Agriculture, Ecosystems and Environment 137 (2010) 348–357

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

Quantifying the impact of land-use change to European farmland bird populations

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ARTICLE INFO

Article history:
Received 6 November 2009
Received in revised form 3 March 2010
Accepted 5 March 2010
Available online 2 April 2010

Keywords:
Agricultural intensification
Agri-environment scheme
European Farmland Bird Index
Land abandonment
Risk assessment

ABSTRACT

The EU has adopted the European Farmland Bird Index (EFBI) as a Structural and Sustainable Development Indicator and a proxy for wider biodiversity health on farmland. Changes in the EFBI over coming years are likely to reflect how well agri-environment schemes (AES), funded under Pillar 2 (Axis 2) of the Common Agricultural Policy, have been able to offset the detrimental impacts of past agricultural changes and deliver appropriate hazard prevention or risk mitigation strategies alongside current and future agricultural change. The delivery of a stable or positive trend in the EFBI will depend on the provision of sufficient funding to appropriately designed and implemented AES. We present a trait-based framework which can be used to quantify the detrimental impact of land-use change on farmland bird populations across Europe. We use the framework to show that changes in resource availability within the cropped area of agricultural landscapes have been the key driver of current declines in farmland bird populations. We assess the relative contribution of each Member State to the level of the EFBI and explore the relationship between risk contribution and Axis 2 funding allocation. Our results suggest that agricultural changes in each Member State do not have an equal impact on the EFBI, with land-use and management change in Spain having a particularly large influence on its level, and that funding is poorly targeted with respect to biodiversity conservation needs. We also use the framework to predict the EFBI in 2020 for a number of land-use change scenarios. This approach can be used to guide both the development and implementation of targeted AES and the objective distribution of Pillar 2 funds between and within Member States. We hope that this will contribute to the cost-effective and efficient delivery of Rural Development strategy and biodiversity conservation targets.

1. Introduction

The accession of 10 new Member States to the European Union (EU) in 2004 and a further two in 2007 brought approximately 60 million ha more farmland under the governance of the Common Agricultural Policy (CAP). CAP came into force in the early 1960s and remains the main agricultural policy tool of the EU. Until recently, the structure of CAP created a protected market with guaranteed prices and it was the driving force behind both the intensification of agriculture and land abandonment that has occurred in the EU15 (the first 15 Member States of the EU) over recent decades (Bignal, 1998; MacDonald et al., 2000; Donald et al., 2002). These land-use and management changes, and associated losses of resources from the agricultural landscape, have led to widespread and significant declines of farmland birds and other wildlife (Fuller et al., 1995; Matson et al., 1997; Green et al., 2005; Gregory et al., 2005). Despite the introduction of the Single Payment Scheme under the 2003 CAP reforms, which decoupled payments to farmers from production levels, there is great concern that accession and exposure to CAP could result in losses of agricultural biodiversity in the new Member States similar to those recorded in the EU15 (EEA, 2004; Donald et al., 2006). This is of particular concern because farmland in the newly joined Member States, mostly former communist countries of central and eastern Europe, supports internationally important populations of many species (Hagemeijer and Blair, 1997; BirdLife International, 2004).

The EU has set a target of halting biodiversity loss by 2010 (European Council, 2001) and, in adopting the European Farmland Bird Index (EFBI) as a Structural and Sustainable Development Indicator, identified farmland bird trends as a proxy for wider biodiversity health on farmland. This indicator, based on data collected under the Pan-European Common Bird Monitoring Scheme

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doi:10.1016/j.agee.2010.03.005

(PECBMS: http://www.ebcc.info/pecbm.html), shows that farmland bird populations have nearly halved across Europe since 1980. Given that farmland of one kind or another comprises approximately 50% of the European land surface, managing the likely agricultural changes in newly joined Member States and the predicted changes in land-use patterns across Europe, driven by factors such as climate change, the introduction and expansion of bio-energy crops, modernisation, specialisation and further land abandonment (MacDonald et al., 2000; Rounsevell et al., 2005, 2006), will be fundamental to achieving this broader biodiversity goal. In this context, land abandonment can be the result of different processes and drivers in different regions, including a loss of grazing or active afforestment. Agri-environment schemes (AES), under which farmers are paid to protect and enhance the environment and to preserve landscape and historical features, represent the main available mechanism to mitigate such impacts and prevent or reduce declines in farmland biodiversity over large areas (Vickery et al., 2004). Funding for agri-environment programmes has been available under CAP since 1992 (EEC Regulation 2078/92) and the provision of such schemes became compulsory within all EU countries in 2003 (Kleijn and Sutherland, 2003). Funding for AES is provided under the Rural Development component (Pillar 2) of the CAP, the budget for which has been set at €87bn for the 2007–2013 programming period. Whilst Pillar 2 funding is distributed between Member States according to agricultural area, agricultural employment and GDP, Member States have a high level of flexibility with regard to the allocation of these funds across a range of rural development measures which include AES but also cover measures for increasing competitiveness of agricultural and forestry sectors, improving the quality of life in rural areas and supporting the diversification of the rural economy. As a consequence, the actual allocation of funds specifically to AES is extremely low in some Member States and may not reflect the health of farmland biodiversity. Changes in the EFBI over coming years are likely to reflect the extent to which AES have been able to both offset the detrimental impacts of past agricultural changes that are driving current biodiversity loss and to deliver appropriate hazard prevention or risk mitigation strategies alongside current and future agricultural change. The delivery of a stable or positive trend in the EFBI is likely to depend firstly on the successful and cost-effective design and implementation of AES that target the key drivers of bird population declines in agricultural landscapes (Kleijn and Sutherland, 2003; Butler et al., 2007) and, secondly, on the appropriate distribution of Pillar 2 funds between and within Member States to ensure that schemes with the potential to deliver resources to significant proportions of farmland bird populations receive greater support.

Butler et al. (2007) published a trait-based risk assessment framework capable of predicting the impacts of agricultural change on biodiversity. Using UK farmland birds as a model system, they showed that a species' response to land-use change could be predicted by assessing the impact of that change on the species' key resource requirements. Furthermore, they used this framework to identify the main drivers of farmland bird population declines in the UK and identified a clear disparity between the sources of risk in the agricultural landscape and patterns of risk mitigation delivery through the agri-environment scheme Entry-Level Stewardship (ELS). Here we show that a similar approach can be used to assess the impacts of agricultural change on farmland bird population trends at a pan-European scale and assess the relative influence of the twenty countries that provide data to the EFBI on its current level. Based on these relationships, we go on to model the likely impacts of four land-use change scenarios on farmland bird populations and the EFBI to 2020.

