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## Executive Summary

7. The executive summary must not exceed 2 sides in total of A4 and should be understandable to the intelligent non-scientist. It should cover the main objectives, methods and findings of the research, together with any other significant events and options for new work.

1. One of the key objectives of agri-environment (AE) schemes is the maintenance and enhancement of biodiversity, and in England there has been particular emphasis on the recovery of declining farmland bird populations. Most previous studies of the impacts of AE schemes on farmland birds have compared densities, or changes in densities, between areas with and without AE measures but have not allowed for concomitant changes in cropping or land use in the wider countryside. In this study, we tested for impacts of AE provision on four red-listed farmland bird species, after allowing for associated changes in land use in the wider landscape.

2. Data on bird population changes between 2003 and 2009 were available from 315 tetrads in South Devon. The study area encapsulates the current breeding range of the ciril bunting in Britain, which has been the subject of targeted conservation action and associated research and monitoring for the last 20 years. The bird data from this area were combined with detailed information on the type and extent of AE provision (extracted from Natural England databases), and the composition of the wider farmed landscape (manually mapped at the field scale in a subset of 56 tetrads during both years). Major changes in land use during this study period included the loss of set-aside and the introduction of the Environmental Stewardship Scheme which includes both the Entry Level Scheme (ELS) and the Higher Level Scheme (HLS). Novel statistical methods were used to identify correlates of changes in bird abundance (at the 1-km square scale), distinguishing between the effects of initial habitat / AE option area, and changes in habitat / AE option area. We also tested for AE option correlates of bird population changes across Countryside Stewardship Scheme (CSS) agreements.

3. The main changes in land use between 2003 and 2009 were a 70% loss of fallow land (reflecting the loss of set-aside in the UK), a 15% reduction in the area of cereal production and an increase in the area of intensively managed grassland. The total extent of AE provision in South Devon increased substantially during this period, with large increases in the area of grassland and length of hedgerow (predominantly under ELS) and wild bird seed mixtures (entirely under HLS). Provision of higher-level winter stubbles also increased with some of the CSS 'special barley ciril bunting option' being replaced by HLS stubbles.

4. Across the entire study area, ciril bunting abundance increased (by 16%) while the abundance of yellowhammers, skylarks and song thrushes all decreased (by 18%, 9% and 15% respectively) between 2003 and 2009. For three of the four species (song thrush being the exception), population changes did not differ according to changes in higher-level AE status (i.e. retained, remained absent, gained or lost). Song thrushes declined most in localities where higher-level AE provision was introduced after 2003.

5. Population changes of ciril buntings and yellowhammers were significantly more positive on land that entered ELS agreements between 2003 and 2009, compared to land that did not. These differences were large (ELS vs. non-ELS: +48% vs. 0% for ciril bunting, -5% vs. -24% for yellowhammer).

6. The fine-scale distributions of breeding birds were positively associated with semi-improved grassland and hedgerows (all four species) and cereals (except song thrush), while song thrushes selected

woodland and gardens. Cirl buntings and song thrushes exhibited a significant preference for semi-improved grassland relative to intensive grassland, while skylarks preferred intensive grassland (probably reflecting high nesting densities in grass silage). There was weak positive selection of higher-level arable grass margins (by cirl bunting, yellowhammer & skylark) and higher-level grassland, while the distribution of skylarks was positively associated with the CSS special barley option (2009 only). All four species preferred non-AE hedges to hedges under higher-level management, which may reflect a short-term reduction in hedgerow suitability following the introduction of restorative management. The strong associations between the distributions of bird territories and semi-improved grassland and hedges, suggests that these habitats may be regionally important in maintaining farmland bird populations.

7. Wider countryside habitats associated with spatial variation in bird density included woodland (negative for buntings & skylark, positive for song thrush), cereals (positive for yellowhammer & skylark, negative for song thrush), intensive grassland (negative for cirl bunting & song thrush; positive for yellowhammer) and semi-improved grassland (positive for song thrush). Densities of cirl bunting and skylark peaked at intermediate ratios of arable-grassland farmland suggesting beneficial effects of a mixed landscape.

8. Skylark was the only species for which there was evidence of association with fallow habitat (mainly set-aside). Skylark densities in 2003, and subsequent reductions in skylark density between 2003 and 2009, were positively related to fallow area in 2003. This provides strong evidence for a negative impact of set-aside loss on the regional abundance of skylarks.

