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THE EFFECT OF AGRI-ENVIRONMENTAL SCHEMES ON FARMLAND BIRDS IN EASTERN SCOTLAND

BY

GERGANA DASKALOVA

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Summary

Farmland ecosystems in the UK are currently experiencing a major biodiversity crisis due to agricultural intensification, with declines apparent across many taxa and in particular, birds. Currently there is a gap in our understanding of how farmland birds respond to conservation strategies, such as agri-environmental schemes (AES), which aim to reverse population declines. Here, I investigated the effects of AES on the abundance of six farmland bird species in Eastern Scotland. I found that during the period of 2003-2015, AES have not had a significant effect on target species' populations, and provide evidence for possible negative responses to AES. By examining the relationships between farm area and avian abundance, I demonstrate that to augment AES benefit, it is not sufficient to only increase the area over which AES are applied. The proportion of farm area that is dedicated to field boundaries and edge habitats also has to be considered, as they are favoured by numerous birds for nesting and foraging. The lack of positive AES effects on bird populations suggests that the declining farmland bird trends in the UK will not be reversed, unless the design, implementation, and targeting of AES is improved. As Scotland implements the new Agri-Environment Climate Scheme in 2016, it is important to evaluate past schemes and make necessary changes to design an optimal conservation strategy for the future.

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I am most thankful for the patience and dedication of my late grandfather who accompanied me on my first birdwatching trips. The time I spent with him looking for hoopoes and golden eagles inspired me to pursue an education in ecology. The memories of my wild childhood in a small village where traditional farming supports a rich avian fauna were instrumental for getting me through the many hours I spent analysing data. Finally, I would like to thank my family for tolerating my bird enthusiasm and supporting me regardless.

List of abbreviations

AES	Agri-environmental scheme
RSS	Rural Stewardship Scheme (operating from 2001 to 2008)
RP	Rural Priorities (operating from 2009 to 2015)
FBL	Farmland Bird Lifeline (operating from 2001, ongoing)
CB	Corn bunting
LI	Linnet
RB	Reed bunting
S	Skylark
TS	Tree Sparrow
Y	Yellowhammer
FBI	Farmland Bird Index
RSPB	Royal Society for the Protection of Birds

Introduction

Globally, farmland ecosystems cover 38% of ice-free land (FAOSTAT 2016) and are currently experiencing a major biodiversity crisis due to agricultural intensification (Kleijn *et al.* 2009). Exponential human population growth and rising consumer demand are predicted to cause increased pressure on agricultural land, thus exacerbating environmental degradation (Green *et al.* 2005). Agricultural intensification has profoundly altered farmlands, resulting in steep declines of numerous taxa and disruptions in water and nutrient cycles (Newton 2004, Geiger *et al.* 2010). Disentangling drivers of farmland biodiversity decline and establishing effective conservation practices without compromising production yields is an urgent issue for both government officials and scientists (Thurpp 2000). Currently there is a gap in our knowledge of how communities and individual species respond to management practices, which hinders effective farmland conservation (Vickery *et al.* 2004). This lack of understanding is especially prominent in the UK, where farmland ecosystems are well studied, recovery plans have been implemented for over 30 years and yet, farmland biodiversity continues to decline at an alarming rate (Kleijn *et al.* 2011).

The majority of land in the UK (71%) is used for agriculture (DEFRA 2015a), supported by the Common Agricultural Policy (CAP) within the European Union (EU) (Donald, Gree & Heath 2001). Decreasing levels of farmland biodiversity have been directly linked to the level of intensification within the British agricultural sector (Chamberlain *et al.* 2000, Benton *et al.* 2002; Robinson & Sutherland 2002). The resulting deterioration of farmland ecosystems threatens the populations of more than two thirds of the floristic and faunal species, associated with farmland habitats (Butler *et al.* 2009). The magnitude of those negative effects is assessed through indicator-based monitoring schemes (Büchs 2003). In the UK, bird population trends are the most common measure of the impact of intensive agriculture on biological communities (Büchs 2003). The suitability of birds as ecosystem indicators is supported by their place within ecosystem food webs, sensitivity to farming practices, ease of observation and identification, as well as the availability of long-term population data (French, Centre & Brathens 2002, but see Landres *et al.* 1988 for a critique).

Farmland birds in the UK are experiencing significant declines in population size, as well as range contractions (Donald, Gree & Heath 2001, Fuller *et al.* 2014, Donald *et al.* 2006), often leading to local extinctions of species (Chamberlain & Fuller 2000, Gates & Donald 2000). Between 1970 and 2014, the Farmland Bird Index (FBI), an indicator of the population trends of 18 bird species associated with farmlands, decreased by 54% (DEFRA 2015b). The general causes of the FBI decline are loss of mixed farming (Siriwardena *et al.* 2000), increased pesticide use (Benton *et al.* 2002), loss of overwintering stubble (Hancock & Wilson 2003), land drainage, and higher stocking rates in pastoral farming (Newton 2004). Specific drivers of decline have also been identified, such as the loss of hedgerows which is particularly detrimental for yellowhammers (Whittingham *et al.* 2005). Further examples include the drainage of wetlands which reduces habitat availability for reed buntings (Newton 2004) and the early mowing of meadows, known to increase breeding failure among corn buntings (Perkins *et al*. 2013). Understanding how farming practices compromise avian ecological requirements has facilitated the implementation of conservation measures, such as agri-environmental schemes.

Agri-environmental schemes (AES) aim to encourage extensive, rather than intensive, land management in order to bring ecosystem and biodiversity benefits (Perkins *et al.* 2011). Around 20% of EU land is under AES management, whereby farmers receive funding for land stewardship, such as provision of set-aside land and creation of wildlife corridors, field margins and hedgerows (Herzog 2005, Kampmann *et al.* 2012). These conservation measures aim to provide birds with food during winter, a safe nesting place, and enough summer food to rear their broods (Butler *et al.* 2009). Species' responses to AES are not uniform across their distribution ranges, thus making the effectiveness of AES contingent on local landscape, environmental conditions and biotic interactions (Donald & Evans 2006, Whittingham *et al.* 2007). The combination of processes on multiple scales (field, farm and landscape) reinforces the importance of rigorous assessment to determine AES contribution towards meeting national (Vickery *et al.* 2004), as well as global biodiversity goals (UNEP 2011).

Evaluations of AES have returned mixed results, with some studies reporting significant benefits for birds (Batary *et al.* 2011, Baker *et al.* 2012, Bright *et al.* 2015), whilst others have found either no, or negative effects (Kleijn & Sutherland 2003, Kleijn *et al.* 2006). Demonstrating a general pattern has been hindered by the many species-specific interactions between AES and population dynamics (Kleijn *et al.* 2006), as well as the presence of non-AES drivers of population change. The latter include extreme weather events, diseases, predation, competition, mating territoriality, and source-sink dynamics (Bradbury *et al.* 2003, Durell & Clarke 2004, Evans 2004). Furthermore, AES effects are contingent on landscape complexity and the area over which AES are implemented (Vickery *et al.* 2004). The interaction between AES performance and surrounding landscape reinforces the importance of investigating AES in all major UK regions, as there could be within-country variation in AES effects on bird populations. Most of the scientific attention so far has focused on farmland conservation in England, whilst Scottish AES have been the subject of relatively fewer studies. As Scotland implements the new Agri-Environment Climate Scheme in 2016, budgeted at £350 million (SRDP 2016), it is important to evaluate past schemes and make necessary changes to design an optimal and cost-effective strategy for the future.

Objectives and research questions

I aim to investigate the effect of agri-environmental schemes on farmland bird populations in Eastern Scotland. The AES during the study period were Rural Stewardship Scheme (2003-2008), Rural Priorities (2009-2015), and Farmland Bird Lifeline (2003-2015). The urgency and importance of studies like this is affirmed by their applied implications for government policies and sustainable land management. In particular, my research will contribute to the knowledge of regional species-specific interactions between avian abundance and land management. The results of my study can inform the optimisation of future schemes, such as the Agri-Environment Climate Scheme. To achieve this, I will investigate the population trends of six farmland bird species across farms with conventional and AES management, addressing the following research questions:

1. Do farms implementing AES support a higher total avian abundance than conventional farms?
2. How does each study species respond to AES management?
3. How is avian abundance (total and of individual species) influenced by farm area?

Research hypotheses

I hypothesise that there will be a positive relationship between AES treatment and total avian abundance, and between farm area and total avian abundance. When taking into account species identity, I hypothesise that species will have different relationships with treatment type and farm area, with both positive and negative interactions present. Those hypotheses will be tested against the null hypotheses of no relationship between treatment type and avian abundance, and farm area and avian abundance.

Predictions

I predict that in Eastern Scotland, AES farms will support a significantly higher total avian abundance than control farms, as AES generally reflect positively on population trends (Wilkinson *et al.* 2012). I have identified differences in ecological requirements among my target species, based on which I predict that species will respond to the schemes in individualistic ways. I predict that the relationship between farm area and abundance will also be species-specific, as some species, e.g. skylark (*Alauda arvensis*), make use of whole fields, whilst others, such as yellowhammer (*Emberiza citronella*), prefer field boundaries, which do not necessarily scale positively with farm area.

If I find support for my null hypotheses, this will indicate that after 13 years of implementation, AES have not had any effect on avian abundance, thus questioning their ability to deliver benefits for birds. If the results are in line with my alternative hypotheses, this will demonstrate the effectiveness of AES in enhancing farmland bird populations in Eastern Scotland.

Methods

Study sites

Study sites represented 53 farms in Eastern Scotland between 2003 and 2015 (Figure 1). The dominant land use types were spring- and autumn-sown cereals, oilseed rape and grasslands for silage or grazing. Mixed farming was the most common practice, with 95% of farms studied here using land for both arable crops and pastoral farming. There were two farm treatments – agri-environmental scheme (AES) and control (conventional farming, no scheme). Several farms changed management strategies during the study period, which was accounted for in statistical analysis by including a treatment type variable for each year of observations. Out of the 53 farms, 43 were under AES management and 26 under control treatment at some point during the study period. All AES were voluntary and competitive (farmers had to apply and be approved to participate), with the specific land management options outlined in Appendix 1. Across all years, mean area was 120 ha (± 4.6 SE) for AES farms and 104 ha (± 11.2 SE) for control farms. The dominant treatment duration was 13 years (56% of farms), with the remaining farms being under the same treatment for three to six years. Overall, I consider treatment duration to be sufficiently long to examine potential effects of treatment type on bird populations.

Farm selection was executed by the RSPB and surveys were undertaken as part of the work of Dr Allan Perkins, prior to the start of this project. AES farms were located within or adjacent to a 2 km square within which corn bunting had previously set breeding territories. Control farms were chosen so as to have similar land use to AES farms and be within 10 km of an AES farm.

Study species

I selected study species based on their suitability as indicators of farmland ecosystem health (French, Centre & Brathens 2002), resulting in the following six bird species – corn bunting (*Emberiza calandra*), linnet (*Carduelis cannabina*), reed bunting (*Emberiza schoeniclus*), tree sparrow (*Passer montanus*), skylark (*Alauda arvensis*) and yellowhammer (*Emberiza citronella*). Their ecology, main threats and conservation efforts are summarised in Appendix 2(a,b). All six species have shown significant nation-wide declines over the last 40 years (Balmer *et al.* 2013), thus making them a focal point of many of the AES policy and biodiversity targets (JNCC & DEFRA 2013, The Scottish Government 2013).

Data collection

Surveys were carried out between May and August of 2003, 2004, 2006, 2008, 2009, and 2015 by RSPB field biologists. Each year, farms were visited three times (70.5% of data points based on three visits) or two times, on calm mornings with good visibility. The number of farms surveyed varied between years, with details on this presented in Appendix 3. Survey routes were designed to pass within 250 m of all points on each farm, predominantly following field boundaries. Survey effort scaled positively with farm size, with larger farms having longer survey routes. During each visit, the locations and behaviour of all individuals from the six study species were recorded on a 1:10 000 map. The maps from the two or three visits were then superimposed on each other to determine the number of territorial males (for yellowhammer, reed bunting, corn bunting and skylark) or breeding pairs (for tree sparrow and linnet). Data on farming practices and farm areas were collected by Dr Allan Perkins. Further details on survey methodology are available on request from Dr Allan Perkins.