2. Methods

2.1. Calculation of EFBI

The PECBMS generates national population indices for 135 bird species (Gregory et al., 2005, 2007, 2008). Supra-national indices for four European regions, North (Finland, Norway & Sweden), West (Austria, Belgium, Denmark, Germany, Holland, Ireland, Switzerland & United Kingdom), East (Czech Republic, Estonia, Latvia, Hungary & Poland) and South (France, Italy, Portugal & Spain), are calculated as the weighted average of a species' trend in the constituent countries; trends are weighted by relative breeding population size of each bird species in each country (taken from BirdLife International, 2004). Pan-European trends are calculated based on the weighted average of regional trends, again based on the relative proportion of the European breeding population found in each region. Multi-species indices, such as the EFBI, are calculated at a regional or pan-European scale by calculating the geometric mean of contributing species' trends. Full details of trend calculations can be found in Gregory et al. (2005) or on the European Bird Census Council website (www.ebcc.info). Our analyses of the impacts of land-use change on farmland bird populations cover the 20 countries that contributed bird population trend data to the EFBI in 2005; a new monitoring scheme was initiated in Bulgaria in 2004 and now contributes to the PECBMS.

2.2. Quantifying risk in current agricultural landscapes

The underlying structure of the risk assessment framework and methods for quantifying risks associated with land-use change have been published in detail (see Butler et al., 2007 and supporting online material (SOM)). In brief, the risk of agricultural change x to species y is defined as the degree of coincidence between the environmental impacts of that change and the resource requirements of that species, adjusted for the species' ecological resilience, defined by the breadth of its resource requirements and its reliance on farmland for those resources. Using these definitions, we developed a risk assessment framework for European farmland bird species. Firstly, we constructed a resource requirements matrix for 54 species by gathering data on their summer and winter diets, summer and winter foraging habitat and nest site location (Cramp, 1998). These 54 species include all those contributing to the EFBI and those with agriculture/grassland listed as one of their habitat classifications in Tucker and Evans (1997) for which sufficient data are available to generate pan-European population trends. A list of species included in our analyses is provided in the SOM. Whilst we assumed that species' resource requirements would not vary significantly between countries, it was considered likely that their reliance on farmland for these resources might. We therefore asked a number of ornithological experts in each country to independently score species as having either a major, moderate or minor reliance on farmland habitat or as not being present as a breeding species. The modal response for each country was used in risk score calculations. If two categories received equal support the higher reliance category of the two was used in calculations. Details of the reliance scores provided are presented Tables S1–S4, Supplementary data in the SOM. The migration strategy and location of wintering grounds of each species were also determined, with species classified as being either resident, partial migrants, full migrants with part or all of their population remaining in Europe or full migrants with over-wintering grounds outside Europe (Cramp, 1998). Wintering grounds of migrant species remaining in Europe were identified at a regional rather than country level because data on the precise wintering locations for most breeding populations are not available.

Table 1
Main environmental impacts, likely to have adverse population scale effects on farmland birds, of the six components of agricultural intensification and two components of agricultural abandonment used in the risk assessment framework validation process. S, summer; W, winter.

Component of agricultural intensification	Key impacts[a]
Spring to autumn sowing	Loss of crop foraging habitat (S & W) Loss of crop seeds (W) Loss of crop nest sites
Loss on non-cropped habitat	Loss of margin foraging habitat (S & W) Loss of hedge foraging habitat (S & W) Loss of margin nest sites Loss of hedge nest sites
Increased agrochemical inputs	Loss of crop invertebrates (above ground) (S & W) Loss of crop plant material (S & W) Loss of crop seeds (S & W)
Land drainage	Loss of margin foraging habitat (S & W) Loss of crop invertebrates (soil) (S & W) Loss of crop invertebrates (above ground) (S & W) Loss of margin invertebrates (soil) (S & W) Loss of margin invertebrates (above ground) (S & W) Loss of margin vertebrates (S & W) Loss of hedge invertebrates (soil) (S & W) Loss of margin nest sites
Switch from hay to silage and earlier harvesting	Loss of crop foraging habitat (S) Reduced nesting success (increased mechanical damage)
Intensified grassland management	Loss of crop invertebrates (soil) (S & W) Loss of crop invertebrates (above ground) (S & W) Loss of crop seeds (S & W) Loss of crop vertebrates (S & W) Loss of crop nest sites Reduced nesting success (increased trampling by stock)
Loss of semi-natural grassland	Loss of crop habitat (S & W) Loss of crop nest sites
Afforestation	Loss of crop habitat (S & W) Loss of crop nest sites

[a] 'Crop' refers to cropped areas of the landscape rather than the actual crop itself, 'margin' refers to non-cropped open habitats in the landscape and 'hedge' refers to structural vegetation elements in the landscape such as hedgerows and trees.

2.3. *Validation of the risk assessment framework*

2.3.1. *Components of agricultural intensification and land abandonment*

To validate the framework, we assessed the impact of eight widespread land-use and management changes, known to have significant detrimental impacts on farmland birds across Europe over recent decades (Laiolo et al., 2004; Donald et al., 2006; Wretenberg et al., 2006; Reif et al., 2008), for their impact on food abundance, foraging habitat availability, nesting habitat availability and nesting success (Table 1). Six of these changes are associated with agricultural intensification – switch from spring to autumn sowing, increased agrochemical inputs, loss of non-cropped habitats, land drainage, the switch from hay to silage and increased stocking densities – and two with land abandonment – loss of semi-natural grassland and active afforestation.

