. Esta constatação tem merecido a atenção das autoridades e dos investigadores, que nos últimos anos se têm dedicado à procura de alternativas que permitam reforçar a eficácia das políticas agro-ambientais europeias, num quadro de crescente restrição orçamental. Esta tese pretende ir ao encontro destas questões, procurando contribuir para o conhecimento dos efeitos das políticas agrícolas em áreas de elevado valor natural, por forma a promover a sua sustentabilidade e, consequentemente, a conservação da biodiversidade agrícola associada.

O trabalho empírico aqui desenvolvido incidiu sobre uma zona agrícola de elevado valor natural do Sul de Portugal que inclui a área de actuação do programa agro-ambiental de Castro Verde, actualmente classificada como Zona de Protecção Especial (ZPE) para as aves, no âmbito da Directiva Aves da rede Natura 2000. Trata-se de uma zona de grande importância para a conservação de diversas espécies ameaçadas de avifauna estepária, que se fixaram nesta região devido à abundância de um habitat de “estepe-cerealífera” que durante décadas dominou a paisagem agrícola da região, proporcionado por um sistema de agricultura tradicional baseado numa rotação de cereais de sequeiro com pousios longos, pastoreados por rebanhos de ovinos com baixa carga animal. Nos últimos anos, este sistema de agricultura tradicional entrou em claro declínio, sendo substituído por sistemas agricultura especializados em produção animal, essencialmente vocacionados para o pastoreio de bovinos ou ovinos, pondo em causa a qualidade do habitat estepário e, consequentemente, a conservação da biodiversidade associada. Para estas transformações terão contribuído as alterações recentes da PAC, com destaque para o desligamento das ajudas directas às culturas arvenses, que incentivavam a cultura dos cereais, e para a continuidade da ajuda ligada às vacas aleitantes e aos pequenos ruminantes (ovinos e caprinos), ocorridas na reforma da PAC de 2003.

Nesta tese procurámos desenvolver uma metodologia baseada em sistemas de produção agrícola para identificar e descrever as transformações agrícolas ocorridas na área de estudo e os factores que determinaram essas transformações, com destaque para o papel das políticas agrícolas. Recorrendo a dados do Sistema Integrado de Gestão e Controlo (SIGC) e do Sistema de Identificação de Parcelas (SIP) provenientes da agência nacional de pagamentos PAC (IFAP – Instituto de Financiamento da Agricultura e Pescas, I.P.), referentes aos anos 2000 a 2010, complementados pelos resultados de um inquérito às explorações agrícolas da área de estudo, estabelecemos uma tipologia de sistemas de produção agrícola com vista a: 1) analisar as principais transformações agrícolas ocorridas na área de estudo ao longo deste período, relacionando-as com as mudanças de políticas verificadas (com destaque para a reforma da PAC de 2003); 2) avaliar o potencial desta abordagem para identificar sistemas de produção agrícola de elevado valor natural, com base nas características de paisagem e em práticas agrícolas associadas que os diferenciam, e; 3) estimar um modelo que permita prever a decisão do agricultor na escolha do sistema de produção agrícola, com base em características biofísicas e estruturais das explorações, e em variáveis económicas e políticas, para avaliar cenários de mudanças de políticas e/ou mercados, com vista a estimar uma curva de oferta de serviços de conservação da biodiversidade.

O trabalho empírico da tese foi organizado em quatro etapas principais, tendo cada uma delas correspondido a um artigo científico, apresentados nos capítulos principais da tese. A primeira etapa (Capítulo 2) foi dedicada à investigação das transformações agrícolas ocorridas na área de estudo ao longo de um período de tempo que envolveu a reforma da PAC de 2003. Com base em dados SIGC/SIP da área de estudo entre 2000 e 2010, foram identificados os principais sistemas de produção agrícola presentes na região ao longo deste período e comparou-se a sua distribuição espacial antes e depois da reforma da PAC de 2003 para detectar as principais alterações ocorridas. Estas foram posteriormente cruzadas com dados de caracterização biofísica, estrutural e política das explorações para avaliar o papel destas variáveis nas mudanças de sistemas de produção verificadas. Os resultados mostraram que as mudanças de sistemas de produção foram efectivamente influenciadas por estas variáveis, onde sobressaiu o facto de a medida agro-ambiental de Castro Verde não se ter revelado significativa neste processo, mostrando como o efeito de alterações nas políticas horizontais (i.e. não locais), como foi a introdução do regime de pagamento único e as inerentes alterações nas ajudas directas às culturas arvenses e às vacas aleitantes e pequenos ruminantes, se sobrepôs aos incentivos oferecidos pelas políticas locais de adesão voluntária.

Na etapa seguinte (Capítulo 3) avaliaram-se os impactos que estas mudanças de sistema de produção tiveram na paisagem agrícola e as suas consequências no valor de conservação da área de estudo. Os sistemas de produção foram caracterizados segundo métricas de paisagem e posteriormente comparados para identificar até que ponto eles podem ser tomados como preditores de padrões de paisagem associados a diferentes níveis de valor natural. Concluiu-se que essa separação não é completa, tendo-se verificado que sistemas de produção distintos podem apresentar padrões de paisagem semelhantes. Foi também possível perceber que alguns sistemas de produção apresentam uma grande variabilidade de padrões de paisagem (i.e., as explorações classificadas num mesmo sistema de produção exibem grande diversidade de padrões de paisagem), enquanto noutros essa variabilidade de paisagens entre explorações é significativamente menor, e que essa variabilidade diminui com o nível de intensidade agrícola do sistema de produção. Esta constatação levou a propor uma terminologia para classificar os sistemas de produção agrícola segundo um gradiente variando entre sistemas “landscape takers”, de baixa intensidade e com elevada variabilidade de padrões de paisagem entre explorações, e sistemas “landscape makers”, mais intensivos e com menor variabilidade dos padrões de paisagem entre explorações. Estes conceitos revestem-se de potencial utilidade no desenho e avaliação de políticas agro-ambientais, uma vez que é de esperar que explorações em sistemas tipo “landscape takers” consigam mais facilmente alterar o seu padrão de uso do solo para cumprir os requisitos de programas agro-ambientais sem que isso implique uma mudança de sistema de produção; por outro lado nas explorações em sistemas tipo “landscape makers” essa capacidade de ajustamento é inferior, por serem menos flexíveis em relação ao padrão de uso do solo, dificultando a sua adesão a programas agro-ambientais sem que isso implique uma mudança de sistema de produção.

A terceira etapa (Capítulo 4) envolveu uma comparação de sistemas de produção em termos de práticas agrícolas de reconhecida relevância para a conservação, desenvolvida com base em inquéritos às explorações. Mostrou-se que os sistemas de produção apresentam diferenças significativas relativamente a práticas agrícolas com impacto na biodiversidade, que os diferenciam em termos de valor de conservação. Mostrou-se também que sistemas de produção com padrões de paisagem relativamente próximos podem encerrar diferenças significativas em temos de práticas agrícolas, de que resultam evidentes diferenças no seu valor de conservação, chamando a atenção para as possíveis vulnerabilidades dos programas agro-ambientais baseados apenas em regras de uso do solo.

Na quarta e última etapa (Capítulo 5) foi estimado um modelo de decisão para simular a escolha do sistema de produção agrícola, com base em variáveis descritivas das explorações (biofísicas, estruturais e políticas locais) e incluindo o efeito dos incentivos económicos veiculados pelas políticas e pelos mercados. Este modelo foi usado para fazer predições sobre o impacto de cenários de políticas e mercados na escolha do sistema de produção mais favorável aos objectivos de conservação da área de estudo, permitindo derivar uma “curva de oferta de serviços de conservação da biodiversidade” que relaciona níveis de pagamento (agro-ambiental) com a extensão dos serviços de conservação prestados, expressa na proporção da área de estudo gerida por sistemas de produção de elevado valor natural, na qual se incluiu uma componente de avaliação de risco baseada em experiências de Monte Carlo para estimar intervalos de confiança a 95%.

A tese termina com uma discussão das principais conclusões (Capítulo 6), onde se avalia em que medida a abordagem adoptada permitiu responder às questões de investigação, se exploram algumas implicações dos resultados alcançados para o desenho de futuras políticas agro-ambientais e se defende a ideia de um sistema de pagamentos agro-ambientais atribuídos às explorações que pratiquem sistemas de produção de elevado valor natural em áreas com interesse para a conservação, automaticamente identificadas com base nos dados do SIGC, cuja implementação poderia caber no âmbito do Pilar 1 da PAC, configurando-se como um “*greening* de segunda geração”, na linha do precedente recentemente aberto pela reforma da PAC de 2013.

Palavras-chave: Sistema de produção agrícola; Política Agrícola Comum; Medidas agro-ambientais; Áreas agrícolas de elevado valor natural; Conservação da biodiversidade

Abstract

In recent years, High Nature Value (HNV) farmland became a priority for biodiversity conservation in Europe. Covering about 1/3 of total agricultural area in Europe, HNV farmland is in decline mainly because of agriculture abandonment/intensification driven by markets and public policies. HNV farmland is mostly provided by low-intensity and traditional farming systems, largely on less productive areas. Given the multiple environmental public goods provided by HNV farming systems, agri-environment schemes were set under the Common Agricultural Policy (CAP) to maintain HNV farmland, although with mixed results. Recent policy changes, including the decoupling of direct payments during the 2003 CAP reform, led to changes in farming systems which often further undermined HNV farmland sustainability. This research aims at evaluating the resilience of HNV farmland under policy change based on a farming systems approach, seeking to contribute to improving the design of agri-environment policies.

Focusing on HNV farmland in southern Portugal, we used farm level data from the Integrated Administration and Control System and the Land Parcel Identification System provided by the national CAP paying agency for the years 2000 to 2010, complemented with farm survey data, to derive a farming systems typology which was used to 1) assess agricultural changes over this time period, related with changes in the policy framework (e.g. 2003 CAP reform); 2) assess the extent to which this approach enables the identification of HNV farming systems, based on landscape features and farming practices, and; 3) model the choice of farming system based on farms' biophysical and structural features and on economic and policy variables, to assess scenarios of market and policy change and derive a supply curve for biodiversity conservation services.

The farming systems approach proved to be a relatively simple way to identify HNV farming systems and a promising tool to improve the cost-effectiveness EU agri-environment policy, suggesting the feasibility and benefits of a CAP direct payment scheme aimed at HNV farming systems.

Keywords: Farming systems; High Nature Value farmland; Common Agricultural Policy; Agri-environment schemes; Biodiversity conservation

Resumo

As áreas agrícolas de elevado valor natural (AAEVN) têm vindo a assumir uma importância crescente para a conservação da biodiversidade Europeia. Abrangendo cerca de 1/3 da superfície agrícola europeia, estas AAEVN têm estado em declínio devido às pressões de intensificação ou abandono agrícola geradas pelos mercados e políticas públicas. As AAEVN estão associadas a sistemas de produção agrícola tradicionais, de baixa intensidade, normalmente encontrados em áreas de menor aptidão agrícola. Reconhecendo a importância dos serviços dos ecossistemas proporcionados por estes sistemas de agricultura de EVN, foram introduzidos programas agro-ambientais no âmbito da Política Agrícola Comum (PAC) para promover a sua sustentabilidade, cujos resultados se têm revelado aquém das expectativas. Alterações políticas recentes, como o desligamento das ajudas directas introduzido pela reforma da PAC de 2003, levaram a mudanças nos sistemas de produção agrícola que agravaram a sustentabilidade destas AAEVN. Nesta tese procurámos avaliar a resiliência destas AAEVN face a alterações de políticas, através de uma abordagem por sistemas de produção agrícola, visando contribuir para a melhoria do desenho das políticas agro-ambientais europeias. Com base num caso de estudo de uma área agrícola de elevado valor natural localizada no Sul de Portugal, recorremos a dados do Sistema Integrado de Gestão e Controlo e do Sistema de Identificação de Parcelas para os anos 2000 a 2010, provenientes da agência nacional de pagamentos PAC, complementados com dados de um inquérito às explorações, para estabelecer uma tipologia de sistemas de produção agrícola que foi usada para 1) avaliar as transformações agrícolas que ocorreram na área de estudo ao longo deste período relacionadas com alterações de políticas (e.g. reforma da PAC de 2003); 2) avaliar o potencial desta abordagem para identificar sistemas de produção agrícola de EVN, com base em padrões de usos do solo e de práticas agrícolas, e; 3) modelar a escolha do sistema de produção com base em características biofísicas e estruturais das explorações e em variáveis políticas e económicas, para avaliar cenários de alterações de políticas e mercados. A abordagem por sistemas de produção revelou-se uma forma relativamente simples e eficaz de identificar sistemas de EVN e uma ferramenta promissora para ajudar a melhorar o custo-eficácia das políticas agro-ambientais europeias.

Palavras-chave: Sistema de produção agrícola; Política Agrícola Comum; Medidas agro-ambientais; Áreas agrícolas de elevado valor natural; Conservação da biodiversidade

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Chapter 1

General introduction

General introduction

This thesis deals with the effects of agricultural policy on high nature value farmland. The research presented here was mostly carried out in the scope of a research project supported by the Portuguese Foundation for Science and Technology (FCT - Fundação para a Ciência e Tecnologia), named «*Effects of Agri-Environment Schemes on Biodiversity: Evaluation of Long-Term Landscape Experiment in Southern Portugal*» (Project AGRIENV – PTDC/AGR-AAM/102300/2008). The project aimed at studying the effects of agri-environment programs for biodiversity conservation, focusing on the agri-environment scheme (AES) of the region of Castro Verde ("Plano Zonal de Castro Verde") as a case study. This is the most important area for steppe-bird conservation in Portugal, but significant changes in farm management in recent years have questioned the maintenance of its conservation value. The search for solutions to safeguard this farmland of high conservation concern, make up the bulk of the research carried out in this thesis. In view of the relevant environmental issues of the case study supporting the empirical work, agricultural services in biodiversity conservation will be used here as a particular subject within the wider issue of agricultural ecosystem services.

In the next paragraphs of this General introduction we start by setting the scene, going through the theoretical background and the conceptual approach of the research, where we first address the issues of conciliating agriculture and biodiversity conservation, leading us to the underlying market failure in the provision of ecosystem services from agriculture and to the need for policies. We then address the recent changes occurred in the Common Agricultural Policy (CAP) of the European Union (EU), outlining how it has sought to cope with that market failure. Following, we present the concepts of High Nature Value farmland and Farming System, which are key concepts within the research, and provide a brief review of the recent modelling approaches to assess how changes in farming systems driven by policy change impact on biodiversity values. Next, we present the research questions underlying the thesis and the general objectives to be achieved, which guided the design of the work plan outlined for the research, concluding with a description of the organization of the thesis.

Theoretical background and conceptual approach

Agriculture and biodiversity: a challenging conciliation

Today's European countryside is the result of centuries of agricultural activity that transformed the natural space in a rich and diverse mosaic of man-made landscapes and habitats, which include agricultural areas, forests, pastures, woodlands and others. More than 70% of the European Union (EU) land is currently managed by agriculture or forestry (Eurostat, 2010), of which about 32% is renowned for its conservation value (Paracchini et al., 2008). The corresponding figure for Portugal increases to about 58% (Paracchini et al., 2008). This transformation forced many species of wildlife to adapt and even lose link to their original natural habitats, becoming reliant on anthropogenic ecosystems, mostly associated with agriculture (Kleijn et al., 2006; Stoate et al., 2009, 2001).

Whereas in vast regions of the world agriculture is still perceived as a threat to biodiversity conservation by promoting, for example, clear cutting of tropical forests for expansion of large-scale agriculture (Barbier, 2004), in Europe and other developed regions much of the biodiversity of interest to conservation depends on the maintenance of certain farming systems, including extensive and traditional systems (Batáry et al., 2015; Bignal and McCracken, 1996; Kleijn et al., 2009; Lomba et al., 2015; Stoate et al., 2009). Along the last decades these systems have shown a declining trend in Europe, due to pressures for agriculture intensification or abandonment driven by social, economic and policy changes (Stoate et al. 2009; Latacz-Lohmann & Hodge 2003; Batáry et al. 2015; Lomba et al. 2015), risking to jeopardize the efforts for conserving farmland biodiversity (Donald et al. 2002; Latacz-Lohmann & Hodge 2003; Reidsma et al. 2006), which is a major target for EU current policy, as claimed by the EU Biodiversity Strategy to 2020 (European Commission, 2011).

These trends, which are apparently negative from a social welfare perspective, are partly explained by the fact that many of the environmental services provided by agriculture are public goods (Beaufoy and Marsden, 2011; BirdLife International, 2011), in the sense that the benefits are directly captured by society in general, without the mediation of a market, making it impossible to compensate their providers. The non-exclusive and non-rival in consumption nature of these public goods prevents the formation of a market for these services, and this failure-to-pay often results in its sub-optimal supply (Cooper et al., 2009). Such market failure justifies and call for state intervention, acting through institutional mechanisms and regulation (e.g. agri-environment payments or cross-compliance) or methods for incorporating value (indirect markets) such as certification schemes of origin or quality, which have been implemented under the Common Agricultural Policy (CAP) (Buckwell, 2009; Santos et al., 2016).

Recent trends in EU agri-environment policy

Recognizing the need for public intervention to tackle market failure in the provision of public goods from agriculture, agri-environment policies were introduced within the CAP during the reform of 1992, when the productivist model that prevailed since its inception in 1962 was questioned due to agricultural surpluses and environmental and budgetary problems. The CAP was then divided into two “pillars”, one directed to support farmers' income (Pillar 1) and the other directed to rural development support (Pillar 2), which included agri-environment schemes “*(…) designed to encourage farmers to protect and enhance the environment on their farmland by paying them for the provision of environmental services [and to] play a crucial role for meeting society's demand for environmental outcomes provided by agriculture.*” (http://ec.europa.eu/agriculture/envir/measures). Since then, the application of agri-environment programmes has been compulsory for Member States in the framework of their rural development plans, whereas remaining optional for farmers. Agri-environment schemes (AES) thereby became the main policy measure to address farmland conservation and environmental issues, through voluntary agreements with farmers, compensating them for loss of income (or the increase in production costs) associated with the maintenance or adoption of certain agricultural practices with environmental objectives (European Commission, 2005). By being included in the Pillar 2 of the CAP, these measures are co-financed by the Member States (MS), which is a policy feature likely to discourage its broader application across MS with budgetary constraints.

The CAP agri-environment policy went largely unchanged during the 2003 reform, although a major change has occurred related to the decoupling of Pillar 1 direct payments and its replacement by the single payment scheme. In Portugal, suckler cows, goat and sheep premia were kept coupled, a decision that was criticized due to expected negative effects in farmland areas of conservation concern (e.g. Brady et al., 2009; Oñate et al., 2007; Reino et al., 2010a; Renwick et al., 2008).

While positive AES outcomes have been acknowledged (e.g. Primdahl et al. 2003; Boatman et al. 2008), a number of studies has questioned their cost-effectiveness, mainly due to high transaction costs undermining farmers’ voluntary participation, high administrative expenditure, poor environmental performance, and failure in safeguarding high conservation-value farming systems (e.g. Kleijn et al. 2001; Kleijn & Sutherland 2003; Siebert, Toogood & Knierim 2006; Defrancesco et al. 2008; Weber 2013; Pe'er et al. 2014; Batáry et al. 2015).

Under an ongoing debate about the need to improve the cost-effectiveness of the EU agri-environment policy (Mettepenningen et al., 2011; Weber, 2013), the next CAP reform in 2013 also left the substance of the agro-environment policy in Pillar 2 mostly untouched, but brought the novelty of the "Greening" of Pillar 1 direct payments, whereby EU farmers receiving area-based payments were required to comply with non-contractual practices for the benefit of the environment and the climate. These included diversifying crops, maintaining permanent grassland and dedicating 5% of arable land to "ecological focus areas" (European Commission, 2013a, 2013b). With this amendment, for the first time in CAP history, measures literally targeted on safeguarding and improving farmland biodiversity were implemented under the Pillar 1, in a far more explicit way than that in former cross-compliance regulations (http://ec.europa.eu/agriculture/envir/cross-compliance). As we shall see at the end of the thesis, this may well represent the dawn of promising future changes in EU agri-environment policy. Such broad-brush agri-environment policies (in the sense that they apply throughout the EU) operating under the CAP Pillar 1 provide the advantage of eliminating, or significantly reducing, transaction costs, in addition to being more attractive to MS because they are typically fully funded by the CAP budget, without the need for co-financing. Nevertheless, their effectiveness in the preservation of farmland biodiversity and agroecosystems was called into question due to poorly specified conservation objectives and low effectiveness of mandatory commitments (Pe'er et al., 2014). So, the search for alternative cost-effective policy measures remained under discussion.

With this purpose, some authors have defended the need to increase the targeting of Pillar 2 AES, finely tailoring them to meet specific conservation objectives identified at the local level (Armsworth et al. 2012). Others proposed the adoption of results-based schemes, rather than management-based schemes (Keenleyside et al., 2014b). Others still advocated the need for agri-environment policies directly supporting farming systems of acknowledged conservation value (e.g. Beaufoy and Marsden, 2011; BirdLife International, 2011; Keenleyside et al., 2014a; Poláková et al., 2011; Weber, 2015), as they seem to be associated with specific agricultural practices and land-use patterns to which biodiversity responds (Bamière et al., 2011; Calvo-Iglesias et al., 2009; Carmona and Nahuelhual, 2010). Such alternatives could prove best suited for deal with necessary trade-offs between scheme precision and administrative costs (Weber, 2015), providing more targeted (and specially more locally fit) management prescriptions than those achieved by broad-brush policies and reducing the costs required by local schemes (Poláková et al., 2011). This thesis follows these authors, seeking to assess how a farming systems approach could be used to support the design of agri-environment policies, providing they are confirmed as good predictors of landscape patterns and conservation-relevant farming practices.

The High Nature Value concept and the farming systems approach

Since the 90's, it was noticed that certain types of farming systems were not only less harmful to the environment, but also positively related to biodiversity, so that some may even be crucial for the survival of species with high conservation value (Andersen et al., 2004). These systems were initially referred to as low-intensity farming systems, which later gave rise to the concept of High Nature Value (HNV) farming proposed by Baldock et al. (1993): "*High Nature Value (HNV) farming systems are predominantly low-intensity systems which often involve a relatively complex interrelationship with the natural environment. They maintain important habitats both on the cultivated or grazed area (for example, cereals steppes and semi-natural grasslands) and in features such as hedgerows, ponds and trees, which historically were integrated with the farming systems. (…) The semi-natural habitats currently maintained by HNV farming are particularly important for nature conservation in the EC because of the almost total disappearance of large scale natural habitats*.". Much of this HNV farmland are now strongly associated with areas of traditional agricultural landscape or marginalized rural areas with low agricultural potential, since in areas with higher agricultural potential the intensification led to their disappearance (Paracchini et al. 2007).

The first work carried out to identify HNV farmland at the European-wide scale was that of Andersen et al. (2004), who used a triple approach based on land-use, biodiversity and farming system indicators. In spite of early implementation problems related to data availability, it was soon recognised that the farming system approach had the strength of providing insights into the particular farming practices delivering HNV farmland. This meant that the farming system approach could be used to monitor short-term changes in HNV farming systems and thus in evaluating pressures for further changes in HNV farmland (Paracchini and Britz, 2010).

Several studies have used farming systems to evaluate the relationships between agriculture and biodiversity conservation, for purposes of identifying HNV farmland, assessing agri-environment policy or analysing landscape changes (e.g. Bamière et al., 2011; Beaufoy et al., 2012; Calvo-Iglesias et al., 2009; Darnhofer et al., 2012; Oñate et al., 2007; Oppermann et al., 2012; Pointereau et al., 2010). The majority of these works relates the concept of farming system with a classificatory analysis built from economic data, such as the Farm Accountancy Data Network (FADN) (Andersen et al., 2004; Reidsma et al., 2006) or data from the Farm Structure survey (FSS) (Pointereau et al., 2007), or based on land use / land cover data, such as CORINE Land Cover (Paracchini et al., 2008). However, the concept of farming system in the context of agricultural economics and rural sociology, is broader than that, referring to an understanding of the complex network of relationships established between land-use, animal rearing, farming practices, crop yields, inputs used, social identities, economic and policy environment or others.

Building on previous works mostly by French authors, such as Malassis or the Count Gasparin, Reboul (1976) defines the concept of farming system as "*Un système de production agricole est un mode de combinaison entre terre, forces et moyens de travail à des fins de production végétale et/ou animale, commun à un ensemble d'exploitations. Un système de production agricole est caractérisé ici par la nature des productions, de la force de travail (qualification) et des moyens de travail mis en œuvre, et par leurs proportions.*" (Reboul 1976, p. 57). Dixon et al. (2001) defined farming system as "*(…) a population of individual farm systems that have broadly similar resource bases, enterprise patterns, household livelihoods and constraints, and for which similar development strategies and interventions would be appropriate.*" (Dixon et al. 2001, p. 2). Ferraton and Touzard (2009) highlight that the analysis of a farming system is to study not only each of the subsystems that compose it, but especially the interactions between them, and also to understand resource allocation choices between the various activities of the farm.

Therefore, describing a farming system involves knowing in detail the production factors (inputs) used and the way they combine and interact to obtain a given yield or combination of various outcomes. Different farming systems use different combinations of inputs to produce distinct combinations of agricultural products. The farming system is thus a consequence of decisions taken by the farmer in a framework of social, economic, cultural, policy, biophysical, among others, drivers and constraints. In these terms, the concept of farming system is potentially impractical as a tool to develop agro-economic studies involving a large number of farms, since it requires the availability of detailed farm-level data that can usually only be obtained through direct enquiries to individual farmers, which is often unfeasible. Seeking to overcame this difficulty, we proposed to investigate if a simplified approach based on a farming system typology built from incomplete (coarse) data on land use and livestock, such as that available in CAP paying agencies, could be used to infer about landscape and livestock patterns, farm management practices and, ultimately, to identify HNV farmland. We refer, in particular, to data from the IACS (Integrated Administration and Control System, for the management of CAP payments) and the associated LPIS (Land Parcel Identification System), which are annually updated at the EU-wide level and feature the significant advantage of including a spatial component at the farm and even parcel levels, provided by the LPIS component, which can be potentially useful for landscape assessment, enabling habitat quality evaluation or the computation of environmental indicators. This is a substantial advantage as compared to other EU data sources, such as the FADN, which uses a farm-type classification system that identifies groups of farm businesses showing similar combinations of economic outputs, which is not appropriate to identify HNV farmland because it does not reflect the land cover characteristics that are critical to HNV, in addition to being sampled data, lacking the comprehensive nature of the IACS/LPIS data.

Assessing the effects of agricultural policies on HNV farmland

Seeking solutions to offset the market failure that has led to the decline of HNV farming systems, many authors have drawn attention to the need to increase knowledge about predictive models that permit scenario-based approaches to explore the links between policy, land-use and biodiversity (Mattison and Norris, 2005; Paracchini and Britz, 2010).

Most of the papers published in this area have used mathematical programming models, econometric models or agent-based models. Concerning the diversity of approaches, Balkhausen et al. (2005) conducted a review on several studies using different predictive models to evaluate the effect of decoupling within the 2003 CAP review, concluding that the main differences in results stem mainly on assumptions, rather than the type of model.

One of the predictive models that have been extensively used by many authors to evaluate the effects of CAP changes on European agriculture and HNV farmland was the CAPRI model (*Common Agricultural Policy Regionalised Impact*) (http://www.capri-model.org). This is a mathematical programming model that uses the GAMS software and which is structured in modules that allow to study the effects of policies in different EU countries, integrating data of agricultural product markets, including the effect of the world market. Paracchini and Britz (2010) used this model to simulate the effect of agri-environment policies in HNV farmland, by integrating the use of an indicator that expresses the probability associated with farming systems to support biodiversity, on a European scale. The CAPRI model was used to produce results on crop distribution, stock density, yields and fertilizer consumption, with an output resolution of 1x1km. Pointereau et al. (2010) also mention the use of the CAPRI model in the work of Paracchini and Britz (2010), highlighting its application as an *ex-ante* evaluation tool to assess the impact of different policy scenarios on HNV farmland.

Mathematical programming models have been widely used in agro-economic modelling, often including environmental or biodiversity conservation objectives. For example, Oñate et al. (2007) resorted to mathematical programming models to compare three alternative policies aimed at supporting the income of representative farms in a cereal steppe region of Spain.

These models typically start from a baseline farm land-use set for the reference year and, through adjustments of yields and inputs, optimize an objective function that maximizes the gross margin of the farms. Fragoso et al. (2008) made use of mathematical programming as a tool for calibrating agricultural supply models with predictive purposes. Bamière et al. (2011) used mathematical programming models with a spatial component, to study the effects of different agri-environment schemes in a Natura 2000 area in France, assessing the outputs at three different spatial levels: the plot, the farm and the landscape level. Armsworth et al. (2012) also resorted to mathematical programming to design a model that takes into account the "biodiversity production" on farms, while also considering the production of conventional agricultural outputs, to estimate the marginal private cost associated with conservation objectives, which they called the "true supply price of biodiversity". Based on these data, they estimated trade-off curves relating improvements in biodiversity with loss of farm income.

Valbuena et al. (2010) used the Net Logo 4.1 software to develop an agent-based model to assess the effect of farmers' decisions in landscape structure in a Dutch region. According to the authors, these models represent a technique that allows to simulate the individual decision-making of farmers, relating it to the space component represented by the farm, allowing to describe, simulate and analyse interactions between farmers and between them and the environment.

Regarding econometric models, we highlight the works presented by Lewis and Plantinga (2007) or Wätzold et al. (2015), for the proximity with the objectives and approaches undertaken in this thesis. In these works, the authors seek to develop methodologies that allow a spatial analysis of the effects of policies on landscape quality, combining econometric models with GIS landscape modelling tools. Our work differs from the earlier as it relies on a farming systems approach based on real-farm observed data, instead of using land use / land cover data, as it was done in those previous works, to develop a model for assessing policy and market scenarios.

Also in the context of econometric models with a spatial component, it should be noted the work of Lesschen et al. (2007), which is a reference in the field. This is a methodological document addressing the main aspects related to the construction of this type of models, from the type of data (economic and spatial data), data mining, regression models, problems with spatially explicit data (e.g. multicollinearity or spatial autocorrelation) and other issues relevant to the spatial analysis of land uses and farming systems. According to Lesschen et al. (2007), spatially explicit analysis enables to consider the effects of farming system or technology changes that are not necessarily related to land use, such as changes in choices of livestock breed or crop variety.

Inspired by these arguments, a spatial econometrics (Anselin, 1999) approach was admitted in the early stages of the thesis, to explore the possibility of spatial autocorrelation in the data. However, the approach came to prove inadequate because, on the one hand, there was a suspicion that the choice of the farming system, despite being conditioned by biophysical and structural features of the farm, which may be spatially explicit, is mainly an economic decision (as it came to be confirmed), which is not dependent on the spatial location (not within our regional scale of analysis, at least). Moreover, the fact that the spatial representation of our dependent variable (the farming system) often corresponded to "multi-polygon" entities (since farms are frequently composed of more than one block of land, sometimes located significantly apart from each other, and where all blocks are mandatorily classified with the same farming system and subject to synchronous changes, whenever they occur), which prevents the practical application of the auto-correlation tests suitable for discrete entities (such as Moran's I, available in R package "spdep" (Bivand et al., 2016)), because these are based on a neighbourhood "joint count" test, drawn from a triangular irregular network linking polygon centroids, which is not appropriate in the case of multi-polygon entities because the centroids often fall outside the boundaries of the farm.

Accordingly, taking into account the limitations and potentialities of the IACS/LPIS data and the conceptual approach to be developed, we opted for a modelling methodology focused on the use of discrete choice econometric models (logistic regression models) to simulate the effect of changes in policy or economic variables (prices) on the choice of farming system, where the spatial component would be used as a tool to assess impacts on HNV farmland, by computing environmental indicators. As the data included repeated observations on the same farms over the years, a panel data structure was anticipated to test for possible individual heterogeneity (Falconer et al., 2001; Greene, 2012; Hynes and Garvey, 2009). Also anticipating the occurrence of heterogeneity in the data due to unobserved variables, such as those controlling farmers' socio-cultural profile, which is a fact known to influence motivations and attitudes towards policies (Burton et al., 2008; de Snoo et al., 2013; Siebert et al., 2006), a latent class model approach was tested to enable dealing with heterogeneity in farmers' preferences regarding the choice of the farming system. (Boxall and Adamowicz, 2002; Greene, 2012). Latent class models have been extensively used in recent stated-preference studies using discrete choice models, including within the context of agri-environment policy evaluation (e.g. Colombo et al., 2009; Garrod et al., 2012; Ruto and Garrod, 2011; Villanueva et al., 2014). Such a model was intended to be applied in the identification of the factors governing the choice of the farming system and to evaluate policy and market scenarios, hoping to fill a gap on the knowledge about the economics of HNV farming systems and to contribute to offset the market failure behind the decline of HNV farmland.

Research question and general objectives of the thesis

With this background, the thesis started from a simple question asking "what are the effects of agricultural policies on biodiversity conservation?", which developed into a more elaborate formulation, asking: "*Does a typology of farming systems built on IACS/LPIS data allows us to identify farms operating HNV farming systems which could therefore be selected to receive an agri-environmental premium payment? What would be the level and type of funding necessary for its sustainability?*". To address these questions, the following objectives were established:

1 – To investigate the effects of policy changes on HNV farming system dynamics, and the role of biophysical (soils, topography, climate...) and structural (farm size, farm fragmentation…) constraints in this dynamics;

2 – To evaluate the relationships between farming systems and landscape patterns, focusing on those landscape features that are relevant for conservation;

3 – To assess to what extent the adoption of farming practices that are relevant for biodiversity conservation are intrinsic and inherent to specific HNV farming systems;

4 – To build a methodology that integrates all these results and model the landscape effects of policy changes, by explaining farmers' decisions regarding the choice of farming system under different policies and economic incentives. To the extent possible, such methodology should have the ability to be reproducible in other locations with comparable conservation problems, using similar data sources.

Organization of the thesis

In addition to this General introduction (Chapter 1) and the final General discussion and conclusion (Chapter 6), this thesis comprises four main chapters (Chapters 2 to 5) presenting the empirical work of the thesis and relating to the four general objectives specified in the preceding section, which generally match the four steps that were outlined in the initial work plan of the research (Figure 1. 1) and which led to four scientific papers that were produced within the research work. The following paragraphs provide a brief description of each of these steps.

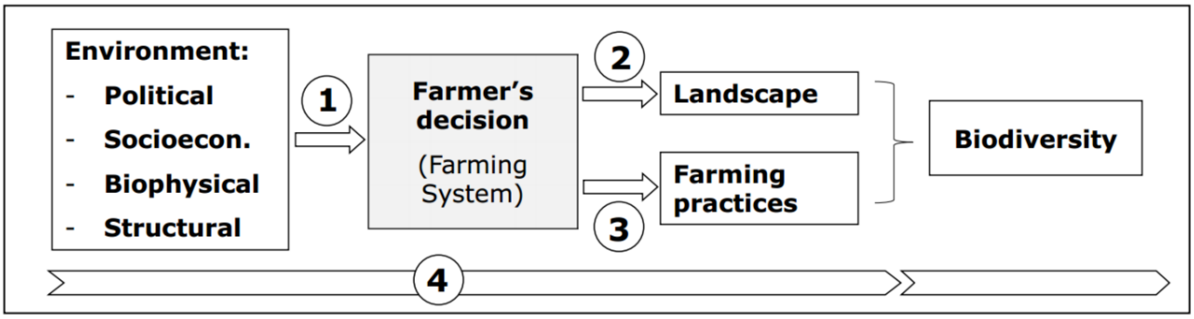

Figure 1. 1 - The work plan outlined for the research, showing the four main steps involved: 1) Drivers and constraints of farming system dynamics; 2) Effects of farming systems dynamics on landscape; 3) Effects of farming systems on farming practices relevant to the conservation of biodiversity; 4) Modelling the effects of policy change on conservation-relevant landscape indicators and farming practices.

Step 1 – Drivers and constraints of farming system dynamics: this task aimed at explaining the effects of policy change on HNV farming system dynamics, under the limitations on production options imposed by biophysical, structural and policy constraints. The analysis relied on geo-referenced data on farmers' annual declarations of crop areas and livestock units in the context of their applications for CAP payments, between 2000 and 2010, gathered from the Portuguese national paying agency (IFAP). Multivariate techniques were used to identify a typology of farming systems that were GIS-mapped for two periods – before and after the 2003 CAP reform – to assess the effects of this major policy change on farming-system transitions. Adding geographic information on biophysical, structural and policy variables, logistic models were used to assess the role of these variables in explaining observed farming-system transitions. Results allowed a full discussion on the resilience of local agri-environment schemes within the context of major horizontal policy changes. The outcome of this step resulted in the publication of the paper "*Modelling farming system dynamics in high nature value farmland under policy change*", published in *Agriculture, Ecosystems and Environment* in 2014, which is presented in Chapter 2.

Step 2 – Effects of farming systems dynamics on landscape: in this step we sought to determine the extent to which different farming systems translate into different landscapes, with the background concern of understanding whether farming system change can change landscapes to a point where these changes can undermine their habitat functions. Landscape modelling software was used to characterize landscape patterns associated to specific farming system and differences between farming system groups were investigated with principal components and linear discriminant analysis. Implications for biodiversity conservation were explored from the results to evaluate and compare the conservation value of the farming systems. This research resulted in the scientific paper presented in Chapter 3 "*Landscape* *makers and landscape takers: links between farming systems and landscape patterns along an intensification gradient*", published in *Landscape Ecology* in 2016.