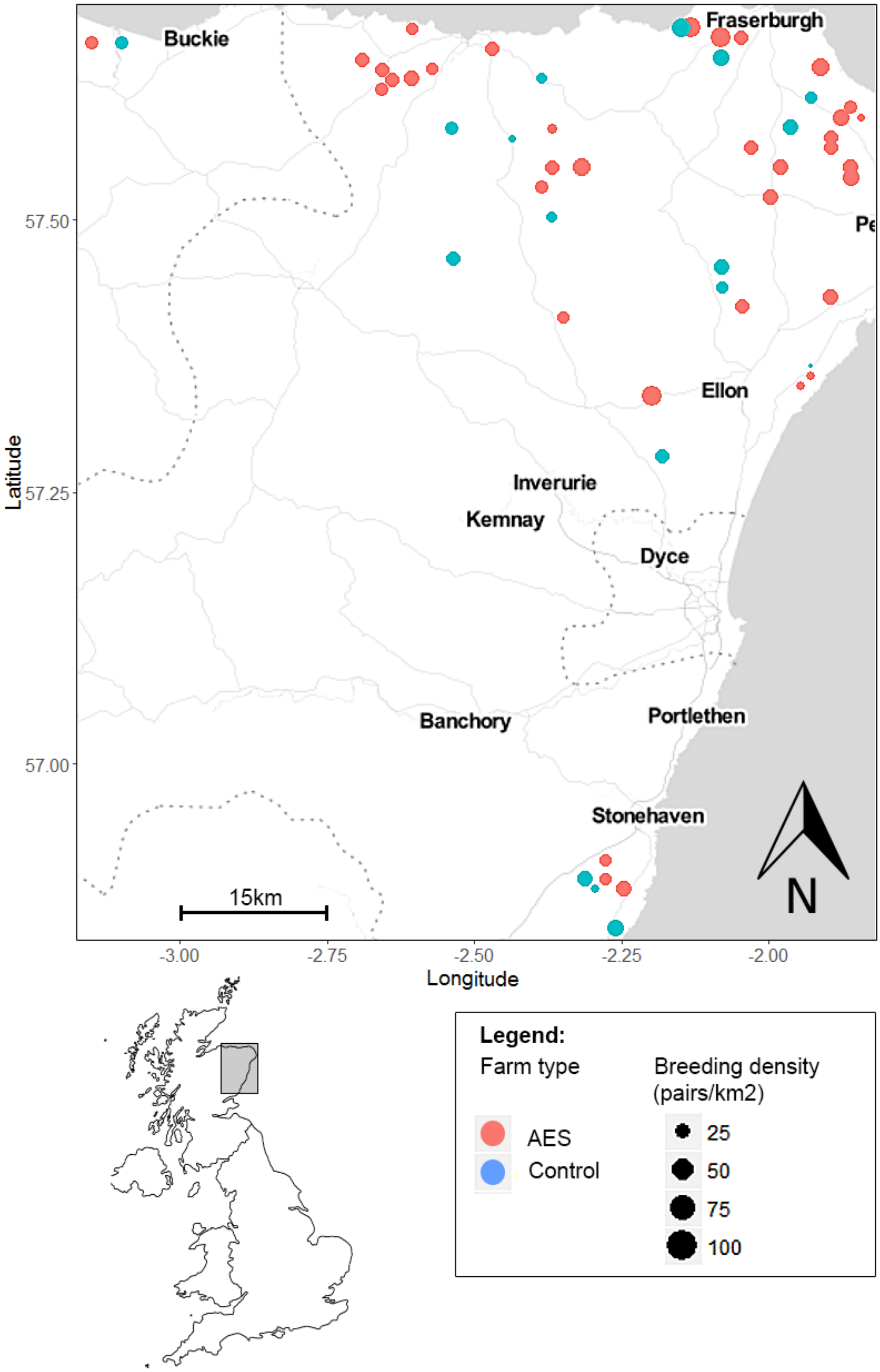

Figure 1. Locations of study sites within Scotland. Point colour indicates treatment type in 2015 and point size correlates positively with the summed up breeding densities of the six study species in 2015.

Data analysis

All statistical analysis was conducted in R v. 3.2.3 (R Core Team 2015). To visualise raw data, I used the package {ggplot2} (Wickham 2009). To create a map of study sites, I used the package {ggmap} (Kahle & Wickham 2016). To determine if avian abundance differed significantly between treatments (Research Question 1), I used a generalised linear mixed effects model based on a Markov chain Monte Carlo (MCMC) Bayesian method, implemented in the package {MCMCglmm} (Hadfield 2010). As the data represent skewed count data, a Poisson distribution was used. The MCMC Bayesian approach was selected because of its suitability for overdispersed data from a Poisson distribution (Hadfield 2015). Further benefits of the MCMC Bayesian methodology include the ability to calculate credible intervals, showing the range within which the probability of the real value of the predicted parameter occurring is 95% (Johnson 1999).

I modelled total avian abundance (the summed up breeding abundances of six study species) as the response variable. All three schemes (RSS, RP and FBL) were coded as the same treatment (AES), as scheme identity was not found to have a significant effect on avian abundance (mixed effects model where scheme identity was a fixed effect, p=0.24, n=154). A qualitative comparison of RSS, RP and FBL further confirmed sufficient similarities between schemes to justify including them as the same treatment (Appendix 1). Number of visits to a farm (two or three) did not have a significant effect on avian abundance and was excluded from the analysis (mixed effects model where number of visits was a fixed effect, p=0.06, n=230). Treatment type (AES vs control), year in which the observations were taken, farm location (latitude and longitude), farm area and treatment duration were included as fixed effects.

I included year as a fixed effect to account for the effect of one year's population on the next, as severe winters and wet cold summers can increase bird mortality, thus affecting the number of individuals available for mating in the following years (Fuller *et al.* 1995). I transformed area data to a logarithmic scale to account for the high variance in farm size (more than one order of magnitude). I centred latitudinal and year data, so that their intercept estimates correspond to the average value of the predictor. Longitudinal data did not need to be centred, as the study sites exhibited little longitudinal variation. Treatment duration was modelled as a fixed effect, with no quadratic term included, as I found no evidence of such a relationship between avian abundance and duration (Appendix 4). The random term in my model, farm identity, accounted for non-independence of data points. I could not include year as a random effect, because there was not enough information in the data to inform the posterior of the year effect.

Previous studies of farmland birds suggest that the type of farming (arable, grazing and mixed) and the season of sowing of crops (spring and autumn) influence avian biodiversity (Newton 2004). Here, 95% of farms practiced mixed farming and 96% of farms sowed crops either in spring, or in both spring and autumn. Farming practices were deemed to be sufficiently similar to justify not including them as fixed effects in the MCMC model. Previous evaluations of the effect of AES on farmland birds adopt a different methodology, in that area is included as an offset, thus effectively modelling density (Perkins *et al.* 2011, Bright *et al.* 2015). This approach follows the assumption of a linear relationship with a slope of one between abundance and area. My data violated this assumption, and I thus consider the offset methodology to be inappropriate for my analysis. I included area as a fixed effect and as such modelled avian abundance whilst accounting for variation in farm area.

To investigate possible crossed interactions between species and treatments, and assess if farmland birds are responding to treatments in a homogeneous or species-specific manner (Research Question 2), I included a species factor, and a species:treatment interaction term in the MCMC model. Here, the response variable abundance represented the individual abundance records for each species.

To determine how avian abundance is influenced by farm area (Research Question 3), I plotted generalised linear models of total avian abundance and farm area, and individual species abundance and farm area, using the package {ggplot2} (Wickham 2009).

Due to the inclusion of several model variables, model predictions were calculated by hand, as well as by using the predict command within {MCMCglmm}. Model predictions were then visualised using the package {ggplot2} (Wickham 2009). The full R script of this analysis is included in Appendix 5.

I used default priors for my MCMC models, as I consider them to be uninformative on fixed and random effects. The models were ran using 100 000 iterations with a thinning factor of 10. MCMC models were validated through visual examination of trace plots and density estimate distributions of model parameters. Further validation was conducted by examining values for autocorrelation between fixed and random terms, with values smaller than 0.1 considered as acceptable levels of autocorrelation, and by investigating estimates for effective sample size of each parameter (Hadfield 2015). To test for model overfitting, I plotted observed versus predicted values and compared their relationship with a 1:1 line, which represents a perfect fit. The closer the predicted ~ observed values relationship is to 1:1, the better the model fit is considered.

Model variables were included based on their ecological significance and known relationships between explanatory variables and the response variable. For my analysis, I worked with the full model, and did not execute any term deletion, which has been criticised due to increased probability of Type 1 errors (Mundry & Nunn 2009). I included area because of the well-documented relationship between area and species abundance (MacArthur & Wilson 2015); the effect of treatment can change with time, thus making duration important; latitude relates to land use history, with agriculture less intensive as latitude increases. Including latitude and longitude reduces the spatial signal and potential autocorrelation. Furthermore, the experimental design randomised the locations of AES and control farms with regards to environmental conditions. Here, spatial autocorrelation was ignored, potentially leading to underestimation of the credible intervals.

I used abundance as a measure of avian biodiversity. Another possible metric is density (territorial males or breeding pairs per km^2). To investigate if choice of biodiversity indicator has an effect on results, I performed supplementary analyses, which modelled density as the response variable, using the package {MCMCglmm} and a Gaussian distribution (Hadfield 2010). I excluded area as a fixed effect from those models, as the area data were incorporated in the calculations of density.

When presenting the results of the MCMC models, I will refer to statistical significance. While this terminology is not consistent with Bayesian statistics, where the correct phrasing would refer to the credible intervals not overlapping zero, for ease of writing and reading, I will use terms such as ‘significant difference’. The p values I will report are MCMC p values. Within the Bayesian community, such practices are already in use (Hadfield 2015). The effect sizes I will present are on a logarithmic scale, the abbreviation CI stands for 95% credible intervals, and b stands for slope estimate.

Results

Total avian abundance on AES and control farms (Research Question 1)

Visualising raw abundance data revealed a trend of higher total avian abundance (summed up breeding abundances of the six study species) on AES farms versus control farms (Figure 2). While avian abundance fluctuated between different years, in general bird populations remained stable and did not show any steep declines or increases on either farm type (Figure 2).

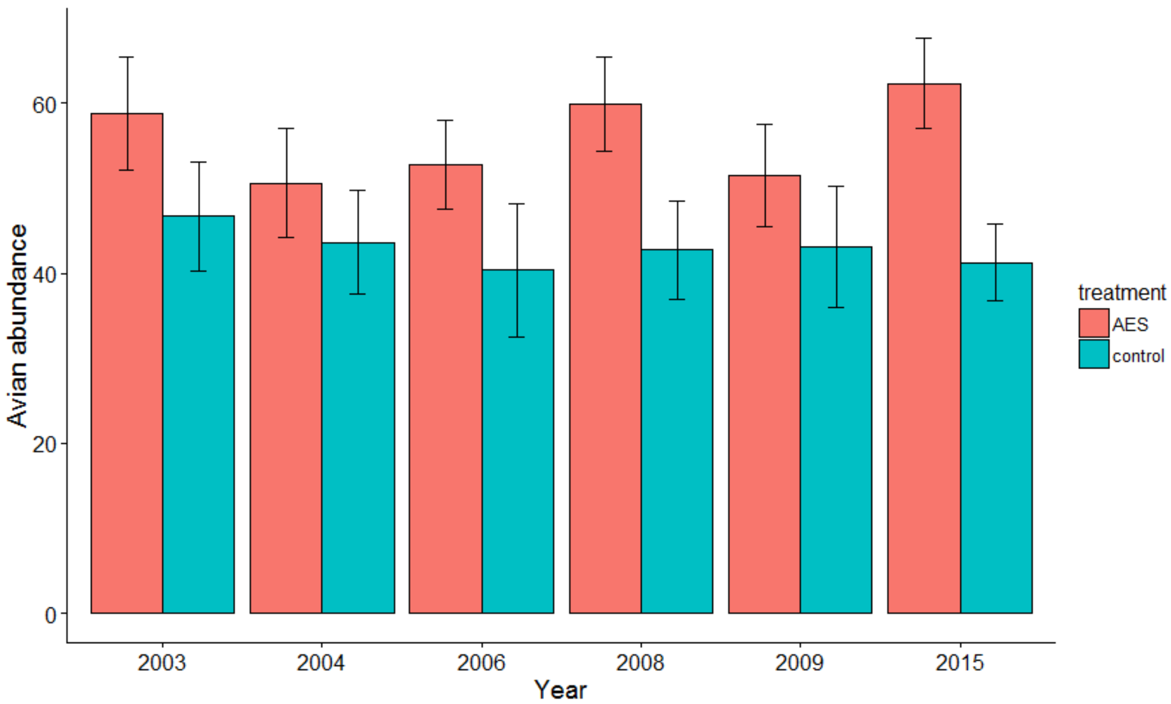

Figure 2. Mean total avian abundance during the study period. Mean values calculated from raw data records. Error bars show standard errors. Avian abundance is measured in number of breeding pairs.