9. Changes in the numbers of cirl buntings and yellowhammers were positively related to changes in the area of summer AE options (mainly ELS grassland). Grazed grassland provides a potentially important source of invertebrate prey for buntings during the breeding season, but further research is needed to understand how ELS grassland management might be benefitting these species.

10. Cirl bunting density (though not changes in density) was positively related to the area of CSS special barley option. This could reflect impacts of the option on cirl bunting abundance that pre-date this study, or efficient local targeting of this option. The only correlate of change in bird abundance at the CSS agreement scale was a positive influence of CSS special barley in the surrounding landscape on change in cirl bunting count, which implies this option may be acting primarily as a winter habitat for this species.

11. A weak negative relationship between skylark density and CSS stubbles in the surrounding landscape in 2003, changed to a strong positive relationship in 2009, implying that population changes were more positive in localities with relatively large areas of CSS winter stubble. The transition to positive associations between the CSS special barley option and skylark abundance at both the fine-scale (see 6 above) and surrounding landscape scale by 2009 is consistent with an increased reliance on this option (both as a wintering and a breeding habitat) following the loss of set-aside.

12. Despite a large increase in the provision of wild bird seed mixtures between 2003 and 2009, only negative associations with (changes in) bird abundance were detected (involving skylark, yellowhammer and song thrush).

13. Regression models were used to predict the amount of AE provision needed to enhance farmland bird abundance. The amount of habitat (per km<sup>2</sup>) needed to increase bird abundance by 20% (assuming no initial provision) was approximately 3-4ha of ELS grassland (cirl bunting & yellowhammer) and at least 3ha of CSS special barley (skylark).

14. Causal factors contributing to the declines of three of the four species in this study are likely to include the loss of set-aside (there was evidence for this in the case of skylark) and other favoured habitats (such as cereals in the case of yellowhammer). Widespread provision of CSS special barley may have buffered cirl buntings from any negative impacts of set-aside loss and probably limited any impacts on skylarks. The large decline in yellowhammer abundance may have been caused by a lack of seed-rich habitat during winter although local losses were not correlated with loss of fallow.

15. Further research should aim to understand the positive population changes of cirl buntings and yellowhammers on grassland recently entered into ELS management. Analyses of the longer-term cirl bunting distribution and AE provision data (available for 1992, 1998, 2003 and 2009) will have greater power to detect and quantify relationships between AE provision and cirl bunting population responses. The current utility of AE options for soil-invertebrate feeding birds like song thrushes also merits investigation.

## Project Report to Defra

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8. As a guide this report should be no longer than 20 sides of A4. This report is to provide Defra with details of the outputs of the research project for internal purposes; to meet the terms of the contract; and to allow Defra to publish details of the outputs to meet Environmental Information Regulation or Freedom of Information obligations. This short report to Defra does not preclude contractors from also seeking to publish a full, formal scientific report/paper in an appropriate scientific or other journal/publication. Indeed, Defra actively encourages such publications as part of the contract terms. The report to Defra should include:
- the scientific objectives as set out in the contract;
  - the extent to which the objectives set out in the contract have been met;
  - details of methods used and the results obtained, including statistical analysis (if appropriate);
  - a discussion of the results and their reliability;
  - the main implications of the findings;
  - possible future work; and
  - any action resulting from the research (e.g. IP, Knowledge Transfer).

### 1. Objectives.

The objectives of this study were to answer the following questions using available spatial data on birds distributions, habitat and agri-environment (AE) provision within the South Devon range of the cirl bunting:

1. Do population changes of four red-listed farmland bird species differ between land managed under agri-environment management and adjacent farmland out with agri-environment agreements?
2. Which habitat features in the landscape have most influence on the fine-scale breeding distributions of farmland birds?
3. Which habitat features (including agri-environment options) have had the greatest influence on bird population changes over the period 2003-2009?
4. Which options within CSS/HLS have had most impact on farmland bird numbers within agreement areas?