Step 3 – Effects of farming systems on farming practices relevant to the conservation of biodiversity: this task was intended to evaluate whether farming practices that are considered most relevant for biodiversity can be predicted based on the specific farming system to which a farm belongs. AES contracts often include an extensive list of detailed specifications related to farming practices, whose complexity ("*red tape*") reduces uptake and/or compliance by farmers, while impairing supervision and control by the official authorities. So, the objective was to investigate the extent to which better results could be achieved with simpler programs focusing directly on farming systems, which implicitly would assure compliance with the relevant farming practices. This would have the advantage of making the programs easier to understand and accept by farmers and, at the same time, easier to monitor by control agencies. To meet these goals, 200 field surveys were conducted with farmers within the study area, to check specific technical aspects of agricultural activity associated with each farming system. The results were presented in Chapter 4, corresponding to the scientific paper "*An applied farming systems approach to infer conservation-relevant agricultural practices for agri-environment policy design*", published in 2016 in *Land Use Policy*.

Step 4 – Modelling the effects of policy change on conservation-relevant landscape indicators and farming practices: this last step integrated results from the previous steps to develop an econometric model to simulate the effects of policy change on biodiversity conservation in HNV farmland. Discrete choice logistic regression models were used to estimate the probability of a farmer to adopt a given (HNV) farming system, depending on a diverse set of explanatory variables, such as policies effects, economic indicators, farmer's profile, farm structural and biophysical features. Latent class and panel data techniques were used to cope with possible heterogeneity in farmers' preferences, resulting from unobserved variables. Model outcomes were used to simulate the effect of policy changes on farming systems within a given region, presenting results in a GIS environment to allow the assessment of the resulting landscape structure and evaluate its interest for conservation. The model was additionally used to relate the level of landscape quality, as a supplier of habitat, with increases in the level of AES payments, leading to the estimation of a supply curve for conservation services. This research was brought together in paper "*A spatially explicit choice model to assess the impact of conservation policy on High Nature Value farming systems*", presented in Chapter 5, to be submitted to *Ecological Economics* in 2016.

References

Andersen, E., Baldock, D., Bennett, H., Beaufoy, G., Bignal, E., Brouwer, F., Elbersen, B., Eiden, G., Godeschalk, F., Jones, G., McCracken, D., Nieuwenhuizen, W., Eupen, M. van, Hennekens, S., Zervas, G., 2004. Developing a High Nature Value Farming area indicator 75.

Anselin, L., 1999. SPATIAL ECONOMETRICS (techreport). Richardson, TX.

Armsworth, P.R., Acs, S., Dallimer, M., Gaston, K.J., Hanley, N., Wilson, P., 2012. The cost of policy simplification in conservation incentive programs. Ecol. Lett. 15, 406–414. doi:10.1111/j.1461-0248.2012.01747.x

Baldock, D., Beaufoy, G., Bennett, G., Clark, J., 1993. Nature conservation and new directions in the EC Common Agricultural Policy: the potential role of EC policies in maintaining farming and management systems of high nature value in the Community. Institute for European Environmental Policy, London.

Balkhausen, O., Banse, M., Grethe, H., Nolte, S., 2005. Modelling the Effects of Partial Decoupling On Crop and Fodder Area and Ruminant Supply in the EU: Current State and Outlook, in: 89th EAAE Seminar, in Parma, Italy. pp. 1–16.

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. doi:10.1016/j.ecolecon.2010.12.014

Barbier, E.B., 2004. Explaining Agricultural Land Expansion and Deforestation in Developing Countries. Am. J. Agric. Econ. 86, 1347–1353.

Batáry, P., Dicks, L. V., Kleijn, D., Sutherland, W.J., 2015. The role of agri-environment schemes in conservation and environmental management. Conserv. Biol. 29, 1006–1016. doi:10.1111/cobi.12536

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.

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

Bignal, E.M., McCracken, D.I., 1996. Low-intensity systems in the conservation of the countryside. J. Appl. Ecol. 33, 413–424. doi:10.2307/2404804

BirdLife International, 2011. High Nature Value Farming - How Diversity in Europe's Farm Systems Delivers for Biodiversity.

Bivand, R., Altman, M., Anselin, L., Assunção, R., Berke, O., Bernat, A., Blanchet, G., Blankmeyer, E., Carvalho, M., Christensen, B., Chun, Y., Dormann, C., Dray, S., Gómez-Rubio, V., Gubri, M., Halbersma, R., Krainski, E., Legendre, P., Lewin-Koh, N., Li, H., Ma, J., Millo, G., Mueller, W., Ono, H., Peres-Neto, P., Piras, G., Reder, M., Tiefelsdorf, M., Yu, D., 2016. Package "spdep" - Spatial dependence: weighting schemes, statistics and models.

Boatman, N., Ramwell, C., Parry, H., Bishop, J., Gaskell, P., Mills, J., Dwyer, J., 2008. A review of environmental benefits supplied by agri-environment schemes. Land Use Policy Group.

Boxall, P.C., Adamowicz, W., 2002. Understanding Heterogenous Preferences in Random Utility Models: A Latent Class Approach. Environ. Resour. Econ. 23, 421–446. doi:10.1023/A:1021351721619

Brady, M., Kellermann, K., Sahrbacher, C., Jelinek, L., 2009. Impacts of Decoupled Agricultural Support on Farm Structure, Biodiversity and Landscape Mosaic: Some EU Results. J. Agric. Econ. 60, 563–585. doi:10.1111/j.1477-9552.2009.00216.x

Buckwell, A., 2009. Public Goods from Private land. Rural Investment Support for Europe.

Burton, R.J.F., Kuczera, C., Schwarz, G., 2008. Exploring Farmers' Cultural Resistance to Voluntary Agri-environmental Schemes. Sociol. Ruralis 48, 16–37. doi:10.1111/j.1467-9523.2008.00452.x

Calvo-Iglesias, M.S., Fra-Paleo, U., Diaz-Varela, R.A., 2009. Changes in farming system and population as drivers of land cover and landscape dynamics: The case of enclosed and semi-openfield systems in Northern Galicia (Spain). Landsc. Urban Plan. 90, 168–177. doi:10.1016/j.landurbplan.2008.10.025

Carmona, A., Nahuelhual, L., 2010. Linking farming systems to landscape change: an empirical and spatially explicit study in southern Chile. Agric. Ecosyst. … 139, 40–50. doi:10.1016/j.agee.2010.06.015

Colombo, S., Hanley, N., Louviere, J., 2009. Modeling preference heterogeneity in stated choice data: an analysis for public goods generated by agriculture. Agric. Econ. 40, 307–322. doi:10.1111/j.1574-0862.2009.00377.x

Cooper, T., Hart, K., Baldock, D., 2009. Provision of Public Goods through Agriculture in the European Union.

Darnhofer, I., Gibbon, D., Dedieu, B., 2012. Farming Systems Research: an approach to inquiry, in: Darnhofer, I., Gibbon, D., Dedieu, B. (Eds.), Farming Systems Research into the 21st Century: The New Dynamic. Springer, pp. 1–26.

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. doi:10.1111/j.1755-263X.2012.00296.x

Defrancesco, E., Gatto, P., Runge, F., Trestini, S., 2008. Factors affecting farmers'participation in agri-environmental measures: A northern Italian perspective. J. Agric. Econ. 59, 114–131. doi:10.1111/j.1477-9552.2007.00134.x

Dixon, J., Gulliver, A., Gibbon, D., 2001. Farming Systems and Poverty - Improving Farmers' Livelihoods in a Changing World. FAO and World Bank.

Donald, P.F., Pisano, G., Rayment, M.D., Pain, D.J., 2002. The Common Agricultural Policy, EU enlargement and the conservation of Europe's farmland birds. Agric. Ecosyst. Environ. 89, 167–182.

European Commission, 2013a. Overview of CAP Reform 2014-2020, Agricultural Policy Perspectives Brief Nº 5.

European Commission, 2013b. Regulation (EU) Nº 1307/2013 of the European Parliament and of the Council of 17 December 2013 establishing rules for direct payments to farmers under support schemes within the framework of the common agricultural policy and repealing Council Regulation. European Parliament and of the Council.

European Commission, 2011. Communication from the Commission to the European Parliament, the Council, the Economic and Social Committee and the Committee of the Regions. Our life insurance, our natural capital: an EU biodiversity strategy to 2020. Brussels.

European Commission, 2005. Agri-environment Measures - Overview on General Principles, Types of Measures, and Application. Directorate General for Agriculture and Rural Development.

Eurostat, 2010. Results on EU land cover and use published for the first time.

Falconer, K., Dupraz, P., Whitby, M., 2001. An Investigation of Policy Administrative Costs Using Panel Data for the English Environmentally Sensitive Areas. J. Agric. Econ. 52, 83–103.

Ferraton, N., Touzard, I., 2009. Comprendre l'agriculture familiale : diagnostic des systèmes de production, Quae. ed. Les presse agronomique de Gembloux.

Fragoso, R.S., Carvalho, M.L., Henriques, P.S., 2008. A programação matemática positiva como instrumento de calibração e prescrição dos modelos de oferta. Economia 28, 1–16.

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. doi:10.1016/j.ecolecon.2012.02.011

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

Hynes, S., Garvey, E., 2009. Modelling Farmers' Participation in an Agri-environmental Scheme using Panel Data: An Application to the Rural Environment Protection Scheme in Ireland. J. Agric. Econ. 60, 546–562. doi:10.1111/j.1477-9552.2009.00210.x

Keenleyside, C., Beaufoy, G., Tucker, G., Jones, G., 2014a. High Nature Value farming throughout EU-27 and its financial support under the CAP. Report Prepared for DG Environment, Contract No ENV B.1/ETU/2012/0035, Institute for European Environmental Policy. London. doi: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. Prepared for the European Commission, DG Environment, Contract No ENV.B.2/ETU/2013/0046, Institute for European Environme.

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. Ecol. Lett. 9, 243–247. doi:10.1111/j.1461-0248.2005.00869.x

Kleijn, D., Berendse, F., Smit, R., Gilissen, N., 2001. Agri-environment schemes do not effectively protect biodiversity in Dutch agricultural landscapes. Nature 413, 723–725. doi:10.1038/35099540

Kleijn, D., Kohler, F., Báldi, A., Batáry, P., Concepción, E.D., Clough, Y., Díaz, M., Gabriel, D., Holzschuh, A., Knop, E., Kovács, A., Marshall, E.J.P., Tscharntke, T., Verhulst, J., 2009. On the relationship between farmland biodiversity and land-use intensity in Europe. Proc. Biol. Sci. 276, 903–909. doi:10.1098/rspb.2008.1509

Kleijn, D., Sutherland, W.J., 2003. How effective are European agri-environment schemes in conserving and promoting biodiversity ? J. Appl. Ecol. 947–969.

Latacz-Lohmann, U., Hodge, I., 2003. European agri-environmental policy for the 21st century. Aust. J. Agric. Resour. Econ. 47:1, 123–139.

Lesschen, J.P., Verburg, P.H., Staal, S.J., 2007. Statistical methods for analysing the spatial dimension of changes in land use and farming systems – LUCC Report Series 7. The International Livestock Research Institute, Nairobi, Kenya & LUCC Focus 3 Office, Wageningen University, the Netherlands.

Lewis, D.J., Plantinga, A.J., 2007. Policies for Habitat Fragmentation : Combining Econometrics with GIS-Based Landscape Simulations. Land Econ.

Lomba, A., Alves, P., Jongman, R.H.G., Mccracken, D.I., 2015. Reconciling nature conservation and traditional farming practices : a spatially explicit framework to assess the extent of High Nature Value farmlands in the European countryside. Ecol. Evol. 1–14. doi:10.1002/ece3.1415

Mattison, E.H. a, Norris, K., 2005. Bridging the gaps between agricultural policy, land-use and biodiversity. Trends Ecol. Evol. 20, 610–616. doi:10.1016/j.tree.2005.08.011

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. doi:10.1016/j.ecolecon.2010.10.007

Oñate, J.J.J., Atance, I., Bardají, 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. doi:10.1016/j.agsy.2006.09.003

Oppermann, R., Beaufoy, G., Jones, G., Oppermann, R., Beaufoy, G., Jones, G., 2012. High Nature Value Farming in Europe. verlag regionalkultur.

Paracchini, M.L., Britz, W., 2010. Quantifying effects of changed farm practices on biodiversity in policy impact assessment – an application of CAPRI-Spat. OECD, pp. 1–16.

Paracchini, M.L., Petersen, J., Hoogeveen, Y., Bamps, C., Burfield, I., Swaay, C. Van, 2008. High Nature Value Farmland in Europe - An Estimate of the Distribution Patterns on the Basis of Land Cover and Biodiversity Data, EUR 23480. ed, Institute for Environment and Sustainability Office for Official Publications of the European Communities Luxembourg. Office for Official Publications of the European Communities. doi:10.2788/8891

Paracchini, M.L., Terres, J.-M., Petersen, J., Hoogeveen, Y., 2007. High Nature Value Farmland and Traditional Agricultural Landscapes – Open Oportunities in the Development of Rural Areas, in: Pedroli, B., Van Doorn, A., De Blust, G., Paracchini, M., Wascher, D., Bunce, F. (Eds.), . LANDSCAPEEUROPE / KNNV, pp. 21–34.

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 (80-.). 344, 1090–1092. doi:10.1126/science.1253425

Pointereau, P., Doxa, A., Coulon, F., Jiguet, F., Paracchini, M.L., 2010. Analysis of spatial and temporal variations of High Nature Value farmland and links with changes in bird populations: a study on France, JRC Scientific and Technical Reports. Joint Research Centre of the European Commission.

Pointereau, P., Paracchini, M.L., Terres, J., 2007. Identification of High Nature Value farmland in France through statistical information and farm practice surveys, Methodology.

Poláková, J., Tucker, G., Hart, K., Dwyer, J., Rayment, M., 2011. Addressing biodiversity and habitat preservation through measures applied under the Common Agricultural Policy, Report Prepared for DG Agriculture and Rural Development, Contract No. 30-CE-0388497/00-44. Institute for European Environmental Policy.

Primdahl, J., Peco, B., Schramek, J., Andersen, E., Oñate, J.J.J., 2003. Environmental effects of agri-environmental schemes in Western Europe. J. Environ. Manage. 67, 129–138. doi:10.1016/S0301-4797(02)00192-5

Reboul, C., 1976Reino. Mode de production et systèmes de culture et d'élevage. Économie Rural. 112, 55–65. doi:10.3406/ecoru.1976.2413

Reidsma, P., Tekelenburg, T., van den Berg, M., Alkemade, R., 2006. Impacts of land-use change on biodiversity: An assessment of agricultural biodiversity in the European Union. Agric. Ecosyst. Environ. 114, 86–102. doi:10.1016/j.agee.2005.11.026

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

Renwick, A., Revoredo-Giha, C., Barnes, A., Jansson, T., Schwarz, G., 2008. Assessment of the impact of partial decoupling on prices, production and farm revenues within the EU. Final Report for DEFRA, Department. ed, Health (San Francisco). SAC.

Ruto, E., Garrod, G., 2011. Investigating farmers' preferences for the design of agri-environment schemes: a choice experiment approach.

Santos, J.L., Madureira, L., Ferreira, A.C., Espinosa, M., Gomez-y-Paloma, S., 2016. Building an empirically-based framework to value multiple public goods of agriculture at broad supranational scales. Land use policy 53, 56–70.

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

Stoate, C., Báldi, 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. J. Environ. Manage. 91, 22–46. doi:10.1016/j.jenvman.2009.07.005

Stoate, C., Boatman, N.D., Borralho, R.J., Carvalho, C.R., de Snoo, G.R., Eden, P., 2001. Ecological impacts of arable intensification in Europe. J. Environ. Manage. 63, 337–365. doi:10.1006/jema.2001.0473

Valbuena, D., Verburg, P.H., Veldkamp, a., Bregt, A.K., Ligtenberg, A., 2010. Effects of farmers' decisions on the landscape structure of a Dutch rural region: An agent-based approach. Landsc. Urban Plan. 97, 98–110. doi:10.1016/j.landurbplan.2010.05.001

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, Agust 2014 5.

Wätzold, F., Drechsler, M., Johst, K., Mewes, M., Sturm, A., 2015. A Novel, Spatiotemporally Explicit Ecological-economic Modeling Procedure for the Design of Cost-effective Agri-environment Schemes to Conserve Biodiversity. Am. J. Agric. Econ. doi:10.1093/ajae/aav058

Weber, A., 2015. Implementing EU co-financed agri-environmental schemes: Effects on administrative transaction costs in a regional grassland extensification scheme. Land use policy 42, 183–193. doi:10.1016/j.landusepol.2014.07.019

Weber, A., 2013. How are public transaction costs in regional agri-environmental scheme delivery influenced by EU regulations? J. Environ. Plan. Manag. 1–23. doi:10.1080/09640568.2013.776950

Chapter 2

Modelling farming system dynamics in High Nature Value Farmland under policy change

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Abstract

Understanding the factors driving changes in farm management is needed for designing policies and subsidy schemes to protect High Nature Value Farmland (HNVF). We describe farming system dynamics in HNVF of southern Portugal, between 2000–2002 and 2008–2010, encompassing a period of major policy transformations introduced by the reform of the Common Agricultural Policy (CAP) of the European Union in 2003. We also assess how farming system dynamics was modulated by structural, biophysical and policy factors constraining agricultural options. Farming systems changed in about 40% of the farmed area during the period of study. Overall, there was a marked transition from arable systems to either specialized livestock or permanent crop systems, involving major declines in the traditional system of dry cereal rotations and sheep grazing. Transitions were influenced by farm size, soil quality and coverage by open oak woodlands, while there was little effect of agri-environment schemes and legal regulations specifically targeted to support the traditional farming system. Despite these changes, agricultural intensity remained essentially stable, though there was a marked decline in land-use heterogeneity with likely negative impacts on biodiversity. Observed changes agree with *ex-ante* impact assessments of the CAP reform in Iberian cereal steppes, which suggested that decoupling of payments from production could promote shifts from the traditional cereal–fallow–sheep system towards specialized livestock grazing systems. Effectively protecting HNFV may thus require a better integration of horizontal policies and agri-environment schemes.

Keywords: Agri-environment schemes; Biodiversity conservation; CAP reform; Farming systems; High Nature Value Farmland

Introduction

In Europe, the concept of High Nature Value Farmland (HNVF) was developed to typify and help safeguarding agricultural systems with biodiversity value (Baldock et al., 1993; Hoogeveen et al., 2004), because many wild species of conservation concern are dependent on habitats created or maintained by low-intensity farming (Kleijn et al., 2009; Bugalho et al., 2011; Doxa et al., 2012). Despite its importance, HNFV has been declining due to rural depopulation, agricultural abandonment and afforestation in marginal farming areas, coupled with intensification in the most productive areas (Stoate et al., 2009). It is generally agreed that agri-environment schemes (AES) and other funding mechanisms implemented under the Common Agricultural Policy (CAP) of the European Union (EU) could contribute for ameliorating these trends. However, the design of effective policies is hindered by a limited understanding of how policies affect farmer decisions, and how these in turn shape farmland landscapes and their value for biodiversity (Baldock et al., 1993; Mattison and Norris, 2005; Beaufoy et al., 2012). Major modifications to EU agricultural policies were introduced by the CAP reform of 2003. The main innovation of this reform was the Single Farm Payment and the associated decoupling of payments from production, whereby farmers were no longer required to maintain production for receiving CAP payments, but had to keep land in good environmental and agricultural conditions (Renwick et al., 2008; Brady et al., 2009). *Ex-ante* conjectures of the consequences of this change for HNVF were contrasting, with some foreseeing positive outcomes because farmers would no longer be forced into intensive farming (BirdLifeInternational, 2003), whereas others anticipated negative effects because decoupling could promote abandonment of low-income farming areas of conservation value (Oñate et al., 2007; Tranter et al., 2007). These processes were thought to be conditional on other CAP mechanisms such as AES, which could support otherwise economically unsustainable farming (Brady et al., 2009). At present, however, there is little information on how agricultural management of HNVF varied during this period of major policy change, and how this variation was affected by AES, farm characteristics and biophysical constraints.

The farming system framework may provide a relatively simple and practical approach to evaluate agricultural changes in HNVF, because it concentrates on groups of farms with similar typology, thereby avoiding the need to detail the multiple idiosyncrasies of a large number of individual farms (Paracchini and Britz, 2010; Darnhofer and Gibbon, 2012). Farms included in the same farming system type have similar resource bases, enterprise patterns, livelihoods and household restrictions, and so they are expected to show similar responses to policy, market and biophysical drivers (Dixon et al., 2001; Ferraton and Touzard, 2009). Furthermore, information on potential biodiversity impacts can be gained by analysing changes in farming systems, because they are associated with specific agricultural practices and land-use patterns to which biodiversity components respond (Calvo-Iglesias et al., 2009; Carmona et al., 2010; Bamière et al., 2011).

This study used farming systems to examine agricultural changes on cereal-steppes of the Iberian Peninsula, during a period (2000–2010) encompassing the CAP reform of 2003. This HNVF type corresponds to extensively farmed, mixed rotational systems of winter cereals, fodder crops and grazed fallow land and pastures, covering over 4.5 million ha in dry areas with low forest cover (Suárez et al., 1997). Cereal-steppes are critical for the conservation of a range of open farmland birds of European conservation concern (Suárez et al., 1997; Bota et al., 2005). The specific objectives of the study were to: (i) quantitatively define a farming system typology based on spatially explicit farm-level data; (ii) estimating farming system dynamics in the period 2000–2010; (iii) modelling farming system dynamics in relation to structural, biophysical and policy constraints to agricultural management; and (iv) evaluating the consequences of farming system dynamics in terms of agricultural intensification and land-use heterogeneity, which are known to influence biodiversity patterns and trends (Benton et al., 2003; Donald et al., 2006). Results were then used to explore the consequences of farming system dynamics for biodiversity conservation in Iberian cereal-steppes, and to discuss potential applications of the farming system concept to improve agri-environment schemes and other agricultural policies.

Methods

The study was conducted in lowland agricultural landscapes of southern Portugal, within about 210,000 ha (Appendix 1). The climate is Mediterranean, with hot dry summers and cold and moderately rainy winters. Despite its relative homogeneity, the study area shows a north-south gradient of decreasing soil quality and reduced availability of irrigation water, which is reflected in the presence of more intensive crops in the north (e.g. irrigated annual crops and olive groves) and a more extensive land-use to the south, dominated for decades by the traditional cereal–fallow–sheep farming system (Bacharel and Pinto-Correia, 1999; Delgado and Moreira, 2000). The study area encompassed the Special Protection Area (SPA) of Castro Verde, designated under EU Directive 92/43/EEC. This is the most important area in Portugal for the conservation of open farmland birds, including globally threatened species such as lesser kestrel *Falco naumanni*, great bustard *Otis tarda*, and little bustard *Tetrax tetrax* (Pinto et al., 2005; Reino et al., 2010; Moreira et al., 2012). Since 1995, following the CAP reform of 1992, most of the SPA has benefited from an agri-environment scheme (AES) specifically targeted at the conservation of open farmland birds through the maintenance of the traditional farming system (Marta-Pedroso et al., 2007). Farms within most of the SPA are thus entitled to AES payments, subject to production commitments that have changed over the years but that generally included maintaining a traditional cereal–fallow rotation, keeping livestock grazing densities below specified thresholds, growing specified crops benefiting steppe birds and keeping watering spots for wildlife. At the same time, these farms were affected by some legal constraints associated with the SPA status, such as restrictions on the plantation of permanent crops or farmland afforestation.

Farm characterization

Farms (n = 2800) were characterized using variables reflecting the dominant agricultural land uses and the stocking rates (Table 2. 1). Dry cereals included mainly wheat and barley. Other annual crops were generally irrigated arable crops (e.g. sunflower, chickpea). Fallows included arable land that was not seeded for one or more seasons, and which was usually grazed by sheep or cattle. Pastures included all fodder crops and pastures (excluding grazed fallows), either permanent or temporary, natural or sown. Permanent crops were mostly olive groves. Livestock density was based on Livestock Units (LU), aggregating animals from different species (cattle and sheep) and ages using standard conversion factors (Appendix 2). Variables were extracted from a spatially explicit database maintained by the Portuguese Ministry of Agriculture, which is based on farmer declarations when applying for CAP payments, and is verified on a random basis by Ministry officers. A farm was assumed to correspond to all the parcels owned by a farmer, though some-times it did not represent a continuous block of land. Variables for each farm were obtained for each year and then averaged for each of two time periods, corresponding to the start (2000–2002) and end of the study (2008–2010), thereby eliminating short term variations in agricultural land uses due for instance to crop rotation. Farms were excluded from analysis if they were not represented in the data sets in at least one year in each of both study periods, and if most of its land was outside the study area in a given year.

Farms were also characterized in terms of structural, biophysical and policy variables reflecting significant constraints to management options (Table 2. 1). Soil quality was estimated from a digital map of soil capacity for agriculture (SROA/CNROA, 2012). Oak woodland cover was estimated by extracting the area occupied by open cork oak *Quercus suber* and holm oak *Q. rotundifolia* woodlands (locally called “montado”) from a digital land cover map (IGP, 2012), and it was used because cutting these oaks is strongly restricted by Portuguese law, constraining the range of farmer management possibilities. The location of the farm inside the SPA of Castro Verde was included because it is associated with legal restrictions to some land-use changes. The adherence of the farmer to AES specifically targeted at open farmland birds was used because it supports the traditional cereal–fallow–sheep system.

Data analysis

Farming system typology was determined by non-hierarchical clustering of the whole dataset, considering both time periods together and the seven agricultural variables (Table 2. 1), using the partition around medoids (PAM) clustering algorithm (Kaufman and Rousseeuw, 1990). In PAM, representative elements of each cluster (medoids) correspond to real observations, rather than centroids or averages. So, a medoid is a representative farm in a cluster, whose average dissimilarity to all the other farms in the same cluster is minimal. Silhouette plots were used to help assess the ideal number of categories to be considered (Rousseeuw, 1987). PAM was implemented with the ‘cluster’ package (Maechler et al., 2012) for R (R Development Core Team, 2011). Based on the results of PAM, each farm in each time period was assigned to a farming system, and the total area occupied by each farming system in each period was estimated and mapped. Overall dynamics in farming system cover between time periods was summarized in a transition matrix (Gergel and Turner, 2002).

A more detailed analysis of factors affecting transitions was then carried out by overlaying a 500-m grid on the study area, and registering the transition type observed at each point. To avoid pseudo-replication only one point was used per farm, and we only used farms for which information was available for both periods. Based on these criteria, we obtained a sample of 722 farms that were used to model farming system dynamics in relation to the five structural, biophysical and policy variables (Table 2. 1) using multinomial logistic regression (Hosmer and Lemeshow, 1989). A minimum sample of 10 observations was required to include a given transition type in the model. Modelling started with all explanatory variables and a backward variable selection was then carried out using likelihood ratio tests. In all models, farming system persistence (lack of change) was used as the reference category. A threshold of $p < 0.10$ was used to identify potentially important variables, as we were more interested in understanding their explanatory role than making predictions. Model fit was assessed by comparing with a null model as an overall significance test (likelihood ratio test), and by using

Nagelkerke's r^2 (Nagelkerke, 1991) and classification accuracy compared to that of a null model classifying all cases as the modal category. Models were run in SPSS for Windows 19.0.0.

Table 2. 1 - Summary statistics of variables used to characterize farms and to model farming system dynamics. UAA = Utilized Agricultural Area.

Variable	Description	Mean ± SD (Min-Max)
Agricultural variables (n = 2800 farms)		
Dry cereals	% of dry cereals in the UAA	29.1 ± 26.4 (0–100)
Fallow land	% of fallows in the UAA	9.8 ± 15.8 (0–100)
Pastures	% of forages and pastures in the UAA	41.5 ± 38.8 (0–100)
Other annual crops	% of other annual crops in the UAA	12.0 ± 19.5 (0–100)
Permanent crops	% of permanent crops in the UAA	7.6 ± 21.9 (0–100)
Livestock density	Livestock units (LU) per ha of fodder area	0.27 ± 0.44 (0–2)
Cattle ratio	% of cattle LU in total LU	14.1 ± 31.4 (0–100)
Structural, biophysical and policy variables (n = 722 farms)		
Farm size	Total UAA of the farm in the initial period. Ordinal coding: 1 (<25 ha), 2 (25–50 ha), 3 (50–150 ha), 4 (150–250 ha), 5 (>250 ha)	3.40 ± 1.34 (1–5)
Soil quality	% of soils with no or few restrictions for agricultural use in the UAA. Ordinal coding: 0 (0%), 1 (0–20%), 2 (20–40%), 3 (>40%)	1.13 ± 1.26 (0–3)
Oak woodland	% of the UAA covered with cork or holm oak woodlands. Ordinal coding: 0 (0%), 1 (0–25%), 2 (25–50%), 3 (>50%)	1.30 ± 1.14 (0–3)
Castro Verde SPA	>50% of the UAA inside or outside (1/0) the Castro Verde Special Protection Area (SPA)	0.35 ± 0.48 (0–1)
AES	Farms adhering or not (1/0) to agri-environment schemes specifically targeted at open farmland birds (i.e., within the AES area of Castro Verde and with more than 50% of the UAA joining agri-environment measures)	0.14 ± 0.35 (0–1)

Agricultural intensity and land-use heterogeneity were estimated for each farming system, based on the characteristics of the corresponding medoid farm. Intensity was estimated using an indicator of farm income per hectare (Turner and Doolittle, 1978), based on the Portuguese database of agricultural standard gross margins (Rosário, 2005). Subsidies were discounted to minimize the effects of changes in income support schemes introduced by the CAP reform of 2003. Heterogeneity was based on the diversity of agricultural land uses computed with Shannon's diversity index (Spellerberg and Fedor, 2003). Intensity and heterogeneity estimates were standardized by dividing their values by the corresponding maximum values. Intensity and heterogeneity dynamics were then estimated by computing a weighted mean of the corresponding indicators for each time period, with weights proportional to the surface occupied by each farming system.

Results

Overall, the UAA of the sampled farms was dominated by pastures (ca. 40%), followed by cereals (ca. 30%), other annual crops (ca. 12%) and fallows (ca. 10%) (Table 2. 1). The average livestock density, mostly sheep, was 0.27 LU/ha of forage area. Six farming systems resulted from the cluster analysis (Table 2. 2). Cluster 1 (Traditional; 17.2% of farms) typically comprised farms with a mosaic of dry cereals, pastures and fallow land, where livestock density was moderate and composed only by sheep. Cluster 2 (Annual Crops; 36.4% of farms) was the most common farming system, including farms with dry cereals and other annual crops occupying about 90% of the land, and with no pastures, no livestock and little fallow land. Cluster 3 (Cattle Grazing; 16.2% of farms) included farms characterized by a large dominance of pastures and little dry cereals, and with a high livestock density dominated by cattle. Cluster 4 (Sheep Grazing; 24.2% of farms) included farms with a crop pattern similar to Cattle Grazing system, but with lower livestock densities and composed strictly by sheep. Cluster 5 (Permanent Crops; 6.0% of farms) corresponded to farms entirely covered by permanent crops, most of which were very intensive olive grove plantations. Cluster 6 (Intensive Sheep; 6.0% of farms) corresponded to farms dedicated to sheep husbandry with livestock densities six times higher than those typical for the region (Table 2. 1 and Table 2. 2), probably relying on rented pastures or stubble crops in neighbouring farms to enable the maintenance of large herds. This farming system was not considered further, because it occupied a very small area (<1%) and it was an outlier in relation to the others prevailing in the study area.

Table 2. 2 - Characteristics of representative farms (medoids) identified for farming systems identified in the study area. n = sample size (farm/period combinations). Variable definition provided in Table 2. 1.

Variable	Cluster 1 *Traditional* (n = 454)	Cluster 2 *Annual Crops* (n = 962)	Cluster 3 *Cattle Grazing* (n = 427)	Cluster 4 *Sheep Grazing* (n = 638)	Cluster 5 *Permanent Crops* (n = 159)	Cluster 6 *Intensive* Sheep (n = 160)
Dry cereals (%)	28.3	57.6	14.2	9.6	0.0	0.0
Fallow land (%)	24.9	9.5	5.9	0.4	0.0	0.0
Pastures (%)	42.4	0.0	77.2	88.1	0.0	100.0
Other annual crops (%)	4.4	31.1	0.0	0.2	0.0	0.0
Permanent crops (%)	0.0	1.8	2.6	1.7	100.0	0.0
Livestock density	0.15	0.00	0.46	0.23	0.00	1.81
Cattle ratio (%)	0.0	0.0	85.1	0.0	0.0	0.0

Farming system dynamics

There were marked changes in cover by different farming systems between 2000–2002 and 2008–2010 (Figure 2. 1). In the former period, Traditional and Cattle Grazing were the dominant farming systems, whereas Cattle and Sheep Grazing were the dominant systems in the later period. This change was caused by marked increases in the area covered by Cattle (34%) and Sheep (50%) Grazing systems, which were matched by drastic reductions in the Traditional (65%) and Annual Crop (54%) systems (Figure 2. 1). Permanent crops were largely absent in the first period and showed the largest proportional increase.

Table 2. 3 - Transition matrix illustrating farming systems dynamics between 2000–2002 and 2008–2010. Each row shows the percentage of area of each farming system persisting (underlined) or changing to another farming system.

Farming systems	2008–2010				
	Traditional	Annual crops	Cattle grazing	Sheep grazing	Permanent crops
2000–2002					
Traditional	<u>15.8</u>	4.4	36.4	40.9	0.9
Annual crops	23.1	<u>36.1</u>	15.7	9.8	14.0
Cattle grazing	0.7	1.5	<u>85.3</u>	9.7	2.1
Sheep grazing	2.4	0.9	20.6	<u>73.5</u>	0.0
Permanent crops	0.0	0.0	0.0	0.0	<u>100.0</u>

There were transitions between nearly all farming systems, though the most important were those from Traditional to either Sheep (ca. 18,000 ha) or Cattle (ca. 16,000 ha) Grazing systems (Table 2. 3). Other relevant transitions were those from Annual Crops to Traditional (ca. 7000 ha), Cattle Grazing (ca. 4700 ha) or Permanent Crop (ca. 4200 ha) systems. Still relevant were the transitions from Sheep to Cattle Grazing (ca. 4000 ha), and the opposite transition (ca. 4600 ha). Cattle grazing (85%) and Permanent Crop (100%) systems were the most persistent during the study period.

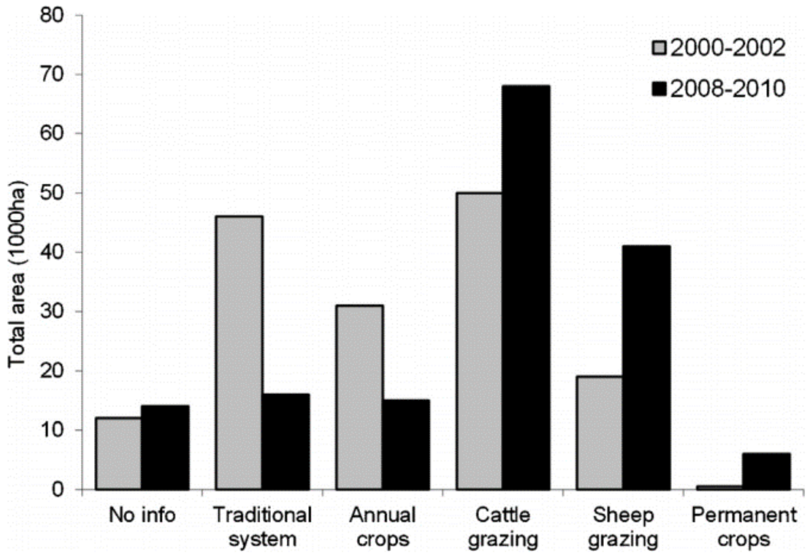

Figure 2. 1 - Total area (×10^3 ha) occupied by farming systems identified in the study area (2000–2002 and 2008–2010).

Drivers of farming system dynamics

Transitions between farming systems were significantly related to structural, biophysical and, to a much lesser degree, policy variables (Figure 2. 2 to Figure 2. 4; Appendix 3). For the Traditional system (Figure 2. 2), transition to the Annual Crop system was most likely in farms with more productive soils (p = 0.027) and larger percentage of oak woodlands (p = 0.022); transition to Cattle Grazing was most likely in larger farms (p = 0.021), and transition to Sheep Grazing was most likely in poor quality soils (p = 0.061). For the Annual Crop system (Figure 2. 3), transition to the Traditional system was most likely in areas with more oak woodlands (p = 0.015) and poorer soils (p = 0.044); transitions to Cattle or Sheep Grazing were most likely in farms with poorer soils (p < 0.001), although the latter was also most likely in farms with a lower percentage of oak woodland (p = 0.030). Transition from Cattle to Sheep Grazing was most likely in poorer soils (p = 0.020), inside the SPA (p = 0.057) and in smaller farms (p = 0.073) (Figure 2. 4). For Sheep grazing, there was enough data to model persistence and transition to Cattle grazing, but no significant effects were detected.

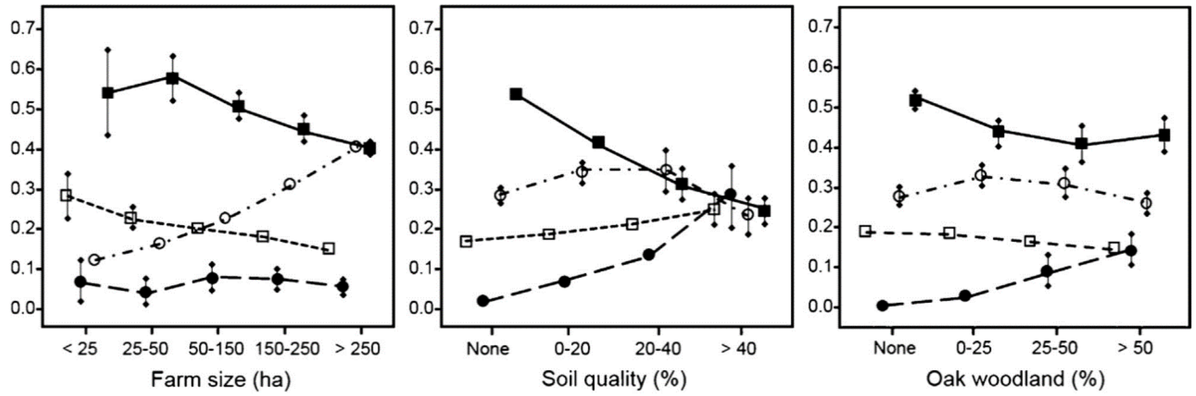

Figure 2. 2 - Estimated transition probabilities (and 95% confidence intervals) in relation to explanatory variables for the Traditional system: persistence (white squares), and transitions to Annual Crops (black dots), Cattle Grazing (white dots) and Sheep Grazing (black squares).