The results of the total avian abundance model revealed that across all survey years, there was no significant difference between total avian abundance on AES and control farms ($p=0.77$, $n=230$). I found no support for my initial prediction of AES increasing total avian abundance, and thus accept the null hypothesis of no relationship between total abundance and treatment type. Farm area had a significant effect on avian abundance ($p<0.001$, $n=230$, $b=0.63$ (CI 0.47 – 0.81)). There was a significant latitudinal trend, with abundance increasing towards the north ($p=0.05$, $n=230$, $b=0.45$ (CI 0.01 – 0.90, a graph of the relationship is included in Appendix 6). This result, however, is to be taken with caution, as there is limited variation in latitudes and considerably fewer farms in the southern regions of the study area. No longitudinal effect on avian abundance was detected ($p=0.53$, $n=230$). Contrary to my initial predictions, avian abundance did not vary significantly among years or with treatment duration (p=0.79 and p=0.12, respectively, n=230). Examining the posterior means of the random term, farm identity, revealed among farm variation (variance=0.12 (CI 0.07 – 0.18), and little within farm variation (variance=0.03 (CI 0.02 – 0.04)). Full model output is presented in Appendix 7.

Species-specific responses to AES and control treatments (Research Question 2)

Visual examination of raw data did not reveal a clear difference between the abundances of individual species on AES and control farms (Figure 3). During the 12 years of this study, corn bunting and reed bunting abundance remained relatively stable, with AES farms supporting slightly more birds (Figure 4). Among the six study species, skylarks suffered the biggest population decline during the study period (Figure 4). Between 2003 and 2015, mean skylark abundance declined by 21.3% on AES farms and by 43.8% on control farms. Yellowhammer and tree sparrow are the only species whose populations are higher in 2015 than in 2003. Linnet numbers fluctuated between years, with no clear pattern apparent.

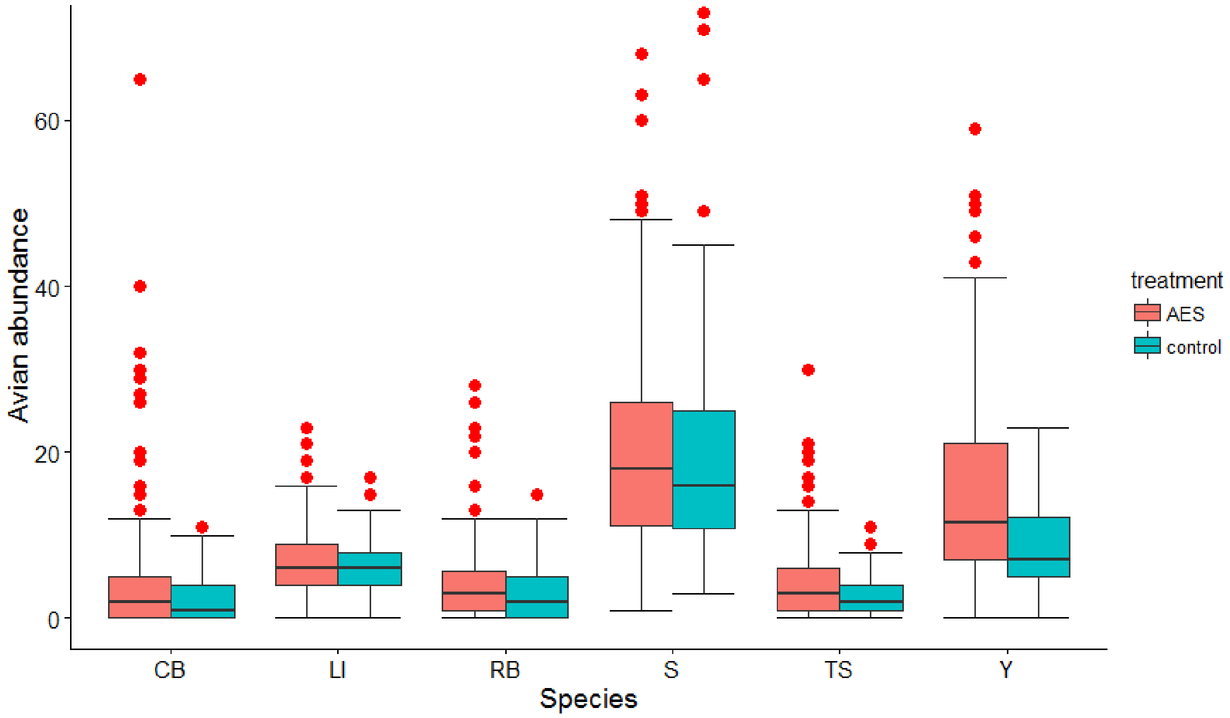

Figure 3. Boxplot comparison of individual species' abundances between AES and control farms. Based on raw abundance data from the entire study period. Error bars represent the distance between 1.5 * the inter-quartile range of the upper hinge and of the lower hinge. Data not falling within the error bars' extend are plotted as points. Avian abundance units are breeding pairs for TS and LI, and number of territorial males for CB, RB, S and Y.

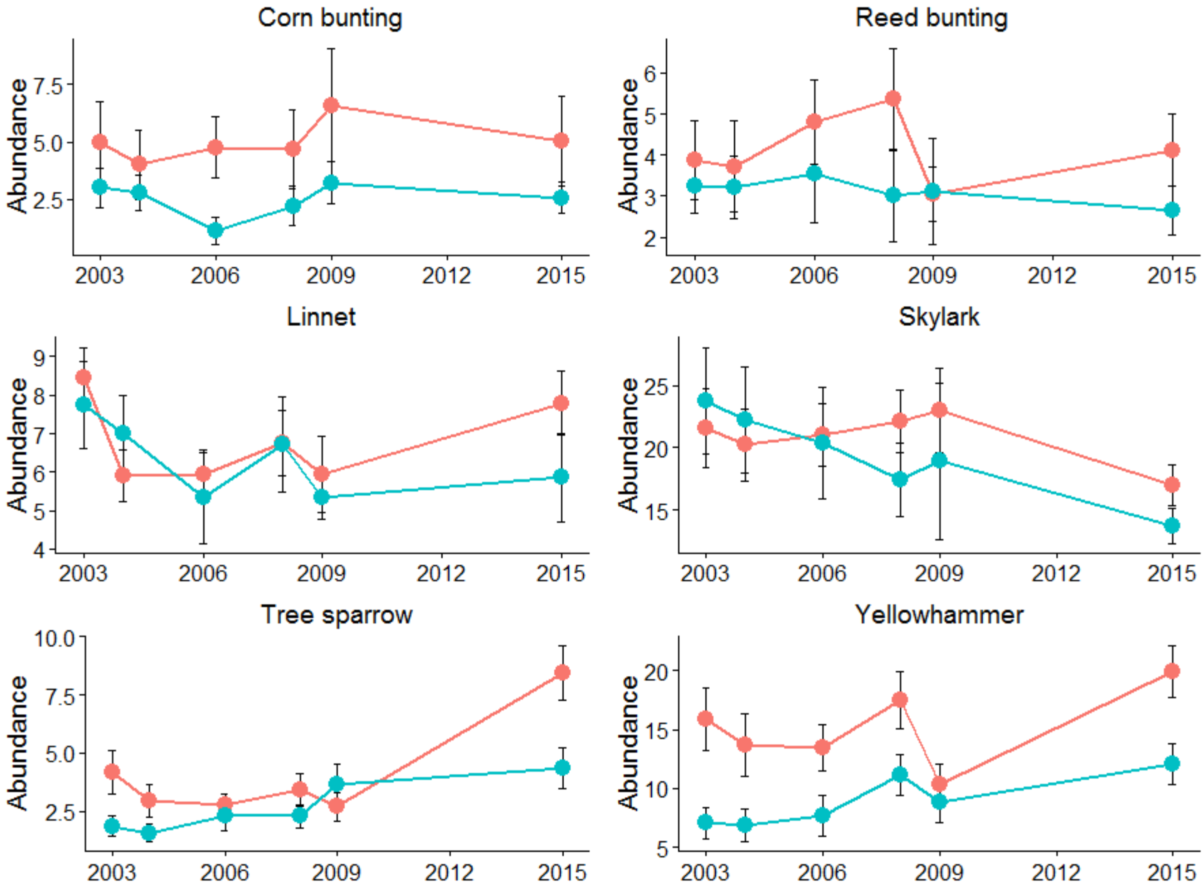

Figure 4. Abundance trends of target species on AES and control farms during the study period. Data points show mean abundance values and error bars represent standard errors, both based on raw abundance records. Red represents AES and blue represents control treatment. Abundance is measured in number of breeding pairs for TS and LI, and in number of territorial males for CB, RB, S and Y.

The model which compared abundance among species revealed that there is no significant difference between individual species' abundances on AES and control farms (p=0.32, b=-0.14 (CI -0.41 – 0.13), n=1380). Investigating the interaction between treatment type and species identity revealed that there is a marginally significant interaction between skylarks and treatment type (p=0.05, b=0.31 (CI -0.01 – 0.62), n=230), and between linnets and treatment type (p=0.05, b=0.32 (CI 0 – 0.64), n=230). These marginally significant values, however, are to be interpreted with caution, as it is possible that they are arising due to chance, especially considering the overlap of credible intervals between AES and control treatments for these two species (Figure 5). The remaining species (corn bunting, reed bunting, tree sparrow and yellowhammer) did not exhibit any significant interactions with treatment type (see Appendix 8 for full model output).

Model predictions confirmed my hypothesis of crossed interactions between species identity and treatment type, as abundance was higher on AES farms for corn bunting, reed bunting and tree sparrow, and higher on control farms for skylark and linnet. However, I found no evidence to support my hypothesis of a positive effect of AES on bird populations, as four species had no significant response to treatment type, with the remaining two (linnet and skylark) being negatively affected (Figure 5). Predicted values for corn bunting, reed bunting and tree sparrow abundances showed little variation based on treatment type (Figure 5). Yellowhammer is the only species for which the model predicted a distinctly higher abundance on AES farms, but this difference in abundance was not significant (p=0.43, n=230, Figure 5). Similarly to the total abundance model, area and latitude had a significant effect on individual species' abundances (p<0.001, b=0.57 (CI 0.41 – 0.77) for area, and p=0.02, b=0.53 (CI 0.07 – 0.99) for latitude, n=1380 for both). Examining the posterior means of the random term, farm identity, revealed relatively little among farm variation (variance=0.11 (CI 0.06 – 0.18). However, in contrast to within farm variation in the total abundance model, here within farm variation was considerably higher (variance=0.38 (CI 0.33 – 0.42)).

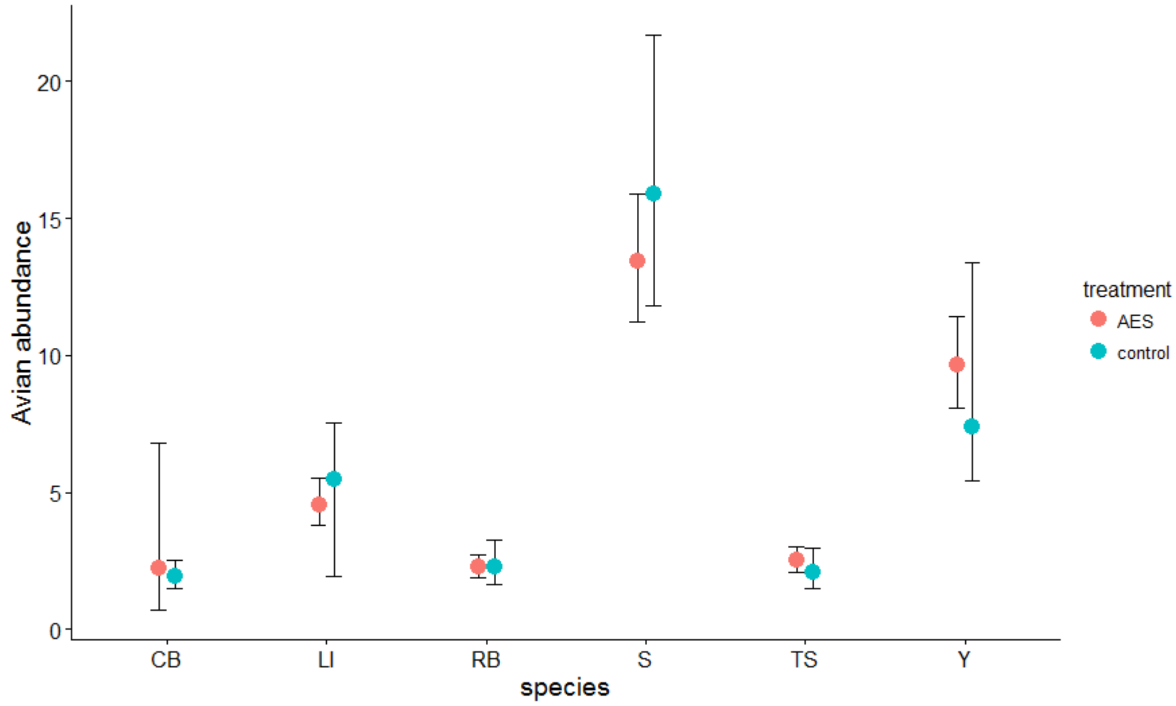

Figure 5. Model predictions for species' abundance across treatment types. To calculate prediction values, explanatory variables were standardised as area=100ha, longitude=-2,5°, treatment duration=5 years. Standardisation values were chosen to match real life farm characteristics. As the year and latitude data were centred prior to running the model, they were not included in the calculations. Data points represent the exponentials of predicted values, to transform from the model scale (logarithmic) to the data scale. Avian abundance is measured in number of breeding pairs for TS and LI, and in number of territorial males for CB, RB, S and Y.

