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Population responses of farmland bird species to agri-environment schemes and land management options in Northeastern Scotland

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Abstract

1. The decline of farmland birds across Europe is a well-documented case of biodiversity loss, and despite land stewardship supported by funding from agri-environment schemes (AES), the negative trends have not yet been reversed.

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2. To investigate the contribution of AES towards farmland bird conservation, we compared abundance of five farmland bird species across twelve years and 53 farms (158 farm years = AES, 72 farm years = non AES) in Northeastern Scotland (UK), a region with relatively mixed farmland.

3. Between 2003 and 2015, on both AES and control farms, skylark (*Alauda arvensis*) showed a non-significant decline, and tree sparrow (*Passer montanus*) and yellowhammer (*Emberiza citrinella*) non-significant increases, whereas reed bunting (*Emberiza schoeniclus*) and linnet (*Carduelis cannabina*) populations remained relatively stable.

4. We did not detect a significant association between AES and avian abundance or population trends for any of these species, but there were positive associations with some AES management options.

5. Possible explanations for the lack of a significant AES-bird abundance association include poor uptake of the best AES options for farmland birds, sub-optimal implementation, spill-over effects from AES onto control farms, and the relatively good state of farmland habitats outwith AES in Northeastern Scotland.

6. *Synthesis and applications.* We documented a weak effect size of participation in agri-environment schemes on farmland bird abundance. We therefore recommend future monitoring studies be designed after consulting a power analysis. Among different land management options, we found that species-rich grasslands, water margins and wetland creation enhanced breeding bird abundance, highlighting the importance of relatively undisturbed herbaceous or grassland vegetation for farmland conservation.

Keywords

agriculture, farmland birds, agri-environment schemes, passerine, low-intensity farmland, spatial statistics, offset, multi-membership

Introduction

Globally, farmland ecosystems cover 38% of ice-free land (FAOSTAT 2018) and are currently experiencing population declines across many taxa due to agricultural intensification (Donald, Gree & Heath 2001, Donald *et al.* 2006, Kleijn *et al.* 2009). In Europe, these declines are addressed through large-scale implementation of agri-environment schemes (AES), which encourage less intensive land management to bring ecosystem and biodiversity benefits (Vickery *et al.* 2004, Batáry *et al.* 2015). Around 25% of EU farmland is under AES (Science for Environment Policy 2017), whereby farmers receive funding for land stewardship, such as provision of fallow land and creation of wildlife corridors (Kampmann *et al.* 2012). AES have been the main tool in European farmland conservation for over 30 years, but evaluations of their performance have returned mixed results (Kleijn *et al.* 2011, Princé *et al.* 2012, Ekroos *et al.* 2014, Batáry *et al.* 2015). Although the schemes can alleviate population declines, they might not always be able to reverse them (Gamero *et al.* 2017).

In the UK, AES have been successful in halting and reversing national population declines where they targeted the recovery of high-priority and range-restricted farmland bird species (e.g., cirl bunting *Emberiza cirlus*, corncrake *Crex crex* and stone curlew *Burhinus oedicnemus*), and have also delivered wider biodiversity benefits (Peach *et al*. 2001, O'Brien *et al*. 2006, MacDonald *et al.* 2012a, 2012b, Wilkinson *et al*. 2012). However, when implementation is poorly targeted, AES have been less effective, with outcomes varying across plant, invertebrate and bird species (Kleijn *et al.* 2006), within specific groups, e.g., farmland birds (Bright *et al.* 2015), and within individual species' ranges (Donald & Evans 2006, Whittingham *et al.* 2007). The spatial heterogeneity in AES performance is in accord with landscape-moderated AES effects, with smaller and/or harder to detect AES benefits in complex landscapes (Batáry *et al.* 2011).

Comprehensive AES audits depend on assessing scheme performance across the full range of landscape complexity, but most AES studies have been in places where farming systems are relatively intensive (e.g., Merckx et al. 2010, Bright *et al*. 2015, but see Santana *et al.* 2017 and Concepción *et al.* 2008 for studies along a land-use intensity gradient). For example, autumn-sown wheat and oilseed rape, two intensively managed crops, cover 44% and 16% of arable land in England, compared with just 10% and 7% in Northeastern Scotland, where less intensively managed spring-sown barley is the dominant crop (55% of arable land, cf. 7% in England) and most lowland farms are mixed arable-livestock enterprises (Scottish Government 2010, FERA 2018). Although some studies have considered regional variation in the efficacy of English AES in high-intensity farmland (e.g., Davey *et al*. 2010), a key remaining question is how AES implementation affects biodiversity in heterogeneous, low-intensity farmland.