### 2. Methods.

#### 2.1. Data collection and processing

Breeding cirl buntings *Emberiza cirlus* and several other priority farmland birds have been surveyed in 2003 (281 tetrads) and again in 2009 (315 tetrads including all those surveyed in 2003). In 2003 and 2009, all tetrads previously occupied by cirl buntings were surveyed along with all tetrads adjacent to previously occupied tetrads. The aim of this intensive sampling was to identify any range expansion of the cirl bunting. Each tetrad was visited twice during the breeding season, once during the period mid-April to the end of May, and once between early June and late August. Tetrads were surveyed by walking all public rights of way including all roads and footpaths, and all public open spaces. Farmland, semi-natural habitats, small woodlands and rural villages were surveyed but larger woodland blocks and towns were not. All singing and territorial behaviour was noted on maps (following Marchant et al., 1990), and a CBC-style mapping procedure was used to determine territory distribution from the two sets of survey maps (following Wotton et al. 2004). Here we only consider data for four farmland bird species that were sufficiently abundant across the study area and whose survey locations were considered likely to have reflected their breeding territories: cirl bunting, yellowhammer *E. citrinella*, skylark *Alauda arvensis* and song thrush *Turdus philomelos*. All four species are red-listed in the UK as a consequence of population declines exceeding 50% over 25 years (Eaton et al, 2009).

During July-August 2003 and 2009, detailed crop and habitat mapping was conducted within a random sample of 56 tetrads. Within each tetrad, all habitats were mapped including woodland and urban areas. The sample of 56 tetrads was stratified to include 35 tetrads that have been continuously occupied by cirl buntings since 1992, and a further 21 tetrads that were colonised by cirl buntings between 1992 and 2003. Habitat recording broadly followed NCC Phase 1 survey methods with some refinements to provide more detail on farmland habitats (Table 2.1). Crop types were identified by eye using binoculars where necessary. Public Rights of Way and open space were used to conduct bird surveys and habitat mapping. Occasionally fields were too distant to identify crop type, and such fields were excluded from all analyses. In 2009, higher resolution base maps were used which allowed

higher resolution and more accurate recording of habitat categories. For example, in 2003 private properties with gardens were often recorded entirely as garden habitat in 2003, while in 2009 such properties were sub-divided into garden and buildings (=unsuitable).

Particular attention was paid to separating improved and semi-improved grass fields. Improved grasslands were those meadows and pastures subjected to heavy grazing, drainage and/or the application of inorganic fertilisers and /or slurry. The following criteria were used in the field to designate grass fields as 'improved': a bright green, lush and even sward dominated by grasses; a low diversity of broad-leaved forbs (typically no more than 3-5 species) common within the sward; more than 50% combined cover of *Lolium perenne*, *Trifolium repens* and other agricultural grasses. 'Semi-improved' grassland is intended to include grass fields modified by drainage, artificial fertilisers and slurry, and intensive grazing. The following criteria were used to designate 'semi-improved' grass fields (hereafter referred to as 'extensive grass'): dull green or brown vegetation, uneven sward containing a mix of grasses including species such as sweet vernal grass *Anthoxanthum odoratum* and crested dogstail *Cynosurus cristatus*; at least five species of broad-leaved forbs were typically common within the sward; *Lolium spp.*, *Trifolium repens* and other agricultural grasses were present but not dominant (i.e. combined cover <50%).

Table 2.1. Habitats recorded during fieldwork in 2003 and 2009, and habitat combinations used in analyses.

Analysis categories	Habitats recorded
Cereals	Winter wheat, winter oats, winter barley, winter cereals (not identified), spring barley, spring barley with undersown peas, spring wheat, spring oats, spring cereal (not identified)
Other crops	Brassicacae, horticulture, linseed, lupins, maize, Miscanthus, oilseed rape, other cereal crop, ploughed field, potato, root crops
Fallow	Fallow, set-aside, stubble
Intensive grass	Improved grazed, improved ungrazed, amenity grass, orchards on grass, newly sown tree plantation
Extensive grass	Semi-improved grazed, semi-improved ungrazed, semi-improved diverse grazed, semi-improved diverse ungrazed, orchard with unimproved grass, horse paddocks
Scrub	Scrub, coastal scrub/grass/heath mosaic
Woodland	Woodland
Gardens	Rural gardens attached to lone houses, small rural villages with gardens, farm buildings & yards
Unsuitable – excluded	Urban/suburban areas, reedbed & marshland, quarry, sewage works, roads, car parks, other developments, open freshwater, sea

Bird distribution (territory centres, usually singing birds recorded as an 8-figure grid reference i.e. to within 10m) and habitat data were digitized using the MAPINFO GIS software (MapInfo Professional version 7.8, ©MapInfo Corporation, 1985-2000), on to an OS 1:10000 map layer (RSPB licence 100021787). Habitat boundaries usually followed the mapped field boundaries except for some patches of scrub and woodland, and a few split fields. Hedge habitats were digitized as linear features on a separate GIS layer. Hedge areas were estimated by assuming a fixed width of 2m for all hedge types (AE and non-AE hedges).