2.3.2. *Risk score calculation*

To reflect within-species differences in reliance on farmland habitat and migration strategy between countries, a pan-European risk score for each species was calculated in three stages. Firstly, we calculated the potential summer and winter risk score accrued by each species in each country if it was present in that season (Stage 1). Individuals from migrant species are not necessarily exposed to the winter risk in the country in which they breed, rather they are exposed to the winter risk in the country or countries in which they over-winter. We therefore calculated the total risk score for breeding populations of each species in a given country by combining their potential summer risk score for that country with their potential winter risk scores for the locations where the breeding birds from that country over-wintered (Stage 2). Finally, we calculated a pan-European risk score as a weighted average based on relative population size in the constituent countries (Stage 3). The details of each stage are outlined below.

2.3.2.1. Stage 1. Using the resource requirements matrix, we identified every species likely to have been adversely affected by the detrimental changes in the quantity and quality of resources associated with each agricultural change and calculated potential risk scores for each species in each country. These risk scores reflect the proportion of a species' resource requirements affected by that change and their reliance on farmland to provide those resources (see Butler et al., 2007 and SOM for full details). To accommodate migration patterns in calculations of total risk (see Stage 2), summer and winter risks were summed separately across the eight agricultural changes at this stage.

2.3.2.2. Stage 2. The potential summer risk score calculated for a given species in a given country was assigned as the summer risk score accrued by that species in that country if the species was recorded as breeding there. The winter risk score accrued by breeding populations of each species in each country was calculated based on migration strategy. For resident populations, the potential winter risk score calculated for the country in which they breed was assigned as the winter risk to which they were exposed. For partially and fully migrant populations, winter risk scores were calculated as the average of winter risk in the region(s) where they over-winter. These regional winter risk scores were calculated as the weighted average of the potential winter risk scores for each constituent country in which the species is known to over-winter.

In these calculations it was assumed that the breeding population from a given country was distributed between the over-wintering regions identified and between the constituent countries within each over-wintering region in proportion to their Usable Agricultural Area (UAA) (http://faostat.fao.org).

2.3.2.3. Stage 3. Total risk scores for each species in each country were calculated by summing summer and winter risk scores. A pan-European risk score was then calculated for each species as the average of its total risk scores in each constituent country, weighted by the relative breeding population size in each country (BirdLife International, 2004). This risk score reflects the pan-European impact of past agricultural intensification and land abandonment on each species.

2.3.3. BASIC and SCALE frameworks

Next, we used General Linear Modelling (GLM) to investigate the relationship between pan-European population trends and pan-European risk score. We would expect the two to be highly correlated if, as we strongly suspect, farming practices and bird population trends are causally linked. As the EFBI reflects population changes since 1980, we only included the 39 species in our dataset for which population trend data from 1982 onwards were available in these analyses. We used 1982 rather than 1980 as the cut-off point as this allowed 5 extra species to be included following instigation of population monitoring in eastern Europe. By exploring the relationship between risk score and population growth rate over the period in which the risk was accrued, it is possible to use parameter estimates from the derived model to make predictions about farmland bird population responses to novel land-use changes (see Section 2.7 below). We refer to this as the BASIC framework.

By assuming that annual population growth rates have been constant between 1980 and 2005, the BASIC framework does not link spatial and temporal variation in risk accrual with population dynamics. It therefore cannot be used to make predictions about population responses to changes in the intensity and extent of existing land-uses. We therefore constructed a second framework (hereafter referred to as the SCALE framework), re-calculating species' risk scores after introducing a scaling mechanism. This was based on the assumption that, whilst there is a potential risk associated with an agricultural change for any given species, the extent to which that agricultural change occurs will influence a species' exposure to that risk and therefore its impact on population trend. Donald et al. (2001) showed that there was a linear relationship between farmland bird trends in a given country and the level of agricultural intensification, indicated by cereal yield, in that country. We therefore assumed that there was a linear relationship between the levels of risk associated with an agricultural change to which farmland birds are exposed and the rate at which that change has occurred. Having identified the species likely to be affected by each component of agricultural change and calculated potential summer and winter risk scores for each country (Section 2.3.2.1), we multiplied the scores for the six changes associated with agricultural intensification by the rate of change of cereal yield (tonnes/ha) and the scores for the two changes associated with land abandonment by the rate of change of UAA (as a proportion of national land area). Rates of change of intensification and land abandonment were calculated between 1980 and 2005, the same period over which pan-European population trends were estimated. We could not scale the risk associated with each individual agricultural change separately because appropriate data were not available. Rates of change were estimated at the regional level because agricultural data were not available for all countries in all time periods; regional rates of change were calculated as the average across constituent countries for which data were available (http://faostat.fao.org) and applied to all countries within each region. Having scaled potential summer and winter risk scores for each species in each country, Stages 2 and 3 of risk score calculations were completed as detailed above.

2.3.4. Alternative model structures

Our risk scoring system assumes that each source of risk has equal weighting in terms of its relationship to population growth. To critically assess this assumption, we constructed a series of more complex, alternative models that decomposed the total risk score into various component parts, allowing the weighting of different sources of risk to vary. Total risk score was decomposed firstly into risk accrued from loss of diet-related resources and nest-related resources, secondly into risk accrued from loss of resources in summer and loss of resources in winter and, thirdly, into risk accrued from intensification-related changes and abandonment-related changes in resource availability. Comparisons of these models using Akaike's Information Criterion (AIC) showed that our assumption was reasonable. When assessing the BASIC framework, the most parsimonious model of population growth rate only included total risk as a predictor variable, with none of the other models receiving substantial support (ΔAIC > 2 in all cases). When assessing the SCALE framework, the most parsimonious model again only included total risk as a predictor variable but the model with risk decomposed into diet- and nest-related risk also received some support (ΔAIC < 2)(see Tables S5 and S6, Supplementary data in SOM for full details). Parameters from these three models were used to make predictions about the impact of land-use change scenarios (see below).