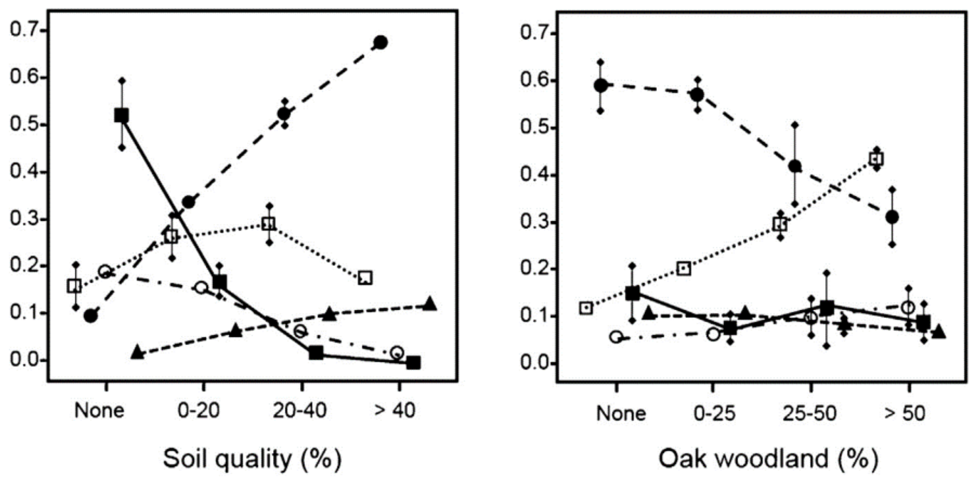

Figure 2. 3 - Estimated transition probabilities (and 95% confidence intervals) in relation to explanatory variables for the Annual Crop system: persistence (black dots), and transitions to Traditional (white squares), Cattle Grazing (white dots), Sheep grazing (black squares), and Permanent Crops (triangles).

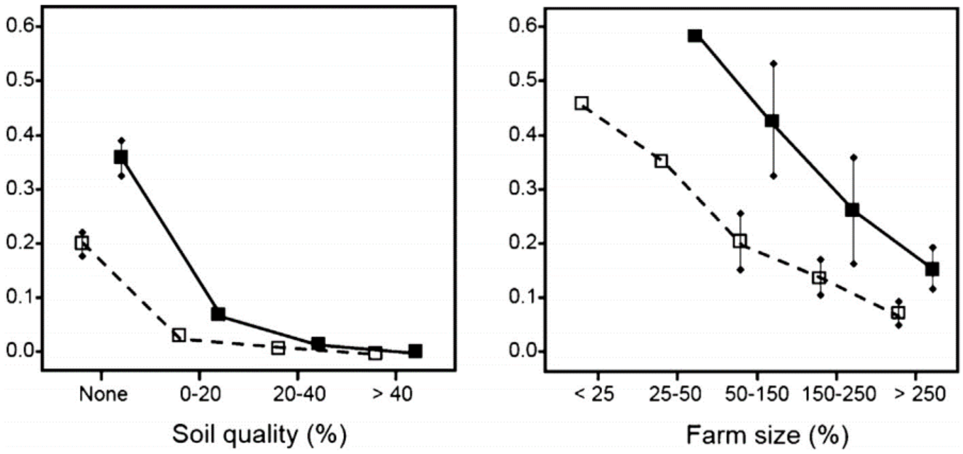

Figure 2. 4 - Estimated transition probabilities (and 95% confidence intervals) in relation to explanatory variables for the Cattle Grazing system: transitions to Sheep Grazing inside (black squares) and outside (white squares) the AES area.

Farming intensity and heterogeneity

Agricultural intensity was much higher for Permanent Crops than for the other farming systems, which all showed broadly similar intensity values (Figure 2. 5).

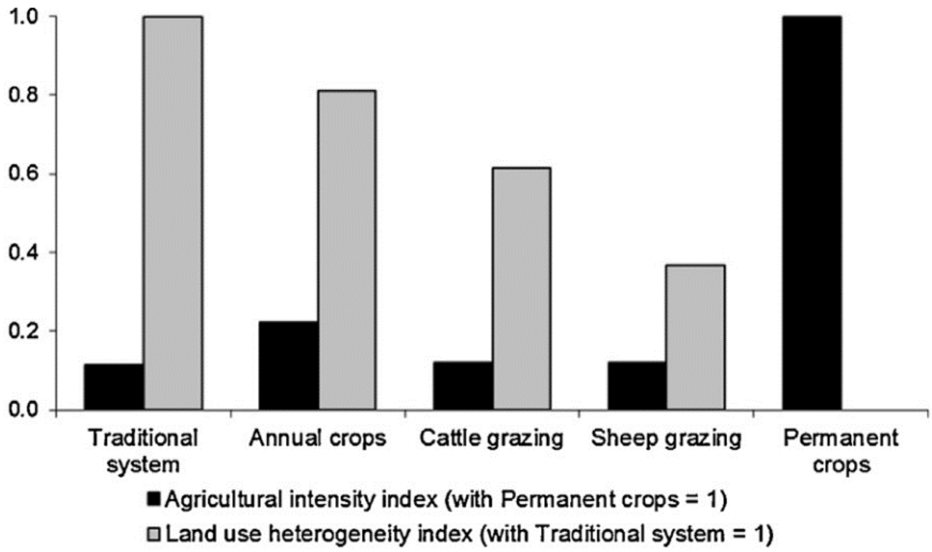

Figure 2. 5 - Indexes of agricultural intensity level and land-use heterogeneity across farming systems identified in the study area.

Land-use heterogeneity was highest for the Traditional and the Annual Crop systems and null in the Permanent Crop system (monoculture). Cattle and Sheep Grazing systems showed intermediate heterogeneity values. Average intensity was very similar in 2000–2002 (0.15 ± 0.05 SD) and 2008–2010 (0.18 ± 0.18 SD), while heterogeneity was much higher in the former (0.74 ± 0.22 SD) than in the later period (0.57 ± 0.23 SD).

Discussion

Our results showed that there was a strong farming system dynamics between 2000–2002 and 2008–2010, which affected about 40% of the farmed area. Overall, there were marked declines in the Traditional and Annual Crop Farming systems, which were matched by strong increases in the Cattle Grazing, Sheep Grazing and Permanent Crop systems. The changes observed were possibly influenced to some extent by the CAP reform of 2003, though confirmation of this hypothesis would require more in-depth socio-economic analysis.

One of the most notorious changes observed in our study was the replacement of cereal-based systems (Traditional and Annual Crops) by specialized livestock farming systems (Sheep and Cattle Grazing). One factor possibly contributing to these changes was the decoupling of payments from production introduced by the CAP reform. The Portuguese government decided to use the option provided by CAP to decouple crop payments, while keeping livestock payments partially or fully coupled for sheep and suckler cows, respectively, which may have created economic incentives for the maintenance of or conversion into specialized livestock systems. A similar effect was predicted in *ex-ante* impact assessments of the effects of the decoupling, which foresaw shifts from the traditional cereal–fallow–sheep system towards livestock grazing systems (e.g. Serrão and Coelho, 2005; Nagy, 2006; Tranter et al., 2007).

The large scale loss of the Traditional system is particularly noteworthy, considering that it was supported by a specific AES and by legal restrictions to land-use conversion within the SPA of Castro Verde. This result thus suggests that the AES was not able to sustain this system despite its value for biodiversity conservation, probably as a consequence of the too low per-hectare AES payments. In particular, the cut on the AES payments decided in the 2001 national revision of the AES programme, from about 80–56 euros/ha, and the expectations then set for the upcoming CAP reform, probably led many farmers not to revalidate their five-year AES contracts that were coming to an end by then. This is supported by the decrease in the number of farms participating in AES between the two time periods, from about 35 to 16%. As a consequence, many farmers probably decided to move to the livestock grazing system, which were likely seen as better economic alternatives compared to the Traditional system, despite the AES support.

Another noteworthy trend was the dramatic increase in the Permanent Crop system, which was nearly absent in 2000–2002. This system encompasses mainly farms that specialized into intensive olive plantations, most of them changing from the Annual Crop system. A possibility to explain this pattern is that the decoupling of subsidy payments left farm management free to follow market oriented strategies (Fragoso et al., 2011), and the high profitability of intensive olive groves likely made them a very attractive alternative for farmers.

Local drivers of farming system dynamics

The overall trends in farming system dynamics uncovered by our study masked considerable variability among farms, which could be explained to a large extent by farm level differences in structural and biophysical features. In fact, factors such as farm size, soil quality and oak woodland cover were critical drivers of farming system transition of individual farms, which at the same time appeared to be little affected by policy factors such as land-use restrictions imposed by the Special Protection Area of Castro Verde, and the incentives made available through AES.

One of the key determinants of farm transitions was soil quality, probably because it influenced production options and thus the choice of the farming system (e.g. Lesschen et al., 2007). Overall, soil quality strongly influenced the dynamics for three farming systems, with poor quality soils favouring transitions to the Traditional and the Livestock Grazing systems (particularly sheep), whereas good quality soils favoured transitions to Annual Crops, especially when coming from the Traditional system.

Farm size showed particularly strong effects in transitions to livestock systems, with larger farms being more likely to turn to Cattle Grazing and smaller ones to Sheep Grazing. This is consistent with livestock farming systems typical to this region, where sheep led by a shepherd graze during the day and is gathered into barns at night, whereas cattle commonly graze in fenced parks where it also remains during the night. This entails important differences in both systems in terms of needs for labour per hectare, with much smaller needs in cattle systems, thus favouring them for larger farms.

Cover by oak woodlands was another important factor, probably because legal restrictions to tree cutting precluded major land-use changes. However, transition patterns were difficult to interpret, as higher percentages of oak woodlands were related to changes from the Traditional to the Annual Crop system, as well as the inverse transition. In the latter case, it is possible that constraints to land-use change due to the presence of oak woodlands could have led landowners to disinvest towards the more extensive Traditional system. Alternatively, absent landowners confronted with structural difficulties imposed by the presence of oak woodlands or unable to further invest in more demanding productions, preferred to reduce activity to the minimum level of compliance with maintenance of the land in good agricultural and environmental conditions in order to receive the single payment (Costa et al., 2011).

Surprisingly, the availability of the Castro Verde AES did not appear a relevant driver of farming system transitions. Influence of a policy driver was only detected in the case of the Special Protection Area, which was found to influence farm transitions from Cattle Grazing to Sheep Grazing systems. Reasons for this effect are unknown, but they may reflect farmers' decisions to move to a less demanding system in an area dominated by poor soils, where other options such as farmland afforestation and conversion to permanent crops were legally restricted. Overall, however, the maintenance or change of farming system was probably highly dependent on farmers' motivations and attitudes in relation to different policies (e.g. Burton et al., 2008; De Snoo et al., 2013), so further research is required to elucidate the psychological and social processes driving farmers' decisions.

Farming intensification and land-use heterogeneity

The farming system transitions observed in our study resulted in an overall decline of within-farm heterogeneity, though there was no tendency for increasing intensification. Loss of heterogeneity was a consequence of farm-level specialization on either Permanent Crop or Livestock Grazing systems, with the concurrent loss of the mosaic of cereal fields, grazed fallows and pastures, and ploughed land associated with the Traditional system (Delgado and Moreira, 2000). In spite of these changes, the overall Intensification level was maintained, probably because the intensity levels of most farming systems in the region were roughly equivalent. The exception was the Permanent Crop system, which was both far more intensive than all the others and had virtually no land-use heterogeneity (i.e. monoculture).

Conservation implications

The High Nature Value Farmland type analysed in this study has a high biodiversity value, mainly due to their importance for a range of farmland birds of conservation concern (Suárez et al., 1997; Bota et al., 2005). Conservation of these species require a diverse agricultural mosaic, which favour the coexistence of species with contrasting habitat requirements (Reino et al., 2010), and species using different habitats during the annual cycle (Moreira et al., 2004). In these circumstances, the changes in farming systems observed during the study period, from the heterogeneous Traditional farming system to the specialized systems based on Livestock Grazing or Permanent Crops, may have had negative implications for farmland bird conservation. A full appreciation of the biodiversity consequences of these changes is still lacking, but it is likely that they will result in declines of species requiring cereal (e.g. corn bunting *Emberiza calandra*, fan-tailed warbler *Cisticola juncidis*, quail *Coturnix coturnix*) and ploughed fields (e.g. short-toed lark *Calandrella brachydactyla*, tawny pipit *Anthus campestris*, and black-eared wheatear *Oenanthe hispanica*), while favouring species associated with grazed pastures (e.g. calandra lark *Melanocorypha calandra*) (Delgado and Moreira, 2000; Reino et al., 2010).

The decline of the Traditional farming system observed in our study occurred despite the operation since 1995 of an AES specifically targeted to favour its persistence within the SPA of Castro Verde. This result underlines the fragility of AES, which can be overshadowed by other drivers of land-use change including large scale horizontal agricultural policies, such as decoupling of support from production, and small scale structural and biophysical factors constraining farmer options. Improving the efficacy of AES thus requires a better integration with other agricultural policies and fine-tuning to meet local specificities. As designing AES at the farm level may be unfeasible in most cases, the farming system approach used in our study may provide a practical alternative, by grouping farms according to their agricultural typology and providing information on the key factors driving major land-use transitions. AES could thus be designed to meet the specificities and constraints of each farming system, thereby optimizing investments on the farming systems that need to be maintained and encouraging transitions benefiting biodiversity in unfavourable farming systems.

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References

Bacharel, F., Pinto-Correia, T., 1999. Land use, nature conservation and regional policy in Alentejo, Portugal. In: Kroenert, R., Baudry, J., Bowler, I., Reenberg, A. (Eds.), Land-Use Changes and their Environmental Impact in Rural Areas in Europe. UNESCO, Paris, pp. 65–79.

Baldock, D., Beaufoy, G., Bennett, G., Clark, J., 1993. Nature Conservation and New Directions in the EC Common Agricultural Policy: the Potential Role of EC Policies in Maintaining Farming and Management Systems of High Nature Value in the Community. Institute for European Environmental Policy, London, 224 pp.

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.

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, Ubstadt-Weiher, pp. 525–535.

Benton, T.G., Vickery, J.A., Wilson, J.D., 2003. Farmland biodiversity: is habitat heterogeneity the key? Trends Ecol. Evol. 18, 182–188.

BirdLifeInternational, 2003. Balancing the Costs: Wildlife and Modern Agriculture. Royal Society for the Protection of Birds/Birdlife International, Sandy, Bedfordshire, 6 pp.

Bota, G., Morales, M.B., Camprodon, J. (Eds.), 2005. Ecology and Conservation of Steppe-land Birds. Lynx Edicions & Centre Tecnològic Forestal de Catalunya, Barcelona, 343 pp.

Brady, M., Kellermann, K., Sahrbacher, C., Jelinek, L., 2009. Impacts of decoupled agricultural support on farm structure, biodiversity and landscape mosaic: some EU results. J. Agr. Econ. 60, 563–585.

Bugalho, M.N., Caldeira, M.C., Pereira, J.S., Aronson, J., Pausas, J.G., 2011. Mediterranean cork oak savannas require human use to sustain biodiversity and ecosystem services. Front. Ecol. Environ. 9, 278–286.

Burton, R.J.F., Kuczera, C., Schwarz, G., 2008. Exploring farmers' cultural resistance to voluntary agri-environmental schemes. Sociol. Ruralis 48, 16–37.

Calvo-Iglesias, M.S., Fra-Paleo, U., Diaz-Varela, R.A., 2009. Changes in farming system and population as drivers of land cover and landscape dynamics: the case of enclosed and semi-openfield systems in Northern Galicia (Spain). Landscape Urban Plan 90, 168–177.

Carmona, A., Nahuelhual, L., Echeverría, C., Báez, A., 2010. Linking farming systems to landscape change: an empirical and spatially explicit study in southern Chile. Agr. Ecosyst. Environ. 139, 40–50.

Costa, M.A.M., Moors, E.J., Fraser, E.D.G., 2011. Socioeconomics, policy, or climate change: what is driving vulnerability in southern Portugal? Ecol. Soc. 16, 28 http://www.ecologyandsociety.org/vol16/iss1/art28/

Darnhofer, I., Gibbon, D., 2012. Farming systems research: an approach to inquiry. In: Darnhofer, I., Gibbon, D., Dedieu, B. (Eds.), Farming Systems Research into the 21st Century: the New Dynamic. Springer, Dordrecht, pp. 1–26.

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.

Delgado, A., Moreira, F., 2000. Bird assemblages of an Iberian cereal steppe. Agr. Ecosyst. Environ. 78, 65–76.

Dixon, J., Gulliver, A., Gibbon, D., 41 pp 2001. Farming Systems and Poverty – Improving Farmers' Livelihoods in a Changing World. FAO and World Bank, Rome and Washington DC ftp://ftp.fao.org/docrep/fao/004/ac349e/ac349e00.pdf.

Donald, P.F., Sanderson, F.J., Burfield, I.J., Van Bommel, F.P.J., 2006. Further evidence of continent-wide impacts of agricultural intensification on European farmland birds, 1990–2000. Agr. Ecosyst. Environ. 116, 189–196.

Doxa, A., Paracchini, M.L., Pointereau, P., Devictor, V., Jiguet, F., 2012. Preventing biotic homogenization of farmland bird communities: the role of high nature value farmland. Agr. Ecosyst. Environ. 148, 83–88.

Ferraton, N., Touzard, I., 2009. Comprendre l'Agriculture Familiale: Diagnostic des Systèmes de Production. CTA, Éditions Quae, and Les Presse Agronomique de Gembloux, Wageningen, Versailles and Gembloux, 123 pp.

Fragoso, R., Marques, C., Lucas, M.R., Martins, M.B., Jorge, R., 2011. The economic effects of common agricultural policy on Mediterranean montado/dehesa ecosystem. J. Policy Model. 33, 311–327.

Gergel, S.E., Turner, M.G. (Eds.), 2002. Learning Landscape Ecology: A Practical Guide to Concepts and Techniques. Springer, New York, 316 pp.

Hoogeveen, Y., Petersen, J.-E., Balazs, K., Higuero, I., 2004. High Nature Value Farm-land – Characteristics, Trends and Policy Challenges. European Environment Agency, Copenhagen, 32 pp.

Hosmer, D.W., Lemeshow, S., 1989. Applied Logistic Regression, 1st ed. Wiley, New York, 307 pp.

IGP, 2012. Carta de Ocupação do Solo - COS' 90 (1:25,000). Portuguese Geographic Institute http://www.igeo.pt/produtos/CEGIG/COS.htm (accessed 05.09.12).

Kaufman, L., Rousseeuw, P.J., 1990. Finding Groups in Data: An Introduction to Cluster Analysis. Wiley, New York, 349 pp.

Kleijn, D., Kohler, F., Báldi, A., Batáry, P., Concepción, E.D., Clough, Y., Díaz, M., Gabriel, D., Holzschuh, A., Knop, E., Kovács, A., Marshall, E.J.P., Tscharntke, T., Verhulst, J., 2009. On the relationship between farmland biodiversity and land-use intensity in Europe. Proc. R. Soc. Lond. B: Biol. Sci. 276, 903–909.

Lesschen, J.P., Verburg, P.H., Staal, S.J., 2007. Statistical Methods for Analysing the Spatial Dimension of Changes in Land Use and Farming Systems – LUCC Report Series 7. The International Livestock Research Institute & LUCC Focus 3 Office, Wageningen University, Nairobi, Kenya, 80 pp.

Maechler, M., Rousseeuw, P., Struyf, A., Hubert, M., Hornik, K., 2012. Cluster: Cluster Analysis Basics and Extensions. R package version 1.14.2. http://cran.r-project.org/web/packages/cluster/index.html (accessed 22.05.12).

Marta-Pedroso, C., Domingos, T., Freitas, H., de Groot, R.S., 2007. Cost–benefit analysis of the Zonal Program of Castro Verde (Portugal): highlighting the trade-off between biodiversity and soil conservation. Soil Tillage Res. 97, 79–90.

Mattison, E.H.A., Norris, K., 2005. Bridging the gaps between agricultural policy, land-use and biodiversity. Trends Ecol. Evol. 20, 610–616.

Moreira, F., Morgado, R., Arthur, S., Great, S., 2004. Great bustard *Otis tarda* habitat selection in relation to agricultural use in southern Portugal. Wildlife Biol. 10, 251–260.

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., 2012. 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.

Nagelkerke, N.J.D., 1991. A note on a general definition of the coefficient of determination. Biometrika 78, 691–692.

Nagy, S., 2006. CAP reform: what is there for bustard conservation? In: Leitão, D., Jolivet, C., Rodríguez, M., Tavares, J.P. (Eds.), Bustard Conservation in Europe in the Last 15 Years. RSPB/BirdLife, Sandy, pp. 115–122.

Oñate, J.J., Atance, I., Bardají, I., Llusia, D., 2007. Modelling the effects of alternative CAP policies for the Spanish high-nature value cereal-steppe farming systems. Agr. Syst. 94, 247–260.

Paracchini, M.L., Britz, W., 2010. Quantifying Effects of Changed Farm Practices on Biodiversity in Policy Impact Assessment – An Application of CAPRI-Spat. OECD, Paris, 16 pp.

Pinto, M., Rocha, P., Moreira, F., 2005. Long-term trends in great bustard (Otis tarda) populations in Portugal suggest concentration in single high quality area. Biol. Conserv. 124, 415–423.

R Development Core Team, 2011. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing http://www.r-project.org (accessed 26.12.11).

Reino, L., Porto, M., Morgado, R., Moreira, F., Fabião, A., Santana, J., Delgado, A., Gordinho, L., Cal, J., Beja, P., 2010. Effects of changed grazing regimes and habitat fragmentation on Mediterranean grassland birds. Agr. Ecosyst. Environ. 138, 27–34.

Renwick, A., Revoredo-Giha, C., Barnes, A., Jansson, T., Schwarz, G., 2008. Assessment of the impact of partial decoupling on prices, production and farm revenues within the EU. In: Final Report for DEFRA. Department. ed, Health (San Francisco), SAC Consulting, Ediburgh, 90 pp.

Rosário, M.S., 2005. Margens Brutas Padrão – Triénio de 2000. Gabinete de Planeamento e Política Agro-Alimentar, Lisboa, 56 pp.

Rousseeuw, P.J., 1987. Silhouettes – a graphical aid to the interpretation and validation of cluster analysis. J. Comput. Appl. Math. 20, 53–65.

Serrão, A., Coelho, L., 2005. Analysing farmers' decision-making process face to the mid-term review of common agricultural policy in the Alentejo dryland region of Portugal. In: 2005 Annual Meeting of the American Agricultural Economics Association, Providence, Rhode Island, 24 pp.

Spellerberg, I.F., Fedor, P.J., 2003. A tribute to Claude Shannon (1916–2001) and a plea for more rigorous use of species richness, species diversity and the Shannon-Wiener Index. Global Ecol. Biogeogr. 12, 177–179.

SROA/CNROA, 2012. Carta de Capacidade de Uso do Solo – 1:25,000. DGADR – Serviço Reconhecimento e Ordenamento Agrário, Lisboa http://www.dgadr.mamaot.pt/cartografia/cartas-solos-cap-uso-digital

Stoate, C., Báldi, 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 22–46.

Suárez, F., Naveso, M.A., Juana, E., 1997. Farming in the drylands of Spain: birds of the pseudosteppes. In: Pienkowski, M.W., Pain, D. (Eds.), Farming and Birds in Europe. Academic Press, London, pp. 297–330.

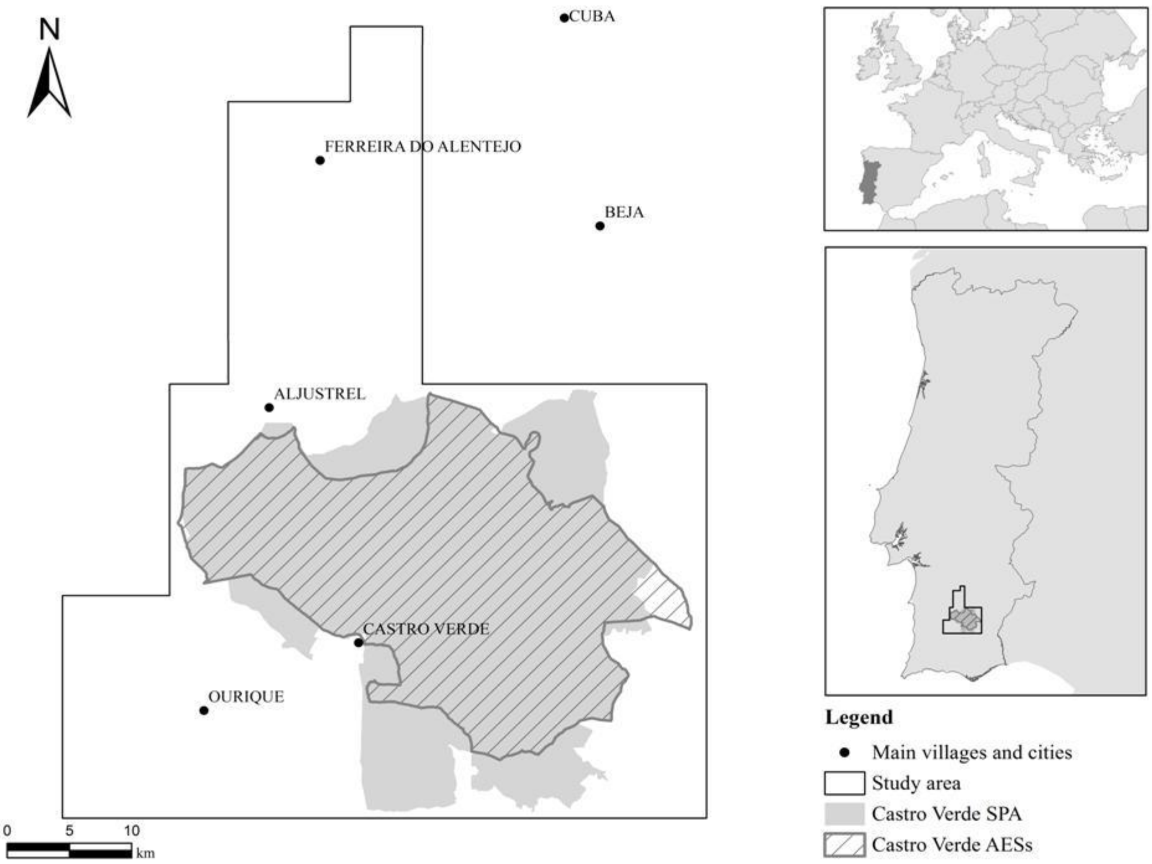
Tranter, R.B., Swinbank, a., Wooldridge, M.J., Costa, L., Knapp, T., Little, G.P.J., Sottomayor, M.L., 2007. Implications for food production, land use and rural development of the

European Union's Single Farm Payment: Indications from a survey of farmers' intentions in Germany, Portugal and the UK. Food Policy 32, 656–671.

Turner, B.L., Doolittle, W.E., 1978. The concept and measure of agricultural intensity. Prof. Geogr. 30, 297–301.

Supplementary information

Appendix 1. Location of the study area in the Castro Verde – Ferreira region of southern Portugal, showing the main urban areas, the Special Protection Area (SPA) of Castro Verde, and the area included in the Agri-environment Scheme (AES) of Castro Verde.



Appendix 2. Coefficients used to convert animals (heads) to Livestock Units.

Species	Age (Years)	Coefficient	Source
Bovine	> 2	1	(a)
	0.5 – 2	0.6	(a)
	< 0.5	0.4	(a)
Dairy cow	-	1.2	(b)
Sheep / Goat	> 1	0.15	(a)
	0.5 – 1	0.07	(b)

(a) PRODER - Portuguese Rural Development Program 2007-2013

(b) Portuguese legislation (Decreto-lei n.º 214/2008, November 10th)

Appendix 3. Detailed results of multinomial logistic regression models of factors affecting farming system dynamics from before (200-2002) to after (2008-2010) the CAP reform of 2003.

3.A. Multinomial logistic model for describing Traditional farming system dynamics

Significance of variables:

Effect	Chi-square [(a)]	df	P
Farm size	8.736	3	0.033
Oak woodland	6.533	3	0.088
Soil quality	15.591	3	0.001

(a) The chi-square statistic is the difference in -2 log-likelihoods between the final model and a reduced model. The reduced model is formed by omitting an effect from the final model. The null hypothesis is that all parameters of that effect are 0.

Parameter estimates (Variables with P<0.100 in bold):

Traditional system changes to [(a)]:		B	Std. Error	Wald	df	P	Exp(B)
Annual crops	Intercept	-3.905	1.443	7.326	1	0.007	
	Farm size	0.189	0.296	0.405	1	0.525	1.208
	Oak Woodland	**0.769**	**0.337**	**5.211**	**1**	**0.022**	**2.158**
	Soil quality	**0.672**	**0.303**	**4.915**	**1**	**0.027**	**1.957**
Cattle grazing	Intercept	-1.162	0.792	2.151	1	0.142	
	Farm size	**0.433**	**0.188**	**5.312**	**1**	**0.021**	**1.542**
	Oak Woodland	0.034	0.193	0.031	1	0.861	1.034
	Soil quality	-0.092	0.221	0.172	1	0.678	0.912
Sheep grazing	Intercept	1.037	0.653	2.526	1	0.112	
	Farm size	0.019	0.163	0.014	1	0.905	1.020
	Oak Woodland	0.045	0.176	0.064	1	0.800	1.046
	Soil quality	**-0.392**	**0.210**	**3.497**	**1**	**0.061**	**0.676**

(a) the reference category is Traditional system (persistence)

Model fit:

Likelihood ratio test: χ^2=33.5, df=9, P<0.001
Nagerkelke pseudo r^2=0.17
Classification accuracy= 47.2%
Null model accuracy = 45.7%

3.B. Multinomial logistic model for describing Annual crops system dynamics

Significance of variables:

Effect	Chi-square [(a)]	df	P
Oak Woodland	17.002	4	0.002
Soil quality	113.797	4	0.000

(a) The chi-square statistic is the difference in -2 log-likelihoods between the final model and a reduced model. The reduced model is formed by omitting an effect from the final model. The null hypothesis is that all parameters of that effect are 0.

Parameter estimates (Variables with P<0.100 in bold):

Annual crops changes to [(a)]		B	Std. Error	Wald	df	P	Exp(B)
Traditional	Intercept	-0.532	0.617	0.741	1	0.389	
	Soil quality	**-0.397**	**0.197**	**4.060**	**1**	**0.044**	**0.672**
	Oak Woodland	**0.491**	**0.202**	**5.885**	**1**	**0.015**	**1.634**
Cattle grazing	Intercept	0.818	0.686	1.419	1	0.233	
	Soil quality	**-1.393**	**0.273**	**26.114**	**1**	**0.000**	**0.248**
	Oak Woodland	-0.133	0.312	0.182	1	0.670	0.876
Sheep grazing	Intercept	2.460	0.628	15.335	1	0.000	
	Soil quality	**-2.430**	**0.397**	**37.507**	**1**	**0.000**	**0.088**
	Oak Woodland	**-0.714**	**0.329**	**4.715**	**1**	**0.030**	**0.490**
Permanent crops	Intercept	-1.658	0.941	3.105	1	0.078	
	Soil quality	-0.027	0.307	0.008	1	0.929	0.973
	Oak Woodland	0.041	0.290	0.020	1	0.889	1.041

(a) the reference category is Annual crops (persistence)

Model fit:

Likelihood ratio test: χ^2=128.5, df=8, P<0.001

Nagerkelke pseudo r^2=0.44

Classification accuracy= 61.5%

Null model accuracy = 53.0%

3.C. Multinomial logistic model for describing Cattle grazing system dynamics

Significance of variables:

Effect	Chi-square [a]	df	P
Farm size	3.249	1	0.071
SPA	3.705	1	0.054
Soil quality	10.335	1	0.001

(a) The chi-square statistic is the difference in -2 log-likelihoods between the final model and a reduced model. The reduced model is formed by omitting an effect from the final model. The null hypothesis is that all parameters of that effect are 0.

Parameter estimates (Variables with P<0.100 in bold):

Cattle grazing changes to [a]		B	Std. Error	Wald	df	P	Exp(B)
Sheep	Intercept	1.211	1.058	1.310	1	0.252	
Grazing	**Farm size**	**-0.439**	**0.245**	**3.221**	**1**	**0.073**	**0.645**
	SPA [outside]	**-0.946**	**0.498**	**3.615**	**1**	**0.057**	**0.388**
	Soil quality	**-1.725**	**0.739**	**5.445**	**1**	**0.020**	**0.178**

(a) the reference category is Cattle grazing (persistence)

Model fit:

Likelihood ratio test: χ^2=19.5, df=3, P<0.001

Nagerkelke pseudo r^2=0.22

Classification accuracy= 83.0%

Null model accuracy = 82.2%

Chapter 3

Landscape makers and landscape takers: links between farming systems and landscape patterns along an intensification gradient

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Landscape makers and landscape takers: links between farming systems and landscape patterns along an intensification gradient

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Abstract

Context. Agricultural intensification is a leading cause of landscape homogenization, with negative consequences for biodiversity and ecosystem services. Conserving or promoting heterogeneity requires a detailed understanding of how farm management affects, and is affected by, landscape characteristics.

Objectives. We assessed relationships between farming systems and landscape characteristics, hypothesising that less-intensive systems act as landscape takers, by adapting management to landscape constraints, whereas more intensive systems act as landscape makers, by changing the landscape to suit farming needs.

Methods. We mapped dominant farming systems in a region of southern Portugal: traditional cereal-grazed fallow rotations; specialization on annual crops; and specialization on either cattle or sheep. We estimated landscape metrics in 241 1-km^2 buffers representing the farming systems, and analysed variation among and within systems using multivariate statistics and beta diversity metrics.

Results. Landscape composition varied among systems, with dominance by either annual crops (Crop system) or pastures (Sheep), or a mixture between the two (Traditional and Cattle). There was a marked regional gradient of local landscape heterogeneity, but this contributed little to variation among systems. Landscape beta diversity declined from the Sheep to the Crop system, and it was inversely related to agriculture intensity.

Conclusions. Less intensive farming systems appeared compatible with a range of landscape characteristics (landscape takers), and may thus be particularly suited to agri-environmental management. More intensive systems appeared less flexible in terms of landscape characteristics (landscape makers), likely promoting regional homogenization. Farming systems may provide a useful standpoint to address the design of agri-environment schemes.

Keywords: Agriculture intensification; Agri-environment schemes; Conservation; Compositional heterogeneity; Configurational heterogeneity; Biodiversity; Ecosystem services; Farming systems; Land use; Mediterranean.

Introduction

Landscape heterogeneity is considered a key factor for biodiversity conservation and ecosystem-service provisioning on farmland (e.g. Benton et al 2003; Bassa et al 2012; Schippers et al 2015). During the past decades, however, farmland landscapes have become ever more homogeneous, which is often associated with the intensification of agriculture, and it is regarded as a major cause for the loss of biodiversity and of number of services such as habitat provisioning, pollination and water quality (e.g. Stoate et al 2001; Tscharntke et al 2005; Kleijn et al 2009; Stoate et al 2009). As a consequence, agri-environment policies and management strategies have been devised to maintain or enhance farmland heterogeneity, involving for instance support to farmers for increasing the amount of natural habitats, the diversity of crop types or land uses (compositional heterogeneity), and the spatial complexity of patch shapes and of their distribution in the landscape (configurational heterogeneity) (Fahrig et al. 2011). In general, however, limited consideration has been given to the influence of farm management strategies on landscape characteristics, and vice versa, though these interactions can impact on farmer’s income and thus limit the practical application of agri-environmental management (Fahrig et al. 2011; Bamière et al. 2011).

Farm management strategies are expected to affect both compositional and configurational landscape heterogeneity (Dunning et al. 1992; McGarigal and Marks 1995), as they shape for instance the type and diversity of crops, the size of fields to enable the use of heavy machinery, the amount and spatial distribution of natural habitats, and the prevalence of non-crop elements such as ponds, hedges, and scattered trees, among others (e.g. Deffontaines et al. 1995; Calvo-Iglesias et al. 2009; Carmona and Nahuelhual 2010). On the other hand, however, farm management options are constrained by a number of biophysical and structural features such as soil quality, the presence of rocky outcrops, and the amount of forest habitats, which are also reflected in landscape pattern (e.g. Persson et al. 2010; Ribeiro et al. 2014). In this context, farm managers may have one of two contrasting approaches, either modifying the landscape to suit their needs (i.e., landscape makers), or adapting farm management to existing landscape constraints (i.e., landscape takers). The first approach is often associated to agriculture intensification, involving for instance levelling of fields, removal of hedgerows and rocks, and drainage of wetlands, to allow for the intensive production of certain crop types (e.g. Tscharntke et al. 2005; Concepción et al. 2007; Ruiz and Domon 2009; Ferreira and Beja 2013; Lomba et al. 2014). The second approach is more associated with low-intensity systems, and implies farming strategies that can be implemented without major changes to landscape characteristics (e.g. Bignal and McCracken 1996; Lomba et al. 2015). Although this dichotomy probably represents the extremes of a continuum of farmer attitudes towards the landscape, it may provide a useful framework to understand how farm management shapes, or is shaped by, the landscape, and how this interferes with the development of agri-environment strategies for safeguarding farmland heterogeneity.