Here, we examined the association between long-term AES implementation and abundance and population trends of five farmland bird species in Northeastern Scotland, an area dominated by relatively low-intensity mixed farming. Specifically, we hypothesised that: 1) AES has a positive effect on species abundance, 2) the effect becomes stronger over time under AES treatment, 3) AES farms have more positive population trends, and 4) AES management options that match species' ecological requirements have a stronger positive effect on abundance. We present a novel application of a multi-membership model, which allows us to estimate the effects of different management options, whilst also accounting for differences in the area they cover and the different combinations in which they are implemented. Finally, we also conducted a post-hoc power analysis to determine the power of our survey design to detect different effect sizes of AES on population trends in low-intensity farmland.

1. Methods

1.1. Study species

We investigated populations of five farmland bird species – linnet (*Carduelis cannabina*), reed bunting (*Emberiza schoeniclus*), tree sparrow (*Passer montanus*), skylark (*Alauda arvensis*), and yellowhammer (*Emberiza citrinella*). All five species are widespread within this region, and were recorded on 80–100% of study farms. Across Scotland (33–119 random 1 km squares), reed bunting, tree sparrow and yellowhammer increased between 1995 and 2016, in contrast with more modest increases or declines on a UK-scale (195–1220 random 1 km squares; Harris *et al.* 2017). Over the same 21-year period, linnet and skylark declined at both Scotland and UK-scales (96–223 and 1252–1803 random 1 km squares, respectively; Harris *et al.* 2017).

1.2. Study sites

Our study sites were 53 lowland farms in Northeastern Scotland (Fig. 1). There were two farm treatments – AES (mean farm area = 120 ha ± SD 56) and control (conventional farming, mean farm area = 103 ha ± SD 58). For details on site selection see Perkins *et al.* 2011. Land-use types were similar between farms, with 95% of study sites supporting mixed farming practices (Supporting Information, Table S1). AES farms participated in the national Rural Stewardship Scheme (RSS, 2001–2008, Scottish Government 2006), its successor Rural Priorities (RP, 2009–2015, SRDP 2014), and Farmland Bird Lifeline (FBL), a local intervention scheme targeting corn buntings. AES management varied between farms, with participants selecting several management options from 33 available in RSS, 49 in RP and 7 in FBL (SI, Table S2). Some farms switched treatment between years due to entering or leaving AES agreements which ran for five years in the national schemes and were renewed on an annual basis in FBL (see SI, Table S1 for yearly sample sizes). The average treatment duration was 5 years ± SD 4 for AES and 4 years ± 3 for control farms. Twenty-one farms remained under AES management for the entire study duration (13 years).

1.3. Data collection

Farms were visited two or three times (70% of data points based on three visits) between May and mid-August of 2003, 2004, 2006, 2008, 2009, and 2015, as part of a corn bunting (*Emberiza calandra*) monitoring project during which other farmland birds were also counted (Perkins *et al*. 2011, 2016). Survey routes scaled positively with farm size and passed within 250 m of all points on the farm, largely following field boundaries. Since the surveys were designed for corn buntings which nest from May to August and favour open landscapes, some late visits (mid-July to August) and survey routes might not have been optimal for all study species, e.g., detection rates for skylarks are low once they stop singing in late summer, and tree sparrows frequently occupy farm woodlands which survey routes often avoided. Nevertheless, the survey design, combined with highly experienced fieldworkers, was considered sufficient to detect a high proportion of birds of each study species. During each visit, bird location and activity were recorded on a 1:10 000 map. The maps were superimposed onto each other to determine the number of breeding birds from clusters of registrations. We chose recording units based on species behaviour in a manner that optimises detectability. For yellowhammer, reed bunting and skylark, we recorded territorial males, based on singing birds. For tree sparrow and linnet, two species which sing less frequently and are semi-colonial, singing birds underestimate abundance, so we recorded numbers of apparently breeding pairs (birds displaying breeding behaviour in suitable nesting habitat).