Repeating the analysis using density as the response variable in the MCMC models revealed similar results to those outlined above, indicating that the general pattern of no effect of AES hold true regardless of which avian biodiversity metric is examined. Full density model outputs are included in Appendix 9(a,b).

Visual examination of trace plots and density estimate distributions of model parameters, and large estimates for effective sample size (all larger than 7500) confirmed model convergence. Examining the values for autocorrelation between fixed and random terms revealed no values higher than 0.1, showing that autocorrelation is not high enough to impede model convergence. Plotting predicted versus observed values revealed a good fit for the total avian abundance model (Appendix 10a). The more complex model with a species factor and a species and treatment interaction was found to perform well for lower abundances, but the relationship plateaued at abundances of around 25, indicating that the model is underpredicting high avian abundances (Appendix 10b).

Relationship between avian abundance and farm area (Research Question 3)

Total avian abundance scaled positively with farm area, confirming that larger farms support more birds (Figure 6). Abundance increased more steeply on AES farms than on control farms, but this trend was not significant (p=0.77, n=230).

Investigating the species-specific relationships between abundance and farm area revealed that target species respond to increases in farm area (on both control and AES farms) in different ways (Figure 7). Skylark and linnet abundance correlated positively with farm area, whereas corn bunting, reed bunting, yellowhammer, and tree sparrow abundance trends remained stable, regardless of increases in farm area (Figure 7).

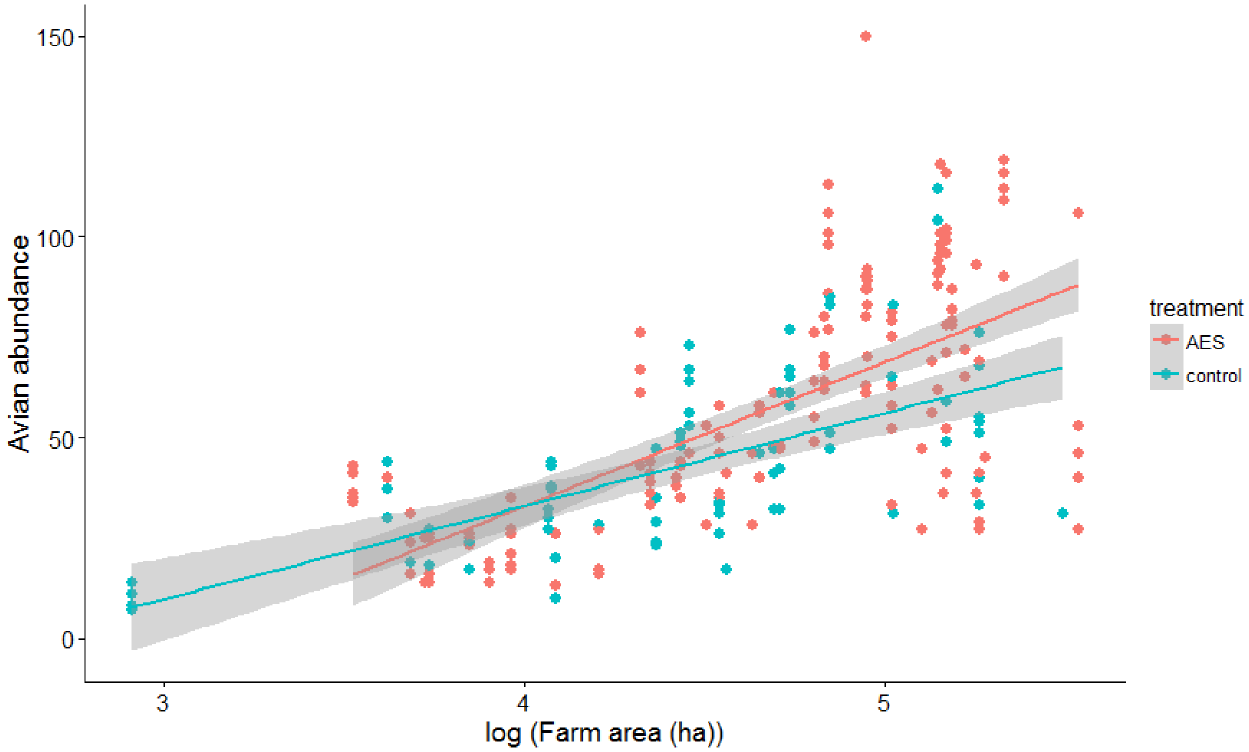

Figure 6. Relationship between total avian abundance and farm area. Data points represent raw data records and the lines show a generalised linear regression fit with 95% confidence intervals. Avian abundance is measured in number of breeding pairs.

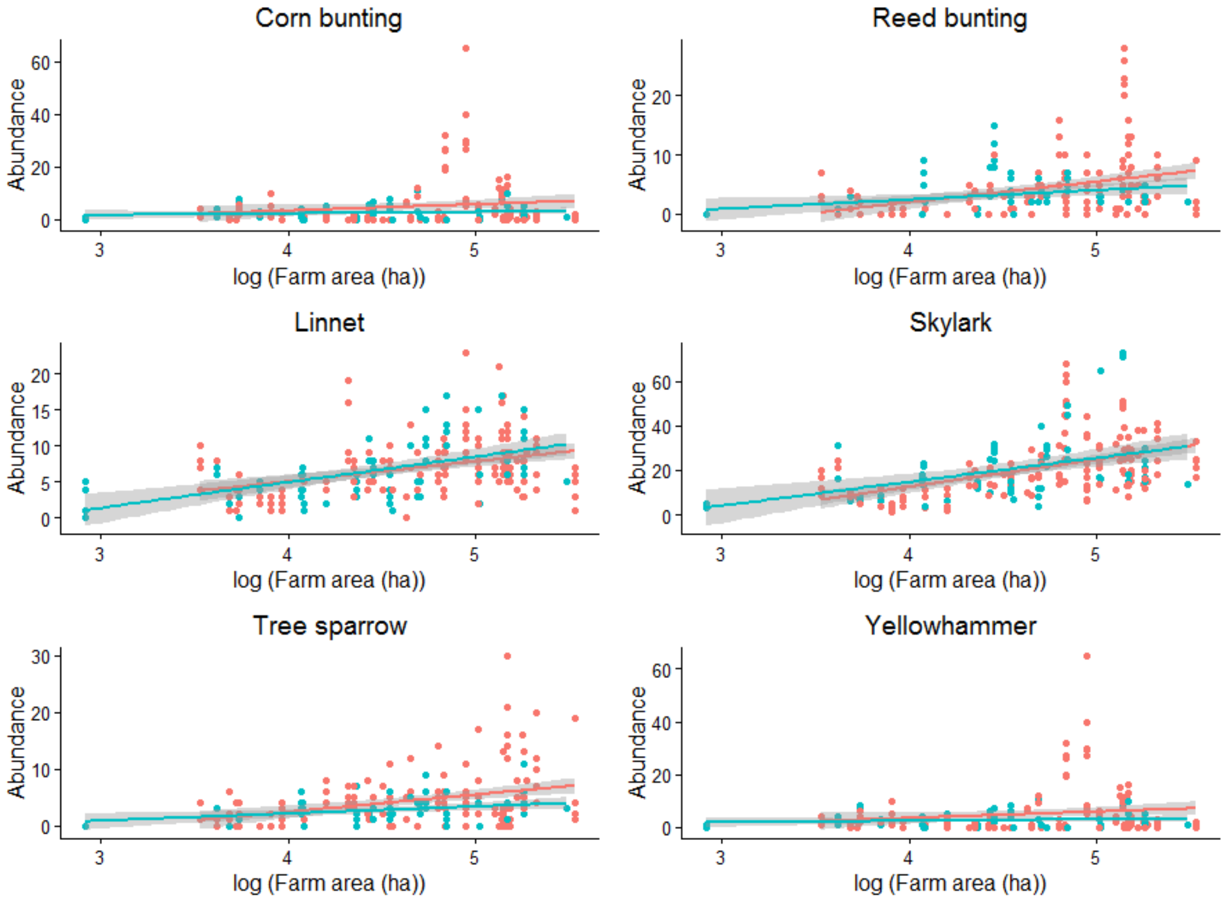

Figure 7. Species-specific relationships between avian abundance and farm area. Data points represent raw data records and the lines show a generalised linear regression fit with 95% confidence intervals. Red represents AES and blue represents control treatment. Abundance is measured in number of breeding pairs for TS and LI, and in number of territorial males for CB, RB, S and Y.

Discussion

Agri-environmental schemes operating from 2003 to 2015 in Eastern Scotland did not have a significant effect on the abundance of six farmland bird species (Figure 5). I found no support for my hypotheses of a positive relationship between total avian abundance and AES treatment, and individual species' abundance and AES treatment. I demonstrate that 'broad and shallow' agro-environmental policies have not lead to an increase in bird abundance after 13 years of implementation in the study area. Furthermore, I provide evidence for possible negative responses to AES, suggesting that AES are not delivering a broad benefit to farmland bird communities in Eastern Scotland. The lack of positive responses to AES could be due to a mismatch between scheme design and ecological requirements of birds, an implementation gap on behalf of farmers, and/or control farms not being as detrimental for birds as previously thought. The species-specific relationships between abundance and farm area are of particular interest, as they demonstrate that increasing area of AES farms does not enhance the populations of most study species (Figure 7). The RSS and RP's lack of positive effect on bird populations (Figure 4, Figure 5) suggests that the declining FBI trend in the UK will not be reversed, unless the design, implementation, and targeting of land stewardship is improved.

I found no evidence for a significant positive effect of AES on any on the six target species (Figure 5), which could be because the schemes are failing to provide key food and nesting opportunities. For example, there were no restrictions on pesticide application on 51 of the 53 AES farms (the remaining two are organic) and chemical use could be compromising invertebrate food availability for birds (Benton *et al.* 2002, Boatman *et al.* 2004). Furthermore, pesticide use could be neutralising AES benefits from stewardship options such as species-rich grasslands (Siriwardena *et al.* 2000, Kleijn *et al.* 2011). Additional ecological shortcomings of AES management include insufficient breeding habitat on AES farms (Bradbury & Kirby 2006, Bradbury *et al.* 2010, Perkins *et al.* 2012). Schemes are also static in their nature and do not account for variation in phenology. For example, AES farms delay mowing meadows until the 1st July to prevent breeding failure of ground-nesting birds, (The Scottish Government 2014), but fledging times vary due to weather and seasonal food availability and designating a set date for mowing does not ensure breeding success (Johst *et al.* 2015).

The mismatch between scheme design and resource needs of study species is most pronounced for skylark and linnet, which were predicted to be more abundant on control farms (Figure 5). Henderson, Vickery & Carter (2004) found that skylark abundance is driven by stubble availability, not farm participation in AES. Furthermore, skylarks do not have a direct benefit from unharvested crops or wild bird seed mixes, as they avoid tall vegetation (Henderson, Vickery & Carter 2004, Newton 2004, Koleček, Reif & Weidinger 2015). Patterns in their abundance might, therefore, be driven by sowing regime, which in turn impacts overwintering stubble availability. It is possible that because control farms do not have to allocate land for prescriptions such as the wild bird seed mix, they have more overwintering stubble fields, thus attracting more skylarks. Similarly, linnet, the other species for which I did not find an effect of AES in this study, has a distinct preference for oilseed rape fields, and its abundance patterns have previously been shown to be driven by oily seed availability, not AES management (Bright *et al.* 2015). The aforementioned mismatches suggest that applying general conservation strategies aimed at a diverse range of species is not successful in delivering biodiversity benefits.