In common with other studies examining the relationships between agriculture and biodiversity and ecosystem services (e.g. Oñate et al 2007; Calvo-Iglesias et al 2009; Pointereau et al 2010; Carmona and Nahuelhual 2010; Bamière et al 2011), the farming-system approach may provide a convenient starting point to describe the interactions between farm management and landscape characteristics, as it helps simplifying the very large range of activities carried out by farmers. The concept of farming system stems from agricultural economics, and it is based on a comprehensive analysis of all farm management activities, including land use, animal husbandry, farming practices, resources involved, and the interdependencies between them, within a given political and socio-economic environment (Reboul 1976). Under this concept, farms may be aggregated based on the similarity of resource bases, production patterns and management strategies, which are likely to impact on the landscape in similar ways, and to show similar responses to biophysical conditions, as well as policy and market changes (Dixon et al. 2001; Ferraton and Touzard 2009). Therefore, it may be hypothesised that each farming system should be associated with a particular set of landscape characteristics, which may be more or less variable within each system depending on agriculture intensity level. Specifically, it may be expected that (i) more intensive farming systems should behave as landscape makers, transforming the landscape to meet their requirements, and thus showing low variability in landscape characteristics across areas farmed using the same system (i.e., low landscape beta diversity); and (ii) less intensive systems should behave as landscape takers, adapting to the constraints imposed by landscape features, and thus showing higher variability across areas under the same system (i.e., high landscape beta diversity).

In this paper we address these issues by assessing how farming systems are related to both landscape compositional and configurational heterogeneity, based on a case study developed in an agricultural landscape of southern Portugal. Using a typology of farming systems developed previously (Ribeiro et al. 2014), we mapped the spatial distribution of farming systems (i.e., a map of landscape units composed of blocks of contiguous farms sharing the same farming system) and characterised landscape patterns at the farming-system level (i.e., a spatial scale intermediate between the farm and the landscape levels), seeking to answer the following questions: (1) are landscape patterns different across farming systems?; (2) if yes, are these differences more associated to landscape composition or configuration?; (3) are there significant differences in landscape variability (beta diversity) across areas within each farming system?; and (4) is this landscape variability associated with the intensity of the farming system? Implications of the results for designing agro-environment policies to retain farmland heterogeneity are then discussed.

Methods

Study area

The study was carried out on a high nature value farmland area with ca. 210,000 ha in southern Portugal (approx. lat.: 38ºN; long.: 8ºW) (Figure 3. 1). The landscape is dominated by lowland agricultural systems, with undulating relief and altitudes ranging between 100-200 m above sea level. Climate is Mediterranean, with hot dry summers and mild rainy winters. The area encompasses the Special Protection Area (SPA) of Castro Verde, designated under EU Directive 79/409/CEE (Birds Directive) to protect populations of globally threatened species such as lesser kestrel (*Falco naumanni*), great bustard (*Otis tarda*) and little bustard (*Tetrax tetrax*) (BirdLife International 2004). The typical landscape is a mosaic of dry cereal crops in rotation with long term fallows, and low density livestock grazing (Delgado and Moreira 2000). Since 1995, part of the area has benefited from an agri-environment scheme under the European

Common Agricultural Policy (CAP) aiming to protect traditional farming systems (Santana et al. 2014). This scheme encourages agricultural practices considered favourable to conservation, including support to traditional rotation of cereals and fallows, the maintenance of low livestock densities, the growth of crops benefiting steppe birds, and the creation and maintenance of wildlife watering spots (Santana et al. 2014). In recent years the traditional farming system has been declining, with many farmers converting to specialized livestock systems, or to more intensive crop systems where soil quality and water availability makes it possible (Ribeiro et al. 2014).

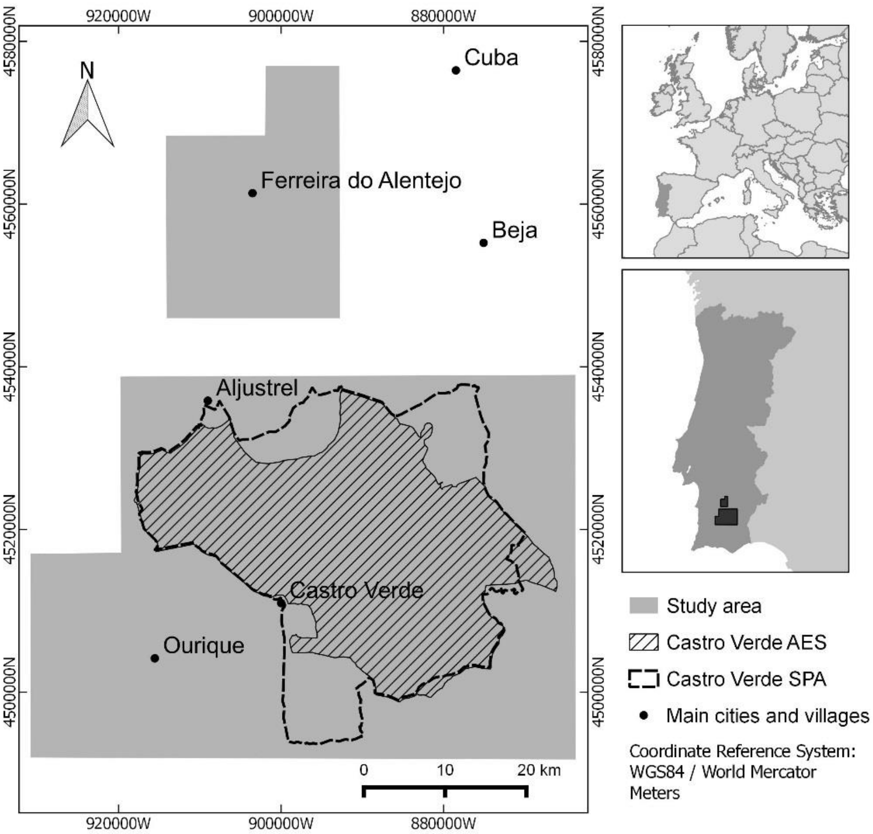

Figure 3. 1 - Location of the study area in southern Portugal, showing the Special Protection Area (SPA) of Castro Verde, the area included in the Agri Environment Scheme (AES) of Castro Verde, and the main urban areas.

Farming systems and landscape patterns

In a previous study using spatially explicit agricultural data at the farm level, Ribeiro et al (2014) identified four main farming systems occurring in the region in 2000-2002: (1) the Traditional system, typically comprising farms with a mosaic of dry cereals and fallows in rotation, and pastures grazed by sheep at moderate densities; (2) the Crops system, including farms with dry cereals and other annual (mostly irrigated) crops, and with no pastures, no livestock and little fallow land; (3) the Cattle system, comprising farms dominated by pastures and small areas of dry cereal, and with high densities of grazing cattle; and (4) the Sheep system, including farms with a crop composition similar to the Cattle system, but with lower livestock densities and composed strictly by sheep.

The intensity of each farming system was estimated following Brookfield (1972, 1993), from estimations of farm income per hectare, calculated by multiplying the unitary gross margin of the different crops and activities (Rosário 2005) by their relative proportions in each farming system. Although agricultural intensity is often measured from the input side (e.g., fertilizers or pesticides consumption per unit of land; Turner and Doolittle 1978; Lambin et al. 2000; Dietrich et al. 2010), here we measured it from the output side (e.g. tons of cereal or beef per unit of land) because data on input consumptions was unavailable, and because it does not involve any presumptions about the effect of inputs on productivity. Since we were comparing farming systems with distinct productions, we used a monetary surrogate to measure and compare outputs across farming systems (Turner and Doolittle 1978; Dorsey 1999).

A digital map of farming systems was produced in a geographical information system, by merging all neighbouring farms classified in the same system (Figure 3. 2). In this way we obtained geographic landscape blocks, each corresponding to groups of farms under a single farming system. A random sample of 1 km^2 circular land plots (ca. 564 m radius) was then extracted from the farming systems map, subject to three criteria: i) each plot should be completely enclosed within the same farming system; ii) plots should not overlap; and iii) a minimum of 30 plots should be extracted per farming system, with no upper limit. A sample of 241 circular land plots was thus obtained, of which 85 were in the Traditional system, 33 in the Crops system, 91 in the Cattle system, and 32 in the Sheep system (Figure 3. 2). Plots were then overlaid on a land use/land cover map (LUC) of the study area, representing the agricultural landscape in the spring of 2001.

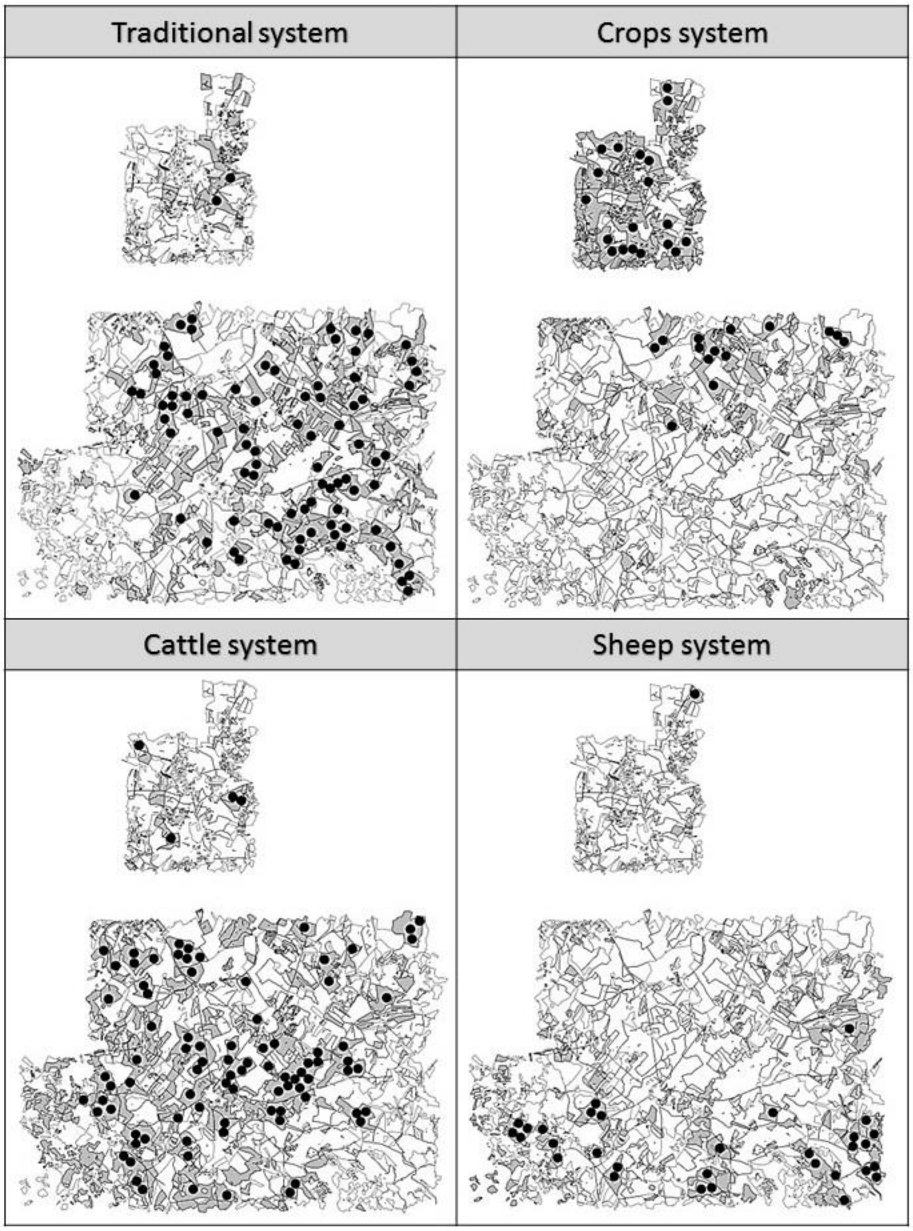

Figure 3. 2 - Distribution of the randomly selected circular land plots (black dots) used to extract landscape variables within the Traditional, Crops, Cattle and Sheep farming systems in the study area (grey areas) in 2000-2002.

Mapping considered 13 LUC categories (Table 3. 1), based primarily on farm parcel level data on agricultural land uses, extracted from a spatially explicit database maintained by the Portuguese Ministry of Agriculture (Ribeiro et al. 2014), and complemented with the following data: (i) shrubland and (ii) “montado” (open cork oak *Quercus suber* and holm oak *Q. rotundifolia* woodlands) cover, obtained from the 1990 digital land cover map (IGP 2012); and (iii) bare soil cover (including ploughed fields), mapped using Landsat ETM+ (Landsat 7) imagery (dated April 1, 2001); (iii). The shrubland and “montado” layers were updated using aerial photographs from 2001, thereby correcting for eventual changes that occurred since

1990. The proportion of the different LUC categories summed more than one, because we considered the land uses (e.g., crops) occurring under the tree canopy of “montado” (e.g., Bugalho et al. 2011).

Table 3. 1 - Summary statistics of variables used to characterize landscape composition and configuration in the 241 circular plots of 1 km^2.

Variable	Description	Mean ± SD (Min-Max)
Landscape composition		
CEREAL	Proportion of cereal crops	0.27 ± 0.23 (0 - 0.99)
FALLOW	Proportion of fallows	0.16 ± 0.19 (0 - 0.91)
BSOIL	Proportion of bare soil	0.01 ± 0.06 (0 - 0.75)
PASTURE	Proportion of permanent pastures	0.38 ± 0.31 (0 - 1.00)
SHRUB	Proportion of shrublands	0.02 ± 0.06 (0 - 0.40)
LEGUM	Proportion of leguminous crops	0.01 ± 0.05 (0 - 0.37)
FORAGE	Proportion of forage crops and temporary pastures	0.01 ± 0.04 (0 - 0.34)
FOREST	Proportion of forest	0.05 ± 0.14 (0 - 0.89)
PERMCROP	Proportion of permanent crops (olive groves)	0.01 ± 0.04 (0 - 0.42)
OTHACROP	Proportion of other annual crops	0.06 ± 0.15 (0 - 0.90)
BUILT-UP	Proportion of built-up areas (roads, buildings)	0.00 ± 0.01 (0 - 0.13)
WETLAND	Proportion of wetland areas (rivers, dams, reservoirs)	0.01 ± 0.03 (0 - 0.34)
MONTADO	Proportion of “montado”	0.22 ± 0.32 (0 – 0.98)
NUSES	Number of different land uses/covers (LUC)	3.9 ± 1.3 (1 – 8)
SDI	Shannon diversity index of LUC	0.87 ± 0.35 (0 – 1.73)
Landscape configuration		
NPATCH	Number of patches	6.0 ± 3.4 (1 – 20)
TEDG	Total edge density (meters/hectare)	63.7 ± 35.0 (0-185.1)
PSCOV	Patch size coefficient of variance (patch size standard deviation divided by the mean patch size)	111.2 ± 46.8 (0-285.9)
AWMSI	Area weighted mean shape index (area weighted sum of each patches perimeter divided by the square root of patch area for all patches and adjusted for the plot, divided by the number of patches)	1.44 ± 0.27 (1 – 2.49)

To describe landscape pattern we used 15 variables quantifying the proportions and diversity of dominant LUC types (landscape composition), and four variables describing configurational heterogeneity (landscape configuration) (Table 3. 1). Variables were selected from a wide range of metrics commonly used in landscape ecology studies (McGarigal and Marks 1995), following the recommendation to avoid redundant or highly correlated variables (e.g. Cushman et al. 2008; Schindler et al. 2008; Bassa et al. 2012). Landscape variables were computed for each of the 241 circular land plots (Table 3. 1), using the Patch Analyst extension to ArcGIS 10 (Elkie et al. 1999).

Data analysis

Landscape variables expressing proportions (Table 3. 1) were arcsin(sqrt(x)) transformed and the remaining were log10(x+1) transformed to improve normality and stabilize variances (McDonald 2009). Overall local landscape patterns were explored using a principal components analysis (PCA) performed on a correlation matrix of the 19 transformed variables. Principal components (PC) with an eigenvalue larger than 1 were retained (Kaiser 1960), and a varimax rotation was applied to simplify and improve the interpretability of the solution. A one-way analysis of variance (ANOVA) was used to test for differences among the mean scores on each PC of plots classified in each farming system.

Linear discriminant analysis (LDA) was then performed on PC to assess the extent to which landscape patterns differed across farming systems, and whether the latter could be predicted from the former. A stepwise forward variable selection procedure was conducted using the Wilks' lambda test, starting with an initial model that includes the variable which separates groups the most, and then adding further variables contingent on the Wilk's lambda criterion (Roever et al. 2014). The procedure was stopped when no additional variable was significant at $p < 0.05$. Prediction accuracy was assessed based on a confusion matrix, implemented with leave-one-out cross-validation. Because sample sizes were uneven among farming systems, we used Cohen's kappa to correct for agreements occurring by chance between observed and predicted categories (Titus et al. 1984).

Inertia ellipses were used to help visualizing the distribution of observations within each farming system along the axes in PCA and LDA plots, considering the default probability of ca. 66% corresponding to a one standard deviation length. To characterise variability in landscape pattern among plots classified in the same farming system, we computed an adimensional beta diversity index (Anderson et al. 2006). Specifically, we computed the average Euclidian distance to group centroids within each farming system using the first two PC coordinates, and then standardized the index by dividing their values by the maximum distance obtained for the four farming systems.

Statistical analyses were implemented in R version 3.1.1 (R Development Core Team 2014) using the “psych” package (Revelle 2014) for the PCA, the “klaR” package (Weihs et al. 2005; Roever et al. 2014) for the Wilks’ lambda test and the “MASS” (Venables and Ripley 2002) for the LDA. The “candisc” package (Friendly and Fox 2014) was used to extract the standardized canonical coefficients of the discriminant functions.

Results

Overall patterns

There was a spatial trend for the Crops system to be predominant in the northern section of the study area, while the Sheep system was more common in the South. Both the Traditional and the Cattle systems were more evenly spread throughout the study area (Figure 3. 2). The sampled landscape plots were dominated by pastures, cereal crops, and fallow fields, which together accounted for more than 80% of the area (Table 3. 1). The average number of different LUC was 3.9 per plot, and the average patch number was six. “Montados” were common in the region, covering ca. 20% of the area of the plots (Table 3. 1).

The PCA returned seven PC with eigenvalues larger than one, together accounting for 69% of the overall variance (Table 3. 2). The first PC (hereafter named “Heterogeneity”) represented a gradient of local landscape heterogeneity, showing a joint increase in the number of patches, edge density, patch size variation, shape complexity (AWMSI), land cover richness and diversity. The second PC (“Specialization”) represented a gradient from landscapes dominated by annual crops (cereals and other annual crops) to landscapes dominated by permanent pastures, thereby separating landscapes associated with crop production from the ones specialized in livestock production. PC3 (“Permanent crops”) was mainly related to increasing proportion of permanent crops (mainly olive orchards), and also of wetlands. PC4 (“Built-up areas”) was associated to the presence of built-up areas, PC5 (“Leguminous crops”) reflected the proportion of leguminous crops, PC6 (“Forage”) represented the proportion of forage crops, and PC7 (“Fallows”) was associated to both fallows and bare soil areas, with opposite coefficient signs. Plot coordinates in PC were significantly different across farming systems (ANOVA, $p < 0.05$), except for PC4 ($p = 0.340$) and PC5 ($p = 0.422$) (Figure S1 in supplementary material).

Table 3. 2 - Standardized loadings of variables on the principal components (PC) retained (eigenvalues > 1) from a principal components analysis of landscape metrics, and rotated using varimax. Variables are sorted by loading values (values > 0.50 in bold) and PC are sorted by explained variance. Variable acronyms are in Table 3. 1 and the ecological interpretation of PC are in Table 3. 3.

Variables	PC1	PC2	PC3	PC6	PC4	PC7	PC5
NPATCH	**0.91**	0.02	0.07	0.09	-0.01	-0.04	0.01
TEDG	**0.91**	0.23	0.07	0.02	-0.05	-0.11	0.01
NUSES	**0.78**	0.08	0.27	0.31	0.15	-0.01	0.06
AWMSI	**0.75**	0.01	0.01	-0.35	0.00	-0.02	0.05
SDI	**0.73**	0.31	0.20	0.13	-0.04	-0.25	0.04
PSCOV	**0.72**	0.04	-0.04	0.13	-0.03	0.12	-0.03
PASTURE	-0.28	**-0.79**	-0.27	0.01	0.22	0.10	0.19
CEREAL	0.08	**0.78**	-0.16	-0.01	-0.01	0.07	0.27
OTHACROP	0.06	**0.53**	0.13	-0.03	0.15	0.00	-0.11
PERMCROP	0.06	0.09	**0.77**	-0.18	0.22	-0.01	0.11
WETLAND	0.12	0.08	**0.72**	0.10	-0.02	0.05	0.05
FORAGE	0.26	-0.11	-0.07	**0.72**	0.05	0.00	0.05
SHRUB	**0.53**	-0.37	-0.11	**-0.55**	-0.08	0.11	-0.03
BUIL-UP	0.18	0.10	0.17	0.19	**0.75**	0.14	-0.05
MONTADO	-0.30	-0.18	0.06	-0.07	**0.54**	-0.17	-0.25
FOREST	0.08	-0.28	0.39	0.34	-0.46	0.11	-0.39
FALLOW	0.23	0.23	-0.05	-0.06	-0.12	**-0.71**	-0.34
BSOIL	0.05	0.15	0.03	-0.07	-0.07	**0.71**	-0.28
LEGUM	0.08	-0.04	0.17	0.05	-0.14	-0.05	**0.78**
Proportion of variance	0.24	0.11	0.08	0.07	0.07	0.06	0.06

The highest average local landscape heterogeneity, revealed by the position of the group centroids along the heterogeneity axis (PC1), occurred in the Traditional system, followed by the Cattle and Crop systems, whereas the lowest heterogeneity was found in the Sheep system (Figure 3. 3A). However, the inertia ellipses suggested that the Sheep system was the most variable spatially, as its landscape plots were found highly scattered along the entire heterogeneity axis (PC1). In contrast, the Crop system presented the lowest dispersion of plots along this axis (Figure 3. 3A).

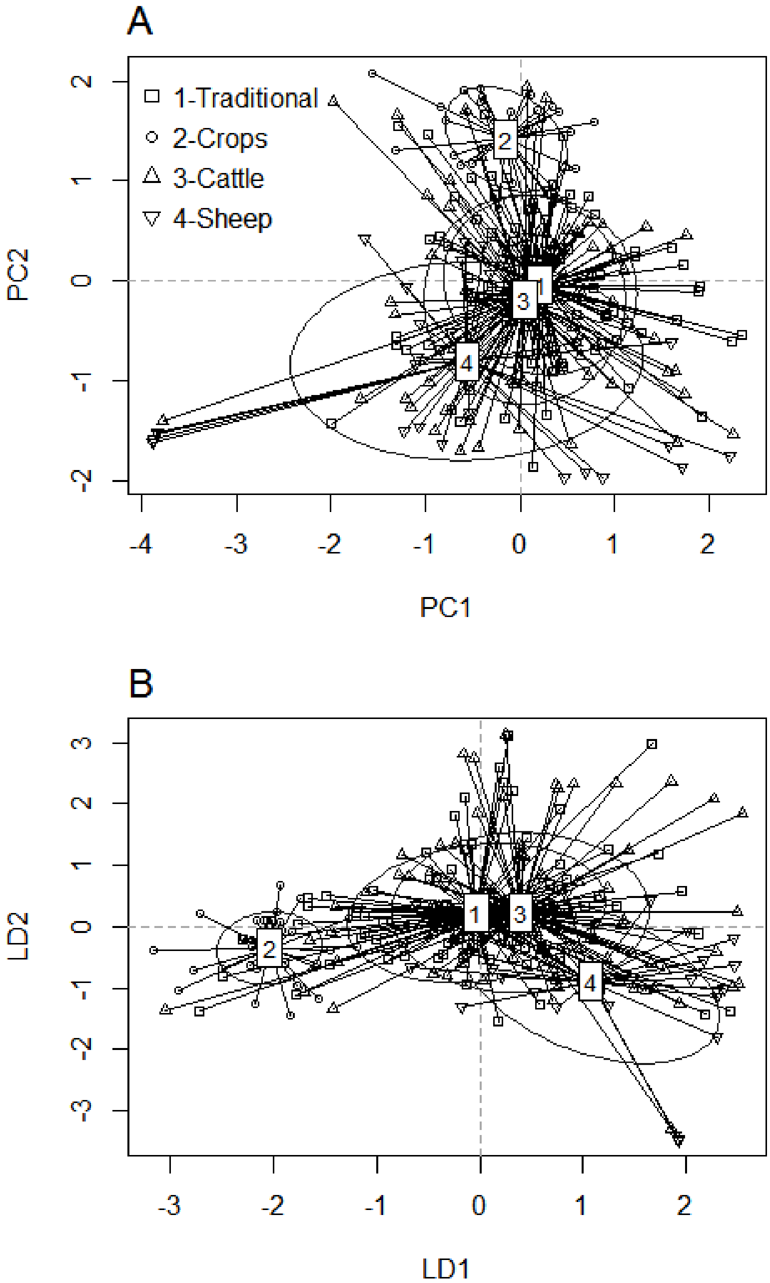

Figure 3. 3 - Scatterplot of the 241 landscape plots in the first two axis extracted from a principal components analysis with varimax rotation (A) and in the first two axis of a linear discriminant analysis (B). The centroids and inertia ellipses are provided for each farming system.

Discrimination of farming systems based on landscape patterns

Variable selection using Wilks' lambda returned five PCs that significantly contributed to the separation of farming systems (Table 3. 3), which were subsequently used in the linear discriminant analysis (LDA). Built-up areas (PC4) and Leguminous crops (PC5) were discarded because they failed to improve the group separation power of the model.

Table 3. 3 - Summary of the Wilk's lambda stepwise forward variable selection test performed on the 7 PC. With a 0.05 threshold for the appropriate p-value of the F-statistic of the partial Wilk's lambda (p value), PC4 and PC5 were discarded.

Variables	Wilks' lambda	F statistics overall	p value overall	F statistics	p value
Specialization (PC2)	0.622	47.932	0.000	47.932	0.000
Forage (PC6)	0.564	26.107	0.000	8.183	0.000
Fallows (PC7)	0.523	19.404	0.000	6.117	0.001
Permanent crops (PC3)	0.486	16.180	0.000	5.914	0.001
Heterogeneity (PC1)	0.453	14.253	0.000	5.660	0.001
Built-up areas (PC4)	0.445	12.094	0.000	1.408	0.241
Leguminous crops (PC5)	0.438	10.543	0.000	1.292	0.278

The first discriminant function (LD1) captured most of the between-group variance (78%), mainly separating the Crops from the other systems (Figure 3. 3B). Specialization (PC2) was the most important variable contributing to LD1 (Table 3. 4), with landscapes associated to the Crops system showing higher proportions of cereals and other annual crops, whereas landscapes associated to the Sheep system had a higher proportions of pastures. The Traditional and Cattle systems were both located close to the origin (Figure 3. 3B), indicating that they were little differentiated by the specialization gradient described by PC2. However, there was a tendency for landscapes in the Traditional system to be closer to the Crops system, due to their higher proportion of cereals. Likewise, the Cattle system tended to be closer to the Sheep system due to their higher proportion of pastures.

The between-group variance captured by the second function (LD2; 16%) was much lower than that of LD1, thereby showing a much lower discrimination ability of farming systems. Nevertheless, LD2 contributed to a weak separation of Traditional and Cattle systems from the Sheep system, with the former being associated with higher local landscape heterogeneity (PC1) and higher proportion of forages (PC6) (Figure S1 in supplementary material). The third discriminant function (LD3) captured a marginal 6% of the between group variance, and it was mainly associated with PC7 (Fallows).

Table 3. 4 - Standardized canonical coefficients of the discriminant functions. Coefficients larger than 0.5 are indicated in bold.

Variable (PC)	LD1	LD2	LD3
Specialization (PC2)	**-0.97**	0.05	0.23
Forage (PC6)	0.14	**0.79**	0.25
Fallows (PC7)	0.28	-0.04	**0.88**
Perm. Crops (PC3)	-0.40	-0.04	-0.07
Heterogeneity (PC1)	-0.08	**0.66**	-0.28

The overall percent agreement rate achieved by LDA predictions with leave-one-out cross-validation was 55.7% (Table S1 in supplementary material), corresponding to an overall Cohen's kappa of 0.35 after correcting for chance agreements. In pairwise classification tables for individual farming systems, the proportions of correct classifications were much higher for the Crops (92.5%; Cohen's kappa = 0.68) and Sheep (87.1%; Cohen's kappa = 0.44) systems, than for the Traditional (65.6%; Cohen's kappa = 0.25) and Cattle (64.3%; Cohen's kappa = 0.24) systems.

Landscape beta diversity and agricultural intensity

Landscape beta diversity was highest for the Sheep system, intermediate for the Cattle and Traditional systems, and lowest for the Crops system (Figure 3. 4). There was a significant inverse relationship between beta diversity and agricultural intensity (R=-0.96, P=0.030), though care should be taken in the interpretation of statistical testing due to small sample sizes.

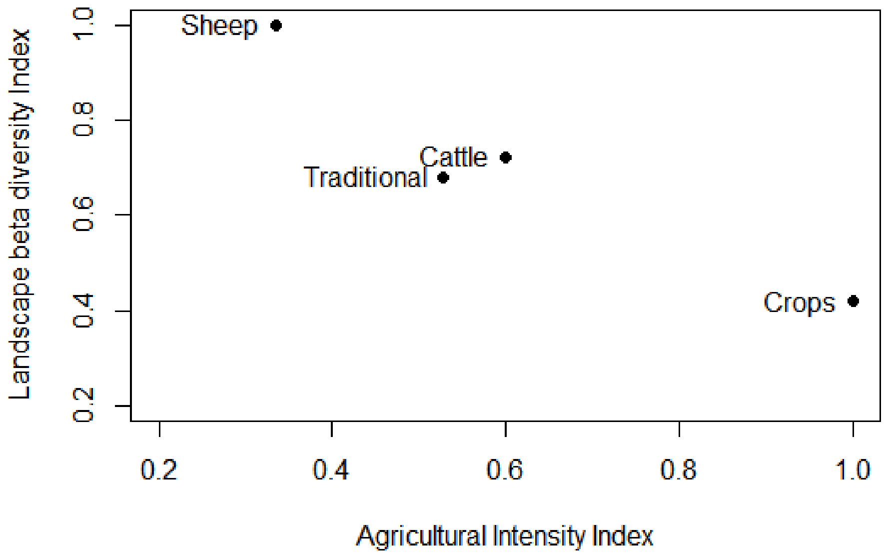

Figure 3. 4 - Relationship between the landscape beta diversity index and the index of agricultural intensification, for four farming systems identified in the study area.

Discussion

Our study pointed out some differences in landscape patterns across farming systems, and that these were far more associated to differences in landscape composition than in configuration. Also, we found differences in landscape beta diversity across farming systems, and that there was an inverse relationship between beta diversity and agricultural intensity. Overall, therefore, our results agree with the idea that farms associated with the more intensive systems may operate under a narrower range of landscape patterns, thereby shaping the landscape to suit their needs. In contrast, less-intensive systems may be compatible with a wider range of landscape patterns, thereby adapting to existing landscape features. These results have implications for agri-environment policies, as they suggest that maintaining landscape patterns to achieve biodiversity and ecosystem service goals require due consideration of the farming systems operating in those landscapes.

Results of our study are in line with previous research showing that farming systems have an influence on landscape pattern and dynamics (e.g. Deffontaines et al 1995; Calvo-Iglesias et al 2009; Carmona et al 2010). In our case, this relationship was mostly a consequence of differences in land uses and cover between systems, mainly associated with the higher amount of cereals and other annual crops in the Crop system, and with the higher amount of pastures in the Sheep system. The Traditional and the Cattle systems had an intermediate position in this gradient, though with stronger affinities between the Traditional and the Crop systems, and between the Cattle and the Sheep systems. This suggests, therefore, that one of the main mechanisms whereby farmers affect landscape pattern is through a decision to produce either crops or livestock, thus leading to a compositional dominance of either cereals and other crops or pastures, or a mix between the two in the Traditional system.

The relatively weak relation between farming systems and local landscape configuration (sensu Fahrig et al. 2011) observed in our study was unexpected, because there was in the study region a dominant gradient from simple to heterogeneous landscapes (with higher number of patches, more variable patch sizes, more complex shaped patches, and higher edge densities). However, farming systems were little differentiated along this landscape gradient, though there was a tendency for the highest local heterogeneity to be found in the Traditional system, followed by the Cattle and Crop systems, and then by the Sheep system. The highest heterogeneity found for the Traditional system was probably a consequence of the mixture of different crop types and pastures characterising the system, which may promote a more complex patchwork of land uses. This is also in line with the major importance for biodiversity conservation of this system, because the diversity of land uses and cover is beneficial for a wide range of species with contrasting requirements, and for species requiring a blend of complementary habitats (e.g. Reino et al. 2010; Santana et al. 2014). It should be noted, however, that variability within each farming system was high, and so high to medium levels of local landscape heterogeneity could be found in areas dominated by any of the farming system.

In line with expectations, there was marked variation in landscape beta diversity among farming systems, with a tendency for a gradient from higher to lower beta diversity to be inversely related to agriculture intensification. The Sheep system is a case in point, corresponding to the less intensive system and the one showing the wider range of landscape patterns. Farmers may thus choose to specialise on sheep production largely irrespective of landscape patterns, either because this production system is not strongly constrained by the landscape or, in alternative, because the farmer’s income is not sufficiently high to allow investment for changing the landscape to meet their needs. In contrast, the Crop system was the most intensive and was associated with the narrowest range of landscape beta diversity. This was probably because specialised crop production demands landscapes with specific characteristics, due for instance to the need of large fields where pivot irrigation systems and heavy machinery can operate. The Traditional and the Cattle systems were intermediate in the beta diversity and intensification gradients, probably because they can accommodate a relatively wide range of landscape patterns, though they may still be able to change the landscape where this is necessary to suit their needs.

Overall, our results are consistent with the landscape taker versus landscape maker dichotomy, with the Sheep system falling in the former category, the Crop system falling in the latter, and the Cattle and Traditional systems probably having an intermediate position. It should be borne in mind, however, that our study is based on a snapshot of farming systems and landscape patterns, and so it is unknown whether the most intensive systems actually changed the landscape to suit their needs (i.e., acted as true landscape makers), or whether they were adopted in landscapes that were already suitable at the outset. Probably both mechanisms were influential, as there is evidence that temporal dynamics in farming systems are shaped by biophysical and structural characteristics of the territory such as soil fertility or the presence of forests limiting agricultural development (Ribeiro et al. 2014). There is also some evidence, however, that the most intensive systems actively change the landscape, by for instance increasing field size, removing hedgerows, ponds, scattered trees and non-crop elements, and reducing crop diversity (e.g. Tscharntke et al. 2005; Concepción et al. 2007; Ruiz and Domon 2009; Ferreira and Beja 2013; Lomba et al. 2014).

Our results have implications for policies and management strategies aiming to retain landscape heterogeneity on farmland, and thus to the conservation of biodiversity and ecosystem services (e.g. Fahrig et al. 2011). We found that more intensive farming systems were associated with reduced landscape beta diversity, thereby suggesting that agricultural intensification may contribute to landscape homogenization at the regional scale. The negative effects of homogenization have been demonstrated in several studies, both in our area (e.g. Reino et al. 2010; Santana et al. 2014) and elsewhere (e.g. Benton et al. 2003; Bassa et al. 2011; Bassa et al. 2012), thereby pointing out the need to avoid regional dominance by a single, intensive farming system. Also, we found that less-intensive systems may occur under a relatively wide range of landscape patterns, suggesting that they may be better able than intensive systems to adapt to landscape constraints. As such, these systems may be more compatible with agri-environment schemes aimed at preserving heterogeneity or to retain important landscape features (e.g., ponds and hedgerows), because these should interfere little with farm management operations. As a consequence of the former two findings, it is likely that in a landscape with a mixture of farming systems with different intensity levels, agri-environment schemes should primarily target at the less intensive ones, with the double objective of promoting landscape features important for biodiversity and ecosystem conservation, and avoiding transitions between less- and more-intensive farming systems (e.g. Ribeiro et al. 2014). The latter may be particularly important, because intensification often results in landscape changes that are damaging to biodiversity and ecosystem services (e.g. Stoate et al 2001; Tscharntke et al 2005; Kleijn et al 2009; Stoate et al 2009), and these may be irreversible through voluntary policies like the agri-environment schemes due to the high costs required to compensate the investments and eventual loss of income by farmers. Furthermore, intensive systems may have limited flexibility to accommodate landscape patterns compatible with biodiversity conservation (e.g. hardly reversible loss of non-crop elements).

Overall, our results underline the importance of considering farming systems as a tool for designing agri-environment policies and to inform landscape management strategies. The farming system approach may be particularly useful because it explicitly recognises the existence of tracts of farmland that are managed similarly and, for this reason, react to the same market and policy changes, which eventually lead to changes on landscape composition and configuration, both at the local and at the regional level (Deffontaines et al 1995; Calvo-Iglesias et al 2009; Carmona et al 2010; this study). As different farming systems often coexist in the same region, cost-effective design of agri-environment schemes should thus be based on a correct identification of dominant farming systems, on their compatibility with environmental management objectives, and on the priority and type of support that should be given to each one for promoting biodiversity conservation and the delivery of ecosystem services. This might provide an opportunity to tailor agri-environment schemes to socio-ecological specificities, without the high costs that would be needed for developing and implementing farm-level management strategies.