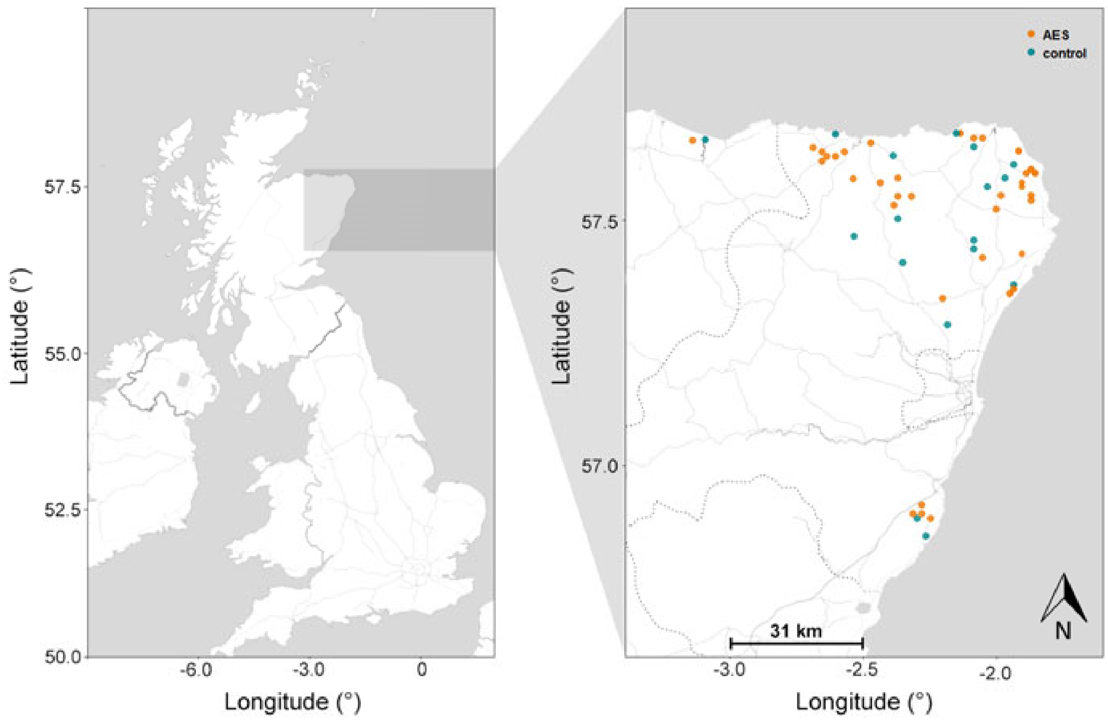

Fig. 1. Locations of study sites within the UK and Northeastern Scotland. Points represent farm treatments in 2015.

1.4. Data analysis

All statistical analyses were conducted in R v. 3.4.2 (R Core Team 2017). Overall bird abundance, trends and AES effects were estimated using generalised linear mixed models (GLMMs), assuming a Poisson distribution (due to the count nature of the data) and using Bayesian inference via the package *MCMCglmm* (Hadfield 2010). GLMMs have been widely used in ecology, particularly because they can efficiently model multiple predictors whilst also accounting for variance introduced by the structure of the data and the experimental design (e.g., records from the same farm and/or year might be more correlated than those among farms/years, Bolker *et al.* 2009, Harrison *et al.*

2018). Models were run for 600 000 iterations, with a burn in period of 20 000 iterations, a thinning factor of 10, and parameter expanded priors which improve model convergence. Convergence was assessed using visual inspection of the autocorrelation in posterior distributions, and the effective sample size was in excess of 1000 for focal parameters.

1.4.1. Overall AES associations with species abundance and number of years in AES

To test overall associations between AES and species abundance, we modelled farm treatment type (AES/control), whilst accounting for the confounding effects of farm area (log transformed and mean centred), location (latitude and longitude mean centred), and survey effort (visits: 2/3). We included latitude and longitude as fixed effects to account for a potential latitudinal land-use intensity gradient in Scotland (decreasing towards the north) and a longitudinal climate gradient going from continental to coastal climate. Farm identity and observation year were treated as random effects to account for some of the spatial and temporal non-independence of data points. It is common for studies assessing temporal trends to include year as a fixed effect but not as a categorical random effect. However, where there are multiple observations for each year, observations made in a single year may be subject to the same year-specific effects arising from drivers not included in the model (e.g., weather). Including year as a random effect takes this pseudoreplication into account, whereas a model that does not include year as a random effect will tend to underestimate standard errors (thereby inflating type I errors) of coefficients estimated for predictor variables that change over time.

We worked with the full models, and did not execute step-wise term deletion and model comparison which have been criticised for increasing the probability of type I errors (Whittingham 2006, Mundry & Nunn 2009). Reported b values are the slope estimate (effect size), and p values refer to MCMC p values. A predictor was considered ‘significant’ when the 95% credible intervals (CI) for the corresponding model parameter did not overlap zero.

We examined whether AES effects become more pronounced as treatment duration increases by including ‘AES years’ as a continuous fixed effect to the models testing for an association between AES treatment and bird abundance. ‘AES years’ was set to zero for farms that never entered AES management, i.e., control farms, and equalled one or above for AES farms, depending on the number of years each AES farm was in a scheme.

1.4.2. Population trends on AES and control farms

We modelled avian population trends during the study period (2003-2015) by adding year (mean centred) as a continuous fixed effect to the models outlined above (one model per species). To determine if population trends differed between AES and control farms, we ran five additional models which also included the year by treatment interaction term as a fixed effect.

1.4.3. Sensitivity of species abundance to specific AES management options

To examine the associations between specific AES management options (SI, Table S2) and overall bird abundance, we conducted an analysis using a multi-membership modelling framework through the package *MCMCglmm* (Hadfield 2010). This analytical approach is particularly appropriate for data with complex structure, where records belong to more than one level of classification (e.g., bird abundance on farms where multiple AES management options were implemented simultaneously). Our model structure included farm area (log transformed and mean centred), location (latitude and longitude mean centred), and survey effort (visits: 2/3) as fixed effects and the proportionate area over which AES land management options were implemented on each farm as a multi-membership random term, and farm identity and observation year as two further random terms. The multi-membership term estimates the random effect of increasing area of each land management option on abundance, assuming all random effects are drawn from a normal distribution with estimated variance. We assessed the associations between the individual options and overall bird abundance from the posterior distribution of each best linear unbiased predictor (BLUP) which signifies the potential increase or decrease in abundance if the proportion of that treatment was 1 (the whole farm).