Mismatches between scheme design and ecological requirements are a common drawback of "broad and shallow" conservation strategies, which consist of simple and general land management options, easily applied on a large scale (Vickery *et al.* 2004). Here, I provided evidence for the lack of positive effects of the "broad and shallow" approach on bird populations in Eastern Scotland during 2003-2015 (Figure 5). Similar results have been obtained by numerous other studies within the UK and in Europe (Chamberlain, Wilson & Fuller 1999, Kleijn *et al.* 2006, Kleijn *et al.* 2011), which has encouraged the development of "narrow and deep" schemes. Such conservation strategies include more targeted and explicit stewardship practices, but are more costly to implement (Vickery *et al.* 2004). "Narrow and deep" AES have been instrumental for the recovery of the cirl bunting (*Emberiza cirlus*), corncrake (*Crex crex*) and stone curlew (*Burhinus oedicnemus*) in England (MacDonald *et al.* 2012a, MacDonald *et al.* 2012b, Wilkinson *et al.* 2012). Evidence that targeting species is necessary to deliver considerable biodiversity benefits is also available from Estonia (Elts & Asko 2012) and Scotland (Perkins *et al.* 2011). The "narrow-and-deep" approaches, however, might have unintended consequences for non-target biota (MacDonald *et al.* 2012a, b).

"Narrow and deep" farmland conservation has enhanced the populations of target species and has had mixed effects on non-target species (MacDonald *et al.* 2012a, b). Schemes focusing on single bird species have been found to deliver benefits for butterflies and bumblebees, but to negatively affect beetles *Carabidae* spp. and forbs (MacDonald *et al.* 2012b). Negative effects of targeted AES could also occur within the same taxa, e.g. when the ecological requirements of the target species are opposite to what non-target species prefer. For example, corn buntings like to nest in tall grass and delayed mowing increases their breeding success (Perkins *et al.* 2013). Skylarks, on the other hand, nest in short grass, and tall vegetation might limit the number of broods they can rear in a given breeding season (Newton 2004). Those trade-offs demonstrate that AES could be failing to deliver a benefit for birds because AES struggle to optimise habitat conditions and nutrition availability for multiple species at once.

Benton, Vickery & Wilson (2003) argue that the problem of mixed responses to schemes can be alleviated by increasing habitat heterogeneity. A variety of nesting and foraging habitats, as well as several nutritional resources available, is suggested to simultaneously accommodate the ecological requirements of numerous species (Butler, Bradbury & Whittingham 2005, Vickery, Feber & Fuller 2009). Enhancing habitat heterogeneity, however, might not bring ubiquitous benefits across farmland communities. Teillard *et al.* (2014) provided evidence for the different effects of habitat heterogeneity on generalist and specialist species. For generalists, diverse habitats provide opportunities to supplement needed resources. Specialists, on the other hand, require a narrow range of resources and respond positively to increasing the area of their specialist habitat, and negatively to introducing different types of habitat (Teillard *et al.* 2014). Although increasing habitat heterogeneity does bring benefits to farmland ecosystems, measures which promote it should be implemented only after cautious environmental audit and consideration of species ecology and behaviour.

Due to mating territoriality, less competitive males could be forced to use sub-optimal habitats (i.e. in this case control farms) (Chamberlain, Wilson & Fuller 1999). Even if breeding birds are declining on control farms, non-breeding birds could be taking their place, thus keeping breeding abundance stable, whilst total abundance is decreasing (Durell & Clarke 2004). Spill-over effects and colonisation by individuals from other areas could be supplementing control farm bird abundance, thus masking possible detrimental effects of intensive farming (Kleijn *et al.* 2011). Furthermore, birds could be compensating for diminished resources by increasing foraging intensity and parental care (Bradbury *et al.* 2003). Finally, increased predation on AES farms could be neutralising the biodiversity benefits they deliver (Josefsson *et al.* 2013). For example, creating field boundaries is beneficial for bird and invertebrates, but those habitats are also used as corridors for predators such as foxes *Vulpes vulpes* and rodents *Rodentia* (Josefsson *et al.* 2013).

Even though the schemes have been operating for 13 years, lags in population responses to environmental change are still possible (Chamberlain *et al.* 2000). On control farms, in particular, the decline caused by intensive farming could be delayed due to a threshold-type relationship between abundance and resource availability (Chamberlain *et al.* 2000, Kyrkos, Wilson & Fuller 2010, Besnard & Secondi 2014). This idea relates to the extinction debt theory, first put forward by Tilman *et al.* (1994), and later discussed in an agro-environmental context by Kleijn *et al.* (2011). Even if birds can initially persist in intensively managed farmlands, if habitat quality and resource availability do not improve, their populations will ultimately go extinct. Consequently, even if the schemes studied here were not found to increase avian abundance, AES management could be serving a protective, rather than enhancing function (Kampmann *et al.* 2012). This phenomenon has been demonstrated in complex landscapes where agriculture is relatively less intensive than in monoculture-dominated agricultural regions (Concepción, Díaz & Baquero 2008, Kampmann *et al.* 2012). The farms in this study fit those criteria, thus suggesting that while AES in Eastern Scotland did not enhance bird populations between 2003 and 2015, they could be preventing the onset of future declines.

Here, I found a relatively small amount of among farm variation in avian abundance (variance=0.12 (CI 0.07 – 0.18) for the total abundance model, and variance=0.11 (CI 0.06 – 0.18) for the species-specific model). The low among farm variation could be indicating that the environmental conditions across farms are similar. In Scotland, climate and topography limit agricultural opportunities and the amount of agricultural intensification that can occur, thus possibly rendering farm conditions similar, regardless of treatment type. Within farm variation was distinctly different between the total abundance model (variance=0.03 (CI 0.02 – 0.04) and the model that compared abundance among species (variance=0.38 (CI 0.33 – 0.42)). The relatively high amount of within farm variation in individual species' abundance during the study period suggests that farm conditions are interacting with year of observation. There is no consistent pattern of certain years having a ubiquitous positive or negative effect on birds, and instead, certain years are good for bird populations on some farms, and detrimental on others. These results are consistent with the idea that processes influencing avian abundance are operating on multiple scales (field, farm and landscape), with possible interactions between farm-specific conditions, environmental conditions during year of observation, and larger-scale landscape factors such as regional amount of intensification.

I presented evidence for species-specific relationships between farm area and abundance (Figure 7). In particular, I demonstrated that only two out of six bird species are more abundant on larger farms, whilst the abundances of the rest remain relatively stable regardless of increases in farm area. Inferences from island biogeography theory (MacArthur and Wilson 2015) have led to the widespread assumption that increasing AES area will lead to an increase in avian abundance (Vickery *et al.* 2004). Whilst large scale AES land stewardship will undoubtedly bring some benefits to birds and biodiversity in general, I found that the relationship between area and avian abundance in Eastern Scotland is not as straightforward as previously assumed. Species which utilise whole fields, such as skylark and linnet, are more abundant on bigger farms

(Figure 7, also demonstrated by Chamberlain, Wilson & Fuller 1999). Boundary and edge specialists, such as corn bunting, reed bunting, tree sparrow and yellowhammer, show similar abundances on small and large farms (Figure 7), as their preferred habitats do not necessarily increase linearly with farm area (Chamberlain, Wilson & Fuller 1999). This finding reinforces the importance of closely matching AES to ecological requirements of birds in order to ensure that the benefit that the schemes deliver does indeed increase when the schemes become more widespread.

In addition to the aforementioned ecological explanations for lack of AES effect on birds, it is also possible that the schemes did not enhance avian populations because of poor scheme implementation on behalf of farmers. Farmers' attitude towards the environment has been shown to influence the quality of AES management (Herzog *et al.* 2005). The funding they receive is based on the quantity rather than quality of land dedicated to certain stewardship measures (Canton *et al.* 2009). Consequently, there is the possibility of adverse selection – farmers use their most fertile land for intensive farming, and enter low quality land they might not even have used otherwise into an AES scheme (Quillérou & Fraser 2010). Furthermore, payments for conservation initiatives such as sowing a wild bird seed mix to provide food for overwintering birds are based on the area of the sowing, not on its yield (Perkins, Maggs & Wilson 2008). If the seeds are sowed on low quality land or not taken care of properly, they could fail to establish and set seed, thus creating an implementation gap (Perkins, Maggs & Wilson 2008).

Sub-optimal AES implementation by farmers stems from lack of engagement with the conservation aspect of AES and farmers viewing of the schemes simply as sources of income. A social study of farmers in Aberdeenshire, Scotland, where most of the farms studied here are based, found that farmers do not connect with the schemes and tend to focus on doing the bare minimum to receive the subsidies (Burton *et al.* 2008). Low farmer engagement is attributed to an absence of opportunities to demonstrate skills and innovate (Burton *et al.* 2008). When their attitude towards AES is purely utilitarian, farmers evaluate land stewardship based only on aesthetics (Burton *et al.* 2008). Such views lead to a preference for land management options which allow farmers to showcase their abilities in front of their peers, for example erecting fences. Conversely, managing scrub habitats, albeit of good conservation value, is disregarded since they are not visually pleasing (Burton *et al.* 2008). As AES management was not found to be embedded within farmer culture (Burton *et al.* 2008), the sustainability of the schemes is uncertain, since farmers can revert back to intensive farming after the five year period of the scheme has elapsed.

Farmer engagement can be encouraged through result-oriented schemes (Matzdorf & Lorenz 2010), where farmers are rewarded for their management via subsidies only after they have met certain conservation targets (Burton & Schwarz 2013). Such schemes have been successfully piloted in Germany (Matzdorf & Lorenz 2010). A particularly beneficial aspect of result-oriented schemes is that they allow farmers to learn from their experience and adapt future actions based on what worked well on their farm (Matzdorf & Lorenz 2010). A major drawback of British AES is that stewardship options remain static for the AES period – if farmers notice that an AES option is not bringing biodiversity benefits, they cannot change their management (Perkins *et al.* 2011). Result-oriented schemes, however, increase the responsibilities of farmers, and rely on them having knowledge to undertake adequate conservation practices (Burton & Schwarz 2013). Unless these schemes are complemented by farmer education and guidance by ecology experts, they could lead to unforeseen effects on farmland ecosystems.

Regardless of how well AES are implemented, it is possible that they affect biodiversity negatively through displacing environmental pressure (Ekroos *et al.* 2014). If setting aside land for wildlife meadows, field margins and hedgerows decreases the overall yield of a farm, this can be an incentive for its owners to manage the remaining land more intensively to gain bigger harvests from it and meet total production goals (Ekroos *et al.* 2014). This effect is well documented for national parks in agricultural areas – even though the parks protect habitats and ecological communities within it, the resulting intensification of the land around it could compromise biodiversity through negative edge effects (Lambin & Meyfroidt 2011). In the context of farmland ecosystems, schemes serve a similar function to a national park, and future evaluations of their effectiveness ought to also consider their indirect effects on non-AES land. Following such assessments, land sparing (increasing yields through intensive farming, so that less land is needed for agriculture), as opposed to land sharing (extensive farming), could be deemed a more appropriate conservation strategy for certain agricultural regions (Mattison & Norris 2005).

While I demonstrated that AES did not have a significant effect on bird populations in Eastern Scotland between 2003 and 2015, the schemes could still be impacting other biota in both positive and negative ways. The effect of extensive management on different taxa has been variable, with mixed results being common (Kleijn *et al.* 2006). Fuentes-Montemayor *et al.* (2011a) found that Scottish AES (RSS and RP) do not have an effect on bats *Pipistrelle* spp. Land management options within those schemes, however, were successful in enhancing moth populations (Fuentes-Montemayor *et al.* 2011b). Other taxa that have benefited from AES in the UK include grasshoppers *Melanoplus* spp. and crickets *Acheta* spp. (Kleijn *et al.* 2006). Pollinators, in particular bumblebees *Bombus* spp., also respond positively to AES, as field margins and species-rich grasslands increase pollen and nectar availability (Lye *et al.* 2009). Holistic evaluations of the effect of land stewardship on multiple taxa have revealed that communities do not respond in a uniform manner (Parish & Sotherton 2004, Marshall, West & Kleijn 2006). Setting appropriate targets, prioritising and resolving trade-offs situations is, therefore, a major challenge for the design and implementation of future AES.

The new Agri-environment Climate scheme has the potential to perform better than its predecessors, RSS and RP, which had no effect on farm-scale abundances of six bird species in Eastern Scotland between 2003 and 2015. The Agri-environment Climate scheme will be spatially targeted, with certain land management options available only in the areas where they will be most cost-effective (The Scottish Government 2016). The scheme can be further improved by taking into account global change processes, such as climate change (Memmott *et al.* 2010). The current species composition of wildlife grasslands, field margins, hedgerows and winter bird seed crops will not necessarily be optimal for farmlands in the context of global warming and increased extreme weather events (Memmott *et al.* 2010). The discrepancies between static policy decisions and dynamic environments can be alleviated through performance evaluation and adaptive management (Memmott *et al.* 2010), thus reinforcing the importance of evaluations such as the one presented here.