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References

Anderson MJ, Ellingsen KE, McArdle BH (2006) Multivariate dispersion as a measure of beta diversity. Ecol Lett 9:683–693

Bamière L, Havlík P, Jacquet F, Lherm M (2011) Farming system modelling for agri-environmental policy design: The case of a spatially non-aggregated allocation of conservation measures. Ecol Econom 70:891–899

Bassa M, Boutin C, Chamorro L, Sans F (2011) Effects of farming management and landscape heterogeneity on plant species composition of Mediterranean field boundaries. Agric Ecosyst Environ 141:455–460

Bassa M, Chamorro L, Sans FX (2012) Vegetation patchiness of field boundaries in the Mediterranean region: The effect of farming management and the surrounding landscape analysed at multiple spatial scales. Landsc Urban Plan 106:35–43

Benton TG, Vickery JA, Wilson JD (2003) Farmland biodiversity: is habitat heterogeneity the key? Trends Ecol Evol 18:182–188

Bignal EM, McCracken DI (1996) Low-intensity systems in the conservation of the countryside. J Appl Ecol 33:413–424

BirdLifeInternational (2004) Birds in the European Union: a status assessment. Wageningen, The Netherlands: BirdLife International. http://www.vwgdepeel.ivnastensomeren.nl/downloads/BOCC_birds_in_the_eu.pdf. Accessed January 2015

Brookfield HC (1972) Intensification and Disintensification in Pacific Agriculture: A Theoretical Approach. Pac Viewp 13:30–48.

Brookfield HC (1993) Notes on the theory of land management. PLEC News Views 1:28–32.

Bugalho MN, Caldeira MC, Pereira JS, et al (2011) Mediterranean cork oak savannas require human use to sustain biodiversity and ecosystem services. Front Ecol Environ 9:278–286

Calvo-Iglesias MS, Fra-Paleo U, Diaz-Varela RA (2009) Changes in farming system and population as drivers of land cover and landscape dynamics: The case of enclosed and semi-openfield systems in Northern Galicia (Spain). Landsc Urban Plan 90:168–177

Carmona A, Nahuelhual L (2010) Linking farming systems to landscape change: an empirical and spatially explicit study in southern Chile. Agric Ecosyst Environ 139:40–50

Concepción ED, Díaz M, Baquero RA (2007) Effects of landscape complexity on the ecological effectiveness of agri-environment schemes. Landsc Ecol 23:135–148

Cushman SA, McGarigal K, Neel MC (2008) Parsimony in landscape metrics: Strength, universality, and consistency. Ecol Indic 8:691–703

Deffontaines JP, Thenail C, Baudry J (1995) Agricultural systems and landscape patterns: how can we build a relationship? Landsc Urban Plan 31:3–10

Delgado A, Moreira F (2000) Bird assemblages of an Iberian cereal steppe. Agric Ecosyst Environ 78:65–76

Dietrich J, Schmitz C, Müller C (2010) Measuring agricultural land-use intensity. Paper presented at HAWEPA 2010 workshop in Halle, Germany, June 28-29

Dixon J, Gulliver A, Gibbon D (2001) Farming Systems and Poverty - Improving Farmers' Livelihoods in a Changing World. FAO and The World Bank, Rome and Washington DC

Dorsey B (1999) Agricultural Intensification, Diversification, and Commercial Production among Smallholder Coffee Growers in Central Kenya. Econ Geogr 75:178–195

Dunning JB, Danielson BJ, Pulliam HR (1992) Ecological processes that affect populations in complex landscapes. Oikos 65:169–175

Elkie P, Rempel R, Carr A (1999) Patch analyst user's manual: a tool for quantifying landscape structure. Ont. Min. Natur. Resour. Northwest Sci. Technol, Ontario

Fahrig L, Baudry J, Brotons L, et al (2011) Functional landscape heterogeneity and animal biodiversity in agricultural landscapes. Ecol Lett 14:101–12

Ferraton N, Touzard I (2009) Comprendre l'agriculture familiale : diagnostic des systèmes de production, Quae. Les presse agronomique de Gembloux, Wageningen

Ferreira M, Beja P (2013) Mediterranean amphibians and the loss of temporary ponds: Are there alternative breeding habitats? Biol Conserv 165:179–186

Friendly M, Fox J (2014) Package "candisc". Visualizing Generalized Canonical Discriminant and Canonical Correlation Analysis. R package version 0.6-5. Available from http://cran.r-project.org (accessed August 2014)

IGP (2012) Carta de Ocupação do Solo - COS' 90 (1:25,000). Portuguese Geographic Institute. Available from http://www.igeo.pt/produtos/CEGIG/COS.htm (accessed 9 May 2012)

Kaiser H (1960) The application of electronic computers to factor analysis. Educ Psychol Meas 20:10.

Kleijn D, Kohler F, Báldi A, et al (2009) On the relationship between farmland biodiversity and land-use intensity in Europe. Proc Biol Sci 276:903–909

Lambin EF, Rounsevell MD a, Geist HJ (2000) Are agricultural land-use models able to predict changes in land-use intensity? Agric Ecosyst Environ 82:321–331

Lomba A, Guerra C, Alonso J, et al (2014) Mapping and monitoring High Nature Value farmlands: challenges in European landscapes. J Environ Manage 143:140–50

Lomba A, Alves P, Jongman RHG, Mccracken DI (2015) Reconciling nature conservation and traditional farming practices: a spatially explicit framework to assess the extent of High Nature Value farmlands in the European countryside. Ecol Evol 1–14

McDonald J (2009) Handbook of biological statistics, University. Sparky House Publishing, Baltimore, Maryland

McGarigal K, Marks BJ (1995) FRAGSTATS: Spatial Pattern Analysis Program for Quantifying Landscape Structure, Gen. Tech. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland

Oñate JJJ, Atance I, Bardají 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

Persson AS, Olsson O, Rundlöf M, Smith HG (2010) Land use intensity and landscape complexity—Analysis of landscape characteristics in an agricultural region in Southern Sweden. Agric Ecosyst Environ 136:169–176

Pointereau P, Doxa A, Paracchini ML (2010) Analysis of spatial and temporal variations of High Nature Value farmland and links with changes in bird populations: a study on France. Joint Research Centre of the European Commission

R Development Core Team (2014) R: A language and environment for statistical computing. In: R Found. Stat. Comput. http://www.r-project.org (accessed February 2014)

Reboul C (1976) Mode de production et systèmes de culture et d'élevage. Économie Rural 112:55–65

Reino L, Porto M, Morgado R, et al (2010) Effects of changed grazing regimes and habitat fragmentation on Mediterranean grassland birds. Agric Ecosyst Environ 138:27–34

Revelle W (2014) Package "psych": Procedures for psychological, psychometric, and personality research. http://www.personality-project.org/r (accessed August 2014)

Ribeiro PF, Santos JL, Bugalho MN, et al (2014) Modelling farming system dynamics in High Nature Value Farmland under policy change. Agric Ecosyst Environ 183:138–144

Roever C, Raabe N, Luebke K, et al (2014) Package "klaR": Classification and visualization. R package version 0.6-12. Available from http://www.statistik.tu-dortmund.de (accessed August 2014)

Rosário MS (2005) Margens Brutas Padrão - Triénio de 2000. Gabinete de Planeamento e Política Agro-Alimentar, Lisboa

Ruiz J, Domon G (2009) Analysis of landscape pattern change trajectories within areas of intensive agricultural use: case study in a watershed of southern Québec, Canada. Landsc Ecol 24:419–432

Santana J, Reino L, Stoate C, et al (2014) Mixed Effects of Long-Term Conservation Investment in Natura 2000 Farmland. Conserv Lett 7:467–477

Schindler S, Poirazidis K, Wrbka T (2008) Towards a core set of landscape metrics for biodiversity assessments: A case study from Dadia National Park, Greece. Ecol Indic 8:502–514

Schippers P, Heide CM Van Der, Peter H, et al (2015) Landscape diversity enhances the resilience of populations, ecosystems and local economy in rural areas. Landsc Ecol 30:193–202

Stoate C, Báldi A, Beja P, et al (2009) Ecological impacts of early 21st century agricultural change in Europe--a review. J Environ Manage 91:22–46

Stoate C, Boatman ND, Borralho RJ, et al (2001) Ecological impacts of arable intensification in Europe. J Environ Manage 63:337–365

Titus K, Mosher JA, Williams BK (1984) Chance-corrected Classification for Use in Discriminant Analysis: Ecological Applications. Am Midl Nat 111:1

Tscharntke T, Klein AM, Kruess A, et al (2005) Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. Ecol Lett 8:857–874

Turner BL, Doolittle WE (1978) The Concept and Measure of Agricultural Intensity. Prof. Geogr. 30:297–301.

Venables W, Ripley B (2002) Modern applied statistics with S, Fourth Edi. Springer, New York

Weihs C, Ligges U, Luebke K, Raabe N (2005) klaR Analyzing German Business Cycles. In: Baier D, Decker R, Schmidt-Thieme L (eds) Data Anal. Decis. Support. Springer, Berlin, pp 335–343

Supplementary information

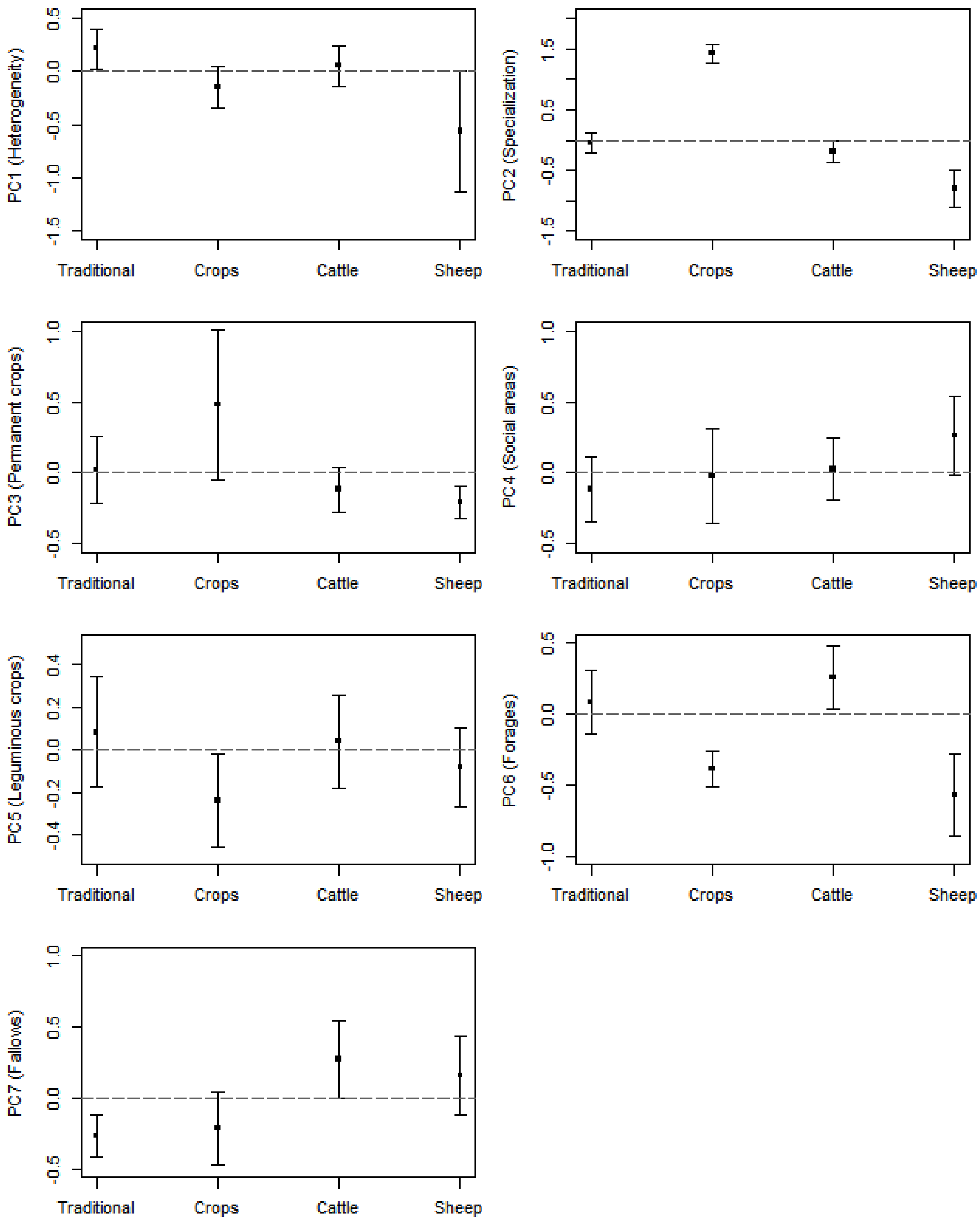

Figure S1. Error bars and 95% confidence intervals for the 7 retained varimax rotated principal components, by farming systems

Table S1. Confusion matrix comparing LDA predictions with the true farming system classifications

	Predicted					Correctly
Observed	Traditional	Crops	Cattle	Sheep	Total	assigned
Traditional	44	9	29	3	85	52%
Crops	3	29	1	0	33	88%
Cattle	31	5	47	8	91	52%
Sheep	8	0	12	12	32	38%

Chapter 4

An applied farming systems approach to infer conservation-relevant agricultural practices for agri-environment policy design

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An applied farming systems approach to infer conservation-relevant agricultural practices for agri-environment policy design

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Abstract

The Common Agricultural Policy (CAP) has shown difficulties in meeting its environmental objectives, namely in supporting biodiversity-friendly farming systems that remain under pressure to intensify or abandon. Proposals to address this have ranged from increasing the focus on highly tailored and targeted agri-environment schemes, to promoting broad-brush policies such as those recently implemented in the Greening of the CAP. Both options have been criticised due to questionable cost-effectiveness. Alternatives based on agri-environment policies oriented to support conservation-relevant farming systems have been suggested, but they have faced operational difficulties related primarily to obtaining the necessary data to define farming system typologies. Here we investigated whether a simplified approach based on a coarse farming system typology built from incomplete data on land-use and livestock, such as that available in CAP paying agencies, could be used to infer on a wider range of conservation-relevant farm management practices and, ultimately, to select the farming systems qualifying for premium payments. Based on data collected by a farm-survey on a High Nature Value farmland area in southern Portugal, we show that some farming systems are consistently associated with conservation-relevant practices related to the use of herbicides, stubble grazing, creation of wildlife plots and early cereal harvest. The traditional system involving the rotational production of cereals and sheep grazing on fallows showed the most favourable balance of land uses and farm management practices with positive conservation effects. Results underlined the potential of farming systems as a framework for developing agri-environment policy.

Keywords: Common Agricultural Policy; Agri-environment schemes; Farming systems; Biodiversity conservation; Farming practices

Introduction

A large part of Europe's biodiversity depends on ecosystems provided by traditional and low-input agricultural systems (Bignal & McCracken 1996; Kleijn et al. 2009). These systems are declining due to agriculture intensification or abandonment, which are driven by social, economic and political changes that have occurred during the last decades (Stoate et al. 2009; Latacz-Lohmann & Hodge 2003; Batáry et al. 2015; Lomba et al. 2015), with negative consequences for farmland biodiversity (Donald et al. 2002; Latacz-Lohmann & Hodge 2003; Reidsma et al. 2006). To support the sustainability of these conservation-relevant farming systems, agri-environment schemes (AES) have been implemented under the Common Agricultural Policy (CAP) of the European Union (EU) (European Commission, 2005). Although AES positive outcomes have been documented (e.g. Primdahl et al. 2003; Boatman et al. 2008), their effectiveness seems to be limited due to high transaction costs undermining farmers' voluntary participation, high administrative expenditure, poor environmental performance, and failure in safeguarding high conservation-value farming systems (e.g. Kleijn et al. 2001; Kleijn & Sutherland 2003; Siebert, Toogood & Knierim 2006; Defrancesco et al. 2008; Weber 2013; Pe'er et al. 2014; Ribeiro et al. 2014; Batáry et al. 2015).

There are essentially two contrasting views to the way agri-environment policies should be designed towards achieving greater effectiveness, generally matching CAP Pillars 1 and 2 (European Commission, 2013a; Poláková et al., 2011). On the one hand, Pillar 1 horizontal policy instruments such as the new Green Direct Payments and its Ecological Focus Areas, were recently proposed under the CAP reform 2014-2020 to safeguard and improve farmland biodiversity, and are now mandatory across the EU (European Commission, 2013a, 2013b). These broad-scope agri-environment policies offer the advantage of eliminating or significantly reducing transaction costs, but their effectiveness in protecting farmland biodiversity and agroecosystems may be limited, due to poorly specified conservation objectives and low effectiveness of mandatory commitments (Pe'er et al., 2014). They are also more attractive to Member States because they are usually fully funded by the CAP budget, without the need for co-financing.

In marked contrast with these broad-brush policy options, some authors advocate increasing the focus on Pillar 2 agri-environment schemes tailored to meet biodiversity conservation objectives at the local level (e.g., small regions or even individual farms), even at the expense of high administrative costs (e.g. Armsworth et al. 2012). Although the environmental benefits of such complex schemes are potentially high, their sustainability is uncertain due to the high transaction costs involved and the downward trend of the EU budget for Rural Development (Mettepenningen et al., 2011; Weber, 2013), in addition to requiring co-financing by member states. Alternatives are thus needed that conveniently address the trade-offs between scheme precision and administrative costs (Weber, 2015), providing more focused management prescriptions than those achieved by the horizontal policies, while reducing the costs required by local-level schemes (Poláková et al., 2011).

The farming system framework might provide a relatively simple and practical approach to deal with this conundrum, by allowing consideration of groups of farms with similar typology, thereby avoiding the need to tackle the multiple idiosyncrasies of a large number of individual farms (Poláková et al., 2011; Poux, 2013). Farms included in the same farming system type often have similar resource bases, enterprise patterns, livelihoods, and household restrictions (Darnhofer et al., 2012; Ferraton and Touzard, 2009; Keating and McCown, 2001) and are expected to show common responses to market and policy drivers (Ribeiro et al., 2014). Also, they tend to be associated with specific agricultural practices and land-use patterns to which biodiversity components respond (Bamière et al., 2011; Calvo-Iglesias et al., 2009; Carmona and Nahuelhual, 2010; Ribeiro et al., 2016). Therefore, it is likely that significant benefits could be achieved through an agri-environment policy based on a farming systems selection criteria, whereby only farms operating a farming system considered beneficial to the conservation objectives would qualify for environmental payments.

Although this idea is potentially attractive, it also has some practical problems that need to be duly considered. An important issue is that developing farming system typologies is a key prerequisite of this approach, but this may be difficult due to the need to identify groups of farms with similar agricultural, economic and sociological characteristics (e.g. Andersen et al., 2004; Pointereau et al., 2010). Typically, the farm-level data needed to develop the typology is obtained through a large number of direct enquiries to individual farmers, but this is costly and time consuming. However, previous studies have shown that operational farming system typologies can be developed from data readily available to CAP paying agencies, derived from farmers' annual subsidy applications (Ribeiro et al., 2014). It is unknown, however, whether such data can capture comparable typologies to that developed from detailed field farm surveys. In particular, it is uncertain whether these broad typologies are associated with particular sets of land-uses and agricultural management practices with conservation relevance, which are usually the target of AES schemes (Batáry et al., 2015).

In this paper we address these issues through a case study developed in cereal-steppe landscapes of southern Portugal, which are representative of a High Nature Value Farmland type that is critical for open farmland birds of European conservation concern (BirdLife International, 2004). In previous studies we used data from the Portuguese CAP paying agency to show that in this region there are at least five main farming systems, which are strongly constrained by biophysical, structural and policy drivers (Ribeiro et al., 2014), and occur under a wide range of landscape conditions (Ribeiro et al., 2016). In here we built on these previous studies, using enquiries to individual farmers to: (i) develop a farming system typology based on farm characterization variables analogous to those available in CAP paying agencies, essentially describing land-uses and livestock husbandry; and (ii) assess how management practices with conservation relevance are associated with particular farming systems. Results were then used to discuss the potential of the proposed farming systems approach as a framework for developing agri-environment policy.

Materials and methods

Study area

The study was conducted in southern Portugal (approx. lat.: 3º 42' N; long.: 8º 05' W), in an agricultural landscape dominated by rainfed, low-intensity farming systems dedicated to cereal production and livestock grazing. The landscape is characterized by a smooth relief, with altitudes ranging ca. 100-200 m above sea level. The climate is Mediterranean, with hot dry summers and moderately cold and rainy winters. The study area limit was adjusted to include 39 local administrative areas encompassing the Special Protection Area (SPA) of Castro Verde, designated under EU Directive 79/409/CEE (Birds Directive). This SPA covers about 80,000 hectares of farmland with high conservation value for several steppe birds of conservation concern, such as the great bustard (*Otis tarda* Linn.), little bustard (*Tetrax tetrax* Linn.) and lesser kestrel (*Falco naumanni* Fleischer) (BirdLife International 2004). The area benefits from an AES set up in 1995 to support traditional farming systems based on extensive rotation of cereals with long term fallows and grazing sheep (Santana et al., 2014), claimed as main providers of the steppe habitat that led to the classification of this area. In recent years there has been a decline of these traditional systems and their replacement by specialized livestock systems, a trend that is partially attributed to recent CAP reforms (Ribeiro et al., 2014) and whose conservation impacts are largely unknown (Reino et al., 2010).

Farm characterization

We conducted direct enquiries to farmers from a sample of 199 farms, between March and May of 2013. The sample to be enquired was selected randomly from the overall set of ca. 350 farmers renewing their annual application to agricultural subsidies at the main farmers' association in the region, located at Castro Verde (Associação de Agricultores do Campo Branco), where most farmers deliver their annual declarations. Each farmer was interviewed personally by a technician, using a structured questionnaire aiming to obtain information on land-uses, livestock husbandry and farm management practices. From this questionnaire we derived a set of variables required to establish the typology of farming systems and to characterise agricultural management (Table 4. 1).

Farm characterization variables focused on land-use and livestock husbandry (Table 4. 1), because these are widely available in governmental agencies paying CAP subsidies (Ribeiro et al., 2014), therefore enabling replication in other locations. Land-use variables included: dry cereals (CEREAL), mainly rainfed wheat and barley but also forage crops, which were considered together because many farmers often use cereal crop for hay production when it did not develop favourably to grain production; other annual crops (OTHCROP), such as sunflower, sorghum or legumes, whether irrigated or rainfed; permanent crops (PCROP), mainly olive groves; fallows (FALLOW), consisting of arable land that was not sown from one to four years to recover soil fertility, typically within a crop rotation scheme which is often used for stock grazing; pastures (PPAST), grazing areas not cultivated for five or more years, encompassing permanent pastures and long term fallows. Livestock production was characterised from variables reflecting stocking density (STOCKDENS) and the relative importance of cattle versus sheep (CRATIO).

Table 4. 1 - Summary statistics of the farm characterization variables, included in the farming system typology, and of the farming practices variables, extracted from the 199 farm questionnaires. See Table S1 in Supplementary Information for a description of related impacts on farmland birds. Agricultural base year from October 2012 to September 2013. % UAA = percentage of the utilized agricultural area. LU = livestock units.

Variable	Code	Description	Mean ± SD (Min-Max)
Farm characterization variables:			
Dry cereals	CEREAL	% UAA with dry cereals	19.9 ± 16.9 (0-100)
Other annual crops	OTHCROP	% UAA with other annual crops	1.0 ± 4.4 (0-55.6)
Permanent crops	PCROP	%UAA with permanent crops	0.8 ± 3.0 (0-20.0)
Fallows	FALLOW	% UAA with fallows	64.4 ± 26.1 (0-100)
Pastures	PPAST	% UAA with pastures	13.9 ± 25.1 (0-100)
Cattle ratio	CRATIO	% of cattle LU in total LU	31.5 ± 43.1 (0-100)
Livestock density	STOCKDENS	LU per hectare of fodder area	0.37 ± 0.44 (0-2)
Farming practices variables:			
Fertilizers	FERTILIZERS	% UAA fertilized	21.4 ± 17.7 (0-100)
Herbicides	HERBICIDES	% UAA treated with herbicides	6.6 ± 13.9 (0-82.6)
Direct drill	DIRCTDRILL	% UAA under direct drill	0.6 ± 5.6 (0-70.8)
Plough	PLOUGH	% UAA ploughed or disked	18.9 ± 18.4 (0-100)
Mechanical operations	N_MECOP	Number of mechanical operations in arable land (typical year)	3.48 ± 1.31 (0-7)
Irrigation	IRRIGAT	% UAA irrigated	1.1 ± 8.4 (0-100)
Stubs not grazed	STUBNGRAZ	% UAA with ungrazed stubbles	3.2 ± 12.4 (0-100)
Improved pastures	PASTPLUS	% UAA with improved pastures	6.4 ± 17.0 (0-100)
Wildlife plots	WLIFEPLOT	% UAA with wildlife food plots	0.8 ± 2.1 (0-17.8)
Conservation buffers	CONSBUFF	% UAA with conservation buffers	7.1 ± 14.6 (0-100)
Stockpiled forages	STOCKFOR	% UAA with stockpiled forages	8.7 ± 13.1 (0-69.6)
Crop rotation	ROTYEARS	Duration of crop rotation (years)	1.82 ± 2.09 (0-5.00)
Early harvest	EARLYHARV	% of the cereal area that is harvested before 31th May	42.0 ± 49.1 (0-100)
Wire fences	WIREFENCE	Meters of wire fences per hectare	31.3 ± 71.6 (0-593.1)

Agricultural management was characterised from 14 variables reflecting farming practices (Table 4. 1), which were selected due to their potential relevance for the conservation of open farmland birds (see Table S1 in Supplementary Information). Fertilizers (FERTILIZERS) and herbicides (HERBICIDES) were considered because they have general negative effects on farmland wildlife. Direct drill or no-till operations (DIRCTDRILL) were included due to the positive effects of such techniques, while ploughing and disking (PLOUGH) provide bare soil used by a number of farmland birds, though they also have negative impacts such as soil erosion and carbon loss. The number of mechanical operations involved in farmland cultivation (N_MECOP; e.g. ploughing, disking, seeding, mowing or harvesting) may cause negative impacts on birds (e.g. mortality or disturbance). Irrigation practices (IRRIGAT) are associated with agricultural intensification and major negative impacts on farmland birds. Ungrazed cereal or forage stubbles (STUBNGRAZ) have a positive effect on birds by providing food sources and avoiding the negative impacts of livestock grazing. Improved pastures (PASTPLUS) are associated with negative impacts emerging from high inputs and livestock densities. Wildlife plots (WLIFEPLOT) correspond mostly to crops sown specifically for the benefit of steppe birds, while conservation buffers (CONSBUFF) are strips of cover crops reducing negative impacts of agricultural practices and providing a variety of ecosystem services, including habitat for farmland birds. Stockpiled forages (STOCKFOR) are forage crops that are allowed to grow and accumulate for later grazing during periods of forage deficit, thereby avoiding the negative impacts of grazing such as trampling or nest destruction. Crop rotation (ROTYEARS) is favourable to farmland birds by promoting a complex mosaic of crops and fallows with different ages. The early harvest of crops (EARLYHARV) is associated with nest destruction and chick mortality. Fencing (WIREFENCE) is important as it can cause high bird mortality due to collisions.

Data analysis

The farming system typology was established considering the seven farm characterization variables reflecting land-uses and livestock husbandry (Table 4. 1). Based on these variables, farms were assigned to groups of farming systems by hierarchical cluster analysis using the Ward's method, which is an agglomerative approach that minimizes the within-group sum of squares, tending to form balanced-sized groups (Borcard et al., 2011; Legendre and Legendre, 1998). The outcome was assessed based on the analysis of the resulting dendrogram and with reference to similar typologies obtained by previous works in the same study area (Ribeiro et al., 2014). We then used one-way ANOVA to identify significant variation among farming systems in the variables characterising land-uses, livestock husbandry and farming practices, applying the Holm's correction on the p-values to account for multiple testing and ensure conservative results (Legendre and Legendre, 1998). To detect where the differences occurred between groups we used Games–Howell (GH) post hoc tests, which allows for heterogeneous variance and uneven sample sizes (Schmitzberger et al., 2005).

To assess whether farming systems could be discriminated in terms of farming practices, we conducted a linear discriminant analysis (LDA). A correlation matrix was computed prior to the analysis to investigate possible collinearity problems between these variables. A stepwise forward Wilk's lambda test procedure was conducted to select those variables that significantly ($p < 0.050$) contributed to group separation (Roever et al., 2014). The accuracy of the prediction was assessed through a confusion matrix, with leave-one-out cross-validation. Since sample sizes were uneven among the farming systems, we used Cohen's weighted kappa to correct for agreements occurring by chance between observed and predicted categories, considering both matches in the main diagonal and off diagonal (Titus et al., 1984). Inertia ellipses were added to the LDA plots to help visualizing the distribution of observations within each farming system along the axes, considering the default probability of ca. 66% corresponding to a one standard deviation length. All statistical analyses were implemented in R version 3.2.2 (R Development Core Team, 2015).

Results

Farming systems characterization

The 199 surveyed farms covered a total of 47,103 hectares of UAA. The dominant land-use was fallow (64.4%), followed by cereal crops (19.9%) (Table 4. 1). Permanent pastures and long-term fallows occupied almost the entire remaining area (13.9%). Other annual crops and permanent crops had a marginal expression (1.0% and 0.8% respectively). Sheep were dominant in total livestock (cattle ratio 31.5%), while overall livestock density was 0.37 LU/ha.

The examination of the dendrogram produced by the cluster analysis (Figure S1 in Supplementary Information) led to the selection of a cut-off point defining five groups, which were consistent with the previous typology defined in the same area using data from the Portuguese CAP paying agency (Ribeiro et al., 2014). The five clusters were named according to the distinctive characteristics of each group (Table 4. 2). The Sheep, Cattle and Intensive grazing systems were identified as livestock specialized systems, differing mainly on the type of livestock and grazing density. The Crops system was acknowledged as specialized in annual crop production, with no livestock. The Traditional system was identified as a mixed system, where annual crop production was complemented with low density livestock grazing, dominated by sheep. The average livestock density characterizing the Intensive grazing system was found abnormally above regional standards, probably indicating that these farms feed their animals by resorting to food purchase (hay, silage or concentrate feed) or to rented pastures or stubble crops in neighbouring farms. Consequently, the estimated value is artificial, not representing a true livestock density in the farm. For this reason, and because it occupied a very small area (< 2%), this cluster was dropped from further analysis, reducing the sample to 180 observations.

All seven farm characterization variables showed significant differences (Holm's adj. $p < 0.050$) among the four retained farming systems, except for OTHCROP and PCROP (Table 4. 2). The proportion of cereals was significantly higher in the Crops system, relatively to the Sheep and Cattle systems (CEREAL GH post hoc $p < 0.001$ and $p = 0.005$, respectively). The Sheep system had the highest proportion of fallows (FALLOW GH post hoc $p < 0.001$) and the lowest proportion was found on the Traditional system (FALLOW GH post hoc $p < 0.001$). The proportion of pastures and long term fallows was significantly higher in the Traditional system (PPAST GH post hoc $p < 0.001$). The Cattle ratio was significantly higher in the Cattle system (CRATIO GH post hoc $p < 0.001$). Stock density was significantly higher in the Sheep and Cattle systems, compared to the Traditional (STOCKDENS GH post hoc $p = 0.003$ and $p < 0.001$, respectively) and Crops systems (STOCKDENS GH post hoc $p < 0.001$ in both cases).

Table 4. 2 - Summary statistics (mean ± SD) of the five farming systems returned by the hierarchical cluster analysis. Variable definition in Table 4. 1. Variables with significant differences ($p < 0.050$) among groups are shaded grey.

Variable	Sheep	Cattle	Traditional	Crops	Intensive grazing[(a)]	ANOVA[(b)]
Number of farms	52	66	31	31	19	-
% UAA	17.2	52.8	12.1	10.8	1.8	-
Average farm size (ha)	155±142	415±288	184±182	164±119	44±60	-
CEREAL (%)	15.2±13.1	20.2±11.8	19.6±25.9	27.9±9.5	19.0±26.7	0.012
OTHCROP (%)	0.5±1.2	0.7±1.7	2.7±10.3	1.5±2.8	0.0±0.0	0.289
PCROP (%)	0.9±3.4	0.2±0.7	0.6±1.5	0.2±0.6	4.0±6.9	0.289
FALLOW (%)	81.8±13.4	66.7±20.4	29.2±23.7	69.6±10.5	57.9±35.6	<0.005
PPAST (%)	1.7±4.0	12.2±19.4	47.9±29.9	0.8±2.6	19.2±36.0	<0.005
CRATIO (%)	0.8±4.2	85.4±21.3	0	0	30.7±43.7	<0.005
STOCKDENS (LU/ha)	0.31±0.19	0.39±0.14	0.15±0.19	0.004±0.02	1.49±0.47	<0.005

(a) System not included in ANOVA
(b) Holm's adjusted p-values

Farming practices

The use of fertilizers (FERTILIZERS; 21.4% of UAA) and soil ploughing (PLOUGH; 18.9%) were the most widespread farming practices, while direct drill (DIRCTDRILL; 0.6%), the sowing of wildlife crops (WLIFEPLOT; 0.8%) and irrigation (IRRIGAT; 1.1%) were the least represented (Table 4. 1). The average number of mechanical operations in the arable land was 3.5 and the average crop rotation lasted for 1.8 years. About 42% of the cereal area on the average farm was harvested before the 31th of May.

The ANOVA tests with Holm's correction revealed statistically significant differences ($p < 0.050$) among farming systems in four out of 14 farming practices (Table 4. 3), namely in herbicide consumption (HERBICIDES), ungrazed stubbles (STUBNGRAZ), wildlife plots (WLIFEPLOT) and cereal area harvested before the 31th of May (EARLYHARV). Pairwise comparisons showed that herbicide consumption was significantly higher in the Crops system than in the Sheep and Cattle systems (HERBICIDES GH post hoc $p = 0.032$ and $p = 0.004$ respectively), and that the proportion of early harvested cereal was significantly higher in the Cattle system compared to all other systems (EARLYHARV GH post hoc $p < 0.001$). Variables STUBNGRAZ and WLIFEPLOT also showed a significant trend for higher values in the Crops system, although no significant differences were found in the pairwise Games–Howell post hoc tests, probably due to their weak representation in the surveyed farms.

Table 4. 3 - Summary statistics (mean ± SD) of the farming systems in terms of farming practices. Variable definition in Table 4. 1. Variables with significant differences ($p < 0.050$) among groups are shaded grey.

Variable	Sheep	Cattle	Traditional	Crops	ANOVA[a]
FERTILIZERS (%)	16.0±13.1	20.7±11.6	22.4±27.5	28.7±11.5	0.063
HERBICIDES (%)	5.7±12.0	4.0±8.1	7.1±20.3	15.2±16.3	0.022
DIRCTDRILL (%)	0.0±0.0	0.8±4.4	2.3±12.7	0.0±0.0	0.733
PLOUGH (%)	13.6±13.6	20.1±12.1	20.0±25.9	19.2±19.0	0.733
N_MECOP (count)	3.5±1.2	3.7±1.1	3.1±1.8	3.8±1.2	0.733
IRRIGAT (%)	0.0±0.0	0.4±1.9	2.8±10.9	0.0±0.0	0.273
STUBNGRAZ (%)	0.3±1.6	0.8±3.9	2.7±11.1	7.4±15.5	0.012
PASTPLUS (%)	3.7±13.2	9.3±17.9	3.9±18.0	4.5±14.3	0.733
WLIFEPLOT (%)	0.5±0.8	0.4±0.8	0.5±1.6	2.1±3.6	0.002
CONSBUFF (%)	6.7±12.5	6.2±12.2	7.3±19.8	11.7±16.6	0.733
STOCKFOR (%)	8.4±12.6	11.4±11.1	10.1±16.8	2.7±12.7	0.170
ROTYEARS (years)	1.71±1.95	2.06±2.20	1.52±2.03	2.68±2.20	0.733
EARLYHARV (%)	27.2±44.0	72.0±44.8	29.0±46.1	16.1±37.4	<0.005
WIRFENCE (m/ha)	44.2±87.5	21.0±26.0	12.5±27.9	11.2±21.7	0.090

(a) Holm's adjusted p-values

High correlation (>0.70) among farming practices was only found between FERTILIZERS and PLOUGH (0.84), and so we discarded the latter from subsequent analysis. In the linear discriminant analysis (LDA), variable selection using Wilk's lambda identified 6 variables contributing significantly ($p < 0.050$) to the separation of farming systems (Table 4. 4). The first linear discriminant axis (LD1) captured 68.1% of the between-group variance and it largely contrasted farming systems with either early harvest (Cattle) or high use of herbicides (Crops) (Table 4. 4, Figure 4. 1). The Traditional and Sheep systems had an intermediate position in this axis (Figure 4. 1). LD2 captured 21.3% of the variance and it was mainly related to the joint increase in early harvest and ungrazed stubbles (Table 4. 4). Farming systems were not clearly separated along this axis. LD3 captured the remaining 10.6% of the between group variance, mainly contrasting farms with either irrigation or fencing. The overall prediction accuracy of the LDA, with leave-one-out cross validation, reached 55.6% of success rate, corresponding to an overall Cohen's weighted kappa of 42.5% after correcting for chance agreements, which is still clearly above the random success probability of 25.0% within the four groups. Comparing the prediction accuracy on individual farming systems, the proportions of correct classifications were much higher for the Cattle (72.7%; Cohen's weighted kappa = 44.4%), Sheep (65.4%; Cohen's weighted kappa = 28.5%) and Crops systems (51.6%; Cohen's weighted kappa = 54.6%), than for the Traditional system (6.5%; Cohen's weighted kappa = 3.4%).

Table 4. 4 - Standardized canonical coefficients of the discriminant functions used to separate four farming systems based on practices relevant for bird conservation. Values > 0.50 are in bold. Variable definition in Table 4. 1.