1.4.4. Spatial autocorrelation analysis

To test for spatial autocorrelation in species' abundance, we ran spatial GLMMs (one per species) through the package *spaMM* (Rousset and Ferdy 2014). We modelled bird abundance as a function of treatment type, farm area (log transformed and mean centred), number of visits to the farm (fixed effects), a random effect with Matérn correlation function based on easting and northing values for each farm, and a fixed smoothness parameter ($v=0.5$) which corresponds to an exponential decay in the correlation between farms. Strong spatial autocorrelation would be consistent with the hypothesis that source-sink dynamics and dispersal may be at play between AES and non-AES farms. The spatial GLMMs demonstrated that spatial autocorrelation between study farms was very low ($rho<0.01$ for all five species, SI, Table S9 and Fig. S1).

1.4.5. Offset analysis

Conservation biologists often fit area as a linear offset to convert counts into densities. If survey effort scales perfectly with area, and abundance increases linearly with survey area (i.e., a slope of 1), estimating density this way requires one fewer degree of freedom. However, survey design, habitat configuration and/or species' ecology may cause the observed slope to depart from 1, and in such cases treating area as an offset will impact on the coefficients of additional predictors included in the model (Helzer & Jelinski 1999). This is especially likely in farmland, where crop fields and grass are separated by narrow strips of semi-natural habitat, meaning that variation in farm size will affect habitat availability for crop-nesting and boundary-nesting species differently. To examine how including area as an offset would impact on our inferences, we repeated the analyses using area as an offset rather than as an estimated coefficient. Slope estimates of log farm area as a fixed effect term in MCMC models showed high variability between species and all except reed bunting were well below 1 (0.48 – 0.72, Table S5), demonstrating that estimated abundance-area relationships on farmland are often non-linear which may be attributable to sampling effort or depend on species' preference for fields vs. edge habitats (SI, Tables S5 and S10). Although the two modelling approaches gave similar results, inclusion of an offset when the relationship with area departs from 1 has potential to generate incorrect inferences for other model terms. Therefore, we recommend that the effect of area should be estimated instead of fixed.

1.4.6. Power analysis

Finally, to investigate the power of our survey design in detecting AES effects on avian abundance and inform future design of similar surveys, we conducted a power analysis using the packages *simr* (Green and MacLeod 2016) and *lme4* (Bates *et al.* 2015), as currently *simr* and *MCMCglmm* are incompatible. We considered effect sizes where AES farms had 10% and 25% more individuals than control farms, and ran the power analysis based on 1000 simulations and 230 observations of bird abundance, with a power threshold of 80%. The *lme4* models estimated bird abundance (Poisson data distribution) as a function of treatment type (AES vs. control), farm area (log transformed and mean centred), latitude and longitude (mean centred), survey effort (visits: 2/3), and included farm identity and year of observations as random effects.

For a summary of all analyses, model structure and tested hypotheses, please see SI, Table S3. The farmland bird data are available from Perkins *et al.* (2017) and the code for all analyses is in Daskalova (2018).

2. Results

2.1. Overall AES associations with species abundance and number of years in AES

We found no significant association between AES and the abundance of five bird species, and model predictions for bird abundance were similar on AES and control farms (Fig. 2a, SI, Table S4). The coefficients for the effect of the AES treatment relative to control were small, ranging from 0.19 to -0.06 on the log scale or a population size difference of +20% for tree sparrow and -6% for skylark (Fig. 2b). There was no significant effect of AES treatment duration on species abundance (SI, Table S7), and model coefficients for the effect of years in AES were very small (Fig. 2c).