2.4. Predicting the EFBI from the population growth model

We assessed the performance of the BASIC framework by using parameter estimates derived from the GLM detailed above to predict the current EFBI and compared this prediction to the actual value. The current EFBI includes 36 species but three of them, *Burhinus oedicnemus*, *Emberiza melanocephala* and *Lanius minor*, have only been incorporated into the index since 2005 as sufficient data to generate accurate population trends only recently became available. These species were therefore excluded from all analyses and predicted and actual EFBI levels were calculated based on the population trends of the remaining 33 species. Population trend data for 23 of these species are available from 1982 onwards and these were included in the set of 39 species used in the validation process outlined above. For these species, a 'leave-one-out' method of jack-knifing was used to calculate predicted annual population growth rates. Each species was excluded from the dataset in turn and parameter estimates from a GLM including the remaining 38 species were used to predict the population growth rate of the excluded species. The remaining 10 EFBI species have been incorporated into the index since 1982 as sufficient data have become available (1989–1 species, 1990–2 species, 1996–7 species; see Table S7, Supplementary data in SOM for details). Parameter estimates from the global model derived in the validation process, including all 39 species, were used to predict population growth rates for these species. Each predicted population growth rate generated in this way is actually the mean of a distribution of possible values described by this mean and its associated standard deviation. To calculate confidence intervals in the predicted EFBI, we first generated a distribution of predicted population growth rates for each species based on the jack-knifed models. Next, we randomly sampled a population growth rate from each distribution and used these growth rates to calculate the predicted EFBI using the same methods that are used to calculate the actual EFBI, with predicted growth rates for the 10 species added since 1982 incorporated into the index in the appropriate year (Gregory et al., 2005). We repeated

this bootstrapping process 1000 times to generate a distribution of predicted EFBIs, from which we estimated the mean. The 25th and 975th EFBI values of the 1000 bootstrapped samples, when ordered by size, were used to estimate the 95% confidence limits of this mean EFBI. In this way, we generated a predicted EFBI with associated confidence limits that could be compared with the observed value associated with past agricultural changes. This process was repeated to assess the performance of the SCALE framework.

2.5. Identifying key sources of risk in current agricultural landscapes

We identified the main sources of risk to farmland birds in current agricultural landscapes using the scores derived from the validation of the BASIC framework. The scores from all 54 farmland bird species were summed to give a total landscape risk score. This was then broken down according to the source of that risk; we calculated the proportion of total risk arising from changes to food abundance, either through the loss of foraging habitat or the loss of prey items in existing habitat, and nesting success, either through the loss of nesting habitat or reduced success in existing habitat, in cropped areas, margins and hedgerows. Note that margin refers to non-cropped open habitats in the landscape and hedgerow refers to structural vegetation elements in the landscape such as hedgerows and trees. Again, this process was repeated using scores derived from the validation of the SCALE framework.

2.6. Assessing the relative contribution of member states to the EFBI

Given that species' pan-European risk scores are related to their pan-European population trends (see Section 3.1), it is possible to estimate the relative contribution of each member state to the current level of the EFBI by assessing its relative contribution to the risk score of each species included in the EFBI. These analyses were undertaken separately on risk scores generated from the validation of the BASIC and SCALE frameworks. To do this, we allocated the risk score of each breeding population to the countries in which that risk had been accrued. Thus the summer risk component of a breeding populations' total risk score was assigned to the country in which it bred whilst the winter risk score component was assigned to the country or countries in which it over-wintered. Since the wintering locations of non-resident populations were only identified at a regional level, their winter risk was distributed between the over-wintering regions identified and between the constituent countries within each over-wintering region in proportion to their UAA. Take, for example, the Swedish breeding population of skylarks. Using the BASIC framework, a total risk score of 17 was calculated for this population, with 12 accrued in the summer and 5 accrued in the winter. However, the Swedish population of skylarks is migratory and its winter risk reflects detrimental changes in resource availability in the West and South region, where it over-winters, rather than changes that have occurred in Sweden. For this population, all of the summer risk but none of its winter risk was assigned to Sweden. Instead, the winter risk was divided between the countries in the West region and South region according to their relative UAA. For each country, we then summed the level of risk contributed to all breeding populations of each species so that, for example, the score derived for skylark in France represented the risk accrued by the breeding population in France plus any winter risk accrued by breeding populations in the other countries as a consequence of France being one of their over-wintering destinations. The sum of every country's score for a given species was identical to that species' pan-European risk score calculated in Section 2.3.2.1. We then calculated the proportional contribution of each country to the pan-European risk score of each species in the EFBI and summed these values across species to identify the relative contribution of each country to the European risk scores of all species in the EFBI and hence to the current level of the EFBI.

Finally, we compared the relative contribution of each country to the EFBI with the allocation of Pillar 2 funding to Axis 2 measures, including AES, during the 2007–2013 budget period. We controlled for UAA to explore the relationship between these two variables at a per unit area scale. A positive relationship would indicate that funding is being directed in a targeted fashion among Member States, with more money available for mitigation measures in countries contributing relatively more to the EFBI, although it would say nothing about the implementation or effectiveness of any such measures.

2.7. Scenario assessment

The BASIC framework can be used to predict the likely impacts on farmland bird populations of novel land-use and management changes, for which no baseline against which the scale of that change can be compared is available. The SCALE framework can be used to predict the likely impacts on farmland bird populations of a change in the intensity or extent of existing agricultural practices and land-uses. Here we assess the impacts of four scenarios of land-use change over the next decade to demonstrate the range of application of the two risk assessment frameworks.

2.7.1. Scenario 1: current conditions persist

This scenario extends current conditions in agricultural landscapes through to 2020, with current rates of intensification and abandonment in each of the four regions maintained for the next decade. Predictions of the expected EFBI under this scenario were made using both the BASIC and SCALE frameworks.

2.7.2. Scenario 2: loss of compulsory set-aside

The removal of set-aside support across Europe following the 2008 CAP 'Health check' is likely to lead to a reduction in the availability of over-wintered stubbles in the agricultural landscape. Any over-wintered stubbles that do persist in the landscape can be expected to have reduced weed seed availability due to associated changes in crop rotation and management. The temporal changes in vegetation structure over the course of the summer will also lead to a reduction in summer foraging habitat and nest site availability due to reduced access. As this scenario represents a novel land-use change we used the BASIC framework to calculate the risk to each species as a consequence of these changes in resource availability.

2.7.3. Scenario 3: accelerated agricultural intensification in the East region

This scenario represents a change in the intensity of existing land-use and practices so the SCALE framework was employed here. We specified that the level of agricultural intensification in the East region, as indicated by cereal yield, would reach the 2005 levels in the West region by 2020. We calculated the rate of change in cereal yield between 2005 and 2020 required to achieve this level and set the intensification scaling factor for the East region to this value. We assumed that no further land abandonment occurred and that cereal yields in the West, North and South regions remained at 2005 levels so the scaling factors for these components were all set to zero.