The extent of AE provision was calculated for each sample square using digitized information summarising agreement details for the CSS in 2003 (provided by Defra) and for the CSS, HLS and ELS in 2009 (provided by Natural England, Aesis database). Each in-field AE option was matched to individual fields on the digitized habitat mapping. The AE option data essentially 'over-wrote' the wider habitat / cropping data relating to the same fields, and the combined AE and wider habitat data were summed for each sample square and exported to Microsoft Excel for analysis. Appendix 1 shows the distribution across South Devon of bird and habitat survey tetrads, along with higher-level and ELS agreements areas.

The compositional analysis (see 2.2.2 below) was conducted at the tetrad scale (n=56), and involved circular buffers being centred on each territory centre to simulate a bird foraging range (i.e. used habitat). For cirl bunting, analyses were conducted for two buffer sizes: 100m and 250m radii around the territory centre. The aim of the 100m buffer was to provide information on habitat selection associated with nest sites and foraging habitats close

to nests, while the 250m buffer provided information on habitat selection at a wider foraging home range (Stevens *et al.* 2002). Foraging home ranges for the other species were based on observations from previous field studies and were defined as a radius of 250m for skylark (Wilson 2001), 100m for song thrush (Peach *et al.* 2004) and 250m for yellowhammer (Morris *et al.* 2001). In many cases, the close proximity of adjacent territories meant that foraging home ranges overlapped. Because all the species considered here defend foraging home ranges, such overlapping radii were merged into a single measure of 'used' habitat for each tetrad in order to avoid the 'double counting' of such areas. MapInfo queries allowed the summation of habitat / AE option areas across buffers within tetrads (=used habitat) and for entire tetrads (=available).

## 2.2. Data analysis

### 2.2.1. Effects of agri-environment status on bird population changes.

We tested whether changes in bird density between 2003 and 2009 differed between the following four AE status categories of land management:

- A: land under higher level agri-environment agreement (CSS or HLS) during 2003 and 2009 (=RETAINED),
- B: land not under higher level land-management during 2003 or 2009 (=ABSENT),
- C: land under CSS management in 2002, but no longer under any form of higher level management by 2008 (=LOST),
- D: land not under CSS management in 2002 but under higher level agreement by 2008 (=GAINED).

Note categories B and C included land that entered ELS management between 2003 and 2009. For each tetrad, land and associated bird distributional data were allocated to one of these four higher level AE status categories. Higher level AE status was defined using agreement level (whole farm) boundaries which will have included a mixture of fields and boundaries under and not under AE management. To allow for any influence of bird territories around the edge of AE agreements, we imposed a 50m buffer around each agreement boundary. Buffers for habitats C and D were removed from the analysis when they overlapped with habitat A buffers. Results from analyses including buffers (presented here) were similar to those where buffers were not included around agreement boundaries. For each species, unoccupied tetrads were excluded from the analysis, and therefore densities relate to occupied tetrads only. Bird counts were then analysed using a generalised linear mixed model (GLMM) in which tetrad was declared as a random factor, year and habitat category (and interaction) as fixed factors, and the natural logarithm of the habitat area as an offset, which effectively converts counts to densities. A significant interaction term would indicate differences in density changes between the four habitat categories.

In a similar set of analyses, we attempted to identify any impact of the introduction of ELS management on bird abundance. To do this we excluded any land under higher-level agreement in 2003 or 2009, and compared changes in bird counts on land that entered the ELS between 2003 and 2009, with changes on land that did not enter the ELS. A 50m buffer was imposed around each ELS agreement. This amounted to a sub-division of category B above, plus the land that was previously allocated to 50m buffers around higher-level agreements (see above).

### 2.2.2. Fine-scale associations between farmland bird breeding distribution and habitat features

Compositional Analysis (Aebischer *et al.* 1993) was used to assess fine-scale habitat associations of territory centres. This technique compares the proportional usage of habitats (buffers are assumed to include all used habitats) in relation to their availability within some defined study area. Analyses were conducted at the tetrad scale (n=56), the habitat within each square being considered 'available' and the sum of the habitat within all buffers being considered as 'used'. Randomisation tests were used to assess whether habitats were used non-randomly (Aebischer *et al.* 1993).