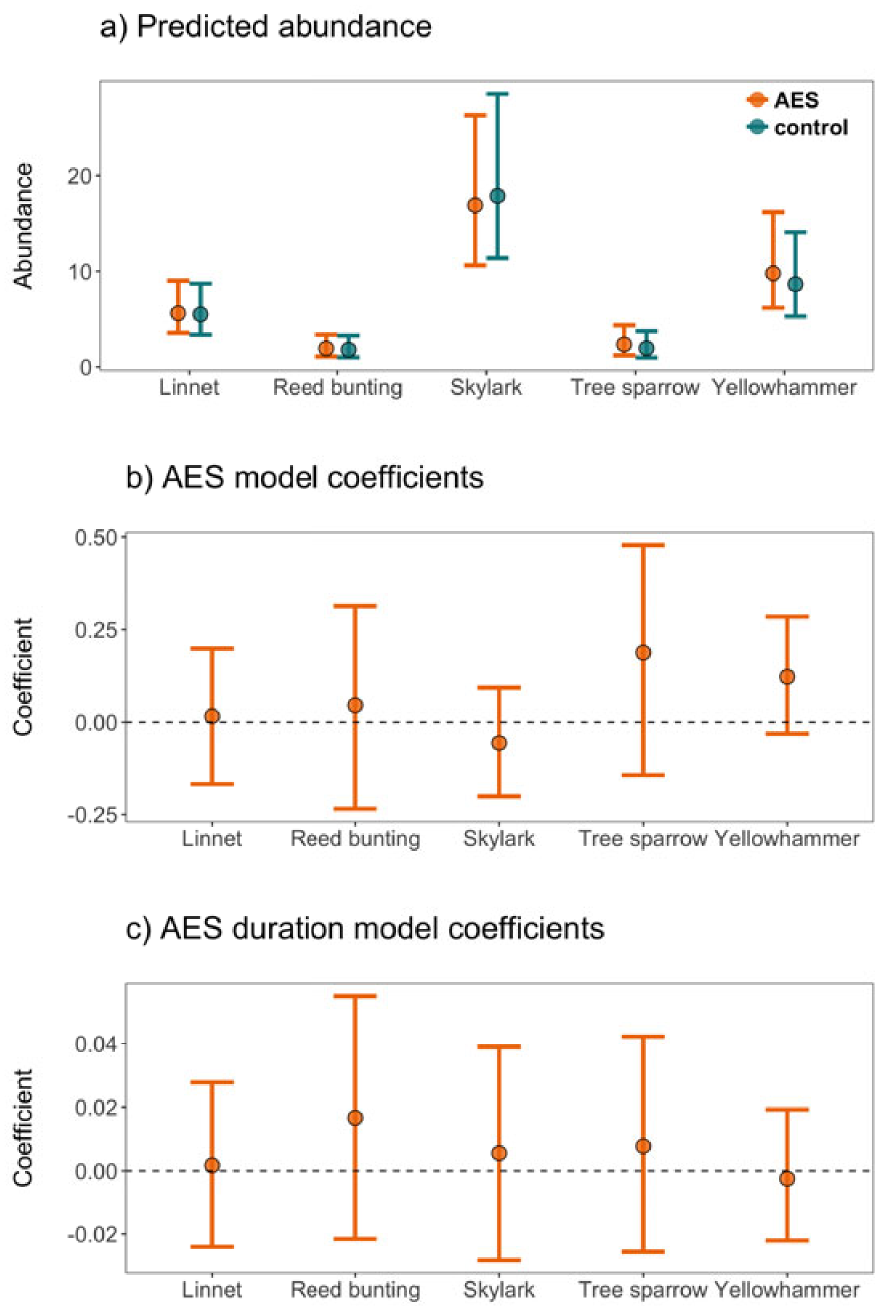

Fig. 2. a) Model predictions for species' abundance across treatment types, model coefficients and 95% credible intervals for b) AES farm treatment relative to control treatment, and c) the effect of number of years in AES on bird abundance. Predicted values on (a) were calculated for an average farm (113ha) that was visited three times in an average survey year, and are presented on the data scale. Coefficients on b) and c) are presented on the logarithmic scale. For full model outputs, see SI, Tables S4 and S7.

2.2. Population trends on AES and control farms

Between 2003 and 2015, the five bird species we monitored did not experience net directional changes. We observed directional albeit non-significant trends for three species – skylark declined in abundance, whereas tree sparrow and yellowhammer abundance increased (Fig. 3, SI, Tables S5 and S6). Overall, we did not detect a significant difference between linear population trends on AES and control farms, although for some species (e.g., skylark between 2004 and 2008), trends on AES and control farms diverged (SI, Table S6).

2.3. Sensitivity of species abundance to specific AES management options

We found few positive (and no negative) associations between bird abundance and particular AES management options (Fig. 4). Linnet, tree sparrow and skylark abundance was not significantly associated with any of the investigated options. Water margins and enhanced riparian buffer areas were positively associated with reed bunting abundance (BLUP=0.47, CI 0.06 – 0.87, i.e., a 60% abundance increase if the entire farm area were devoted to this option), and creation and management of species-rich grass (BLUP=0.12, CI 0.01 – 0.23, i.e., 13% increase) and wetlands (BLUP=0.08, CI 0.02 – 0.13, i.e., 8% increase) were positively associated with yellowhammer abundance. (Fig. 4, SI, Table S8).