AES shortcomings yet to be addressed include the lack of precise targets, and the mismatch between the scales of available targets (national and EU-wide) and of AES implementation (local) (Kleijn *et al.* 2011). AES objectives have been defined as to enhance farmland biodiversity, reverse population declines and ensure the provision of ecosystem services (The Scottish Government 2016). The Agri-environment Climate scheme includes further aims regarding carbon sequestration and carbon reduction commitments (The

Scottish Government 2016). However, no quantitative objectives, or timelines for these goals have been formally set (Kleijn *et al.* 2011). Determining the contribution of AES towards national biodiversity targets such as Biodiversity 2020, and towards reversing the declining FBI trend remains a process largely based on inferences. As Scotland implements its new AES, it is crucial for policy makers and scientists to address mismatches between scheme design and species ecological requirements, implementation gaps, as well obstacles to adequate AES evaluation.

Study limitations

The limitations of this investigation stem from practical and logistical issues, with the added complication of surveying real functioning farms. All data were collected prior to the start of this investigation, with the focus of monitoring corn bunting populations. No richness data were available and consequently, no biodiversity indices, such as Simpson's or Shannon's, and Sørensen–Dice similarity index, could be calculated. Due to limitations in resources, farms could not be surveyed every year, with a considerable time gap between 2009 and 2015. I, therefore, cannot make any conclusions regarding population trends during that timeframe, and it is possible that the lack of data masks variability and meaningful fluctuations, such as responses to harsh winters. Farm selection was hindered by the availability of functioning farms, in particular control farms. As AES gain popularity, finding control farms becomes an increasingly arduous task. Further restrictions applied to 2009, when due to resource limitations, a subset of farms was surveyed (Appendix 3).

The data set was not big enough to allow for year to be included as a random effect and model convergence was not achieved when the random term was added. No information regarding bird abundance before the schemes were implemented is available, which meant that I could not designate appropriate priors in the MCMC generalised mixed effects models. The lack of priors in turn hindered the design of a model with both random slopes and random intercepts, which would have allowed for each farm to have its own relationship between area and abundance. Evaluating model fit (Appendix 9a,b) revealed that the species-specific model performs well up to medium abundances, but is underpredicting avian abundances higher than 25. This could be due to the presence of unknown factors which have an effect on avian abundance, but were not included in analysis, as is often the case with ecological Poisson data. Further statistical challenges include working with overdispersed and zero-inflated data. The MCMC Bayesian methodology addresses these issues through using an additive model to tackle overdispersion and zero-inflation (Hadfield 2015), thus removing the deleterious effects of those issues in this analysis.

Directions for future studies

A particularly interesting direction for future studies is the interaction between scheme performance and landscape context (Hiron *et al.* 2013). Gabriel *et al.* (2010) have suggested a methodology for addressing the spatial distribution of control and AES farms – a hotspot versus coldspot framework, where hotspots indicate an aggregation of AES farms, and coldspots signify that the AES farm is predominantly surrounded by conventional intensive farming. Including a hotspot/coldspot categorical variable as a fixed effect in a mixed effects model can be used to answer questions regarding AES benefit amplification, when AES farms are closer together and create a patch of continuous habitat. Furthermore, the coldspot effect can be used as a proxy for source-sink dynamics, as the results can determine if an AES farm among intensive farms is a safe haven, i.e. a source, or if the avian biota is so damaged that the AES has no effect, i.e. there are no birds to protect and the entire area is a population sink. Knowledge of the landscape interaction can then be used for spatial prioritisation of scheme implementation across the UK.

A national-scale study of the effects of schemes on multiple taxa, in addition to birds, can help overcome the mismatch in scales of biodiversity goals (national and EU) and AES implementation (localised conservation actions) (Kleijn *et al.* 2011). Large scale environmental audits before and after scheme implementation, can be used to study β diversity on farmland ecosystems, as well as processes of biotic homogenisation (Doxa *et al.* 2012). Studies over large spatial and temporal scales have the potential to disentangle causal and correlational factors and to determine not only if biodiversity is increasing or decreasing, but if species composition is changing (Dornelas *et al.* 2014). The results from such investigations will be paramount for our understanding of global change processes and their impacts on biodiversity, particularly in the context of a habitat type as wide-spread as farmlands.

Conclusion

This study advances our understanding of how farmland bird populations respond to different land management practices and provides an evaluation of the main conservation strategy for reversing the FBI decline – agri-environmental schemes. First, I demonstrated that agri-environmental management has not led to significant increases in farmland bird abundance in Eastern Scotland between 2003 and 2015. Second, I investigated species-specific responses to treatment types and contrary to my initial expectations, found evidence for a negative effect of AES on skylark and linnet. The lack of AES benefit on avian biodiversity could be due to several reasons: the schemes are missing the key ecological requirements of bird species, there are gaps in scheme implementation on behalf of farmers, or the habitat provided by control farms is not as degraded as in other parts of the UK. Third, I provided evidence for individualistic abundance-area relationships for farmland birds and showed that to augment AES benefit, it is not sufficient to only increase the total area of AES implementation, but the proportion of field boundaries and edge habitats also has to be considered. I conclude that during 2003-2015, AES have not enhanced the populations of six bird species, but acknowledge that the schemes could be having a positive effect on other biota, such as plants, invertebrates, bats and small mammals.

In Europe, and across the UK, there have been mixed effects of AES implementation, thus asserting the importance of surrounding landscape and biogeographical processes such as source-sink relationships, and behaviour and distribution patterns. Tackling the loss of farmland biodiversity is a complicated task due to the many stakeholders involved, and the complexity of the ecological systems. Spatial prioritisation and implementation optimisation are paramount for the advance of farmland conservation. Although targeted AES have been successful in the recovery of individual bird species, a long lasting benefit of AES across different taxa on a large scale is yet to be documented. If AES are to reverse the declining FBI trend, more ecological and monitoring expertise ought to be involved in the processes of decision-making and implementation. Assessing farm biodiversity a priori, following adaptive management practices, and regular assessment of achievements, will form the base of a successful farmland conservation strategy.

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Appendices

Appendix 1. Land management options (LMO) available under Rural Stewardship Scheme (RSS), Rural Priorities (RP) and Farmland Bird Lifeline (FBL). The three schemes are similar in design, with 14 out of 15 LMO shared between at least two schemes. LMO included here represent prescriptions which are thought to have an effect on farmland birds, and the rest of the LMO were excluded from this table. Adapted from The Scottish Government (2006), SRDP (2014) and FBL data from Dr Allan Perkins.

LMO	RSS	RP	FBL
1. Creation and Management of Species-Rich Grassland	YES	YES	YES
2. Creation and Management of Wetland	YES	YES	NO
3. Conversion and Maintenance of Organic Farming	NO	YES	NO
4. Extensive Management of Mown Grassland for Birds	YES	YES	YES
5. Extensive Management of Mown Grassland for Corn Bunting	NO	YES	YES
6. Management of Extended Hedges	YES	YES	NO
7. Management of Grass Margin or Beetlebank in Arable Fields	YES	YES	YES
8. Management of Habitat Mosaics	NO	YES	NO
9. Management of Hedgerows	YES	YES	NO
10. Management of Open Grazed Grassland for Birds	YES	YES	YES
11. Management of Species-Rich Grassland	YES	YES	YES
12. Management of Water Margin	YES	YES	NO
13. Management of Wetland	YES	YES	NO
14. Scrub and Tall Herb Communities	NO	YES	NO
15. Unharvested Crops	YES	YES	YES

Appendix 2(a). Ecological and conservation profiles of corn bunting, linnet and reed bunting. Information adapted from Balmer *et al.* (2013) and DEFRA (2015b). Photographs by Gergana Daskalova, except linnet photo, which is by Calvin Smith (Creative Commons License).

Species	Ecology and conservation
Corn bunting (*Emberiza calandra*) 	Corn buntings are granivorous birds, which supplement their diet with insects during summer and when raising their brood. They are associated with field boundaries, and grassland habitats. In the UK, corn buntings are resident breeders, and have suffered a 91% population decline over the last 40 years. Agricultural intensification and the early cutting of grasslands is the main threat to their populations, as they do not have enough time to raise their young.
Linnet (*Carduelis cannabina*) 	Linnets are granivorous birds, associated with farmland and field boundaries. They are partial migrants, moving in flocks during the winter. In the last 40 years, their populations in the UK have decreased by 60%. Their decline is associated with the loss of suitable breeding and feeding grounds, especially the scarcity of wildflower seeds.
Reed bunting (*Emberiza schoeniclus*) 	Reed buntings' diet consists of seeds, supplemented by invertebrates during the breeding season. They are resident breeders, associated with farmland and wetland habitats. British reed bunting populations have decreased by 37% over the last 40 years, with the causes of the decline less studied compared to other farmland birds. Possible reasons include the loss of wetlands and the increased use of pesticides.

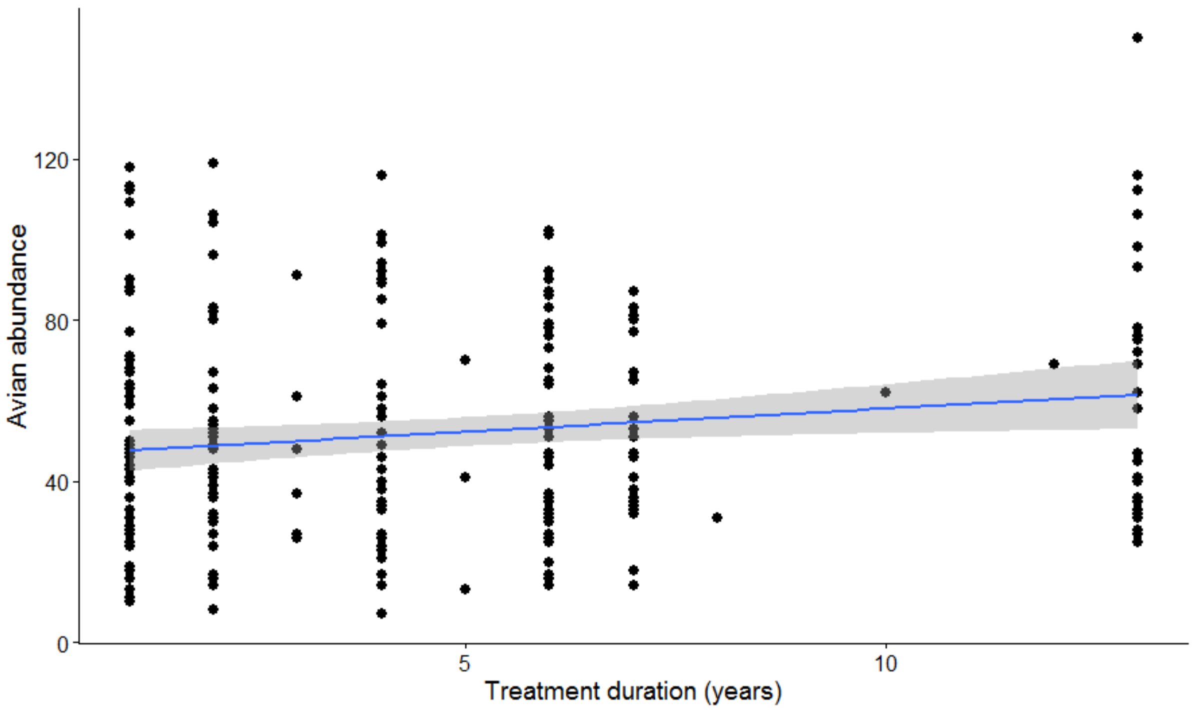
Appendix 2(b). Ecological and conservation profiles of tree sparrow, skylark and yellowhammer. Information adapted from Balmer *et al.* (2013) and DEFRA (2015b). Photographs by Gergana Daskalova.

Species	Ecology and conservation
Tree sparrow (*Passer montanus*) 	Tree sparrows are granivorous birds, which rely on invertebrates to feed their young. They are resident breeders, associated with farmland and open woodland habitats. Their UK populations have suffered a 90% decline, mostly due to the decrease in food availability, and loss of suitable nesting habitat.
Skylark (*Alauda arvensis*) 	Skylarks are granivorous during the winter, and insectivorous during the summer. They are resident breeders, whose preferred habitat is grasslands. In the last 40 years, skylarks have declined by 60% across the UK, mostly due to the switch from spring-sown to autumn-sown cereals, the loss of overwintering stubble, and a decreased food supply due to intensive agriculture.
Yellowhammer (*Emberiza citrinella)* 	Yellowhammers are granivorous birds, which supplement their diet with invertebrates during the summer. Yellowhammers prefer grassland and farmland habitats, in particular field boundaries. In the UK, where they are resident breeders, their populations have declined by 55% over the last 40 years. This negative trend is attributed to the decline in food availability during winter and summer, and the loss of hedges.