Variable	LD1	LD2	LD3
EARLYHARV	**-0.75**	**-0.63**	0.19
HERBICIDES	**0.54**	-0.10	0.12
WLIFEPLOT	0.35	-0.33	0.13
IRRIGAT	-0.30	0.26	**-0.82**
STUBNGRAZ	0.30	**-0.58**	0.08
WIRFENCE	-0.02	0.43	**0.59**

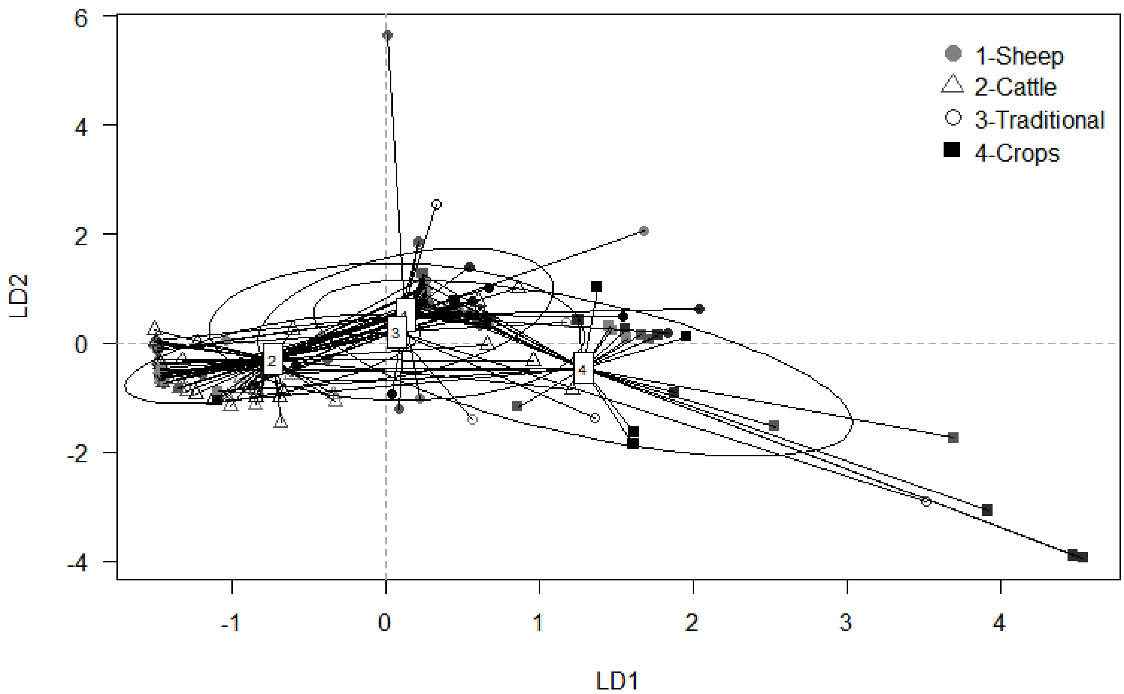

Figure 4. 1 - Scatterplot of the surveyed farms in the first two linear discriminant axis. The centroids and inertia ellipses are provided for each farming system.

Discussion

Our results suggest that a farming systems typology built on partial farm-level data, such as that available in CAP paying agencies, can be used to infer key farm management practices of high conservation relevance, and therefore provide relevant information for conservation managers and policy makers. The farming systems showed statistically significant differences in farm characterization variables (land-use and livestock patterns) and farming practices variables with high impact on biodiversity (herbicide consumption, ungrazed stubbles, wildlife plots and early harvest). These differences may help to identify which farming systems have higher conservation value and therefore potentially qualify for financial support through agri-environment schemes.

The Crops and the Cattle systems showed the strongest links with farming practices known to have direct negative effects on farmland birds in the study area (herbicide consumption and the date of cereal harvest, respectively). The Traditional system displayed a moderate stock density and a more varied landscape mosaic, with a balanced distribution of the UAA between cereals, pastures, fallows and small patches of other crops, providing landscape heterogeneity which is a valued feature for farmland biodiversity (Benton et al., 2003) and a landscape mosaic favourable to steppe bird conservation (Delgado and Moreira, 2000; Moreira et al., 2004; Reino et al., 2010). Accordingly, this system was identified as representing the traditional farming system that dominated the local landscape for decades and provided the high valued steppe habitat for several bird species of conservation concern (Ribeiro et al., 2014; Santana et al., 2014), and therefore it could qualify as a prime target for a local conservation program.

Our study also showed that it is possible that similar agricultural land-uses entail significant differences in farming practices key for conservation when held under distinct farming systems, which is a rather unexplored issue in the literature with potential implications for policy design. A case in point is that of rainfed cereals which are typically harvested in late June for grain production but, if weather conditions do not favour the growth of the crop, it can be harvested in late May for hay production. As a consequence, harvest may overlap the breading season of farmland bird species of conservation concern that nest in cereal fields, causing nest destruction and chick mortality (Beja et al., 2014). This farming practice was found particularly associated to the Cattle system which is dominant in the study area, thereby implying that most of the cereals area in the region (52%) are affected by this farming practice. Interactions between agriculture and conservation such as these are hardly detectable by conventional landscape analysis techniques often used in AES design (e.g. Batáry et al., 2015; Engel et al., 2008; Fahrig et al., 2011; Ribeiro et al., 2016), which would likely classify all dry cereal area indistinctively and assigned with equal conservation value.

The farming system typology built in this study was consistent with similar typologies produced by a previous study that used data from farmers' annual applications for CAP payments within the same study area (Ribeiro et al., 2014), suggesting that the framework can be easily reproduced on other locations with comparable conservation problems, using similar CAP paying data. Such a framework would potentially improve the cost-effectiveness of agri-environment policies since qualifying farms could be automatically selected on a yearly basis, avoiding the need for contracting long-term subscription agreements with individual farmers, as with many actual AES (Batáry et al., 2015), which would thus be encouraged to remain in the premium paid farming systems. In addition, it is conceivable that the framework would fit under CAP Pillar 1, thereby avoiding co-financing by Member States and relieving funds for Pillar 2 Rural Development policies. Moreover, aside from financing the preliminary studies to identify the farming systems to target, no significant extra-costs should emerge, as the required data to implement the framework is already available in the farm-level databases maintained by the CAP paying agencies in Member States, and also because significant cost-savings could be expected from the inherent simplification of the administrative burden and the easing of field inspections.

Overall, our results suggest that farming systems may be a cost-effective instrument for mitigating tension between development and biodiversity conservation, by providing a comprehensive tool to specify patterns of farmland use, livestock husbandry and farming practices, potentially useful to support the design of selective payment schemes targeted on sustainable farming systems, particularly on areas of special conservation concern. This conclusion tends to support those arguing that the CAP agri-environment policy could be planned based on simpler farm-level eligibility criteria (e.g. Beaufoy and Marsden, 2011; Poláková et al., 2011; Poux, 2013; Schmitzberger et al., 2005), such as operating pre-selected high conservation-value farming systems, as illustrated by the Traditional system in our study area.

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References

Andersen, E., Baldock, D., Bennett, H., Beaufoy, G., Bignal, E., Brouwer, F., Elbersen, B., Eiden, G., Godeschalk, F., Jones, G., McCracken, D., Nieuwenhuizen, W., Eupen, M. van, Hennekens, S., Zervas, G., 2004. Developing a High Nature Value Farming area indicator. HNV farming project - Final Report. Institute for European Environmental Policy.

Armsworth, P.R., Acs, S., Dallimer, M., Gaston, K.J., Hanley, N., Wilson, P., 2012. The cost of policy simplification in conservation incentive programs. Ecol. Lett. 15, 406–414. doi:10.1111/j.1461-0248.2012.01747.x

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. doi:10.1016/j.ecolecon.2010.12.014

Batáry, P., Dicks, L. V., Kleijn, D., Sutherland, W.J., 2015. The role of agri-environment schemes in conservation and environmental management. Conserv. Biol. 29, 1006–1016. doi:10.1111/cobi.12536

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

Beja, P., Schindler, S., Santana, J., Porto, M., Morgado, R., Moreira, F., Pita, R., Mira, A., Reino, L., 2014. Predators and livestock reduce bird nest survival in intensive Mediterranean farmland. Eur. J. Wildl. Res. 60, 249–258. doi:10.1007/s10344-013-0773-0

Benton, T.G., Vickery, J.A., Wilson, J.D., 2003. Farmland biodiversity: is habitat heterogeneity the key? Trends Ecol. Evol. 18, 182–188. doi:10.1016/S0169-5347(03)00011-9

Bignal, E.M., McCracken, D.I., 1996. Low-intensity systems in the conservation of the countryside. J. Appl. Ecol. 33, 413–424. doi:10.2307/2404804

BirdLife International, 2004. Birds in the European Union: a status assessment., Wageningen. ed, Wageningen, The Netherlands: BirdLife International. Wageningen, The Netherlands: BirdLife International.

Boatman, N., Ramwell, C., Parry, H., Bishop, J., Gaskell, P., Mills, J., Dwyer, J., 2008. A review of environmental benefits supplied by agri-environment schemes. Land Use Policy Group.

Borcard, D., Gillet, F., Legendre, P., 2011. Numerical Ecology with R, Use R! Springer, New York. doi:10.1007/978-1-4419-7976-6

Calvo-Iglesias, M.S., Fra-Paleo, U., Diaz-Varela, R.A., 2009. Changes in farming system and population as drivers of land cover and landscape dynamics: The case of enclosed and semi-openfield systems in Northern Galicia (Spain). Landsc. Urban Plan. 90, 168–177. doi:10.1016/j.landurbplan.2008.10.025

Carmona, A., Nahuelhual, L., 2010. Linking farming systems to landscape change: an empirical and spatially explicit study in southern Chile. Agric. Ecosyst. … 139, 40–50. doi:10.1016/j.agee.2010.06.015

Darnhofer, I., Gibbon, D., Dedieu, B., 2012. Farming Systems Research: an approach to inquiry, in: Darnhofer, I., Gibbon, D., Dedieu, B. (Eds.), Farming Systems Research into the 21st Century: The New Dynamic. Springer, pp. 1–26.

Defrancesco, E., Gatto, P., Runge, F., Trestini, S., 2008. Factors affecting farmers'participation in agri-environmental measures: A northern Italian perspective. J. Agric. Econ. 59, 114–131. doi:10.1111/j.1477-9552.2007.00134.x

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

Donald, P.F., Pisano, G., Rayment, M.D., Pain, D.J., 2002. The Common Agricultural Policy, EU enlargement and the conservation of Europe's farmland birds. Agric. Ecosyst. Environ. 89, 167–182.

Engel, S., Pagiola, S., Wunder, S., 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecol. Econ. 65, 663–674. doi:10.1016/j.ecolecon.2008.03.011

European Commission, 2013a. Overview of CAP Reform 2014-2020, Agricultural Policy Perspectives Brief N.o 5.

European Commission, 2013b. Regulation (EU) No 1307/2013 of the European Parliament and of the Council of 17 December 2013 establishing rules for direct payments to farmers under support schemes within the framework of the common agricultural policy and repealing Council Regulation. European Parliament and of the Council.

European Commission, 2005. Agri-environment Measures - Overview on General Principles, Types of Measures, and Application. Directorate General for Agriculture and Rural Development.

Fahrig, L., Baudry, J., Brotons, L., Burel, F.G., Crist, T.O., Fuller, R.J., Sirami, C., Siriwardena, G.M., Martin, J.-L., 2011. Functional landscape heterogeneity and animal biodiversity in agricultural landscapes. Ecol. Lett. 14, 101–12. doi:10.1111/j.1461-0248.2010.01559.x

Ferraton, N., Touzard, I., 2009. Comprendre l'agriculture familiale : diagnostic des systèmes de production, Quae. ed. Les presse agronomique de Gembloux.

Keating, B. a., McCown, R.L., 2001. Advances in farming systems analysis and intervention. Agric. Syst. 70, 555–579. doi:10.1016/S0308-521X(01)00059-2

Kleijn, D., Berendse, F., Smit, R., Gilissen, N., 2001. Agri-environment schemes do not effectively protect biodiversity in Dutch agricultural landscapes. Nature 413, 723–725. doi:10.1038/35099540

Kleijn, D., Kohler, F., Báldi, A., Batáry, P., Concepción, E.D., Clough, Y., Díaz, M., Gabriel, D., Holzschuh, A., Knop, E., Kovács, A., Marshall, E.J.P., Tscharntke, T., Verhulst, J., 2009. On the relationship between farmland biodiversity and land-use intensity in Europe. Proc. Biol. Sci. 276, 903–909. doi:10.1098/rspb.2008.1509

Kleijn, D., Sutherland, W.J., 2003. How effective are European agri-environment schemes in conserving and promoting biodiversity ? J. Appl. Ecol. 947–969.

Latacz-Lohmann, U., Hodge, I., 2003. European agri-environmental policy for the 21st century. Aust. J. Agric. Resour. Econ. 47:1, 123–139.

Legendre, P., Legendre, L., 1998. Numerical Ecology, 2nd Englis. ed, Developments in Environmental Modelling. Elsevier, Amsterdam.

Lomba, A., Alves, P., Jongman, R.H.G., Mccracken, D.I., 2015. Reconciling nature conservation and traditional farming practices : a spatially explicit framework to assess the extent of High Nature Value farmlands in the European countryside. Ecol. Evol. 1–14. doi:10.1002/ece3.1415

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. doi:10.1016/j.ecolecon.2010.10.007

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

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 (80-.). 344, 1090–1092. doi:10.1126/science.1253425

Pointereau, P., Doxa, A., Coulon, F., Jiguet, F., Paracchini, M.L., 2010. Analysis of spatial and temporal variations of High Nature Value farmland and links with changes in bird populations: a study on France, JRC Scientific and Technical Reports. Joint Research Centre of the European Commission.

Poláková, J., Tucker, G., Hart, K., Dwyer, J., Rayment, M., 2011. Addressing biodiversity and habitat preservation through measures applied under the Common Agricultural Policy, Report Prepared for DG Agriculture and Rural Development, Contract No. 30-CE-0388497/00-44. Institute for European Environmental Policy.

Poux, X., 2013. Biodiversity and agricultural systems in Europe: drivers and issues for the CAP reform. Institut du développement durable et des relations internationales.

Primdahl, J., Peco, B., Schramek, J., Andersen, E., Oñate, J.J.J., 2003. Environmental effects of agri-environmental schemes in Western Europe. J. Environ. Manage. 67, 129–138. doi:10.1016/S0301-4797(02)00192-5

R Development Core Team, 2015. R: A language and environment for statistical computing. [WWW Document]. R Found. Stat. Comput. URL http://www.r-project.org (accessed 12.26.11).

Reidsma, P., Tekelenburg, T., van den Berg, M., Alkemade, R., 2006. Impacts of land-use change on biodiversity: An assessment of agricultural biodiversity in the European Union. Agric. Ecosyst. Environ. 114, 86–102. doi:10.1016/j.agee.2005.11.026

Reino, L., Porto, M., Morgado, R., Moreira, F., Fabião, A., Santana, J., Delgado, A., Gordinho, L., Cal, J., Beja, P., 2010. Effects of changed grazing regimes and habitat fragmentation on Mediterranean grassland birds. Agric. Ecosyst. Environ. 138, 27–34. doi: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. doi:10.1016/j.agee.2013.11.002

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

Roever, C., Raabe, N., Luebke, K., Ligges, U., Szepannek, G., Zentgraf, M., 2014. Package “klaR”: Classification and visualization [WWW Document]. URL http://www.statistik.tu-dortmund.de

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. doi:10.1111/conl.12077

Schmitzberger, I., Wrbka, T., Steurer, B., Aschenbrenner, G., Peterseil, J., Zechmeister, H.G., 2005. How farming styles influence biodiversity maintenance in Austrian agricultural landscapes. Agric. Ecosyst. Environ. 108, 274–290. doi:10.1016/j.agee.2005.02.009

Siebert, R., Toogood, M., Knierim, A., 2006. Factors affecting european farmers’ participation in biodiversity policies. Sociol. Ruralis 46, 318–340. doi:10.1111/j.1467-9523.2006.00420.x

Stoate, C., Báldi, 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. J. Environ. Manage. 91, 22–46. doi:10.1016/j.jenvman.2009.07.005

Stoate, C., Boatman, N.D., Borralho, R.J., Carvalho, C.R., de Snoo, G.R., Eden, P., 2001. Ecological impacts of arable intensification in Europe. J. Environ. Manage. 63, 337–365. doi:10.1006/jema.2001.0473

Titus, K., Mosher, J.A., Williams, B.K., 1984. Chance-corrected Classification for Use in Discriminant Analysis: Ecological Applications. Am. Midl. Nat. 111, 1. doi:10.2307/2425535

Weber, A., 2015. Implementing EU co-financed agri-environmental schemes: Effects on administrative transaction costs in a regional grassland extensification scheme. Land use policy 42, 183–193. doi:10.1016/j.landusepol.2014.07.019

Weber, A., 2013. How are public transaction costs in regional agri-environmental scheme delivery influenced by EU regulations? J. Environ. Plan. Manag. 1–23. doi:10.1080/09640568.2013.776950

Supplementary information

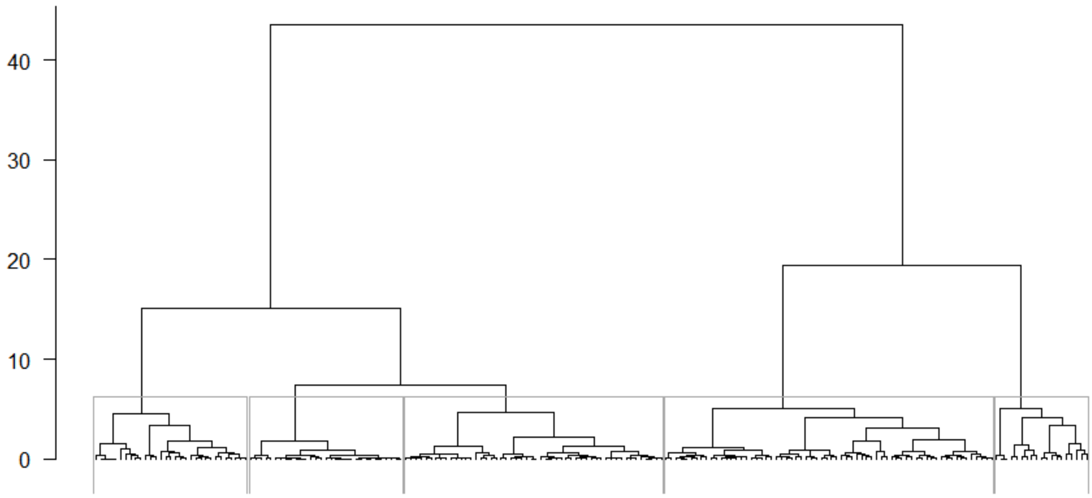

Figure S1. Dendrogram derived from hierarchical cluster analysis using the Ward's method to define the farming systems typology. Grey lines identify the cut-off point for the five clusters solution.

Table S1 - Main farming practices of Mediterranean farming systems (crops, livestock and mixed systems) and their effects on farmland birds.

Farming practice (signal of impact)	Impacts description	References
Inorganic fertilization (-)	Inorganic fertilizers, in particular nitrogen fertilizers, promote greater growth of some plants (e.g. grass) at the expense of others (e.g. many broad-leaved plant species). This results in reduced plant diversity and modified plant composition of grassland, which can influence bird diversity. High levels of fertilizer application can cause the densification of crop structure, preventing its use by birds as nesting or foraging areas.	Stoate *et al.* 2001; Newton 2004
Pesticide usage (-)	Pesticide misuse (including fungicides, insecticides and herbicides) can have direct negative effects on birds, causing reproductive failures and enhanced mortality. Indirectly, they can also risk biodiversity by operating through the food chain and by reducing the availability of food resources, nesting sites and cover areas. The negative side-effects of pesticide usage can be worsened by local conditions (e.g. soil, topography, drainage) and season. For example, spring and summer use of herbicides can reduce broad-leaved weed abundance, compromising the supply of insects and seeds for birds to feed.	McLaughlin & Mineau 1995; Stoate *et al.* 2001; Boatman *et al.* 2004; Kirk, Lindsay & Brook 2011
No-till / direct drilling (+)	Minimal soil disturbance was related to increased abundance of earthworms and weed seeds near soil surface, providing food for birds, and to earlier nesting of farmland bird species, prolonging their nesting season.	Huggins & Reganold 2008; Stoate *et al.* 2009

Farming practice (signal of impact)	Impacts description	References
Ploughing (-/+)	Ploughing is considered a highly destructive mechanical operation, affecting soil invertebrate populations by physical destruction, microbial biomass loss and causing direct plant desiccation, which may result in a loss of biodiversity as biomass and biodiversity are often positively correlated. However, ploughed fields (together with other bare soil areas) have also been described as favourable habitats for some steppe bird species of conservation concern.	Stoate *et al.* 2001, 2009; Moreira *et al.* 2007; Durán Zuazo & Rodríguez Pleguezuelo 2008; Leitão, Moreira & Sborne 2010
Mechanical operations (-)	Agricultural mechanical operations such as ploughing, disking, seeding, mowing and harvesting can cause disturbance and farmland bird mortality, although data on mortality rates are scarce or non-existent.	Tews, Bert & Mineau 2013
Irrigation (-)	Frequently associated with agriculture intensification, with many negative effects on the environment, including the simplification of landscape, loss of crop rotation and fallows, increase in input applications including fertilizers, pesticides and machinery operations and passes. Intensive irrigated systems are therefore often associated with high pesticide, heavy metals and nutrients concentration levels in soil, which can lead to diffuse contamination on other ecosystems. In European Mediterranean regions these effects can be aggravated by the fact that irrigation is higher in spring-summer crops – coinciding with the breeding season for many farmland bird species – when rainfall is minimal, and because of the negative impacts of irrigation infrastructures for water management (dams, water reservoirs and channels) that alter the hydrological cycle.	Stoate *et al.* 2001; Creamer *et al.* 2010
Stubble burning (-)	In addition to many other negative environmental effects, stubble burning can be damaging for birds by reducing insect food sources, particularly for ground nesting birds when fire occurs during the nesting period.	Andreae 1991; Crofts 1999; Middleton, Holsten & van Diggelen 2006

Farming practice (signal of impact)	Impacts description	References
Improved pastures (+/-)	Promotes the use of inorganic fertilizers and the sowing of pasture species (grasses in combination with legumes), occasionally complemented with soil conditioners (e.g. liming) to raise the productivity of native pastures and its protein and energy values. High rates of fertilizer inputs (especially Nitrogen) have been associated with declines in overall bird species diversity. The denser and more uniform structure of the sward in improved pastures may reduce its suitability for ground-nesting and ground-feeding birds, while allowing for increased stock densities which result in greater destruction of eggs and chicks.	Newton 2004; Wilson, Whittingham & Bradbury 2005; Pointereau, Doxa & Paracchini 2010
Conservation buffers (+)	Strips of vegetation maintained to influence ecological processes. Provide food, nesting, shelter and escape cover. Promote bird abundance and diversity, along with nest density. Buffers planted with native grasses and legumes provide significant habitat for grassland birds.	United States Department of Agriculture 2000; Lovell & Sullivan 2006; Bentrup 2008
Stockpiled forage (+)	Stockpiled forage is forage that is allowed to grow and accumulate for later grazing, to cover for periods of forage deficit. Literature on the benefits of stockpiling forages for biodiversity is scarce or non-existent. But since stockpiling forage is accomplished by removing grazing animals from a pasture for a period of time during the growing season, we can infer that, when it matches the breeding season of birds, it can be potentially beneficial for farmland birds, particularly ground-nesting species, because it removes all negative impacts of livestock grazing above mentioned.	Lacefield *et al.* 2006; Barnhart 2010

Farming practice (signal of impact)	Impacts description	References
Livestock grazing (-/+)	Ecological effects of livestock grazing usually comprise i) defoliation, referring to the removal by animals of some or all of the above ground parts of plants, more or less selectively; ii) trampling, which affects both the structure and the botanical composition of the grassland; and iii) manuring, concerning the nutrient recycling in the grassland ecosystem. The three processes depend on a number of factors, including livestock type and density, soil type, topography, grazing period, season of year and rainfall. Sheep and cattle grazing differ greatly: cattle usually produce a structurally diverse sward of benefit to invertebrates, while sheep typically graze swards to a uniformly low height. While extensive grasslands are often considered beneficial to carbon sequestration and to biodiversity, overgrazing can result in a permanent reduction in biodiversity. Increasing livestock density changes habitat structure, sward height and composition, and reduces plant flowering and seeding, together with insects' abundance, many of which are eaten by birds. It also leads to greater disturbance of nesting birds and increased trampling of nest contents, causing egg and chick destruction. Livestock grazing can be beneficial to fen conservation, preventing invasion of shrubs and trees, but in areas like the Mediterranean open oak woodlands cattle grazing of the undercover can reduce the regeneration of trees.	Acharya et al., 2012; Allison and Grayson, 1999; Middleton et al., 2006; Newton, 2004; Pointereau et al., 2010; Poláková et al., 2011; Stoate et al., 2001
Crop rotation (+)	Refers to the sequentially growing of different crops in the same field. Indirectly benefits biodiversity from the reduced dependence on chemical inputs and promotes habitat diversity and landscape heterogeneity, benefiting several species of farmland birds.	McLaughlin & Mineau 1995; Stoate *et al.* 2001; Henderson *et al.* 2009; Christensen *et al.* 2012; Moreira *et al.* 2012

Farming practice (signal of impact)	Impacts description	References
Early mowing/ /cutting (-)	Mowing and cutting have been regarded as more appropriate to manage the biomass of fens and meadows for biodiversity, as compared to livestock grazing. But earlier harvesting dates, like those associated with cutting cereals for hay production instead of grain (or silage instead of hay), may cause a timing overlap between these operations and the breeding season, causing great destruction of eggs and chicks of ground nesting species.	Jefferson 1999; Newton 2004; Middleton et al. 2006; Butler et al. 2010; Swallow et al. 2012; Beja et al. 2014
Stock fencing (-)	Livestock wire and mesh fences are known to cause direct bird mortality as a result of collisions. Fences are also blamed to cause landscape fragmentation, habitat loss and induce avoidance behaviours in farmland birds.	Boone & Hobbs 2004; Hayward & Kerley 2009; Campbell & Ryder 2010; Hovick *et al.* 2014

Supplementary References

Acharya, B.S., Rasmussen, J., Eriksen, J., 2012. Grassland carbon sequestration and emissions following cultivation in a mixed crop rotation. Agric. Ecosyst. Environ. 153, 33–39. doi:10.1016/j.agee.2012.03.001

Allison, C., Grayson, B., 1999. Grazing, in: Crofts, A., Jefferson, R. (Eds.), The Lowland Grassland Management Handbook. pp. 1–84.

Andreae, M., 1991. Biomass burning: its history, use, and distribution and its impact on environmental quality and global climate, in: Levine, J.S. (Ed.), Global Biomass Burning: Atmospheric, Climatic and Biospheric Implications. MIT Press, Cambridge, Massachusetts, pp. 3–21.

Barnhart, S.K., 2010. Stockpiled forages: a way to extend the grazing season. Iowa State University.

Beja, P., Schindler, S., Santana, J., Porto, M., Morgado, R., Moreira, F., Pita, R., Mira, A., Reino, L., 2014. Predators and livestock reduce bird nest survival in intensive Mediterranean farmland. Eur. J. Wildl. Res. 60, 249–258. doi:10.1007/s10344-013-0773-0

Bentrup, G., 2008. Conservation buffers - Design Guidelines for Buffers, Corridors, and Greenways, General Technical Report SRS-109. Ashville, NC.

Boatman, N.D., Brickle, N.W., Hart, J.D., Milsom, T.I.M.P., Morris, A.J., Murray, A.W.A., Murray, K.A., Robertson, P.A., 2004. Evidence for the indirect effects of pesticides on farmland birds. Ibis (Lond. 1859). 146, 131–143.

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

Butler, S.J., Boccaccio, L., Gregory, R.D., Vorisek, P., Norris, K., 2010. Quantifying the impact of land-use change to European farmland bird populations. Agric. Ecosyst. Environ. 137, 348–357. doi:10.1016/j.agee.2010.03.005

Campbell, R.W., Ryder, G.R., 2010. Migrant Western Sandpipers Entangled on Barbed Wire Fences at Boundary Bay , British Columbia 7, 284–288.

Christensen, H., Becheva, S., Meredith, S., Ulmer, K., 2012. Crop Rotation - Benefiting farmers, the environment and the economy. Pesticide Action Network Europe, Friends of the Earth Europe, IFOAM EU Group, APRODEV.

Creamer, R.E., Brennan, F., Fenton, O., Healy, M.G., Lalor, S.T.J., Lanigan, G.J., Regan, J.T., Griffiths, B.S., 2010. Implications of the proposed Soil Framework Directive on agricultural systems in Atlantic Europe - a review. Soil Use Manag. 26, 198–211. doi:10.1111/j.1475-2743.2010.00288.x

Crofts, A., 1999. Burning, in: Crofts, A., Jefferson, R. (Eds.), The Lowland Grassland Management Handbook. English Nature/The Wildlife Trusts, pp. 1–11.

Durán Zuazo, V.H., Rodríguez Pleguezuelo, C.R., 2008. Soil-erosion and runoff prevention by plant covers. A review. Agron. Sustain. Dev. 28, 65–86. doi:10.1051/agro:2007062

Hayward, M.W., Kerley, G.I.H., 2009. Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes? Biol. Conserv. 142, 1–13. doi:10.1016/j.biocon.2008.09.022

Henderson, I.G., Ravenscroft, N., Smith, G., Holloway, S., 2009. Effects of crop diversification and low pesticide inputs on bird populations on arable land. Agric. Ecosyst. Environ. 129, 149–156. doi:10.1016/j.agee.2008.08.014

Hovick, T.J., Elmore, R.D., Dahlgren, D.K., Fuhlendorf, S.D., Engle, D.M., 2014. Evidence of negative effects of anthropogenic structures on wildlife: a review of grouse survival and behaviour. J. Appl. Ecol. 51, 1680–1689. doi:10.1111/1365-2664.12331

Huggins, B.D.R., Reganold, J.P., 2008. No-Till: the Quiet Revolution. Sci. Am. 70–77.

Jefferson, R.G., 1999. Mowing and cutting, in: Crofts, A., Jefferson, R.G. (Eds.), The Lowland Grassland Management Handbook. pp. 1–27.

Kirk, D.A., Lindsay, K.E., Brook, R.W., 2011. Risk of Agricultural Practices and Habitat Change to Farmland Birds. Avian Conserv. Ecol. 6. doi:10.5751/ACE-00446-060105

Lacefield, G., Smith, R., Henning, J., Johns, J., Burris, R., 2006. Stockpiling for Fall and Winter Pasture. Cooperative Extension Service.

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.

Lovell, S.T., Sullivan, W.C., 2006. Environmental benefits of conservation buffers in the United States: Evidence, promise, and open questions. Agric. Ecosyst. Environ. 112, 249–260. doi:10.1016/j.agee.2005.08.002

McLaughlin, A., Mineau, P., 1995. The impact of agricultural practices on biodiversity. Agric. Ecosyst. Environ. 55, 201–212. doi:10.1016/0167-8809(95)00609-V

Middleton, B. a., Holsten, B., van Diggelen, R., 2006. Biodiversity management of fens and fen meadows by grazing, cutting and burning. Appl. Veg. Sci. 9, 307–316. doi:10.1658/1402-2001(2006)9[307:bmofaf]2.0.co;2

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

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., 2012. 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. doi:10.1371/journal.pone.0038876

Newton, I., 2004. The recent declines of farmland bird populations in Britain: An appraisal of causal factors and conservation actions. Ibis (Lond. 1859). 146, 579–600. doi:10.1111/j.1474-919X.2004.00375.x

Pointereau, P., Doxa, A., Coulon, F., Jiguet, F., Paracchini, M.L., 2010. Analysis of spatial and temporal variations of High Nature Value farmland and links with changes in bird populations: a study on France, JRC Scientific and Technical Reports. Joint Research Centre of the European Commission.

Poláková, J., Tucker, G., Hart, K., Dwyer, J., Rayment, M., 2011. Addressing biodiversity and habitat preservation through measures applied under the Common Agricultural Policy, Report Prepared for DG Agriculture and Rural Development, Contract No. 30-CE-0388497/00-44. Institute for European Environmental Policy.

Stoate, C., Báldi, 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. J. Environ. Manage. 91, 22–46. doi:10.1016/j.jenvman.2009.07.005

Stoate, C., Boatman, N.D., Borralho, R.J., Carvalho, C.R., de Snoo, G.R., Eden, P., 2001. Ecological impacts of arable intensification in Europe. J. Environ. Manage. 63, 337–365. doi:10.1006/jema.2001.0473

Swallow, S.K., Anderson, C.M., Uchida, E., 2012. The Bobolink Project: Selling Public Goods from Ecosystem Services Using Provision Point Mechanisms. Zwick Cent. Food Resour. Policy Work. Pap. Ser.

Tews, J., Bert, D.G., Mineau, P., 2013. Estimated Mortality of Selected Migratory Bird Species from Mowing and Other Mechanical Operations in Canadian Agriculture Estimation de la mortalité aviaire par le fauchage et d’autres opérations mécanisées utilisées en agriculture au Canada. Avian Conserv. Ecol. 8, 8. doi:10.5751/ACE-00559-080208

United States Department of Agriculture, 2000. Conservation buffers to reduce pesticide losses. doi:http://www.in.nrcs.usda.gov/technical/agronomy/newconbuf.pdf

Wilson, J.D., Whittingham, M.J., Bradbury, R.B., 2005. The management of crop structure: A general approach to reversing the impacts of agricultural intensification on birds? Ibis (Lond. 1859). 147, 453–463. doi:10.1111/j.1474-919x.2005.00440.x

Chapter 5

A spatially explicit choice model to assess the impact of conservation policy on High Nature Value farming systems

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A spatially explicit choice model to assess the impact of conservation policy on High Nature Value farming systems

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Abstract

High Nature Value (HNV) farming is declining in Europe due to pressures to agriculture intensification or abandonment, compromising biodiversity conservation. Agri-environment policy implemented under the Common Agricultural Policy (CAP) have had difficulties in addressing this problem, leading to the recent emergence of policy proposals that directly target the sustainability of HNV farming systems. However, there is a lack of knowledge about the economics of these HNV farming systems, hampering their practical use in policy design. In this paper, we approach these problems seeking to understand what factors govern the choice of the farming system, with an emphasis on the role that economic incentives play in this decision. Based on a case study of an HNV farmland area in southern Portugal with high conservation value for steppe birds, we used data from the EU Integrated Administration and Control System (IACS) combined with spatially explicit data from the Parcel Identification System (LPIS) covering years 2000 to 2010, to derive a farming systems' typology. Using logistic regression choice models with latent class and panel data, we related the observed farming system dynamics to the biophysical and structural characteristics of farms, and with the evolution of the policy and economic environment, particularly the reform of the CAP in 2003. The model was used to simulate policy and market scenarios on the choice of farming system, allowing an innovative approach to derive a supply curve for biodiversity conservation services bounded by a 95% confidence envelope derived from a Monte Carlo experiment, and to assess the impacts of market and policy scenarios on environmental indicators at the landscape scale. We concluded that economic incentives can easily outcome biophysical and structural constraints, and demonstrated the usefulness of the farming systems approach and the suitability of the framework to support the design of innovative, cost-effective agri-environment policy.

Keywords: Discrete choice modelling; Common Agricultural Policy; Agri-environment schemes; Scenario assessment; Farming systems; HNV farmland

Introduction

The concept of High Nature Value (HNV) farmland was introduced in the early 1990s to demonstrate how the conservation of European biodiversity depends on the maintenance of traditional and low-input farming systems (Baldock et al., 1993; Beaufoy et al., 1994). Despite their importance, HNV farming systems are declining due to social, economic and policy pressures for either the intensification or the abandonment of agriculture (Stoate et al., 2009; Oppermann and Paracchini, 2012). This is compromising the European Union (EU) objectives for biodiversity conservation, such as those established under the EU Biodiversity Strategy to 2020 (European Commission, 2011). Furthermore, this reveals a failure of the Common Agricultural Policy (CAP) in safeguarding farmland biodiversity (Henle et al., 2008; Pe'er et al., 2014).

In a recent report for the European Commission, Keenleyside et al. (2014a) called for more effective ways to support HNV farmland under the CAP, suggesting an approach based on payments to farms in HNV areas or operating HNV farming systems. Operational challenges related to lack of data or indicators to identify HNV farmland or farming systems are however recognized (Keenleyside et al., 2014a). Suggestions to increase focus on policies supporting HNV farming characteristics have also been made elsewhere (e.g. Beaufoy and Marsden, 2011; BirdLife International, 2011; Poláková et al., 2011; Weber, 2015). However, there is a clear lack of research on the economic aspects of HNV farming, to establish the level and type of funding necessary for its sustainability (Keenleyside et al., 2014a). Indeed, several studies addressing the cost-effectiveness of agri-environment policies have been published in recent years, but few have focused on the economic sustainability of HNV farmland or farming systems. Most analysis aimed at estimating the costs for farmers to participate in agri-environment schemes (AES) (e.g. Oñate et al., 2007; Bamière et al., 2011; Barraquand and Martinet, 2011; Wätzold et al., 2015), or to assess the farmer's willingness to accept a compensatory payment for management options benefiting the environment (e.g. Ruto and Garrod, 2009; Christensen et al., 2011; Buckley et al., 2012). 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 in HNV farmland (Keenleyside et al., 2014b). However, the payment calculations are based on the same principles set out in the EU Regulations, which provide compensations for additional costs or income forgone 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 (e.g. 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 (e.g. Calvo-Iglesias et al., 2009; Carmona and Nahuelhual, 2010; 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). One possible illustration would be using the concept of greening the Pillar 1 of the CAP by granting a top-up payment to farmers who operate within the boundaries of the farming systems associated with HNV farmland in a specific region (Ribeiro et al., 2016a). The implementation of this concept requires: (1) identifying these HNV farming systems for different regions across the EU; (2) calculating the required payment level to ensure sufficient uptake by farmers. Although the underlying idea of an agri-environment policy aimed at supporting HNV farming systems may sound interesting, the factors driving the farmer's decision in choosing the farming system are not well known, 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, aiming at: 1) investigating the factors that influence the choice of farming system by farmers; 2) developing a framework to simulate, on a spatially explicit basis, the effects of different policy and market scenarios on HNV farmland. Using logistic regression based on farm-level spatio-temporal data, we estimated a discrete choice model to analyse the farmer's decision when opting for a particular farming system, subject to biophysical, structural, policy and economic drivers and constraints. The framework enabled the outline of empirical supply curves for conservation services (Jack et al., 2009; Lewis and Wu, 2014; Santos et al., 2008), relating levels of payment per hectare paid to farmers operating HNV farming systems with the amount of farmland managed under such systems. Additionally, the spatial nature of the model was evaluated for its potential to provide landscape-scale indicators of environmental quality in the study area under different policy scenarios. These approaches are largely new to the literature and provide new perspectives in assessing the impact of policy changes in HNV farmland.