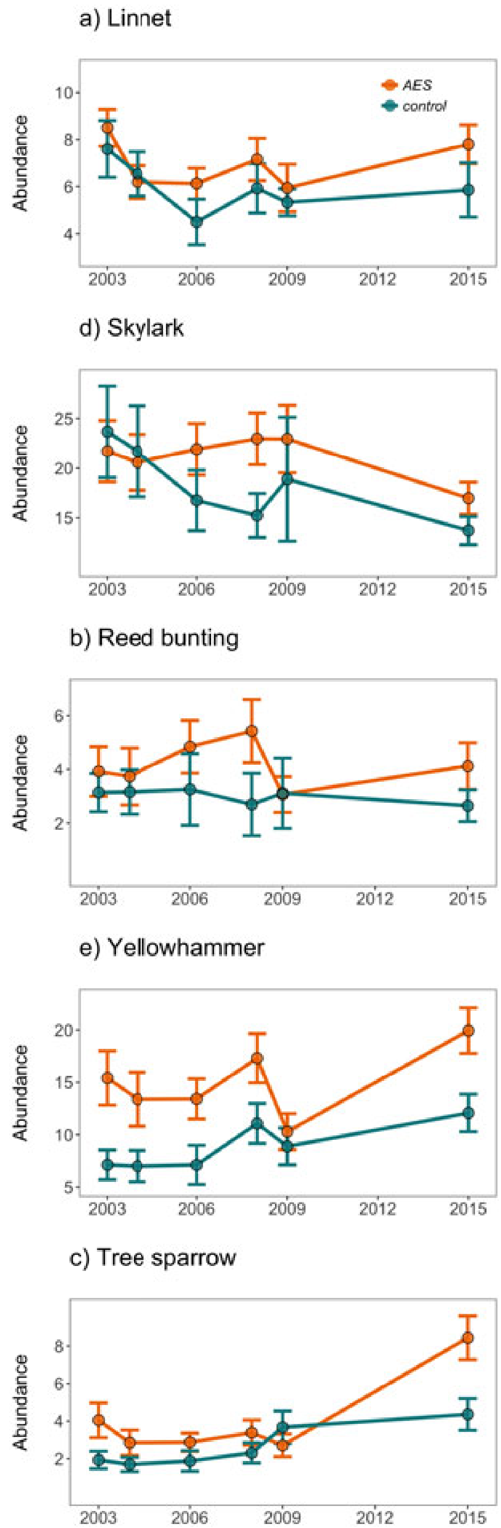

Fig. 3. a-e. Population sizes between 2003-2015 of five farmland bird species in Northeastern Scotland, based on raw data, not controlling for farm size. AES farms tended to be larger in size (mean 120 ha ± 4.6 SE across all years) than control farms (mean 104 ha ± 11.2 SE across all years).

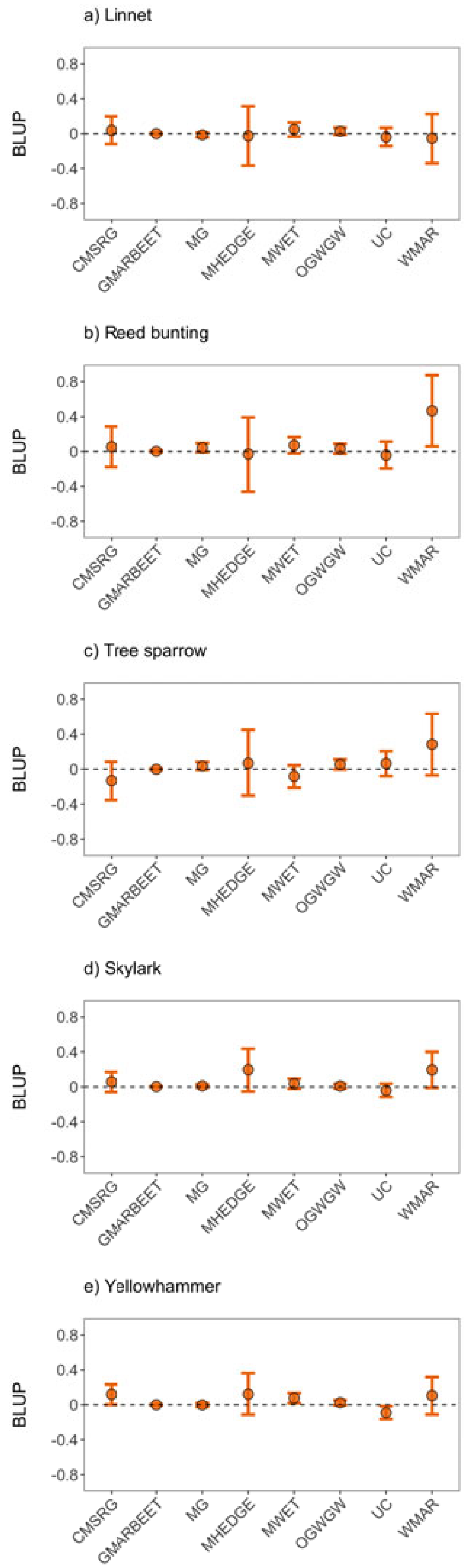
a) Linnet
b) Reed bunting
c) Tree sparrow
d) Skylark
e) Yellowhammer
BLUP
0.8
0.4
0
-0.4
-0.8
CMSRG
GMARBEET
MG
MHEDGE
MWET
OGWGW
UC
WMAR