Appendix 3. Number of AES and control farms surveyed in each study year. Due to resource limitations, only a subset of farms could be surveyed in 2009. The growing popularity of AES constrained the number of control farms available for surveying.

year	AES farms	Control farms
2003	23	16
2004	25	14
2006	30	9
2008	25	14
2009	17	8
2015	34	14

Appendix 4. Relationship between avian abundance and treatment duration. Based on the weak positive trend, I consider the inclusion of a quadratic duration term in the mixed effects models unnecessary. Data points represent all records for total avian abundance (raw data), n=230. Line represents a generalised linear model fit with 95% confidence intervals.



Appendix 5. R script for statistical analysis conducted as part of this investigation.

```
# R script written by Gergana Daskalova 1/5/16

# Analysing abundance data of six farmland bird species provided by Dr Allan Perkins
# Determining if avian abundance varies across two farm treatments (agri-environmental scheme and control)


# Loading the necessary packages

library(MCMCglmm)
library(Rmisc)
library(dplyr)
library(ggplot2)
library(ggmap)
library(gridExtra)
library(ggmap)
library(dismo)
library(rgdal)
library(gpclib)
library(rgeos)
```

```
# Importing the data and checking that it imported alright

   # Total avian abundance (summed up abundances of six species)

abundance <- read.csv2("C:/Users/user/Desktop/DISSERTATION DREAMS/Core csv files/abuntot.csv")
head(abundance)
tail(abundance)
summary(abundance)

   # Individual species' abundance

spintAC <- read.csv2("C:/Users/user/Desktop/DISSERTATION DREAMS/Core csv files/spintAC.csv")
head(spintAC)
tail(spintAC)
summary(spintAC)
```

```
# Visualising raw total abundance data

abun_hist <- ggplot(abuntot, aes(x=abundance))  +  geom_histogram(binwidth=10, colour="black",
fill="white")  +  geom_vline(aes(xintercept=mean(abundance)),  color="red", linetype="dashed", size=1)

abun_hist <- abun_hist +labs(x="Total avian abundance of six farmland bird species", y="Count")  +
theme(axis.text.x = element_text(size=12), axis.text.y = element_text(size=12),  axis.title.x =
element_text(size=14), axis.title.y = element_text(size=14))
abun_hist

   # Skewed count data, therefore a Poisson distribution will be used
```

```
# Making a bar chart of raw total abundance data to visualise how abundance varies across treatments and across years

absum <- summarySE(abuntot, measurevar="abundance", groupvars=c("treatment","year"))
absum$year_factor <- as.factor(absum$year)

abunraw <- ggplot(absum, aes(x=year_factor, y=abundance, fill=treatment)) +
geom_bar(position=position_dodge(), stat="identity", colour="black") +
geom_errorbar(aes(ymin=abundance-se, ymax=abundance+se), width=.2, position=position_dodge(.9))

abunraw <- abunraw + labs(x="Year", y="Avian abundance") + theme_classic() + theme(axis.text.x =
element_text(size=12), axis.text.y = element_text(size=12), axis.title.x = element_text(size=14),
axis.title.y = element_text(size=14))
abunraw     # AES has slightly larger abundances, but not a stark difference

# Visualising individual species' abundance raw data

   # Making a boxplot comparing abundaces among species across treatments

     abun_sp_box <- ggplot(spintAC, aes(species, abundance, fill=treatment, dodge=treatment)) +
     stat_boxplot(geom ='errorbar') + geom_boxplot(outlier.colour = "red", outlier.size = 3) +
     labs(x="Species", y="Abundance (territorial males or pairs)") + theme(axis.text.x =
     element_text(size=12), axis.text.y = element_text(size=12), axis.title.x = element_text(size=14),
     axis.title.y = element_text(size=14))
          abun_sp_box     # Big overlap between abundances on AES and control

   # Making line graphs showing how the abundance of each species varies across treatments and
years.

     CB <- filter(spintAC, species=="CB")
     CBsum <- summarySE(CB, measurevar="abundance", groupvars=c("treatment","year"))


     B1 <- ggplot(CBsum, aes(x=year, y=abundance, colour=treatment)) +
     geom_errorbar(aes(ymin=abundance-se, ymax=abundance+se), colour="black", width=.2) +
     geom_line(size=.9) + geom_point(size=4) + theme_classic() + labs( y="Abundance", title="Corn
     bunting") + theme(axis.text.x = element_text(size=12), axis.text.y = element_text(size=12),
     axis.title.x = element_text(size=14), axis.title.y = element_text(size=14))  +
     scale_x_continuous(breaks=seq(2003, 2015, 3)) + theme(axis.title.x = element_blank(),
     legend.position="none")

    # Graphs for the remaining five species were created in identical ways to the above, substituting CB
     for S, etc. Note for future analyses: I can make a function to optimise the graph making process.
```

```
# Centering latitudinal and year data for total abundance model to avoid forcing it to predict values for e.g. year 0,1,2,3 all the way up to 2003. Year and latitude data were centered for all models.
     # Their intercept estimates correspond to the average value of the predictor

    abuntot$yearcenter <- abuntot$year - mean(abundance$year)
    abuntot$latcenter <- abuntot$Lat - mean (abundance$Lat)
```

```
# Running the MCMC mixed effects model predicting total abundance

   # pr=TRUE so that I can get predicted values using the predict command later
   # nitt=100000, which I consider to be enough iterations
   # I used default priors, as no data on abundance prior to treatment implementation are available
   # I used default thinning factor (10), every ten iterations are stored

abun_tot <- MCMCglmm(abundance ~ treatment + log(area) + duration + yearcenter + latcenter + Long,
random=~farm, data=abuntot, pr=TRUE, nitt=100000, family="poisson")
summary(abun_tot)

   # Model validation - the same procedure was followed for every model presented here.

       # Checking that the effective sample sizes are large (in summary output)

       # Autocorrelation, values below 0.1 acceptable

         autocorr(abun_tot$Sol)
         autocorr(abun_tot$VCV)

      # Examining trace plots and density estimate distributions of model parameters to confirm
       convergence

         plot(abun_tot$VCV)
         plot(abun_tot$Sol)

   # Checking assumptions

      # Agri-environmental scheme identity has no effect on total avian abundance

            # Importing dataset of AES abundances where RSS, RP and FBL (the three schemes) are a
             categorical variable

            abun2scheme <- read.csv2("C:/Users/user/Desktop/DISSERTATION DREAMS/Core csv
             files/abun2scheme.csv")

            abun_scheme <- MCMCglmm(abundance ~ log(area) + duration + yearcenter + latcenter +
             Long + Scheme, random=~farm,  data=abun2scheme, nitt=100000, family="poisson")
                          summary(abun_scheme)          # No significant effect of scheme identity

      # Number of visits has no effect

            abuntot$visits_f <- as.factor(abuntot$visits)

            abun_tot3 <- MCMCglmm(abundance ~ treatment + log(area) + duration + yearcenter +
             visits_f + latcenter + Long, random=~farm, data=abuntot, nitt=100000, family="poisson")
                          summary(abun_tot3)               # No significant effect of number of visits

     # These assumptions were tested in all models presented here
```

Running the MCMC mixed effects model predicting individual species' abundance

```
abun_sp <- MCMCglmm(abundance ~ treatment + log(area) + yearcenter + species + treatment:species
+ duration + latcenter + Long, random=~farm, pr=TRUE, nitt=100000, data=spintAC, family="poisson")
summary(abun_sp)
```

Plotting predicted values for each species across treatments - values calculated by hand

```
# Importing the predicted values data

    predictSP <- read.csv2("C:/Users/user/Desktop/DISSERTATION DREAMS/Core csv
    files/predictSP.csv")

    pd <- position_dodge(0.2)          # So that the error bars on graphs don't overlap

    pred_plot <- ggplot(predictSP, aes(x=species, y=mean, colour=treatment, group=treatment))+
    geom_errorbar(aes(ymin=down, ymax=up), colour="black", width=.2, position=pd) +
    geom_point(position=pd,size=4) + theme_classic() + labs(x="species", y="Avian abundance") +
    theme(axis.text.x = element_text(size=12), axis.text.y = element_text(size=12), axis.title.x =
    element_text(size=14), axis.title.y = element_text(size=14))
    pred_plot
```

Examining the relationship between farm area and abundance

```
# Using stat_smooth(method=glm), because data are not normally distributed
# Fitting a generalised linear regression with 95% confidence intervals
```

For total abundance

```
area_abun_tot <- ggplot(abundance, aes(x=log(area), y=abundance, colour=treatment)) +
 geom_point(size=2.5)

area_abun_tot <- area_abun_tot + labs(x="log (Farm area (ha))", y="Avian abundance") +
geom_point() + stat_smooth(method=glm) + theme_classic() + theme(axis.text.x =
element_text(size=12), axis.text.y = element_text(size=12),  axis.title.x = element_text(size=14),
axis.title.y = element_text(size=14))
area_abun_tot
```

For individual species, procedure was repeated for each species

```
A1 <- ggplot(CB, aes(x=log(area), y=abundance, colour=treatment)) + geom_point(size=0.5)
A1 <- A1+ labs(x="log (Farm area (ha))", y="Abundance", title="Corn bunting") + geom_point() +
stat_smooth(method=glm) + theme_classic() + theme(axis.text.x = element_text(size=10),
axis.text.y = element_text(size=10), axis.title.x = element_text(size=12), axis.title.y =
element_text(size=12)) + theme(legend.position="none")
A1
```

 # Investigating the relationship between avian abundance and treatment duration to determine if I should include a quadratic term

```
abun_dur <- ggplot(abundance, aes(x=duration, y=abundance)) + geom_point(size=2.5)
abun_dur <- abun_dur + labs(x="Treatment duration (years)", y="Avian abundance")
abun_dur <- abun_dur + stat_smooth(method=glm) + theme_classic() + theme(axis.text.x =
element_text(size=12), axis.text.y = element_text(size=12),  axis.title.x = element_text(size=14),
axis.title.y = element_text(size=14))
abun_dur       # Relationship has a very shallow slope, no quadratic term needed
```

 # Investigating the latitudinal pattern in avian abundance (since there was a significant latitudinal effect)

```
abun_lat <- ggplot(abundance, aes(x=Lat, y=abundance)) + geom_point(size=2.5)
abun_lat <- abun_lat + labs(x="Latitude", y="Avian abundance") + stat_smooth(method=glm) +
theme_classic() + theme(axis.text.x = element_text(size=12), axis.text.y = element_text(size=12),
axis.title.x = element_text(size=14), axis.title.y = element_text(size=14))
abun_lat
```