### 2.2.3. Effects of habitat and agri-environment provision on bird population changes

We used changes in bird counts at the 1km square scale to identify correlates of population change related to habitat extent in 2003 and changes in habitat extent between 2003 and 2009. These analyses were restricted to 196 1-km squares in the 56 tetrads for which we had data quantifying both wider habitat extent and the extent of agri-environment provision. Our approach was to screen potential predictors of changes in bird counts in four successive stages. First, we considered abundance of the same species in the surrounding eight 1km squares in 2003 and distance to the coast (high tide line) as two non-habitat related factors that might have influenced changes in bird counts. Second, we tested whether bird counts were related to the extent, or changes in extent, of nine major habitat types (woodland to fallow, Table 3.1) and the proportion of the farmed habitat that was arable (where 'Mixed'= arable / (arable+grass)). Third, after allowing for these potential influences of wider countryside habitat, we considered whether bird counts, and changes in bird counts, were related to the extent of agri-environment provision in 2003, or to changes in the extent of that provision between 2003 and 2009. Agri-

environment measures were combined into fourteen categories providing similar ecological resources for farmland birds (HL hedge to AE winter; Table 3.1). This combining of AE options attempted to distinguish between measures providing nesting habitat (e.g. hedgerows), invertebrate-rich foraging habitat (e.g. low-input grassland, arable field margins) and seed-rich foraging habitat (e.g. low input cereal stubbles, wild bird seed mixes). Fourthly, we considered possible effects of AE provision of winter food (wild bird seed mixes and cereal stubbles) in the surrounding landscape on breeding season bird counts within 1km squares. The bird species considered here are known to be relatively mobile during winter and populations are therefore potentially sensitive to habitat provision on a wider scale. The analysis of surrounding winter habitat was considered two spatial scales: the total amount of HL plus ELS food-rich habitat in the surrounding eight 1km squares (i.e. within a 3x3 block) and in the surrounding twenty-four 1 km squares (i.e. within a 5x5 block). Two habitat categories were considered: winter stubbles (including CSS options OS1, OS2, OS3, special barley; HLS options HF6 & HF15; ELS options EF6, EF15) and wild bird seed mixes (including CSS option WM1, HLS options HF2, HF3 & HF12 and ELS option EF2).

The influence of each potential predictor (usually a habitat extent variable) of bird abundance was tested using the following GLMM:

$$\text{Log}(\text{count}_{ij} / \text{area}) = \text{tetrad}_k + \text{square}_i + \text{year}_j + \text{hab03}_i + \text{hab03}_i * \text{year}_j + \text{hab}_{ij}$$

where  $\text{count}_{ij}$  is the bird count in square  $i$  during year  $j$ , tetrad and square are random factors (square being nested within tetrad). Year is a fixed factor,  $\text{hab03}_i$  is the habitat area within the square in 2003,  $\text{hab}_{ij}$  is the habitat area in square  $i$  during year  $j$ . Total area of suitable habitat within the square is fitted as an offset term. The first habitat term ( $\text{hab03}_i$ ) tests for a linear relationship between bird density (in both years) and habitat area in 2003. The second term ( $\text{hab03}_i * \text{year}_j$ ) tests whether the slope of this relationship differs between 2003 and 2009 (i.e. whether changes in bird counts were correlated with habitat area in 2003), and the third term ( $\text{hab}_{ij}$ ) tests whether changes in bird counts between 2003 and 2009 were related to changes in habitat extent over the same period. In order to keep conclusions about change in habitat area conservative, habitat area in 2003 ( $\text{hab03}_i$ ) was always retained during model development, but the other habitat terms were dropped if non-significant.

Within each of the four model-building stages (described above), each of the potential predictors of abundance was screened, and the most significant retained in the GLMM. The remaining predictors were then screened again in the presence of the retained predictor, and this 'step-up' process was continued until no further predictors were significant.