Fig. 4. a-e Posterior means and 95% credible intervals for best linear unbiased predictors (BLUPs) of bird abundance under different land management options. Numbers after management option codes indicate sample size (number of farms). CMSRG (18) – creation/management of species-rich grass, GMARBEET (35) – grass margins and beetlebanks, MG (9) – mown grassland for wildlife, MHEDGE (42) – creation/management of hedgerows, MWET (21) – creation/management of wetlands, OGWGW (18) – open grazed or wet grassland for wildlife, UC (32) – wild bird seed mix/ unharvested crops, WMAR (30) – water margins and enhanced riparian buffer areas. See SI, Table S2 for details on land management and Table S8 for all model outputs.

3.4 Power analysis

A *post-hoc* power analysis revealed that our survey design (53 farms surveyed in 6 years (2003, 2004, 2006, 2008, 2009 and 2015), with 62–79% of farms in AES in any one year) had sufficient power (i.e., > 80% probability of detecting a significant effect) to detect strong AES associations should they have been present (e.g., AES having 25% higher overall bird abundance than the control treatment) for two out of five study species (skylark, yellowhammer, for which more records were available, since they were more common, SI, Table S11). Distinguishing subtler AES effects for all study species (e.g., AES having only 10% higher bird abundance than the control treatment), however, poses a challenge, as detecting such small effects sizes and effects for rarer species would require monitoring of a substantially larger sample size of farms with equal representation of AES and control treatments.

Discussion

We found no association between AES participation and bird abundance or population trends at the farm scale in low-intensity mixed farmland in Northeastern Scotland. Between 2003 and 2015, skylark declined on both AES and control farms, whereas tree sparrow and yellowhammer increased, but these overall trends were not significant and no species experienced net directional change. The lack of a significant overall AES association could be due to several factors, including lack of additionality of AES in low-intensity farmland, increases on conventional farms due to spill-over effects from AES farms into the wider countryside, land-use legacy, and/or poor selection and implementation of AES management options. When examining AES in detail, we found positive associations between bird abundance and specific land management options that met species' ecological requirements, in particular reed bunting and water margins, and yellowhammers and species-rich grasslands, highlighting the importance of farm- and field-scale targeting of management within AES.

First, there may have been weak contrast in habitat provision for birds between AES and control farms in the study's heterogeneous low-intensity landscape where cereals are predominantly spring-sown and overwinter stubbles are common(around 70%, Scottish Government 2010). Here, AES options providing winter seed food may not have the same level of population effect as demonstrated for more intensively managed arable landscapes (e.g., Gillings *et al*. 2005). The moderating effect of landscape complexity on species' responses to AES is driven by metapopulation dynamics (Durell & Clarke 2004), and habitat and resource availability on farms and the surrounding area (Concepción *et al.* 2008, Batáry *et al.* 2011). Implementing AES in high-intensity farmland creates a high ecological contrast, thus increasing their additionality, and resulting in an easier to detect relative effect (Scheper *et al.* 2013, Hiron *et al.* 2013, Josefsson *et al.* 2017). When implemented in low-intensity farming landscapes that tend to be more complex, AES might lack additionality and their effects might be harder to detect (Concepción *et al.* 2008). Our power analysis confirmed that the monitoring in this study was adequate to detect relatively large effects of AES (e.g., 25% higher overall abundance on AES than control farms), but underpowered for detecting weaker effects, and effects for patchily distributed species such as reed bunting (SI, Table S11). We recommend that the design of future studies monitoring AES efficacy is informed by power analysis, such that they have power to detect weak effects, in particular when assessing AES policies in low-intensity farmland.

Second, overspill from AES farms into the wider landscape can boost bird abundance on control farms, making potential AES effects hard to detect in a comparative study (Kleijn *et al.* 2011). Many farmland birds are vagrant during the winter and move in response to feeding opportunities, so it is possible that some individuals breeding on control farms use resources on AES farms. Previous winter monitoring of our study sites supports this, with flock sizes using AES unharvested crops often far exceeding local breeding numbers (Perkins *et al.* 2008), whilst colour-ringing revealed corn bunting movements up to 15–20 km between wintering and breeding sites (RSPB unpublished data). Another Scottish study showed that tree sparrows and yellowhammers ranged over several kilometres during winter (Calladine *et al.* 2006). We detected very little spatial autocorrelation of abundance among our study sites, suggesting few summer movements between the sampled farms, but the limited number of very close farms constrained our ability to accurately measure spatial autocorrelation over short distances. Complete survey coverage over larger contiguous survey areas (e.g., 100 km^2) could enable stronger testing of spill-over effects, especially if AES farms were entirely surrounded by controls, but this would require full control over which farms participated in AES.