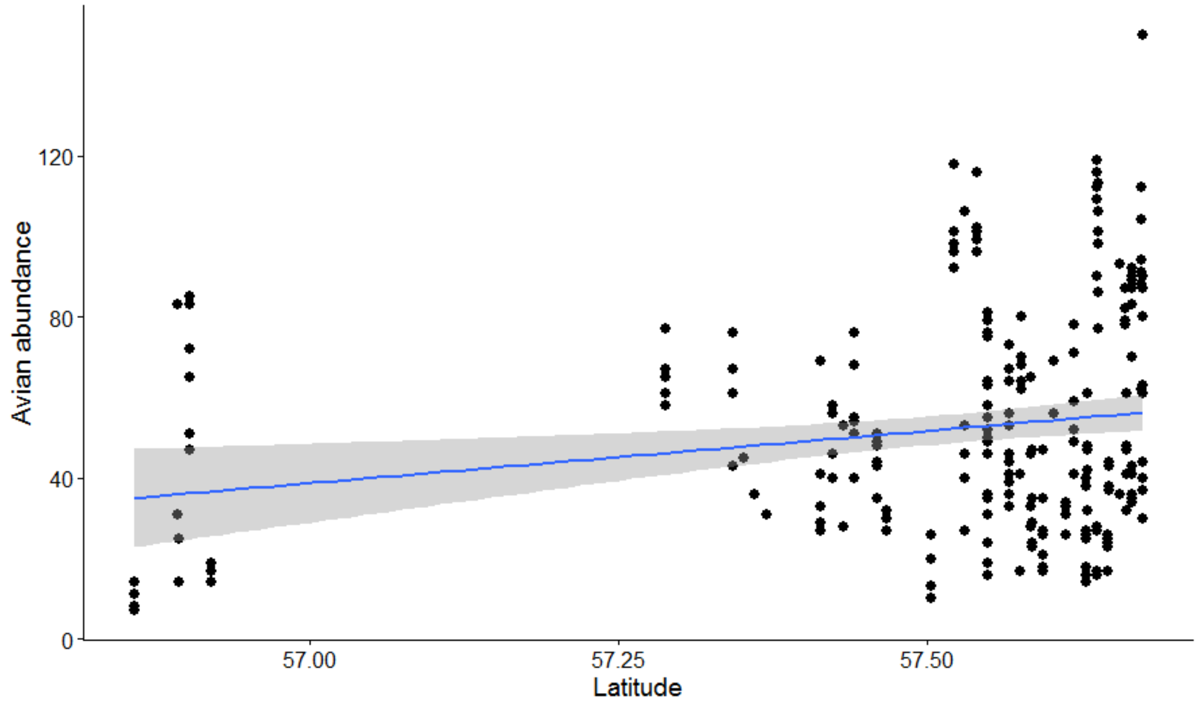
 # Testing for overfitting by plotting observed vs predicted values

```
# Extracting predicted values

# For total abundance model

predtot <- predict(abun_tot, type="response", marginal=NULL, interval="confidence")
write.csv(predsp, file="predtot.csv")

# In excel add observed values to the csv file with the predicted values
# Import it back into R (for future analyses this could all be done in R)

fit_abun <- qplot(observed, predicted, data = overfit2) + geom_smooth(colour = "red") +
theme_classic() + labs (x="Observed values", y="Predicted values") + theme(axis.text.x =
element_text(size=12), axis.text.y = element_text(size=12),  axis.title.x = element_text(size=14),
axis.title.y = element_text(size=14))
fit_abun <- fit_abun + geom_abline(intercept = 0, slope = 1, size=1.2, colour="blue")
fit_abun

# Same procedure was repeated to test the individual species' abundance model for overfitting
```

```
# Offsets are often used when evaluating the effect of AES on bird populations
# Modelling abundance with area as an offset effectively models density
# Offsets assume a 1:1 relationship between area and abundance, which is violated here (see area-abundance graphs)
# I won't use offsets, but will still investigate the AES effect on density to determine if using density gives different results compared to using abundance with farm area as a fixed effect

    # Running a total breeding bird density model with a Gaussian distribution

    den_tot <- MCMCglmm(density ~ treatment + duration + yearcenter + latcenter + Long,
    random=~farm, data=densityAC, nitt=100000, pr=TRUE, family="gaussian")
        summary(den_tot)       # No AES effect, similar to total abundance model

    # Running an individual species' density model with a Gaussian distribution

    den_sp <- MCMCglmm(density ~ treatment + species + treatment:species + duration + yearcenter
    + latcenter + Long, random=~farm, data=denACsp, nitt=100000, pr=TRUE, family="gaussian")
    summary(den_sp)     # No overall effect of AES, negative interaction between AES and skylark,
    similar to individual species' abundance model
```

```
# Making a map of study sites

   # Importing farm location data

   map2 <- read.csv2("C:/Users/user/Desktop/DISSERTATION DREAMS/CSV files/map2.csv")

   coordinates(map2) <- c("Long", "Lat")  # set spatial coordinates
   crs.geo <- CRS("+proj=longlat +ellps=WGS84 +datum=WGS84")  # geographical, datum WGS84
   proj4string(map2) <- crs.geo  # define projection system of data
   map.wgs84 <- spTransform(map2, CRS("+init=epsg:4326"))

   b <- bbox(map.wgs84)
   b[1, ] <- (b[1, ] - mean(b[1, ])) * 1.05 + mean(b[1, ])
   b[2, ] <- (b[2, ] - mean(b[2, ])) * 1.05 + mean(b[2, ])

   map.wgs84.f <- fortify(map.wgs84, region = "ons_label")
   map.wgs84.f <- merge(map.wgs84.f, map.wgs84, by.x = "id", by.y = "ons_label")

   farm_map <- ggmap(get_map(location = b, source="stamen", maptype = "toner-lite",
   crop = T, zoom = 9)) + geom_point(data=map2, aes(x=Long, y=Lat, colour=treatment, size=density))

   farm_map <- farm_map + labs(x="Longtitude", y="Latitude")  + theme(axis.text.x =
   element_text(size=12), axis.text.y = element_text(size=12), axis.title.x = element_text(size=14),
   axis.title.y = element_text(size=14))
   farm_map

  # All of the map code above has to be run together, then the farm coordinate data has to be imported
   again, so that it is in the right class.

   map2 <- read.csv2("C:/Users/user/Desktop/DISSERTATION DREAMS/CSV files/map2.csv")

  # Now running only the farm_map ggmap code again creates the map
```

Appendix 6. Latitudinal pattern in avian abundance. Data points represent all total abundance records (raw data) and line represents a generalised linear regression fit with 95% confidence intervals. There was a significant latitudinal trend, with abundance increasing towards the north (p=0.05, n=230, effect size=0.45 (CI 0.01 – 0.90). In Scotland, agricultural intensity decreases northwards, thus making it possible that the higher avian abundance at higher latitudes is due to less intensive farming practices. However, because of the small latitudinal range in the data investigated here, this result is to be interpreted with caution. A large-scale latitudinal study (e.g. comparing England and Scotland) can provide better insight into the relationship between avian abundance and latitude.



Appendix 7. Model parameters of a mixed effects model, predicting total avian abundance (the summed up breeding abundances of six target species). Farm was the random term in the model, with the rest of the variables included as fixed effects. Intercept refers to the first variable in each category, following an alphabetical order – here AES (agri-environmental scheme treatment). Significant effects are presented in bold.

	Posterior mean	Lower 95% CI	Upper 95% CI	Effective sample size	pMCMC
Intercept	1.01	-0.06	2.14	10731	0.07
treatmentcontrol	0.02	-0.09	0.13	8281	0.77
log(area)	0.63	0.47	0.81	9700	**<0.001**
Duration	0.02	0	0.04	9271	0.12
Yearcenter	0	-0.02	0.02	9011	0.79
Latcenter	0.45	0.01	0.90	9512	**0.05**
Long	0.10	-0.21	0.44	9290	0.53
Farm (among farm variation)	0.12	0.07	0.18	8406	-
Units (within farm variation)	0.03	0.02	0.04	5854	-

Appendix 8. Model parameters of a mixed effects model, predicting individual species abundances across treatment types. Farm was the random term in the model, with the rest of the variables included as fixed effects. Intercept refers to the first variable in each category, following an alphabetical order – here AES (agri-environmental scheme treatment) and CB (corn bunting). Significant effects are presented in bold.

	Posterior mean	Lower 95% CI	Upper 95% CI	Effective sample size	pMCMC
Intercept	-1.60	-2.78	-0.48	9700	**0.01**
treatmentcontrol	-0.14	-0.41	0.13	8065	0.32
log(area)	0.57	0.41	0.77	9700	**<0.01**
Duration	0.01	-0.02	0.04	9032	0.59
Yearcenter	0.01	-0.01	0.04	8858	0.39
Latcenter	0.53	0.07	0.99	9700	**0.02**
Long	0.11	-0.22	0.43	9700	0.49
speciesLI	0.72	0.54	0.91	8254	**<0.01**
speciesRB	0.03	-0.16	0.22	8357	0.72
speciesS	1.80	1.62	1.97	8154	**<0.01**
speciesTS	0.12	-0.07	0.31	8128	0.21
speciesY	1.47	1.29	1.64	8081	**<0.01**
treatmentcontrol:speciesLI	0.32	0	0.64	8143	**0.05**
treatmentcontrol:speciesRB	0.14	-0.18	0.50	7830	0.41
treatmentcontrol:speciesS	0.31	0.01	0.62	8389	**0.05**
treatmentcontrol:speciesTS	-0.05	-0.39	0.31	7655	0.78
treatmentcontrol:speciesY	-0.13	-0.44	0.19	8340	0.43
Farm (among farm variation)	0.11	0.06	0.17	8658	-
Units (within farm variation)	0.38	0.33	0.43	5458	-

Appendix 9(a). Results of examining the effect of AES on total breeding bird density (number of breeding pairs/territorial males of six target species per km^2). Similarly to the abundance models presented above, there was no significant difference in total bird density between AES and control farms (p=0.59, b=1.35 (CI -3.49 – 6.16), n=230). Here, however, among and within farm variation are considerably higher than in the total abundance model (Appendix 7).

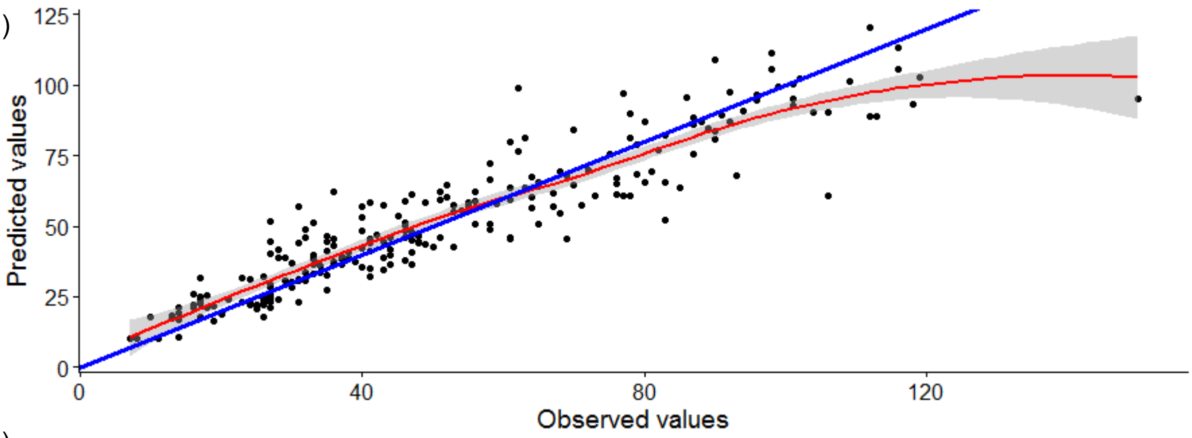
	Posterior mean	Lower 95% CI	Upper 95% CI	Effective sample size	pMCMC
Intercept	62.93	22.98	103.01	9700	**<0.01**
treatmentcontrol	1.35	-3.49	6.16	9700	0.59
Duration	0.52	-0.48	1.48	9700	0.29
Yearcenter	0.06	-0.79	0.85	9700	0.9
Latcenter	16.88	-7.66	38.43	9700	0.15
Long	8.26	-8.8	25.70	9700	0.33
Farm (among farm variation)	344.3	203.2	502.7	9700	-
Units (within farm variation)	110.6	88.58	135.5	9162	-

Appendix 9(b). Results of examining the effect of AES on individual species' density. Density was measured in number of breeding pairs per km^2 for LI and TS and in number of territorial males per km^2 for CB, RB, S and Y. Similarly to the model that compared abundance among species, individual species' density did not vary significantly across treatments (p=0.40, b=-0.81 (CI -2.75 – 1.03), n=1380). Skylark was the only species which had a significant interaction with treatment type (p<0.01, b=4.62, (CI 2.20 – 6.99), n=230), with its density predicted to be higher on control farms.

	Posterior mean	Lower 95% CI	Upper 95% CI	Effective sample size	pMCMC
Intercept	6.69	-0.06	13.47	9700	**0.05**
treatmentcontrol	-0.81	-2.75	1.03	9700	0.40
Duration	0.03	-0.20	0.28	9700	0.78
Yearcenter	0.03	-0.17	0.24	9700	0.22
Latcenter	2.35	-1.58	6.14	8952	0.22
Long	1,27	-1.62	4.06	9700	0.36
speciesLI	2.14	0.73	3.48	9700	**<0.01**
speciesRB	-1.02	-2.42	0.34	9802	0.15
speciesS	13.92	12.50	15.3	9700	**<0.01**
speciesTS	-0.73	-2.1	0.6	10246	0.3
speciesY	8.8	7.4	10.18	9700	**<0.01**
treatmentcontrol:speciesLI	2.06	-0.33	4.45	9700	0.09
treatmentcontrol:speciesRB	1.12	-1.23	3.56	10247	0.36
treatmentcontrol:speciesS	4.62	2.20	6.99	9700	**<0.01**
treatmentcontrol:speciesTS	0.42	-1.95	2.79	9700	0.73
treatmentcontrol:speciesY	-2.21	-4.62	0.15	9700	0.07
Farm (among farm variation)	8.37	4.79	12.9	9700	-
Units (within farm variation)	38.24	35.31	41.15	9853	-

Appendix 10. Comparison of predicted and observed values for (a) total avian abundance and (b) individual abundances of each species. Blue line shows a 1:1 relationship, representing a perfect model fit, and red line represents the observed model fit with 95% confidence intervals. Total avian abundance model (n=230) shows a good fit, with slight underprediction at high abundances. Individual species' abundance model (n=1380) performs well at low to medium abundance values, but is underpredicting high avian abundances.

(a)



(b)