2.7.4. Scenario 4: continued land abandonment across Europe

This scenario represents a change in the scale of existing land-use and practices so the SCALE framework was again employed here. We assessed the consequences of three different levels of land abandonment on farmland bird populations. Specifically, we calculated the rate of change in UAA which would result in a 5%, 10% and

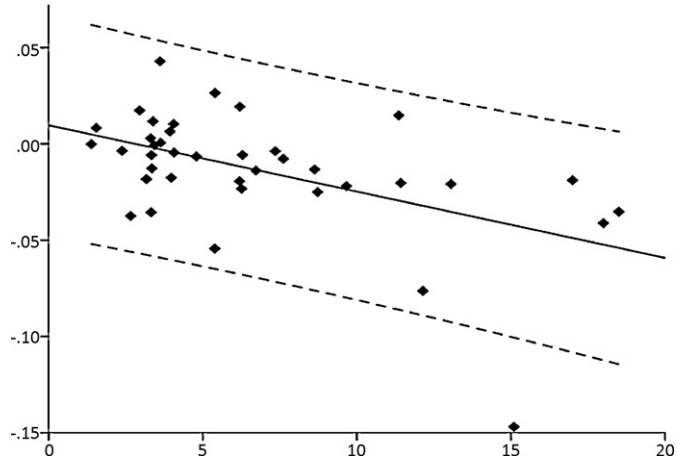

Fig. 1. The relationship between total risk score, derived from validation of the BASIC framework against past agricultural change and species' annual population growth rate between 1980 and 2005. Solid black line shows fitted model for all species ($y = 0.01 - 0.003x$, $R^2 = 0.26$), dashed lines show 95% confidence intervals.

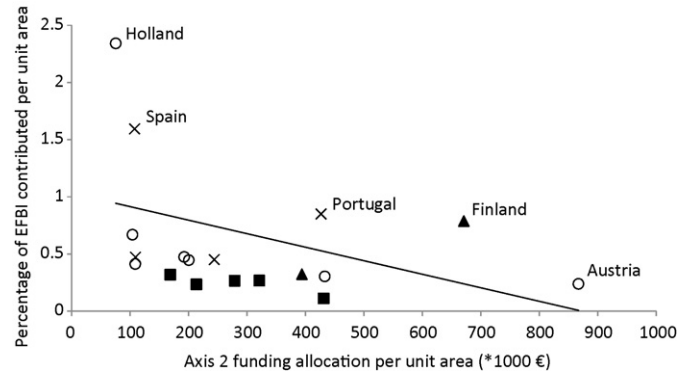

Fig. 2. Relationship between the relative contribution of a country to the 2005 level of the EFBI estimated using the BASIC framework and its Axis 2 funding allocation for the 2007–2013 programming period. Symbols represent regions: West (○); North (▲); East (■); South (X). Solid line shows fitted model for all countries ($y = 1.0327 - 0.0012x$, $R^2 = 0.21$). Particular countries are highlighted with reference to the text. *Note*: Norway and Switzerland are not included because they are not EU Member States so do not receive CAP funding.

15% loss in UAA in each country by 2020. For each level of abandonment, we set the abandonment scaling factor in each region as the average rate of change across constituent countries. We assumed that cereal yields in each region remained at 2005 levels and set the scaling factor for this component to zero.

For Scenario 1, predicted population growth rates for each species between 1980 and 2005, derived from the validation process described above, were applied from 2005 to 2020. For the other three scenarios we followed the same procedure used in the validation of the BASIC and SCALE frameworks discussed above (Section 2.3, Stages 1–3) to derive predicted population growth rates for each species. Firstly, we identified all species likely to be affected under each scenario and quantified the level of impact on them based on the expected changes in resource availability. To predict the population trends of species in the resultant landscape, the risk assessment score calculated for each scenario was added to the species' score derived from the validation process; the scores from the validation process represent the levels of risk in current landscapes into which the scenario changes are introduced. Parameter estimates from the GLMs derived during the validation process detailed in Section 2.3 above were then used to calculate predicted population growth rates in the resultant agricultural landscapes. Finally, predicted EFBIs in 2020 were calculated based on population trend estimates derived from predicted population growth rates in the current landscape (1980–2005) and the predicted population growth rates in the altered landscape (2006–2020). Confidence limits for the predicted EFBIs were generated using the bootstrapping procedure described above.

3. Results

3.1. Validation of BASIC and SCALE frameworks

Risk scores derived from the assessments of the environmental effects of agricultural intensification and land abandonment across Europe using both the BASIC and SCALE framework were significantly related to the annual population growth rates of farmland bird species between 1980 and 2005, respectively. Higher risk scores were associated with species with negative population growth rates and therefore experiencing population declines (BASIC framework: $F_{1,39} = 13.0$, $P = 0.001$, Fig. 1; SCALE framework: $F_{1,39} = 6.98$, $P = 0.01$). The predicted EFBIs, based on population changes between 1980 and 2005, were 0.65 (±SE: 0.52 and 0.81, BASIC framework) and 0.66 (±SE: 0.52 and 0.83, SCALE framework) compared to the actual EFBI over the same time period of 0.64, respectively.

3.2. Key sources of risk in current agricultural landscapes

The validation of both the BASIC and SCALE frameworks suggested that more than 76% of the total risk accrued by the 54 farmland bird species assessed is associated with detrimental changes that have occurred in the cropped area of agricultural landscapes, with three-quarters of this linked to reductions in the quantity or quality of food resources and one quarter with reduced nesting success. The majority of the remaining risk was associated with a reduction in the quantity or quality of food resources in field margins (Table 2).

3.3. Relative contribution of Member States to EFBI

The relative contribution of each country to the position of the EFBI in 2005, based on the risk accrued by each EFBI species in that country, is shown in Table 3. For 12 of the 33 EFBI species, including widespread and emblematic species such as *Alauda arvensis*, *Emberiza citrinella* and *Vanellus vanellus*, risk is accrued in all 20 countries whilst for a further 12 species risk is accrued in each of the four regions if not in every constituent country of each region. Details of the relative contribution of each country to the risk score of each species are provided in Tables S8 and S9, Supplementary data in the SOM. However, results suggest that agricultural change in Spain had the greatest impact on the EFBI, more than three times the impact of agricultural change in Poland, the second ranked country. For six EFBI species, more than 90% of their risk has been accrued in Spain (*Calandrella brachydactyla*, *Galerida theklae*, *Lanius senator*, *Melanocorypha calandra*, *Oenathe hispanica*, *Petronia petronia*), with a further two species (*Anthus campestris* and *Sturnus unicolor*) accruing more than 75% of their risk there.