Third, the land-use legacy of both AES and controls can influence species' response to newly introduced management practices. For example, control farms entering AES will likely support less biodiversity during the first years of the scheme due to delayed effects of conventional agricultural practices (e.g., pesticide build up in the system, Morris *et al.* 2005, but see Perkins *et al.* 2011 for quick population responses to targeted AES options such as delayed mowing and winter seed provision). Conversely, when an AES farm reverts back to conventional management, permanent farm features such as hedgerows or ponds created during the scheme continue to benefit biodiversity, thus potentially inflating bird abundance relative to 'true' controls that have never participated in AES. In our study, 14 farms switched management which we could only partially account for by assigning a treatment factor for each farm during each survey year. Nevertheless, there were 21 farms which remained under AES management for the entire duration of the study (13 years). Previous studies with shorter monitoring durations (e.g., Perkins *et al.* 2011, seven years) and Bright *et al.* 2015, five years) have documented AES effects on similar farmland bird species, suggesting that our monitoring duration of 13 years is sufficient time to allow any potential effects of AES management to manifest themselves. We did not have information about farm management and bird abundance prior to 2003, so our study had weak 'before-after' design. However, post-hoc analysis of the 12 farms that switched from control to AES revealed no significant differences in bird abundance before/after AES had been implemented (SI, Table S12).

Crucially, AES offer farmers a wide range of management options to choose from, and deployment varied substantially between our study sites. We found that bird abundance varied significantly with the area deployed for particular AES options, suggesting that schemes were more effective when option deployment was well-matched to species' ecological requirements (Fig. 4). For example, reed bunting abundance was enhanced by the creation of water margins which provide tall dense vegetation next to watercourses, the species' preferred nesting habitat (Brickle & Peach 2004). AES options can also benefit farmland biodiversity through invertebrate food resource provision (e.g., Vickery *et al.* 2009, McHugh *et al.* 2016), and this probably explained the positive associations between creation and management of species-rich grasslands and wetlands options with yellowhammer abundance. Further examples of the conservation benefits of targeting AES options to species' ecology include corn bunting population increases in the FBL scheme following delayed mowing of meadows (Perkins *et al.* 2011). However, farmers tend to choose AES options that maximise financial income whilst minimising disruption to their current management (Burton & Schwarz 2013, Josefsson *et al.* 2013). In most AES, farmers are paid according to the quantity rather than quality of land dedicated to certain stewardship measures, so they have little incentive for ensuring optimal option deployment (Canton *et al.* 2009, Quillérou & Fraser 2010). An alternative approach that has been successful in Germany is result-oriented schemes, where farmers are paid for management only after certain conservation targets are met (Matzdorf & Lorenz 2010). Adopting a similar practice in future UK schemes may encourage better implementation of AES options and increase their effectiveness.

Overall, we did not detect an overall association between AES and species abundance or population trends. The five farmland bird species we studied in Northeastern Scotland did not experience net directional change. Four species showed signs of stable or increasing populations, with just skylark exhibiting a non-significant decline. The regional population trends we documented for linnet, skylark and yellowhammer (no net change, i.e., stable populations) contrast with UK-wide declining trends from 1995 to 2016 (Harris *et al.* 2017), and might reflect greater resource provision for granivorous passerines in more diverse and less intensive mixed farming systems. We suggest that lack of AES associations was due to low additionality of schemes relative to conventional farmland habitats during the breeding season.

Management recommendations

There were specific management options such as species-rich grasslands, water margins and wetland creation that appeared to enhance bird abundance. These options all provide similar habitat, i.e., relatively undisturbed herbaceous or grassland vegetation that is not managed for agricultural production. Undisturbed vegetation can provide safe nest sites for ground-nesting birds like yellowhammer and reed bunting, and also insect-rich foraging areas during the breeding season. For farms lacking such habitat, AES are currently the main incentive for farmers to create them. Similarly, AES unharvested crops provide bespoke seed food resources for farmland birds during winter, and although we failed to detect associations with breeding abundance, heavy usage during winter (Perkins *et al*. 2008) supports our recommendation for farmers to routinely select this management option. To improve scheme effectiveness, we recommend better targeting and management of AES options at the farm and field-scale to match the ecological requirements of target species, and to use AES to fill 'resource gaps' where specific habitats are lacking. Finally, we recommend that future AES studies of bird population change carefully consider landscape context and likely effect sizes, and use power analyses to design monitoring schemes that will provide adequate tests of whether these expensive policies are having their intended effect.

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Author contributions

AJP and HEM designed the field surveys and collected the data. GND, ABP, MB and AJP contributed to the conception and design of the statistical analyses which GND conducted with support from ABP.

GND wrote the first draft of the manuscript and all authors contributed to revisions. All authors approve the manuscript for publication.

Data accessibility

Data available via the Edinburgh University DataShare repository http://dx.doi.org/10.7488/ds/2275 (Perkins, 2017). Code for statistical analyses is available via Zenodo 10.5281/zenodo.1473732. (Daskalova, 2018).

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