Methods

Study area

The study focused on an extensive HNV farmland area in the south of Portugal, covering ca. 180,000 hectares (Figure 5. 1). The area is characterized by open fields, smooth relief and typical Mediterranean climate, with hot dry summers and moderately rainy cold winters. For decades the landscape was dominated by rainfed cereal crops in rotation with long-term fallows grazed by sheep (Delgado and Moreira, 2000), which provides a cereal-steppe habitat of great importance for several species of farmland steppe birds of conservation concern (BirdLife International, 2004; Oñate et al., 2007). The area encompasses the Special Protection Area (SPA) of Castro Verde, classified under the EU Directive 79/409/CEE (Birds Directive), where an AES is operating since 1995 to support the low-intensive cereal-fallow-sheep farming system. During the study period (2000-2010), there were major farming system changes attributed to the 2003 CAP reform and the resulting decoupling of direct payments, concomitant with the Portuguese authorities' decision to keep a direct payment on suckler cows and sheep (Ribeiro et al., 2014). Changes included a decline in the low-intensive rotational system, and its replacement by specialized livestock systems (cattle and sheep), with possible negative impacts on conservation (Reino et al., 2010; Ribeiro et al., 2014; Santana et al. 2014).

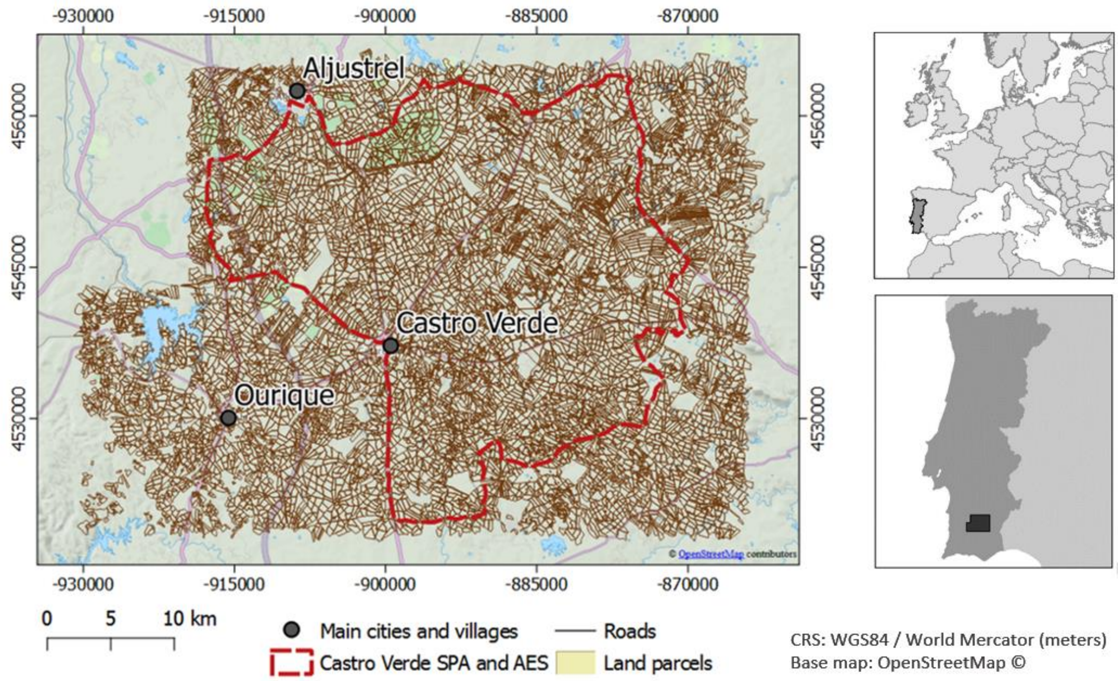

Figure 5. 1 - Location of the study area in the south of Portugal, showing the farm-parcel structure and the Special Protection Area (SPA) of Castro Verde where an agri-environment scheme (AES) is in operation since 1995.

Farming systems identification

We identified farming systems in the study area using farm-level data from the EU Integrated Administration and Control System (IACS), which are collected on a yearly basis from the farmer declarations when applying for CAP payments, 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). Data were provided by the Portuguese national CAP paying agency, for every year from 2000 to 2010. The typology of farming systems was derived by cluster analysis performed on farm characterization variables describing land-use and livestock extracted from these data (methodological details in Annex A of the Supplementary Information). Each farm in each year was assigned to a farming system, thereby providing information on farming system transitions over time.

Biophysical and structural drivers and constraints of farming system choice

Each farm was characterised using biophysical (soil quality, terrain slope and average annual rainfall) and structural features (farm size, farm spatial fragmentation, and oak woodlands) (Table 5. 1), expected to influence farming system choice (Falconer, 2000; National Research Council, 2010; Keenleyside et al., 2014b; Ribeiro et al., 2014). These variables varied spatially but were constant over time within the study period. Soil quality (SOIL) represented the proportion of soils with good aptitude for agricultural use, and it was computed by intersecting a digital map of soil capacity for agriculture (SROA/CNROA, 2012) with the farm parcels layer derived from the LPIS. Terrain slope (SLOPE) indicated the average slope at each farm, extracted from a digital elevation model. Annual rainfall (RAIN) was estimated as the average annual precipitation in each farm, computed by intersection with a raster layer of interpolated values from local weather stations (INMG, 1991) (Figure S1, Annex B, in Supplementary Information). Farm size (UAA) was the total area of all parcels declared by the same farmer. Farm spatial fragmentation was quantified with the Januszewski index (JANUS), calculated from LPIS data for each farm by dividing the squared root of its total area by the sum of the squared roots of all land blocks, where a land block is a set of contiguous parcels belonging to the farm. The index ranges from 0 to 1, with higher values indicating a high degree of farmland consolidation (Januszewski, 1968; Simion, 2008), which has been associated with higher agricultural productivity (Chukwukere Austin et al., 2012). Oak woodlands (OAK) are areas occupied by cork oak *Quercus suber* and holm oak *Q. rotundifolia*, which constrain the range of farm management decisions because cutting these oaks is strongly restricted by Portuguese law. The proportion of oak woodland in each farm was estimated by intersecting the farms layer with a digital land cover map (IGP, 2012) (Figure S1 in Annex B in Supplementary Information). We have also considered policy constraints to farmer’s decisions, by including a dummy variable (SPA) indicating whether the farm was inside or outside the SPA of Castro Verde, and thus whether there were legal restrictions to the expansion of irrigation infrastructures and of permanent crops (Santana et al., 2014).

Table 5. 1 - Summary statistics for the biophysical, structural and policy farm characterization variables (n = 7883)

Variables	Code	Mean ± SD (Min-Max)
Biophysical		
Proportion of good quality soils for agricultural use in the UAA (%)	SOIL	0.15 ± 0.26 (0.00-1.00)
Average slope in the UAA (degrees)	SLOPE	3.07 ± 0.89 (1.29-12.60)
Average annual rainfall (10^3 mm)	RAIN	0.55 ± 0.06 (0.43-0.68)
Structural		
Utilized agricultural area (km^2)	UAA	1.78 ± 2.13 (0.10-22.07)
Proportion of oak woodlands in the UAA (%)	OAK	0.26 ± 0.30 (0.00-1.00)
Januszewski index (adimensional)	JANUS	0.73 ± 0.22 (0.23-1.00)
Policy		
Location regarding the Special Protection Area of Castro Verde (In=1; Out=0)	SPA	0.45 ± 0.50 (0.00-1.00)

Policy and market drivers

The effects of market and policy drivers on farmer decisions was considered by including in modelling the gross income ratio for each farming system and each study year t (GIR^t), which was quantified as the unit gross income of a reference farming system GI_{RFS}^t in relation to that of the alternative farming system GI_{AFS}^t:

$$GIR^t \ = \ GI_{RFS}^t / GI_{AFS}^t$$

This variable was selected, because we expected that farm management changes in the study area were mostly driven by temporal variation in gross income from sale revenues and direct subsidies, considering that the cost structure was not as subject to annual fluctuations, which is an assumption roughly inspired by the “standard output” approach of EU Regulation 1242/2008. Because different activities 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. However, a higher *GIR* means higher relative profitability of the RFS, if the income/cost ratios of each activity do not vary significantly. The reference farming system was selected based on prior knowledge about the HNV farming system in our region, which was named as the “Traditional” farming system in previous studies (Ribeiro et al. 2014, 2016a, 2016b).

A dynamic variable reflecting the difference between GIR^t and GIR^{t-n} (*GIRdif*) was also included in modelling, to account for the farmers' perception on the recent trend in the relative profitability of the farming systems. Positive values in *GIRdif* indicate an upward trend in the profitability of the RFS, compared to alternative systems. This variable assigns a temporal dimension to the model, allowing it to be used in predictions for scenarios of market price or policy changes. The most adequate time lag, *n*, was examined by an adjustment-time analysis of reactions in farming systems to significant changes in the policy framework, namely the 2003 CAP reform, so it could be one or more years.

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 Q_i is the average regional yield per hectare of activities *i* (e.g. wheat, sunflower, cattle, sheep), A_i is the area occupied (land shares) by activity *i*, P_i^t is the producer price of activity *i* in year *t*, and S_i^{t+1} is the value of the direct subsidy in year *t+1* for activity *i*. For simplicity, we assumed that the average yields and areas of each activity within each farming system were constant during the study period. We considered subsidies in year *t+1*, because farmers usually make their production decisions in a given year with a reasonable knowledge on the direct payments to be paid next year. 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 S3 in Annex C in Supplementary Information).

Discrete choice model design

To investigate how farmers react to changes in prices and economic incentives by shifting farming system, we adopted a modelling procedure based on logistic regression (Hosmer and Lemeshow, 1989). The farming systems were specified as a categorical dependent variable. The independent variables included the biophysical, structural and policy constraints (Table 5. 1), and the economic variables GIR^t and *GIRdif*. We also considered an additional lagged variable identifying the farming system assigned to the farm in the previous time point (FSlag) to account for the effect of the track record in choosing the farming system. The logistic model can assume the binomial or multinomial form, depending on the number of categories in the farming system typology.

Despite widespread use of logistic regression in discrete choice modelling, it is limited in that it assumes independence of irrelevant alternatives (IIA) and it is unable to accommodate heterogeneous preferences (McFadden and Train, 2000). However, these are likely to occur in our case study, due to uncontrolled variables such as the socio-cultural profiles of farmers, which may influence motivations and attitudes towards policies (Siebert et al., 2006; Burton et al., 2008; de Snoo et al., 2013). To overcome these limitations and enable handling heterogeneity in farmers' preferences, we used a latent class model approach. Latent class models deal with heterogeneity in preferences by assuming that individuals can be grouped into a discrete distribution of segments or classes, which are internally homogeneous but heterogeneous with each other (Boxall and Adamowicz, 2002; Greene, 2012). Latent class models have been widely used in recent stated-preference studies using discrete choice models, including within the context of agri-environment policy evaluation (e.g. Colombo et al., 2009; Ruto and Garrod, 2011; Garrod et al., 2012; Villanueva et al., 2014). Additionally, since data included repeated observations on the same farms over time, a panel data structure was anticipated, related to individual-specific heterogeneity in preferences, which are taken to be constant over time (Greene, 2012).

To select the final model, a stepwise-like procedure was conducted, 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 reaching a model where all variables were significant at the 95% confidence 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 (James et al., 2013).

All statistical analysis 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).

Assessment of policy and market scenarios

Several policy and market scenarios involving changes in the economic incentives can be assessed with the model, in terms of their likely impacts on farming systems. This include, e.g., 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. Overall, all such changes will ultimately be reflected on the value of the GIR variable, and thereby influence the farmer's choice of farming system. Therefore, we simulated the implications of changes in the GIR variable as an integrative value representing the outcome of possible different scenarios.

To provide a measure of risk in the simulation of scenarios, we estimated a 95% confidence interval based on a Monte Carlo experiment, using 1000 random trials extracted from a multivariate normal distribution, using the model coefficients as the means vector and the corresponding 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.

Simulations results were also used to assess the impact of these scenarios on environmental indicators known to be relevant within the study area, such as the average stocking rate or the proportion of early-harvested cereals (Ribeiro et al., 2016a). Taking advantage of the spatial component of the data (LPIS), we mapped the predicted distribution of farming systems to allow the calculation of landscape metrics potentially usable as indicators of habitat quality. Landscape metrics computed at the farming system level have been made before (Ribeiro et al., 2016b), but research data providing references and benchmarking for habitat evaluation on these grounds are largely absent (Keenleyside et al., 2014b), including within the context of cereal-steppe habitat, which is key in the study area. Therefore, by way of illustration we used a landscape composition metric expressing the proportion of the area covered by the reference HNV farming system (P_RFS), and two simple configurational landscape metrics providing the mean patch area (MPA) and the number of patches (NPATCH) of the RFS land cover class, to assess the impact of policy and market scenarios on HNV farmland. Based on previous studies on the farmland birds in the study area (Santana et al. 2014, and references therein), we expected that higher values of P_RFS, MPA and NPATCH would be indicative of better habitat quality for species of conservation concern. Landscape metrics were computed using the “SDMTools” R package, version 1.1-221 (Vanderwal et al., 2015).

Results and Discussion

Farming systems typology

The cluster analysis performed on the IACS data identified five farming systems occurring within the study area between 2000 and 2010 (Annex A in Supplementary Information). These included 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 operating a low-intensity cereal-fallow rotation with low-density sheep grazing (the Traditional system). This was interpreted as representing the HNV farming system primarily responsible for the provision of the conservation relevant cereal-steppe habitat in the study area, and was selected as the reference farming system (RFS) in the model.

Major farming system dynamics were observed in the study area between 2000 and 2010 (Figure 5. 2). These changes happened mainly between 2003 and 2007, conceivably as a result of the 2003 CAP reform, tending to stabilize thereafter. This suggests that there is an adjustment period to policy changes of this magnitude that can take about 3 to 4 years to complete, during which major farming systems transitions can be observed. This is a potentially relevant issue for policy assessment, although surprisingly understated in the literature.

The main trend was the decline of the mixed (Traditional) and arable crops systems, and the increase of livestock specialized systems which, by the end of the study period, covered ca. 90% of the UAA. The two specialized crop farming systems (Annual crops and Permanent crops) were poorly represented in the area and they were nearly absent by the end of the study period. Its future growth in the study area is unlikely due to restrictions on new plantations of permanent crops imposed by the regulations of the SPA, and to the lack of irrigation water required to expand arable crops. For these reasons, the two crop specialized systems were not considered in the development of the choice model.

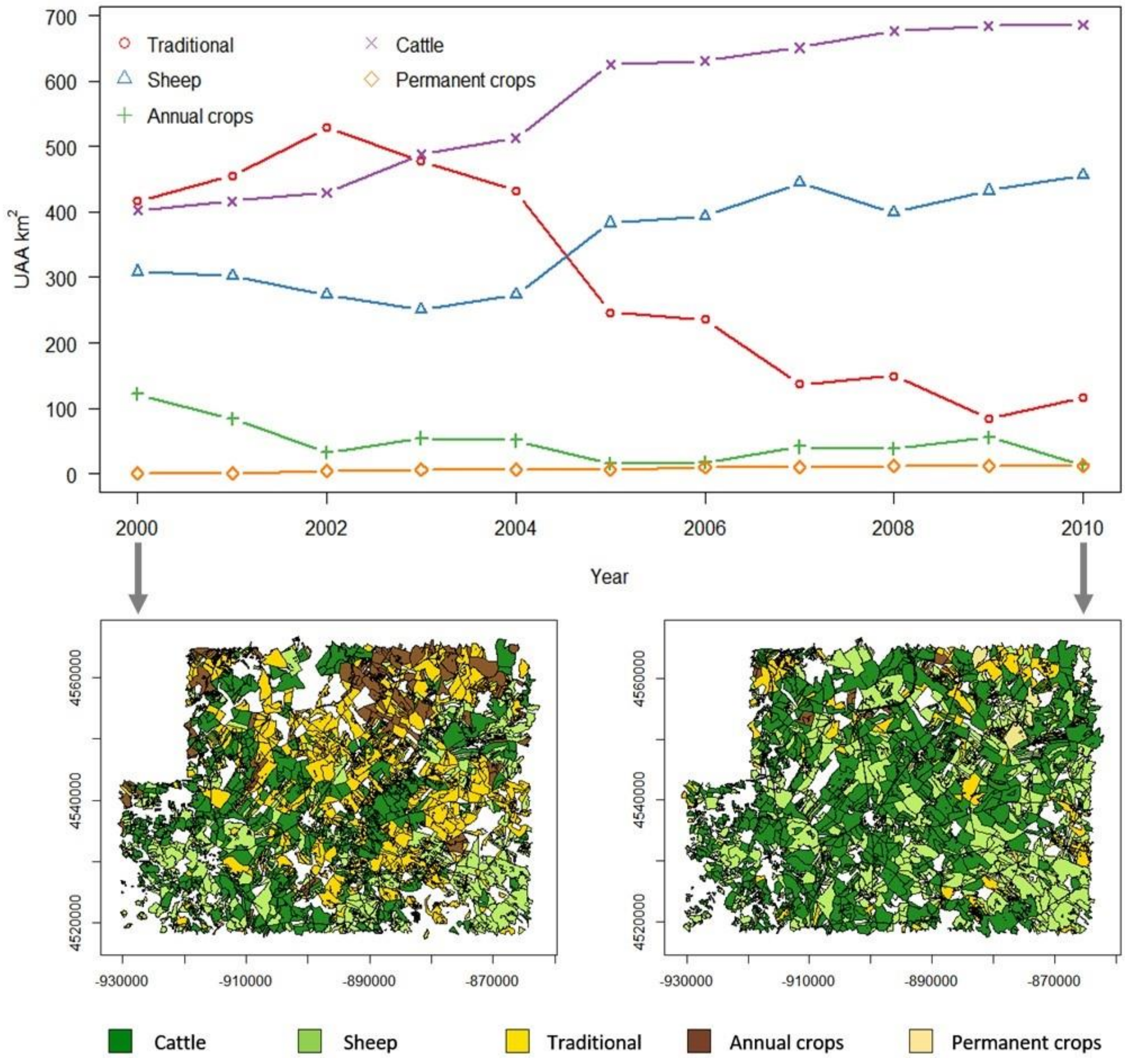

Figure 5. 2 - Temporal variation in the percentage of the utilized agricultural area (UAA) occupied by each farming system between 2000 and 2010 (upper panel) in the Castro Verde region (southern Portugal), and spatial distribution of farming systems at the start (lower left panel) and end (lower right panel) of the study period.

Economic effects of policy and market changes

Changes in market prices and policies occurring during the study period significantly affected the gross income ratio (GIR) expressing the relative profitability of the RFS (Traditional system) when compared to the alternative Cattle and Sheep farming systems (Figure 5. 3). However, the GIRs were largely stable in 2000-2003; they declined sharply in 2003-2007; and then declined at a lower rate between 2007-2010.

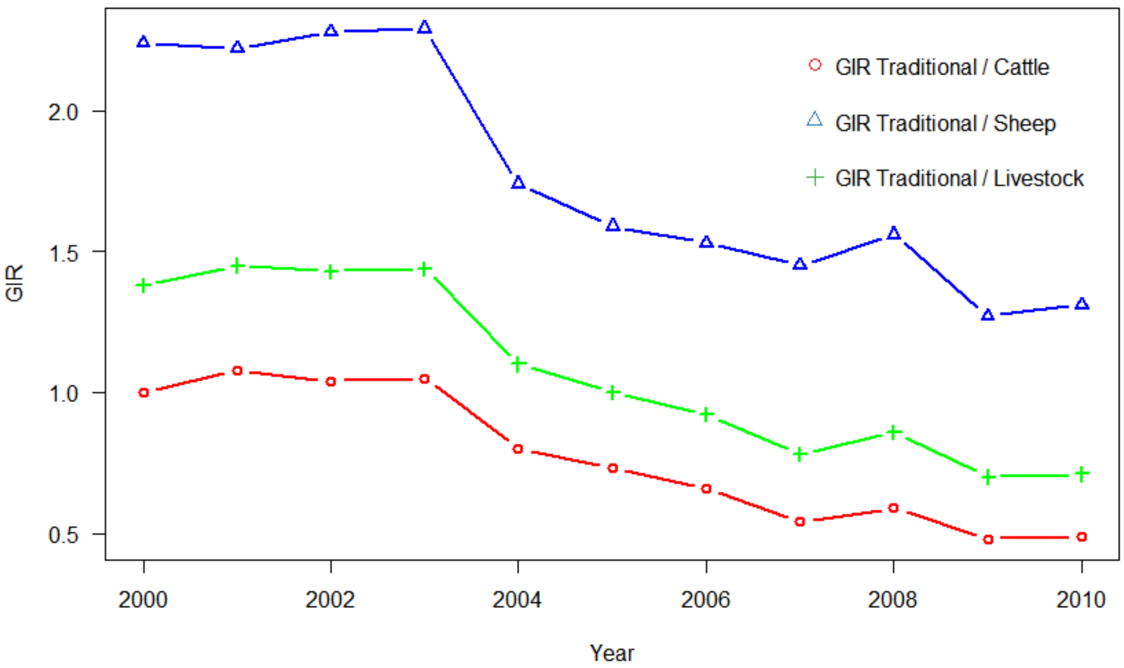

Figure 5. 3 - Changes in gross income ratio (GIR) between the Traditional farming system and the Cattle, Sheep and the composite Livestock systems during the study period. The value is above 1 when the gross income for the Traditional system is higher than the alternative systems, and below 1 otherwise.

The correlation between the GIR variables for the Traditional/Cattle and Traditional/Sheep was very high (0.98), due to the simultaneous decoupling of payments to annual crops (affecting the numerator) and possibly also due to high temporal correlation between the producer prices of cattle and sheep products (in the denominator). Because this was a potential source of collinearity problems in model estimation, the Cattle and Sheep farming systems were merged into a new composite Livestock system whose GIR variable resulted from averaging the gross income time-series of the two preceding systems, before recalculating the GIR.

Although some loss of relevant information may have resulted from this decision, since both systems entail specific problems for conservation (Ribeiro et al., 2016a, 2016b), it is likely that the key conservation implications were not lost, since they are mostly related to the confrontation between the HNV Traditional system and the alternative livestock specialized systems. In any case, the framework could be applied without great additional complexity to a multinomial application, comparing more than two alternative farming systems, including situations where more than one HNV farming system had to be safeguarded.

The farming system discrete choice model

As a consequence of the abovementioned results, we estimated a binomial choice model where the dependent variable was set to value 1 for the Traditional (reference) farming system, and 0 for the (alternative) Livestock specialized system. Given our findings on the farm management adjustment period resulting from significant policy changes, we assumed a time-period of three years for the lagged variables, and fixed the same time frame for the observations to use in model estimation, so that the data could capture primarily the effects of structural adjustments, and discard small annual fluctuations. Under these circumstances, a subset of the 2000-2010 database including only years 2001, 2004, 2007 and 2010 was selected to estimate the model, comprising a total of 1648 observations.

The variable selection procedure used to achieve the final model led to reject variables OAK, SPA and SLOPE for lack of significance at 95% confidence ($p < 0.050$) (Table S4 in Annex D in Supplementary Information). The analysis of the optimal model selection criteria (AIC and BIC) (Table S5 in Annex D in Supplementary Information) resulted in a model with only one latent class, because this simple model consistently showed the lowest BIC, while AIC was nearly identical in models with either one or two classes, but much lower than the three-class models confidence (Table S4 in Annex D in Supplementary Information). Also the panel data effect proved nonsignificant (sigma $p = 0.954$ in the single class model with panel data random effects) and therefore the final model resulted in a binomial logistic regression with no latent classes and no panel data effects (Table 5. 2). From this outcome we conclude that the independent variables were able to capture most of the heterogeneity in the data, suggesting that farmers in the study area are more homogeneous in their preferences, attitudes and motivations towards economic incentives conveyed by policies and markets, than initially expected (Siebert et al., 2006; Burton et al., 2008; de Snoo et al., 2013), although this can be related to the fact that the farms with less than 10 ha were excluded from the sample, which may have contributed to this lack of heterogeneity (Annex A in Supplementary Information).

Table 5. 2 - Binomial logistic model for farming system choice (n = 1648)

	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); AIC = 1079.9; BIC = 1123.2

Variable GIR presented a high positive coefficient, indicating that the likelihood (odds ratio) of choosing the Traditional system increases with its own relative profitability. The GIRdif variable also had a positive coefficient, showing that a positive recent trend in the GIR variable promotes the choice of the Traditional system. The estimated coefficients of these two variables suggest that these farmers behaved in accordance with rational expectations and profit maximization and that the choice of the farming system is mainly an economic decision. The FSlag variable showed a positive coefficient, indicating that the effect of the status quo farming system was relevant in the choice of farming system, evidencing a resistance-to-change effect. This may be related to investment costs or risk related to changes of farming system (e.g. fencing the parcels when changing from Traditional to a specialised livestock system). Good quality soils favoured the choice of the Traditional system, as shown by the positive coefficient of the SOIL variable. Conversely, higher rainfall tended to support the choice of the specialized Livestock system, probably related to higher forage yields, as indicated by the negative sign of the coefficient associated to the RAIN variable. The negative sign of the coefficient of the UAA variable indicates that larger farms were more likely to opt for the Livestock system. Land fragmentation was unfavourable to the Livestock system choice, as revealed by the negative sign in the JANUS coefficient, probably due to greater difficulty in grazing management or higher fencing (investment) costs.

The use of a ratio variable comparing the gross incomes of the Traditional system and the alternative systems as a way to jointly enter the effects of economic incentives into the model, proved to be highly suitable for simulating a wide range of scenarios of policy and market changes. These scenarios could result from changes in CAP regulations, or from a variety of other causes such as 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 could be overcome with information about the unit cost associated to each farming system, allowing the estimation of net profit, but this would very significantly increase the data requirements to implement the framework. This wouldn't be a problem per se if these data were readily available, which was not the case.

Finally, the model presented high predictive accuracy, showing a rate of correct predictions of 86.7% in the validation estimates performed with the train and test sets.

Model predictions for different policy and market scenarios

The model predictions to assess the impact of market and policy scenarios were outlined in a supply curve for biodiversity conservation services for the study area, and environmental indicators were calculated for three points along the curve (Figure 5. 4). The results 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 to lose area for livestock systems, reducing from the ca. 10% of total study area in 2010 to less than 4% in the next time-period (2013). The environmental impacts of such scenario are substantial, as the decline in the Traditional system and its replacement by livestock systems lead to a significant increase in stocking density and early-harvested cereals (Ribeiro et al., 2016a), with effects at the landscape scale by likely fragmentation of the cereal-steppe habitat mosaic, scattered in small patches (Figure 5. 4). To prevent this decline, an economic incentive equivalent of ca. 80 Euros/ha promoting the Traditional system would have to occur, whether provided as a result of 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 132 Euros/ha for the Traditional system to take up to ca. 50% of the study area, and ca. 157 Euros/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 90 Euros/ha, which would result in ca. 8% of the study area under the Traditional system – almost the double than in the status quo scenario.

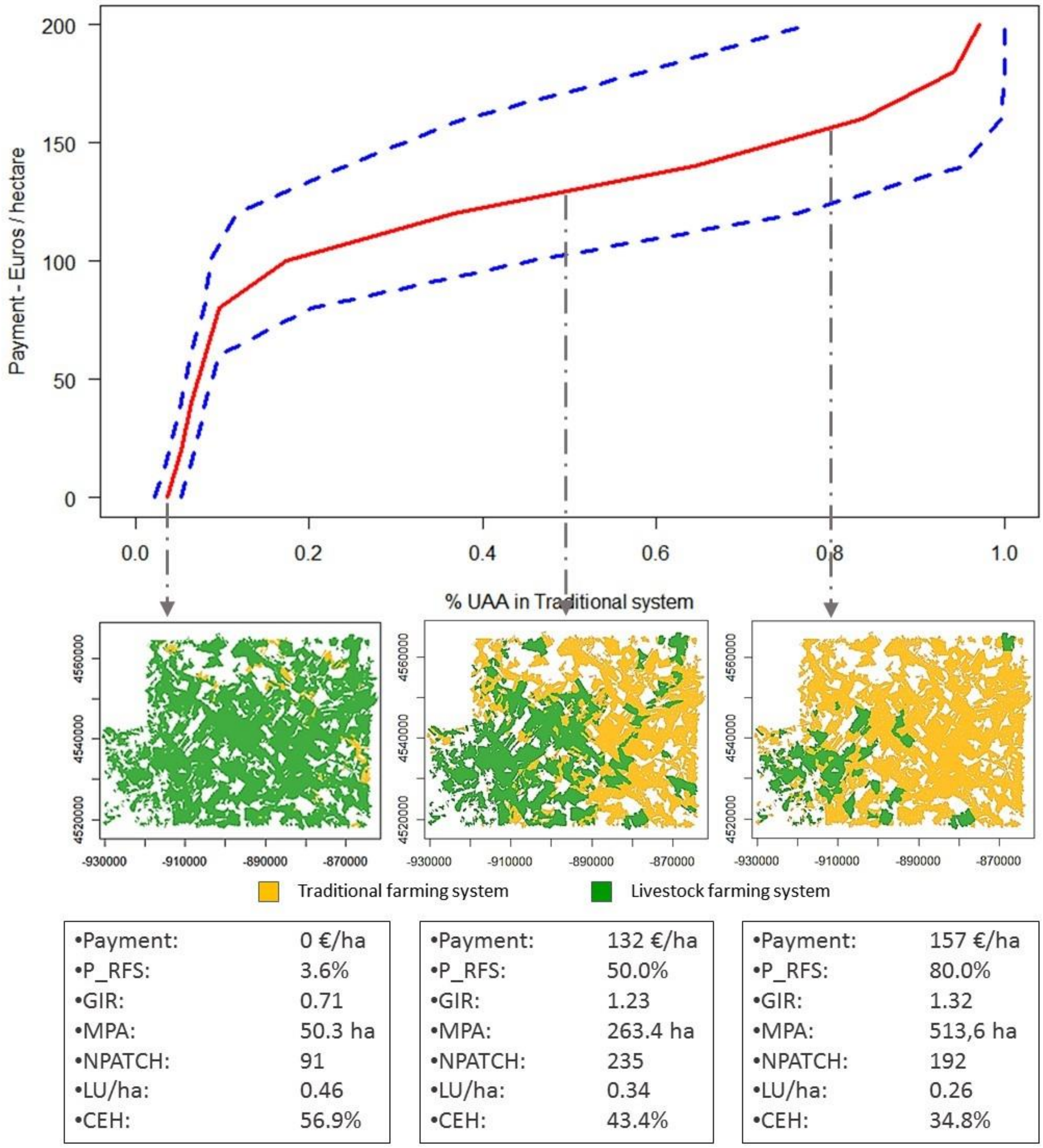

Figure 5. 4 - Supply curve for biodiversity conservation services in the study area (solid line) bounded by 95% confidence intervals (dashed lines), based on model predictions from 2010 to 2013 with Monte Carlo simulations, relating increasing levels of economic incentives towards the Traditional system with the proportion of the study area managed under this farming system. The spatial arrangement of farming systems and environmental indicators are presented for three points in the curve, representing the status quo scenario (left), an intermediate scenario (central) where 50% of the UAA would be managed under the Traditional system, and a maximization scenario (right) where this value would raise up to 80%. 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 two landscape configuration metrics showing the mean patch area (MPA) and the number of patches (NPATCH). The economic incentive (Payment) and the corresponding value for the GIR variable are also provided.

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%, showing 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. 85 Euros/ha, instead of the above mentioned 132 Euros/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).

Overall appraisal of the framework as a tool for conservation policy assessment

The framework developed in this study proved to be suitable to support the design of new agri-environment policy targeted on HNV farming systems, in accordance with previous suggestions (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, however, required to fit the pertinent conservation issues, such as identifying the appropriate RFS (those which are close to the HNV farmland habitat to be conserved). 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 actually 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 environmental payments) using a farming system approach to support HNV farmland across the EU with much lower transaction costs (Ribeiro et al., 2016a). In fact, transaction costs are often a major cause for farmers' low uptake of AES (Muradian et al., 2010; McCann, 2013; Pannell et al., 2013). Private transaction costs (for farmers) for participating in AES have been estimated at over 40 Euros/ha (Mettepenningen et al., 2009; Wätzold et al., 2015), which can offset a significant share of the environmental payment.

The proposed approach can, moreover, be seen as 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 regulation, 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.

Results showed that local agri-environmental policies within the Special Protection Area 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 uptake of the Castro Verde AES does not imply following a specific farming system, but only 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 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 particular 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).

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, in an efficient and reliable manner. 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 proved 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.

Acknowledgments

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References

Baldock, D., Beaufoy, G., Bennett, G., Clark, J., 1993. Nature conservation and new directions in the EC Common Agricultural Policy: the potential role of EC policies in maintaining farming and management systems of high nature value in the Community. Institute for European Environmental Policy, London.

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. doi:10.1016/j.ecolecon.2010.12.014

Barraquand, F., Martinet, V., 2011. Biological conservation in dynamic agricultural landscapes: Effectiveness of public policies and trade-offs with agricultural production. Ecol. Econ. 70, 910–920. doi:10.1016/j.ecolecon.2010.12.019

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.

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

BirdLife International, 2011. High Nature Value Farming - How Diversity in Europe's Farm Systems Delivers for Biodiversity.

BirdLife International, 2004. Birds in the European Union: a status assessment., Wageningen. ed, Wageningen, The Netherlands: BirdLife International. Wageningen, The Netherlands: BirdLife International.

Boxall, P.C., Adamowicz, W., 2002. Understanding Heterogeneous Preferences in Random Utility Models: A Latent Class Approach. Environ. Resour. Econ. 23, 421–446. doi: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. Policy 24, 101–109. doi:10.1016/j.envsci.2012.07.022

Burton, R.J.F., Kuczera, C., Schwarz, G., 2008. Exploring Farmers' Cultural Resistance to Voluntary Agri-environmental Schemes. Sociol. Ruralis 48, 16–37. doi:10.1111/j.1467-9523.2008.00452.x

Calvo-Iglesias, M.S., Fra-Paleo, U., Diaz-Varela, R.A., 2009. Changes in farming system and population as drivers of land cover and landscape dynamics: The case of enclosed and semi-openfield systems in Northern Galicia (Spain). Landsc. Urban Plan. 90, 168–177. doi:10.1016/j.landurbplan.2008.10.025

Carmona, A., Nahuelhual, L., 2010. Linking farming systems to landscape change: an empirical and spatially explicit study in southern Chile. Agric. Ecosyst. … 139, 40–50. doi:10.1016/j.agee.2010.06.015

Christensen, T., Pedersen, A.B., Nielsen, H.O., Mørkbak, M.R., Hasler, B., Denver, S., 2011. Determinants of farmers' willingness to participate in subsidy schemes for pesticide-free buffer zones — A choice experiment study. Ecol. Econ. 70, 1558–1564. doi:10.1016/j.ecolecon.2011.03.021

Chukwukere Austin, O., Chijindu Ulunma, A., Sulaiman, J., 2012. Exploring the Link between Land Fragmentation and Agricultural Productivity. Int. J. Agric. For. 2, 30–34. doi:10.5923/j.ijaf.20120201.05

Colombo, S., Hanley, N., Louviere, J., 2009. Modeling preference heterogeneity in stated choice data: an analysis for public goods generated by agriculture. Agric. Econ. 40, 307–322. doi:10.1111/j.1574-0862.2009.00377.x

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. doi: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. doi:10.1016/S0167-8809(99)00114-0

European Commission, 2011. Communication from the Commission to the European Parliament, the Council, the Economic and Social Committee and the Committee of the Regions. Our life insurance, our natural capital: an EU biodiversity strategy to 2020. Brussels.

Falconer, K., 2000. Farm-level constraints on agri-environmental scheme participation: a transactional perspective. J. Rural Stud. 16, 379–394.

Falconer, K., Dupraz, P., Whitby, M., 2001. An Investigation of Policy Administrative Costs Using Panel Data for the English Environmentally Sensitive Areas. J. Agric. Econ. 52, 83–103.

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. doi:10.1016/j.ecolecon.2012.02.011

Greene, W.H., 2012. Econometric Analysis, Seventh Ed. 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. doi:10.18637/jss.v028.i04

Henle, K., Alard, D., Clitherow, J., Cobb, P., Firbank, L., Kull, T., McCracken, D., Moritz, R.F.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. doi:10.1016/j.agee.2007.09.005

Hosmer, D.W., Lemeshow, S., 1989. Applied logistic regression. Wiley, New York.