Unexpectedly, there is actually a negative relationship between Axis 2 budget allocation and the relative contribution of each Member State to the level of the EFBI in 2005, although the relationship is only significant for SCALE framework risk scores (Fig. 2; BASIC: $r = -0.25$, $N = 18$, $P = 0.31$; SCALE: $r = -0.49$, $N = 18$, $P < 0.04$). The inclusion of national co-financing to Axis 2 measures, in addition Pillar 2 allocations, does not alter the direction of this relationship (BASIC: $r = -0.32$, $N = 18$, $P = 0.2$; SCALE: $r = -0.37$, $N = 18$, $P = 0.13$).

Table 2
The relative distribution of risk accrued from different sources.

Landscape component[a]	Source of risk	Proportion of total risk accrued	
		BASIC framework	SCALE framework
Crop	Reduction in quantity or quality of food resources	0.59	0.59
	Reduction in nest site availability and success	0.19	0.18
Margin	Reduction in quantity or quality of food resources	0.10	0.11
	Reduction in nest site availability and success	0.05	0.05
Hedgerow	Reduction in quantity or quality of food resources	0.03	0.03
	Reduction in nest site availability and success	0.04	0.04

[a] 'Crop' refers to cropped areas of the landscape rather than the actual crop itself, 'margin' refers to non-cropped open habitats in the landscape and 'hedge' refers to structural vegetation elements in the landscape such as hedgerows and trees.

Table 3
Relative contribution of the twenty countries that provide data for the EFBI to its 2005 level, based on the proportion of risk accrued by each EFBI species in each country.

RANK	BASIC framework		SCALE framework	
	Country	Proportion of total risk (%)	Country	Proportion of total risk (%)
1	Spain	37.82	Spain	41.23
2	Poland	11.82	France	13.96
3	France	11.62	Germany	8.07
4	Italy	5.78	United Kingdom	7.20
5	Hungary	5.75	Italy	6.63
6	United Kingdom	4.96	Holland	4.51
7	Germany	4.46	Poland	4.23
8	Holland	3.54	Portugal	3.22
9	Portugal	2.80	Denmark	1.81
10	Finland	2.08	Finland	1.79
11	Latvia	1.60	Hungary	1.37
12	Sweden	1.56	Ireland	1.31
13	Czech Republic	1.53	Sweden	1.03
14	Denmark	1.05	Austria	0.78
15	Ireland	0.98	Belgium	0.57
16	Norway	0.91	Switzerland	0.56
17	Estonia	0.66	Latvia	0.55
18	Austria	0.44	Norway	0.54
19	Belgium	0.36	Czech Republic	0.40
20	Switzerland	0.29	Estonia	0.22

3.4. *Impacts of land-use change scenarios on EFBI*

Our models predicted that the EFBI will continue declining if current conditions persist across European agricultural landscapes. By 2020, it was predicted to fall by around a quarter to between 0.41 and 0.50, depending on the model of population growth rate used (Table 4). Each of the other scenarios explored introduced additional risk into the agricultural landscapes, by reducing the abundance and/or availability of key resources, and the predicted EFBI in 2020 under all three was therefore lower than that predicted if current conditions were to persist. In light of the loss of compulsory set-aside, 43 of the species included in these analyses, including 28 of the 33 EFBI species, can be expected to experience reduced population growth rates. Indeed, the predicted EFBI in 2020 under this scenario was 8% lower than that predicted if current conditions persist. Accelerated agricultural intensification in east Europe was predicted to have a large detrimental impact on the EFBI, with 2020 levels expected to be between 20% and 25% lower under this scenario than if current conditions persist. Higher rates of land abandonment led to greater reductions in the EFBI, with each 5% decline in the UAA predicted to cause a 2–2.5% lowering of the EFBI by 2020 (Table 4). Details of the predicted population growth rate for each species under each scenario are provided in Table S10, Supplementary data in the SOM.

Table 4
Predicted EFBI in 2020 derived from risk scores associated with continued current management (Scenario 1), the loss of compulsory set-aside (Scenario 2), accelerated agricultural intensification in east Europe (Scenario 3) and continued land abandonment (Scenario 4). Mean predicted EFBI values and 95% confidence limits (in parentheses) were generated from three alternative models of population growth rate (see text for details). The predicted EFBI for 2005, generated from the three alternative models, is also shown. For comparison, the EFBI in 2005 calculated from actual population growth rates was 0.64.

Model	Predicted EFBI in 2005	Predicted EFBI in 2020					
		Scenario 1	Scenario 2	Scenario 3	Scenario 4		
					5% loss of UAA	10% loss of UAA	15% loss of UAA
BASIC – total risk	0.65 (0.52–0.81)	0.50 (0.37–0.63)	0.46 (0.34–0.60)	–	–	–	–
SCALE – total risk	0.66 (0.52–0.83)	0.50 (0.38–0.64)	–	0.39 (0.30–0.51)	0.48 (0.37–0.64)	0.47 (0.34–0.62)	0.46 (0.34–0.60)
SCALE – diet-related risk plus nest related risk	0.60 (0.47–0.74)	0.41 (0.31–0.55)	–	0.30 (0.23–0.39)	0.40 (0.30–0.53)	0.39 (0.29–0.52)	0.38 (0.29–0.49)

4. Discussion

We have shown that a trait-based framework, whereby the impacts of land-use and management change are defined in terms of consequential changes in the abundance and availability of key foraging and nesting resources, can be used to assess and quantify the likely risk of agricultural change to farmland birds at a pan-European scale. Our assessment of the impacts of recent agricultural intensification and land abandonment across Europe showed that more than three-quarters of the risk accrued by the species assessed was associated with detrimental changes in resource availability in the cropped area of agricultural landscapes. Our analyses also suggest that agricultural changes in Spain are likely to have the greatest influence on the level of the EFBI and that the levels of financial support allocated for environmental management across Member States do not reflect the relative levels of risk in their agricultural landscapes.