#### 2.2.4. Correlates of bird population changes on individual CSS agreements

Analyses were restricted to 145 CSS agreements that were active in 2003 and still active in 2009. Some CSS agreements active in 2003 expired before 2009 and were replaced by HLS or ELS agreements, or there was no following agreement. The numbers of CSS agreements contributing to each analysis (i.e. at least 1 territory was counted in at least one year) were 51 (cirl bunting), 73 (yellowhammer), 81 (skylark) and 76 (song thrush). Ordinal logistic regression was used to identify correlates of bird population change, and was used in preference to Poisson regression as it overcomes the problem of zero counts during the first year (which was a common feature of these data). Bird population changes were categorised on three-five point ordinal scales which for cirl bunting were: 3=gained 6-8 pairs, 2=gained 2-4 pairs, 1=gained or lost 1 pair or no change, 0=lost 2-4 pairs. Five point scales were adopted for yellowhammer and skylark (large decline, small decline, no change, small increase, large increase), and a three-point scale for song thrush (small decline, no change, small increase). These ordinal categories of population change were the dependent variables for the ordinal logistic regression (following Peach et al. 2001).

The analysis recognized the following CSS options:

hedge management (total area 19.5ha, number of agreements providing the option 38),  
grassland management (2964.6ha, 140), arable summer options (see Table 3.1; 23.1ha, 24),  
arable winter options (wild bird seed mixes; 21.3, 14),  
scrub management (13.2ha, 8),  
special barley stubbles (560.8ha, 93)  
CSS stubbles (15.6ha, 3).

The latter two options were summed to produce a single CSS stubble category. The analysis proceeded in three stages. First, we tested for any influence of the number of pairs of the species in question within a buffer zone extending 1km around each agreement in 2003 (population changes may have been more positive in agreements that were surrounded by larger initial populations). Second, we screened each of the five CSS option area variables (above) for any influence on population changes. The most significant predictor was retained in the model and the remaining options screened for any additional influence. This step-up procedure continued until no

additional variables had any significant influence on change in count status. Thirdly, we tested for any additional influence of the availability of CSS stubbles and/or winter options in the eight (3km\*3km block) and twenty-four (5km\*5km block) 1km-squares surrounding the 1km square containing the CSS agreement (to test for any influence of winter habitat in the surrounding landscape). When conducting this third stage, we also tested for interactions between the area of winter option provided at the agreement and wider scales.

### 3. Results

#### 3.1. Changes in habitat composition and agri-environment provision between 2003 and 2009

Changes in habitat composition and AE provision in the 56 mapped tetrads is summarised in Table 3.1. Some of the measured changes in habitat area are probably artefacts of more accurate, higher resolution mapping in 2009. For example, buildings were often included in the 'garden' category of 2003 but were excluded as unsuitable habitat in 2009. The increased resolution in 2009 also partly accounts for the apparent increase in woodland area, although there was genuine planting of new woodland (deciduous and coniferous) and succession of scrub in several areas (correlation between change in scrub vs. change in woodland area:  $n=194$  1-km squares,  $r = -0.30$ ,  $P<0.0001$ ). The apparent loss of hedges largely reflects a transition into ELS agreements (change in hedge vs. change in AE hedge:  $r = -0.89$ ,  $P<0.0001$ ) rather than any substantive hedgerow destruction. The large apparent increase in intensive grassland is probably partly inconsistent habitat recording (possibly exacerbated by damper soil conditions in 2009 promoting lush grass growth), and partly genuine reflecting a switch from cereal production (change in cereal vs. change in intensive grassland:  $r = -0.32$ ,  $P<0.0001$ ) and other crops ( $r = -0.27$ ,  $P<0.0001$ ). June agricultural census data suggest a genuine increase in total area of agricultural grassland in Devon during this period (ca. 339,000 ha in 2000; 361,000ha in 2005; 381,000 in 2009). Most (75%) of the large apparent decrease in area of extensively-managed grassland is explained by entry into HLS and ELS (change in extensive grass vs. change in total AE grass:  $r = -0.68$ ,  $P<0.0001$ ; Table 3.1), the remainder probably (inconsistently) being recorded as intensively-managed grass. Average rates of fertilizer inputs onto grassland in England and Wales declined by 32% between 2003 and 2009 (Table B2.1 in Defra 2011) which suggests no general intensification of grassland management in South Devon.