Hynes, S., Garvey, E., 2009. Modelling Farmers' Participation in an Agri-environmental Scheme using Panel Data: An Application to the Rural Environment Protection Scheme in Ireland. J. Agric. Econ. 60, 546–562. doi:10.1111/j.1477-9552.2009.00210.x

IGP, 2012. Carta de Ocupação do Solo - COS' 90 [WWW Document]. URL http://www.igeo.pt/produtos/CEGIG/COS.htm (accessed 5.9.12).

INMG, 1991. Normais climatológicas da Região de "Alentejo e Algarve", correspondentes a 1951-1980. Instituto Nacional de Meteorologia e Geofísica. O Clima de Portugal, Fascículo XLIX, Volume 4 - 4ª Região. Lisboa.

Jack, B.K., Leimona, B., Ferraro, P.J., 2009. A Revealed Preference Approach to Estimating Supply Curves for Ecosystem Services: Use of Auctions to Set Payments for Soil Erosion Control in Indonesia. Conserv. Biol. 23, 359–367. doi:10.1111/j.1523-1739.2008.01086.x

James, G., Witten, D., Hastie, T., Tibshirani, R., 2013. An Introduction to Statistical Learning - With Applications in R. Springer.

Januszewski, J., 1968. Index of land consolidation as a criterion of the degree of concentration. Geogr. Plonica 14, 291–296.

Kaufman, L., Rousseeuw, P.J., 1990. Finding Groups in Data: An Introduction to Cluster Analysis. Wiley, New York.

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

European Commission, DG Environment, Contract No ENV.B.2/ETU/2013/0046, Institute for European Environment.

Lewis, D.J., Wu, J., 2014. Land-Use Patterns and Spatially Dependent Ecosystem Services: Some Microeconomic Foundations. Int. Rev. Environ. Resour. Econ. 1–42. doi:10.1561/101.00000069

Maechler, M., Rousseeuw, P., Struyf, A., Hubert, M., Hornik, K., 2012. Cluster: Cluster Analysis Basics and Extensions. R package version 1.14.2.

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

McFadden, D., Train, K., 2000. Mixed MNL Models for Discrete Response. J. Appl. Econom. 470, 447–470.

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. doi:10.1016/j.ecolecon.2010.10.007

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

Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P.H., 2010. Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services☆. Ecol. Econ. 69, 1202–1208. doi:10.1016/j.ecolecon.2009.11.006

National Research Council, 2010. Toward Sustainable Agricultural Systems in the 21st Century, National A. ed. The National Academies Press, Washington, D.C. doi:10.17226/12832

Oñate, J.J., Atance, I., Bardají, 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. doi: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.J., Roberts, A.M., Park, G., Alexander, J., 2013. Improving environmental decisions: A transaction-costs story. Ecol. Econ. 88, 244–252. doi: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 (80-.). 344, 1090–1092. doi:10.1126/science.1253425

Poláková, J., Tucker, G., Hart, K., Dwyer, J., Rayment, M., 2011. Addressing biodiversity and habitat preservation through measures applied under the Common Agricultural Policy, Report Prepared for DG Agriculture and Rural Development, Contract No. 30-CE-0388497/00-44. Institute for European Environmental Policy.

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. R Found. Stat. Comput. URL http://www.r-project.org (accessed 12.26.11).

Reino, L., Porto, M., Morgado, R., Moreira, F., Fabião, A., Santana, J., Delgado, A., Gordinho, L., Cal, J., Beja, P., 2010. Effects of changed grazing regimes and habitat fragmentation on Mediterranean grassland birds. Agric. Ecosyst. Environ. 138, 27–34. doi: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. doi: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. doi: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. doi:10.1007/s10980-015-0287-0

Rousseeuw, P.J., 1987. Silhouettes - A graphical aid to the interpretation and validation of cluster analysis.pdf. J. Comput. Appl. Math. 20, 53–65.

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. doi: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. doi:10.1111/conl.12077

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. ed. Instituto da Conservação da Natureza e da Biodiversidade, Lisboa.

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

Simion, G., 2008. Geographical Analysis of the Land Fragmentation Process Based on Participatory Mapping and Satellite Images . Case Studies of Ciorogârla and Vănătorii Mici From the Bucharest Metropolitan Area. J. Stud. Res. Hum. Geogr. 2, 83–94.

SROA/CNROA, 2012. Carta de Capacidade de Uso do Solo. DGADR - Serviço Reconhecimento e Ordenam. Agrário.

Stoate, C., Báldi, 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. J. Environ. Manage. 91, 22–46. doi:10.1016/j.jenvman.2009.07.005

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., 2015. A Novel, Spatiotemporally Explicit Ecological-economic Modeling Procedure for the Design of Cost-effective Agri-environment Schemes to Conserve Biodiversity. Am. J. Agric. Econ. doi:10.1093/ajae/aav058

Weber, A., 2015. Implementing EU co-financed agri-environmental schemes: Effects on administrative transaction costs in a regional grassland extensification scheme. Land use policy 42, 183–193. doi:10.1016/j.landusepol.2014.07.019

Supplementary information

Annex A

Farming systems in the study area

Farming systems typology

The choice models tested in the present paper used a categorical dependent variable that identified the farming system operated by each farm in a given year, between 2000 and 2010, within the study area. The farming system typology used in the classification was developed by means of a non-hierarchical classification technique, using the partition around medoids (PAM) clustering algorithm (Kaufman and Rousseeuw, 1990). Under this approach, a real observation (a medoid) is selected to represent a cluster, instead of using centroids or group means, corresponding to a representative farm whose average dissimilarity to all the other farms in the same cluster is minimal. The number of clusters (farming systems) to be considered was assessed using silhouette plots (Rousseeuw, 1987) and with reference to similar typologies obtained by previous works in the same study area (Ribeiro et al., 2014). PAM was conducted with the ‘cluster’ package (Maechler et al., 2012) for R (R Development Core Team, 2015).

Table S1. Summary statistics of farm characterization variables used to model farming system (n = 7883; UAA = Utilized Agricultural Area).

Variable	Description	Mean ± SD (Min-Max)
Dry cereals	% of dry cereals in the UAA	21.5±25.2 (0-100)
Other annual crops	% of other annual crops in the UAA.	2.4±10.1 (0-100)
Fallows	% of fallows in the UAA.	11.1±21.1 (0-100)
Pastures	% of forages and pastures in the UAA.	63.1±35.3 (0-100)
Permanent crops	% of permanent crops in the UAA	1.9±9.4 (0-100)
Livestock density	Livestock units (LU) per hectare of fodder area.	0.47±0.55 (0-2)
Cattle ratio	% of cattle LU in total LU.	30.4±41.9 (0-100)

The PAM was performed on a data base of farm characterization variables describing land-use and livestock patterns during the study period (2000 – 2010), extracted from the Portuguese Common Agricultural Policy (CAP) paying agency (Table S1). These data are collected on a yearly basis through farmer declarations when applying to CAP payments, under the European Union (EU) Integrated Administration and Control System (IACS), and comprise a spatially explicit component related to the Land Parcel Identification System (LPIS). The set of land parcels declared by a single farmer in a given year, whether contiguous or not, was considered a farm. Farms under 10 hectares were excluded from the analysis to minimize misclassifications resulting from the effects of crop rotation, which in small farms can lead to very contrasting land-use patterns in successive years. These farms represented less than 0.5% of the study area. The resulting total number of observations was 7883 (combinations of farm/year) for the 11 years of the study period. Therefore, the number of farms in the study area averaged ca. 716 across the time period.

The PAM procedure returned 5 clusters, which were interpreted as farming systems and named on the basis of their main characteristics (Table S2). The Sheep and Cattle systems were identified as livestock specialized systems, mainly differing on the type of livestock and grazing density. The Traditional system was acknowledged as a mixed system, where annual crop production was complemented with low density sheep grazing. This system was identified as a potential representative of the low-intensity cereal-fallow-sheep system that dominated the local landscape for decades, providing the conservation relevant cereal-steppe habitat that lead to the classification of the area as a Special Protection Area under the EU Directive 79/409/CEE (Birds Directive) and to the implementation of an agri-environmental scheme aimed at safeguarding this high nature value farmland (Delgado and Moreira, 2000; Ribeiro et al., 2014). Both the Annual and Permanent crops systems were identified as specialized in crop production, with no livestock, with the former dedicated to arable crops and the later to permanent crops, mostly olive groves.

Table S2. Summary statistics of the five farming systems returned by the PAM cluster analysis. Variable definition in Table S1.

Variable	Sheep	Cattle	Traditional	Annual crops	Permanent crops
Dry cereals (%)	6.2	13.9	49.5	46.1	0
Other annual crops (%)	0	0	0	35.4	0
Fallows (%)	1.5	0	30.2	13.6	0
Pastures (%)	92.3	85.7	20.1	4.9	0
Permanent crops (%)	0	0.4	0.3	0	100
Livestock density (LU/ha)	0.22	0.64	0.19	0	0
Cattle ratio (%)	0	77.4	0	0	0

Annex B

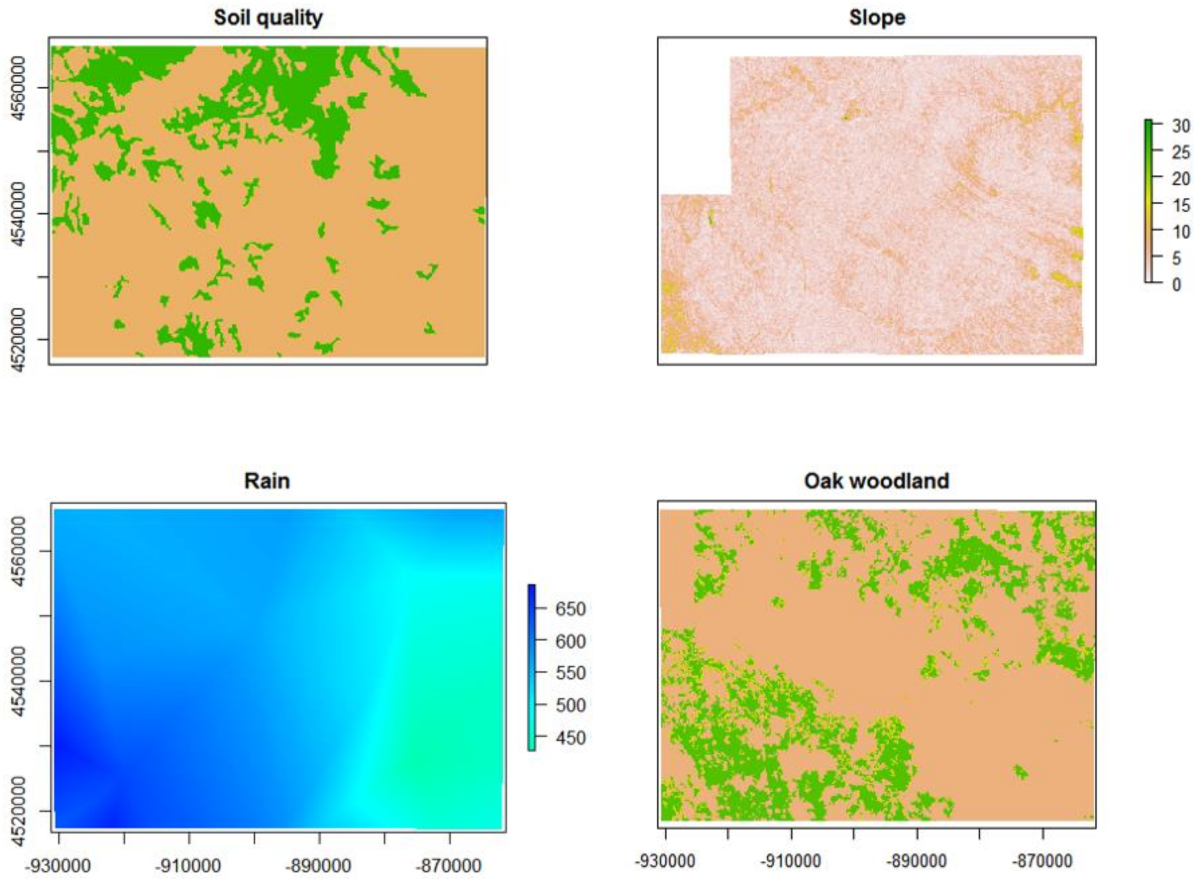

Figure S1. Spatial distribution of the biophysical (soil quality, slope and rain) and structural (Oak woodland) independent variables tested in the models, within the study area.

Soil quality – spatial distribution of soils with good agricultural potential (in green)[1]; Slope – terrain slope, values in degrees (source: personal computation based on a digital elevation model); Rain – average annual precipitation, in mm, computed by interpolating the values from local weather stations[2]; Oak woodland – spatial distribution of Oak woodland areas (in green)[3].

[1] SROA/CNROA, 2012. Carta de Capacidade de Uso do Solo. DGADR - Serviço Reconhecimento e Ordenamento Agrário.

[2] INMG, 1991. Normais climatológicas da Região de "Alentejo e Algarve", correspondentes a 1951-1980. Instituto Nacional de Meteorologia e Geofísica. O Clima de Portugal, Fascículo XLIX, Volume 4 - 4ª Região. Lisboa.

[3] IGP, 2012. Carta de Ocupação do Solo - COS' 90 [WWW Document]. URL http://www.igeo.pt/produtos/CEGIG/COS.htm (accessed 5.9.12).

Annex C

Table S3. Agricultural producer prices and direct payments between 1999 and 2011.

	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Agricultural prices													
Common wheat (€/kg)	0.16	0.14	0.16	0.15	0.15	0.17	0.22	0.21	0.18	0.20	0.14	0.15	0.15
Durum wheat (€/kg)	0.18	0.16	0.19	0.18	0.19	0.19	0.23	0.21	0.20	0.27	0.23	0.17	0.20
Sunflower (€/kg)	0.22	0.19	0.28	0.28	0.22	0.22	0.27	0.32	0.32	0.49	0.25	0.28	0.32
Vegetables for industry (€/kg)	0.10	0.10	0.11	0.13	0.13	0.13	0.12	0.13	0.08	0.09	0.09	0.10	0.08
Olives (€/kg)	0.81	0.75	0.67	0.72	0.76	0.89	1.47	1.75	0.40	0.38	0.32	0.30	0.29
Calf 3 to 6 months (€ per live animal)	244.43	243.50	222.94	244.20	243.97	218.09	220.57	266.21	362.14	338.85	362.41	368.37	402.50
Lamb up to 28 kg (€/kg live weight)	2.76	2.82	3.33	3.10	3.13	2.97	2.79	2.83	2.66	2.66	2.81	2.73	2.78
Direct payments													
Arable crops (€/ton)	62.00	63.00	63.00	63.00	63.00	63.00	63.00	0	0	0	0	0	0
Set-aside (€/ton)	62.00	63.00	63.00	63.00	63.00	63.00	63.00	0	0	0	0	0	0
Durum wheat supplement (€/ha)	344.50	344.50	344.50	344.50	344.50	344.50	313.00	0	0	0	0	0	0
Suckler cow - base (€ per animal)	200.00	200.00	200.00	200.00	200.00	200.00	200.00	200.00	200.00	200.00	200.00	200.00	200.00
Suckler cow - additional (€ per animal)	30.19	30.19	30.19	30.19	30.19	30.19	30.19	30.19	30.19	30.19	30.19	30.19	30.19
Suckler cow - extensification premium <=1,4 LU/ha (€ per animal)	100.00	100.00	100.00	100.00	100.00	100.00	100.00	0	0	0	0	0	0
Sheep - base (€ per animal)	21.00	21.00	21.00	21.00	21.00	21.00	21.00	10.50	10.50	10.50	10.50	10.50	10.50
Sheep - additional (€ per animal)	7.00	7.00	7.00	7.00	7.00	7.00	7.00	3.50	3.50	3.50	3.50	3.50	3.50
Olive oil (€/kg)	0.13	0.13	0.13	0.13	0.13	0.13	0.13	0	0	0	0	0	0

Sources: Agricultural prices – Portuguese National Institute of Statistics (INE – Instituto Nacional de Estatística - http://www.ine.pt); Direct payments: Portuguese Cap paying agency (IFAP – Instituto de Financiamento da Agricultura e Pescas - http://www.ifap.min-agricultura.pt).

Annex D

Table S4. Outputs of the latent class model with panel data, including all candidate independent variables (for 1, 2 and 3 latent classes)

	1 Latent class model					2 Latent classes model									
						Class 1					Class 2				
	Estimate	Std. error	z value	Pr(>|z|)		Estimate	Std. error	z value	Pr(>|z|)		Estimate	Std. error	z value	Pr(>|z|)	
(Intercept)	-1.325	1.113	-1.191	0.234		1.149	2.022	0.568	0.570		-39.068	1206.570	-0.032	0.974	
GIR	6.158	0.705	8.738	<0.001	***	7.546	1.301	5.798	<0.001	***	39.637	1213.187	0.033	0.974	
GIRdif	4.084	1.035	3.944	<0.001	***	5.609	1.983	2.829	0.005	**	8.935	352.413	0.025	0.980	
UAA	-0.133	0.042	-3.127	0.002	**	-0.285	0.093	-3.080	0.002	**	-0.152	0.099	-1.534	0.125	
AES	0.112	0.170	0.660	0.509		0.239	0.284	0.844	0.399		-0.040	0.448	-0.089	0.930	
SPA	-0.192	0.170	-1.131	0.258		-0.360	0.281	-1.284	0.199		-0.360	0.466	-0.774	0.439	
SOIL	1.625	0.308	5.278	<0.001	***	1.421	0.471	3.014	0.003	**	3.677	1.072	3.431	0.001	***
OAK	-0.357	0.319	-1.119	0.263		-0.432	0.521	-0.828	0.408		-0.298	0.774	-0.386	0.700	
SLOPE	-0.023	0.112	-0.208	0.835		-0.281	0.181	-1.556	0.120		0.750	0.393	1.906	0.057	.
RAIN	-8.953	1.630	-5.494	<0.001	***	-11.656	2.746	-4.244	<0.001	***	-10.626	4.600	-2.310	0.021	*
JANUS	-0.882	0.386	-2.287	0.022	*	-1.254	0.639	-1.963	0.050	*	-1.313	1.056	-1.244	0.214	
FSlag	2.488	0.172	14.503	<0.001	***	0.747	0.483	1.549	0.121		15.617	369.583	0.042	0.966	
Class size (n)	1648					678					970				
Class size (%)	100					41.1					58.9				
Log-Likelihood	-530.6985 (df=12)					-517.6736 (df=25)									
AIC	1085.4					1085.3									
BIC	1150.3					1220.5									

Significance codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’

Table S4. (cont.)

	3 Latent classes model														
	Class 1					Class 2					Class 3				
	Estimate	Std.Error	z value	Pr(>|z|)		Estimate	Std.Error	z value	Pr(>|z|)		Estimate	Std.Error	z value	Pr(>|z|)	
(Intercept)	-78.939	38.764	-2.036	0.042	*	138.365	74.060	1.868	0.062	.	-3.921	1.866	-2.101	0.036	*
GIR	43.783	21.781	2.010	0.044	*	14.548	10.350	1.406	0.160		7.776	1.157	6.723	<0.001	***
GIRdif	7.348	6.946	1.058	0.290		-148.959	79.791	-1.867	0.062	.	6.136	1.635	3.754	<0.001	***
UAA	-0.563	0.466	-1.208	0.227		-12.444	6.886	-1.807	0.071	.	-0.128	0.074	-1.721	0.085	.
AES	-3.744	2.242	-1.670	0.095	.	-4.853	2.907	-1.669	0.095	.	0.493	0.282	1.751	0.080	.
SPA	-0.451	1.604	-0.281	0.778		5.963	4.286	1.391	0.164		-0.457	0.275	-1.666	0.096	.
SOIL	52.189	29.499	1.769	0.077	.	-12.467	6.050	-2.061	0.039	*	2.014	0.559	3.601	<0.001	***
OAK	-3.674	2.677	-1.372	0.170		-12.382	7.174	-1.726	0.084	.	-0.550	0.504	-1.090	0.276	
SLOPE	7.581	3.707	2.045	0.041	*	-14.188	7.001	-2.027	0.043	*	0.154	0.192	0.804	0.422	
RAIN	-57.851	38.800	-1.491	0.136		-223.557	117.598	-1.901	0.057	.	-7.078	2.510	-2.820	0.005	**
JANUS	-2.440	4.598	-0.531	0.596		-65.361	33.715	-1.939	0.053	.	0.289	0.643	0.449	0.653	
FSlag	54.787	29.286	1.871	0.061	.	23.888	11.344	2.106	0.035	*	1.453	0.347	4.190	<0.001	***
Class size (n)	766					212					670				
Class size (%)	46,5					12,9					40,7				
Log-Likelihood	-499.0301 (df=38)														
AIC	1074.1														
BIC	1279.5														

Significance codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' '

Table S5. Outputs of the latent class model with panel data, including only the significant independent variables, with 1 and 2 latent classes

	1 Latent class model					2 Latent classes model									
						Class 1					Class 2				
	Estimate	Std. error	z value	Pr(>|z|)		Estimate	Std. error	z value	Pr(>|z|)		Estimate	Std. error	z value	Pr(>|z|)	
Intercept	-1.101	1.003	-1.101	0.272		-5.688	8.027	-0.709	0.479		0.080	1.983	0.040	0.968	
GIR	6.226	0.827	7.532	<0.001	***	10.224	8.787	1.164	0.245		7.535	1.533	4.916	<0.001	***
GIRdif	4.194.	1.061	3.954	<0.001	***	1.327	2.888	0.459	0.646		6.122	2.450	2.498	0.012	*
FSlag	2.497	0.167	14.970	<0.001	***	5.920	3.164	1.871	0.061	.	0.821	0.680	1.207	0.227	
SOIL	1.632	0.314	5.202	<0.001	***	-0.115	0.102	-1.127	0.260		-0.266	0.109	-2.444	0.015	*
RAIN	-9.542	1.738	-5.490	<0.001	***	-1.209	1.091	-1.108	0.268		-1.332	0.691	-1.928	0.054	.
UAA	-0.130	0.039	-3.313	0.001	***	2.676	0.933	2.868	0.004	**	1.545	0.527	2.930	0.003	**
JANUS	-0.885	0.392	-2.252	0.024	*	-13.089	5.480	-2.388	0.017	*	-11.188	2.780	-4.024	<0.001	***
Class size (n)	1648					1219					429				
Class size (%)	100					74.0					26.0				
Log-Likelihood	-531.95 (df=8)					-531.11 (df=17)									
AIC	1079.9					1080.2									
BIC	1123.2					1172.1									

Significance codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’

Chapter 6

General discussion and conclusion

General discussion and conclusion

This thesis was set out to explore the effects of agricultural policy on HNV farmland and to assess the feasibility of a farming systems approach as a framework for developing agri-environment policy. The relevance of these issues comes from the recognition of insufficient cost-effectiveness of the CAP agri-environmental policies, which have not proved capable of halting the continuing loss of HNV farmland, undermining the conservation of European biodiversity (Stoate et al. 2009; Latacz-Lohmann & Hodge 2003; Batáry et al. 2015; Lomba et al. 2015). On this basis, we sought to answer the following research questions: "*Does a typology of farming systems built on IACS/LPIS data allows us to identify farms operating HNV farming systems which could therefore be selected to receive an agri-environmental premium payment? What would be the level and type of funding necessary for its sustainability?*".

To address these questions, we focused on a case study of an agricultural area of high importance for biodiversity conservation in Portugal, renowned for hosting large communities of steppe birds with conservation status. Using IACS/LPIS data for years 2000 to 2010 provided by the Portuguese CAP paying agency (IFAP – Instituto de Financiamento da Agricultura e Pescas), a general framework based on a farming systems approach was developed to analyse the farmland transformations occurring during this time period, seeking to identify their main drivers and constraints and to evaluate their effects on its natural value. Special attention was given to the role of economic incentives in the choice of farming system, aiming to assess different market and policy scenarios.

The main empirical findings of the research are chapter specific and were summarized within the respective empirical chapters (Chapters 2 to 5). In the next section we synthesize and discuss the main findings to evaluate whether the abovementioned thesis-level research objectives and research questions were met and duly answered respectively. We then discuss the potential policy implications of our findings and conclude by highlighting the limitations of the proposed framework and providing recommendations for future research.

Synthesis and main conclusions

The overall feasibility and usefulness of our methodological framework was demonstrated, as we showed that a farming system typology derived from land use and livestock data, such as those readily available in CAP paying agencies across the EU, can be used, in an effective and practical way, as a tool to develop studies integrating agro-economic and ecological aspects, allowing to infer about HNV landscape features and farming practices and therefore to help identify HNV farming systems.

Despite some differences in the reference period, or in the delimitation of the study area or even in the data source used (e.g. IACS/LPIS vs. farm survey data), which inevitably lead to differences in the baseline data used in the different empirical exercises presented in Chapters 2 to 5, we found a high consistency in the farming systems typologies achieved among them: in all exercises it was possible to identify the HNV Traditional system, two livestock specialized farming systems (the Cattle and Sheep systems) and the existence of crop specialized systems (whether annual crops and/or permanent crops, depending on the above mentioned premises).

The research started by identifying the main farming systems in the study area between 2000 and 2010 (Chapter 2). By comparing their spatial distribution and representativeness before and after the 2003 CAP reform, we showed how changes in horizontal broad-brush policies, such as CAP direct payments, can easily undermine the effects of local agri-environment policy, such as the AES of Castro Verde, as was revealed by the decline of the Traditional system, offset by the rise of specialized livestock systems, a result that calls for much greater care in the conciliation of CAP Pillar 1 and Pillar 2 policies. At the same time, we conducted an analysis that allowed us to conclude that both biophysical, structural and policy variables were drivers of the observed farming system dynamics.

To find out whether these farming system transitions led to a decrease in the natural value of the study area, we focused on investigating the existence of a relationship between farming systems and landscape patterns, and if these differences could be used to identify HNV farming systems (Chapter 3). We concluded that such links between farming systems and landscape patterns do exist in some cases, but not in others. A paradigmatic case was illustrated by the Traditional and Cattle systems, which were shown to present similar landscapes patterns, potentially providing them identical access to agri-environment schemes highly dependent on land-use regulations, irrespective of potential differences in their natural value related to technological aspects (the *farming practice* dimension, which was not explored in this chapter).

We found a relationship between agricultural intensity and landscape heterogeneity (regarded as a measure of diversity of landscape patterns), leading us to propose a terminology establishing the two extremes of a continuum from “landscape makers” to “landscape takers” farming systems. We highlighted the potential usefulness of these concepts in policy design, as they suggest that it may be easier to take farms in less intensive farming systems to change their land-use patterns to comply with the requirements of an AES, while remaining within the same farming system, whereas for farms in more intensive farming systems, which are less flexible in relation to landscape features, those land-use adjustments could be far more difficult (i.e. costlier), as eventually they would require change of farming system.

On a different approach, but also aiming to assess how the farming system dynamics observed during 2000-2010 impacted on the natural value of the study area, we focused on investigating links between farming systems and farming practices of conservation concern (Chapter 4). Our results confirmed that our farming systems approach can be used to infer conservation-relevant farming practices, which were able to separate farming systems that were not separated by the landscape analysis (the Traditional and the Cattle systems), while joining farming systems that were separated in terms of landscape pattern but did not differ significantly in many farming practices (the Traditional and the Sheep systems).

The fact that the Traditional and the Cattle systems presented similar landscape patterns but distinct farming practices, assigning them a clear differentiation in conservation value, exposed the weaknesses of designing AES focusing on land-use rules, and the strength of a farming systems approach, as it proved able to simultaneously address both land-use and technological issues of conservation concern. Additionally, because these two farming systems showed similar agricultural intensity levels, we concluded that HNV farmland sustainability is not only threatened by trends for agricultural intensification or abandonment, as is often claimed (e.g. Stoate et al. 2009; Latacz-Lohmann & Hodge 2003; Batáry et al. 2015; Lomba et al. 2015), but also by changes between farming systems with similar intensity levels but distinct specialization levels and much different conservation value. This results calls for a more complex view of different dimensions of farming systems related to their conservation value: at least specialization in addition to intensity.

These findings confirmed the suitability of a farming systems approach to support the identification of HNV farming systems, whether based on landscape patterns or agricultural practices, supporting the idea of using such a farming systems approach as a basis for designing agri-environment policies. Based on these findings, we were able to positively respond to the first part of our research question: "*Does a typology of farming systems built on IACS/LPIS data allows us to identify farms operating HNV farming systems which could therefore be selected to receive an agri-environmental premium payment?*".

Building on the idea of using a farming systems approach as a basis for designing agri-environment policy, the last empirical chapter of the thesis focused on assessing the role of economic incentives provided by policy or markets in the choice of farming system (Chapter 5). The discrete choice model developed in this chapter showed how the choice of farming system is primarily an economic decision, although influenced by biophysical and structural factors. Using the model to assess policy and market scenarios, we were able to derive a supply curve for biodiversity conservation services in the study area, which can be used to determine the payment level required to achieve a certain level of conservation (measured by the extent of HNV farmland). These outcomes allowed us to meet the second part of our research question: "*What would be the level and type of funding necessary for its [HNV farming] sustainability?*".

Policy implications

Overall, our findings support the idea of an agri-environment policy focused on HNV farming systems, by having shown that: i) the typology of farming systems can be successfully built from IACS/LPIS data; ii) these farming systems can be interpreted to identify those with HNV characteristics, which would therefore be eligible for premia payments; iii) it is possible to relate the level of an agri-environment payment targeted on HNV farming systems with the extent of conservation service provided. This means that the proposed farming systems approach can be used to simulate the supply side of a market for biodiversity conservation services which, to identify the optimum level of service, would have to be completed by information on the demand side about the social willingness to pay for improvements in the environmental service (demand curve) or, as this info is often unavailable and as a second best alternative, an administrative selection of the conservation level admitted as socially desirable.

As shown in Chapters 4 and 5, the implementation of an agri-environment policy focused on supporting targeted HNV farming systems within the CAP could provide a convenient compromise between highly targeted AES and broad-brush horizontal policies, by eliminating or significantly reducing both private and public transaction costs of local-level schemes, and enhancing the precision of horizontal policies through a careful selection of the farming systems to support in each HNV farmland area. We therefore argued that such an agri-environment policy focused on HNV farming systems could profit from the precedent set by the recent CAP "Greening" and be valuably included under the CAP Pillar 1, as a "second generation Greening" measure to be considered in the forthcoming CAP revision (expected to start negotiations in 2018). By being shaped under CAP Pillar 1, this proposal would benefit from not requiring co-financing, and therefore find better acceptance from MS wishing to expand their farmland conservation policies, avoid the increasingly difficult funds-transfer from Pillar 1 to Pillar 2, release Pillar 2 funds to support other rural development policies, and would help prevent contradictions between Pillars' 1 and 2 policies such as those found in our case study. Besides, by operating on a quasi-automatic farm selection procedure, it would eliminate the need to establish contracts with farmers, thus avoiding the high transaction costs of existing voluntary AES. The administrative costs for managing such a measure would probably not exceed those currently incurred in controlling and verifying the single payment applications, as these are the baseline data for the entire framework. Given the increasing difficulties in strengthening the budget of CAP Pillar 2, measures such as the one proposed, recommending the transfer of environmental commitments from Pillar 2 to Pillar 1 instead of fund transfers in the opposite direction, could be a solution to improve the CAP environmental performance, without burdening the overall CAP budget.

The implementation of such a policy framework would require local/regional level field work to identify the HNV farming systems to be supported in each HNV farmland. This depends both on getting to know the farmland biodiversity components of higher conservation value, and their expected responses to landscape patterns and agricultural management practices. The farming system typology would require regular updates, possibly during the preparation of CAP reviews, to allow the entry of upcoming farming systems and discard those no more existing, which would probably correspond to an amount of work comparable to what is currently devoted to designing AES. While a consolidated EU-level mapping of HNV farmland may still be lacking, the measure could be triggered in Natura 2000 farmland or other nationally designated protected areas, paying a “HNV greening payment” to all farms operating preselected HNV farming systems, since the framework is easily reproducible across the EU, wherever IACS/LPIS data are available. Finally, the framework also showed the advantage of providing a more reliable methodology for estimating the level of the agri-environment payment, because it is based on farmers’ observed behavioural data, rather than estimations of AES participation costs or loss of income, as required by current CAP regulations. Therefore, the proposed framework to design agri-environment policy aimed at HNV farming systems seems arguably very promising in contributing to improve the overall cost-effectiveness of EU agri-environment policy.

This thesis is probably the first contribution to develop a methodology to design agri-environment policy focused on HNV farming systems sustainability, an idea that although previously suggested (e.g. Beaufoy and Marsden, 2011; Poux, 2013), had never been developed in detail so far.

Limitations of the framework and recommendations for future research

Despite the many advantages shown by the proposed framework, some limitations were identified, mostly related to shortcomings in the baseline data, particularities of the study area and difficulties in assessing the actual impacts of farming system changes on biodiversity, some of which call for future work.

The benefits of the baseline data (IACS/LPIS) were clearly evidenced, as these are a key feature to the all framework. Nevertheless, some technical problems were detected while using them, related to: 1) frequent changes in the nominal list of categories describing land uses / crop covers in the IACS data, usually related to adjustments in regulations, which can make it difficult to track a particular class over a number of years; 2) in areas dominated by large holdings, as in our case study, farmers often declare more than one land use / crop cover within the same land parcel, hindering the connection to LPIS data and forcing to consider only the dominant use, which inevitably implies an artificial simplification of the landscape mosaic and bias the computation of landscape metrics; 3) IACS data does not provide any information on possible complementarities between farms, in particular with regard to the rental or sale of pastures, which can mislead the computation of farms' stock density, and; 4) IACS data also provides no significant information to support a careful characterization of farmers' socioeconomic profile, which would be valuable to include in the farming system choice models.

All the empirical work was developed on an agricultural area with particular biophysical, structural and policy characteristics, and specific conservation problems, which may have determined the design of the framework. In fact, this is a relatively biophysically homogeneous area, dominated by large property and with specific restrictions on farmland management driven by local policies. For this reason, the general validity of the framework should be tested by reproducing it in regions with distinct characteristics, particularly in areas of small property, with greater climate, soils and orographic variability, where agriculture is often a secondary activity and therefore probably with distinct responses to economic incentives, as would be the case of many less favoured areas across the EU, often encompassing significant shares of HNV farmland (e.g. Keenleyside et al., 2014a; Pointereau et al., 2010a). Its implementation in common property areas, often found in remote mountain areas with high conservation value, would also provide an additional challenge to the framework, which would be worth testing in the future. The latent class approach, which in our case study did not produce effective results, should be considered in these future replications, as it is possible for farmers' socioeconomic heterogeneity to be higher in these regions.

An important limitation of our study was that we were not able to relate the observed farming system dynamics to actual impacts on species of conservation concern, which would enable to assess tangible results of farmland biodiversity conservation policy. This was mainly due to a lack of scientific support enabling such interpretation of results, despite this being a long studied area, featuring an important collection of environmental monitoring data (e.g. farmland bird counts) which, however, proved inadequate for the intended purpose (e.g. due to inadequate specification of land uses in the sampled areas or temporal mismatches between agricultural data and bird counts). In fact, many species of conservation concern in the area are medium to large birds with large home ranges and complex behaviour (e.g. lek mating system), whose population responses at landscape scales are still badly known. Even for smaller habitat specialists, relevant questions such as minimum patch sizes or response to fragmentation are still unanswered. As an alternative to our inability to directly link the conservation values to landscape and agricultural management changes, we resorted to habitat quality indicators inferred from landscape metrics and farming practices to assess the biodiversity potential of the different farming systems, which is still indirect evidence. This thesis therefore highlights the need to develop studies to strengthen the link between farming systems (or benchmarks on farming systems spatial distribution patterns) and specific biodiversity indicators, to assess the actual impacts of farming systems changes (FAO, 2011; Keenleyside et al., 2014a, 2014b; Tallis et al., 2008).

Before concluding, it is also worth leaving the suggestion for future work to reconsider the spatial approach, particularly if the limitations described in Chapter 1 can be duly taken into account.

Finally, it is worth noting that although the framework was developed and tested within the background of biodiversity conservation, its application to other agroecosystem services should not present major difficulties, as long as a link between farming systems and the provision of the ecosystem services in question can be established.

References

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

FAO, 2011. Payments for Ecosystem Services and Food Security. Food and Agriculture Organization of the United Nations.

Keenleyside, C., Beaufoy, G., Tucker, G., Jones, G., 2014a. High Nature Value farming throughout EU-27 and its financial support under the CAP. Report Prepared for DG Environment, Contract No ENV B.1/ETU/2012/0035, Institute for European Environmental Policy. London. doi: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. Prepared for the European Commission, DG Environment, Contract No ENV.B.2/ETU/2013/0046, Institute for European Environme.

Latacz-Lohmann, U., Hodge, I., 2003. European agri-environmental policy for the 21st century. Aust. J. Agric. Resour. Econ. 47:1, 123–139.

Pointereau, P., Doxa, A., Coulon, F., Jiguet, F., Paracchini, M.L., 2010. Analysis of spatial and temporal variations of High Nature Value farmland and links with changes in bird populations: a study on France, JRC Scientific and Technical Reports. Joint Research Centre of the European Commission.

Poux, X., 2013. Biodiversity and agricultural systems in Europe: drivers and issues for the CAP reform. Institut du développement durable et des relations internationales.

Stoate, C., Boatman, N.D., Borralho, R.J., Carvalho, C.R., de Snoo, G.R., Eden, P., 2001. Ecological impacts of arable intensification in Europe. J. Environ. Manage. 63, 337–365. doi:10.1006/jema.2001.0473

Tallis, H., Kareiva, P., Marvier, M., Chang, A., 2008. An ecosystem services framework to support both practical conservation and economic development. Proc. Natl. Acad. Sci. 105, 9457–9464. doi:10.1073/pnas.0705797105