4.1. Drivers of decline and targeting resource provision

The delivery of a stable or positive trend in the EFBI is likely to depend on two factors. Firstly, it requires the development and implementation of AES that target the key drivers of farmland bird declines and secondly, requires that AES with the potential to deliver resources to significant proportions of farmland bird populations receive greater support (Kleijn and Sutherland, 2003). We have shown that detrimental changes in the quantity and quality of foraging and nesting resources in the cropped area of agricultural landscapes have the greatest impact on farmland bird population dynamics. The recent loss of support for compulsory set-aside appears likely to exacerbate this as it is expected to lead to a further decline in cropped area resources. Indeed our predictions of farmland bird trends under this scenario showed that, by 2020, the EFBI might be 8% lower than if current conditions persist in agricultural landscapes. Our models suggest that accelerated agricultural intensification in the East region is likely to have a strong negative effect on EFBI level. Poland and Hungary in particular hold relatively large populations of many farmland bird species (BirdLife International, 2004) so detrimental changes in the population trajectory of EFBI species in these countries, brought about by declines in resource abundance and availability, can have a strong influence on the EFBI level. Further land abandonment can also be expected to have detrimental effect on population trends and, therefore, the EFBI level. Whilst the impact of land abandonment on the EFBI does not appear as influential as that of further agricultural intensification, it should be noted that land abandonment is likely to be associated with specific regions of certain countries rather than being more uniformly distributed within and between countries as might be expected from agricultural intensification. Thus, at a local level, land abandonment may have particularly detrimental impacts on farmland bird populations and may well be associated with declines in other indicators of rural development, such as employment. It should also be noted that any detrimental effects of land abandonment are likely to be more delayed than the effects of intensification. Although our model is validated against land-use changes and population trends over a 25 year period (1980–2005), it is therefore possible that the impacts of any land abandonment later in that period may not have been reflected in population trajectories by 2005. As a consequence, predicted population responses to future land abandonment may be conservative.

Delivering key foraging and nesting resources back into the cropped area of agricultural landscapes appears fundamental to reversing current declines of farmland birds. This should be reflected in AES design and implementation as they are the main policy instruments for delivering such resources over large areas (Vickery et al., 2004). AES are developed through a careful balancing of ecological, socio–economic, administrative and political interests (Buller et al., 2000). Whilst trade-offs between conflicting demands will inevitably be made during this process, it is important that the emphasis on delivering beneficial management to the cropped area is not lost if AES are to be successful. Crucially, it is not enough to get the scheme design right if the implementation strategy negates this. For example, in England, where a loss of resources in the cropped area has also been highlighted as the key driver of farmland bird declines, assessments of the ELS scheme have shown that, whilst the range of management options available is weighted towards resource delivery in the cropped area, the freedom given to land-owners and managers to select their preferred management options has resulted in the main emphasis of current agreements being on margin and hedgerow management (Butler et al., 2007; Boatman et al., 2007). This failure to target the key drivers of decline has led to predictions that, in its current format, ELS is unlikely to deliver its biodiversity conservation objectives (Butler et al., 2007).

The order of countries in Table 2 is determined by the breeding population size, reliance on farmland and migration strategy of each species and reflects the relative sensitivity of the EFBI to land-use change, both historic and future, in each country. Whilst most EFBI species accrue risk across Europe, our results suggest that agricultural changes in contributing countries do not have an equal impact on the EFBI. The current level of the EFBI is driven to a great extent by risk accrued in countries with large farmed areas, which support large populations of farmland birds, and in countries in the South region. It is evident that land-use and management change in Spain has a particularly large impact on the EFBI. Two key factors drive this result. Firstly, Spain has a large farmed area and holds over 90% of the European populations of eight species included in EFBI. Population dynamics of these species are therefore largely driven by changes in resource availability in Spain but, due to the methods used to calculate the EFBI, their influence on its level is the same as that of species whose population dynamics are dictated by changes in resources availability in a number of countries. Secondly, many species with breeding populations across Europe over-winter in the South region, including Spain. Land-use or management change in Spain therefore contributes winter risk to the breeding populations of many species in many countries. Whilst detrimental changes in the quantity or quality of foraging and/or nesting resources can have a relatively large impact on the EFBI, the converse is also true. Resource provision in Spain is likely to have a disproportionately large positive impact on the EFBI, compared to similar levels of resource provision in other countries.

The negative relationship between the funds allocated to Axis 2 measures by each Member State and their relative contribution to the level of the EFBI is worrying. Indeed, we had expected the converse relationship i.e. that those countries with relatively high risk per unit area would have increased levels of funding per unit area available to mitigate or offset this risk. This result suggests that the distribution of Pillar 2 funding, both between and within countries, is poorly targeted with respect to biodiversity protection and conservation needs; note that there is a strong positive correlation between the total Pillar 2 funding allocated to a country and the amount that country allocates to Axis 2. It is particularly concerning that Spain and Holland, which make the highest contribution to the EFBI per unit area, have two of the lowest allocations per unit area to Axis 2 measures. Conversely, Austria and Finland make relatively low contributions to the level of the EFBI but are allocating the highest levels of funding per unit area to Axis 2 measures (Fig. 2). It should be stressed that our analyses are based on Axis 2 budget allocation, which includes funds for AES but also includes a number of other measures with “environmental” objectives. The relationship between relative contribution to the EFBI and Axis 2 funding does not necessarily mean that a similar relationship between each country’s contribution to the EFBI and

money it allocates to AES specifically exists. Furthermore, at the scale of these analyses, we cannot pass judgement on the effectiveness of the design and implementation of schemes within each country. However, we believe that increased recognition of the differences in the relative contribution of countries to the EFBI and the likely scale of the response of farmland bird populations to environmentally-beneficial management in the allocation of Pillar 2 funds both between and within countries will contribute to the cost-effective and efficient delivery of the EU's Rural Development strategy. In particular, our results suggest that additional funding for AES in Spain and Holland, whether through an increase in the overall allocation of Pillar 2 funds or via a national re-distribution of existing Pillar 2 money between the four Axes, would be particularly beneficial.

4.2. Model assumptions and data limitations

Trait-based frameworks have successfully been used to assess the impact of agricultural change on a range of taxonomic groups, including farmland birds, arable broadleaf plants and bumblebees at a national level (Butler et al., 2007, 2009). The pan-European approach reported here relies fundamentally on the same principle as these models, i.e. that the impact of a given change on a given species can be quantified by assessing the proportion of the species' key resource requirements detrimentally affected by that change. However, to work at a continental rather than a national scale we had to both broaden existing assumptions and make a number of additional ones. These reveal key data limitations and areas requiring further research that are discussed in detail below.

Firstly, whilst we allowed each species' reliance on farmland to provide their key resources to vary between countries, we assumed that the set of resource requirements identified for each species was appropriate for all countries. Whilst it is possible that there is some intra-species variation in, for example, diet composition between countries, this is likely to operate at a much finer scale of detail than the broad categories of resource requirements defined in our risk assessment framework and therefore will not significantly influence the results presented. Due to a paucity of data for some species, migration strategies for non-resident species that over-winter in Europe were also defined relatively coarsely; wintering locations were identified to a regional level and it was assumed that the breeding population from a given country was distributed between the over-wintering regions identified and between the constituent countries within each over-wintering region on the basis of farmed land area. Whilst we believe this assumption is justified for the analyses presented here, further research to accurately identify the wintering location of the breeding populations of all species in each country would allow more accurate predictions to be made.

It is important to note that our assessment of the impact of past agricultural intensification and land abandonment during the validation of both the BASIC and SCALE frameworks focused solely on the biodiversity risks of land-use and management changes and did not take into account any potential benefits to biodiversity associated with these changes (Robinson et al., 2001). Furthermore, the structure of the BASIC framework assumes that, within a country, the entire population of each species is exposed to the impacts of each change assessed and also that the scale and impact of each change on resource availability has been the same between countries. However, the timing and scale of agricultural change can vary both between and within countries, driven by policy reform, technological advances, socio–economic and environmental factors (Mattison and Norris, 2005). The SCALE framework overcomes these limitations to some extent by accounting for regional variation in the scale of agricultural intensification and land abandonment but there are currently insufficient data available to make country- and change component-specific adjustments for the scale and intensity of changes. Despite these assumptions, we were able to accurately predict the current level of EFBI using our model, suggesting that it captures the key mechanisms driving population dynamics in European farmland bird populations. Including a scaling factor did not alter the predictive capability of the SCALE framework compared to the BASIC framework, probably due to the coarseness of the scaling factor we had to use, but the parallel development of both frameworks allows the potential impacts of a greater range of land-use and management changes to be assessed. Future developments of this approach to reflect within- and between-country differences are likely to improve the accuracy of model predictions.

5. Conclusions

Whilst not all AES are designed with biodiversity protection as their fundamental goal, it is either one of multiple objectives or the principle objective of many schemes. These schemes are currently the main policy tool for mitigating the detrimental environmental impacts of modern agriculture and for delivering the resources on which farmland biodiversity relies back into agricultural landscapes. We demonstrate that a trait-based approach can be used to quantify the impacts of land-use and management change on farmland birds, frequently used as a proxy for wider biodiversity, at a pan-European scale. By identifying the key drivers of population change in agricultural landscapes and the relative contribution of each country to the level of EFBI, our results can be used to guide both the development and implementation of targeted AES and the objective distribution of Pillar 2 funds between and within Member States. We hope that this will contribute to the cost-effective and efficient delivery of Rural Development strategy and biodiversity conservation targets.

Acknowledgements

The following people provided species reliance scores: Ainars Aunins, Chris Bailey, Stuart Benn, Richard Bradbury, Brian Caffrey, Kenna Chisholm, Koen Devos, Ian Dillon, Paul Donald, Virginia Escandell, Rob Field, Martin Flade, Ruud Foppen, Lorenzo Fornasari, Derek Gruar, Marco Gustin, Mark Hancock, Henning Heldbjerg, Sergi Herrando, Magne Husby, Frédéric Jiguet, Marek Jobda, Will Kirby, Jaroslaw Krogulec, Lars Lachmann, Monika Laurits, Domingos Leitão, Hywel Maggs, Riho Marja, Ricardo Martins, Oscar Merne, Pete Moore, Tony Morris, Szabolcs Nagy, Ivar Ojaste, James Pearce-Higgins, Allan Perkins, Janis Priednieks, João Rabaça, Goetz Rheinwald, Jiri Reif, Hans Schmid, Martin Scott, Rosemary Setchfield, Henk Sierdsema, Karel Stastny, Sören Svensson, Juha Tiainen, Norbert Teufelbauer, Risto Väisanen, Chris van Turnhout, Zdenek Vermouzek, Zoltan Waliczky, Richard Winspear, Will Woodrow. Thanks to the individuals and organisations responsible for national trend data collection and analysis; Norbert Teufelbauer, Michael Dvorak, Christian Vansteenwegen, Anne Weiserbs, Jean-Paul Jacob, Anny Anselin, Thierry Kinet, Antoine Derouaux, Svetoslav Spasov, Jiří Reif, Henning Heldbjerg, Michael Grell, Andres Kuresoo, Jaanus Elts, Risto A. Väisänen, Fréderic Jiguet, Johannes Schwarz, Martin Flade, Tibor Szep, Olivia Crowe, Dick Coombes, Lorenzo Fornasari, Elisabetta de Carli, Ainars Aunins, Ruud P. B. Foppen, Magne Husby, Przemek Chylarecki, Barbara Archita, Geoff Hilton, Ana Meirinho, Juan Carlos del Moral, Ramón Martí, Virginia Escandell, Åke Lindström, Sören Svensson, Hans Schmid, David G. Noble, Mike Raven and Andrew Joys. Thanks also to Jana Škorpilová, Alena Klvaňová, Arco van Strien, Adriaan Gmelig Meyling, Tomáš Telenský, Ian Burfield, Fiona Sanderson, Paul Donald, Gareth Morgan, Norbert Schaffer, Sue Armstrong-Brown, David W. Gibbons and Anne Teller for assistance and support. The PECBMS is sup-

ported financially by the European Commission and RSPB. The Institute for European Environmental Policy provided the CAP expenditure data.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.agee.2010.03.005.

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