The large (70%) reduction in fallow land (Table 3.1) reflects the introduction of a zero percentage rate of set-aside from 2007-08. The compulsory rate in England in 2003 was 10% although the actual rate was 15.6% (Langton, 2006). Our data suggest that fallow land accounted for 11.5% of total cereal plus fallow area in South Devon in 2003 (Table 3.1). By 2009, much of this former fallow land had probably gone into cereal production (change in fallow vs. change in cereal:  $r = -0.29$ ,  $P<0.0001$ ) or into intensive grassland (change in fallow vs. change in intensive grass:  $r = -0.17$ ,  $P<0.018$ ). The large reduction in fallow area provides a useful test of the prediction that loss of set-aside from the agricultural landscape could have serious negative consequences for farmland bird populations (Gillings et al. 2010).

Table 3.1 highlights some interesting changes in AE provision in South Devon during a period when CSS was replaced, in 2005, by the ELS and HLS. Generally there was an expansion of AE provision in South Devon with large increases in the extent of grassland under both higher level (an increase of 251 ha) and ELS (823ha) agreements (Table 3.1). Although there was a small reduction in the area the CSS special barley option (of ca. 24ha), this was more than offset by an increase in the area of CSS and HLS cereals with following stubbles (ca. 83ha in total) and ELS stubbles (ca. 25ha). There were also substantial increases in the area of wild bird seed mixes (HL arable winter) and arable field margins (HL arable summer, Table 3.1). The apparent reduction in higher level hedge management reflects a net shift from CSS to ELS management.

#### 3.2. Effects of agri-environment status on bird population changes

Between 2003 and 2009 across the entire study area, cirl bunting counts increased by 16% ( $P(\text{year})=0.0089$ ,  $n=137$  occupied tetrads), while counts of yellowhammer, skylark and song thrush all declined significantly (by 18%, 9% and 15% respectively;  $n=192$ , 216 & 223 tetrads respectively;  $P(\text{year})=0.0001$ , 0.0046 and 0.0001 respectively; Table 3.2). For three of the four species (song thrush being the exception) temporal changes in densities did not differ significantly between higher-tier AE-status categories (i.e. non-significant year\*AE-status terms; Table 3.2). The decline in song thrush density was greatest in areas where AE status was gained (Fig. 3.1d). Three of the four species (cirl bunting, yellowhammer & skylark) occurred at significantly lower density on land where higher-tier AE management was lacking in both years (Fig. 3.1) implying effective targeting of AE resources. Densities of cirl buntings were highest on land that lost AE status although the numbers of birds involved was modest (Table 3.2). Cirl bunting density increased significantly between 2003 and 2009 on land where higher-tier AE was retained or absent, but not on land where AE was lost or gained (Fig. 3.1a). Densities of yellowhammer declined significantly in all AE-status categories (Fig. 3.1b), while densities of skylark and song thrush declined most in areas higher-tier AE was gained (Figs. 3.1c & 3.1d).



Table 3.1. Changes in habitat composition and AE provision in 194 1-km squares surveyed in 2003 and 2009. The numbers of 1-km squares containing each habitat/option and total habitat area are listed. Areas of the main habitat categories (woodland to fallow) were estimated by direct observation in the field (after subtracting AE option areas), while areas of AE categories (HL hedge to AE winter) were extracted from NE databases.

Habitat	No. 1km squares 2003	Area 2003 (ha)	No. 1km squares 2009	Area 2009 (ha)	% change squares	% change area	Comments (see section 3.1)
Woodland	173	1086.6	185	1223.6	+6.9	+12.6	Part genuine, part resolution change
Scrub	174	561.6	181	529.8	+4.0	-5.7	Genuine succession to woodland
Garden	175	815.0	181	741.4	+3.4	-9.1	Over-recorded in 2003
Hedge	187	301.9	182	282.9	-2.7	-6.3	Reflects entry of non-AE hedges to ELS
Intensive grass	173	2121.2	179	2702.8	+3.4	+27.4	Part genuine, part inconsistent recording?
Extensive grass	192	5406.6	189	3966.8	-1.6	-26.6	Substantial transfer of grassland to ELS/HLS
Cereal	171	2748.0	158	2333.9	-7.6	-15.1	Genuine local switch to intensive grass & other crops
Other crop	125	880.0	131	921.3	+4.8	+4.7	Genuine increase
Fallow	91	358.8	39	108.1	-57.1	-69.9	Zero % set-aside rate from 2007/08
HL hedge (A)	85	18.3	25	12.4	-71.6	-32.2	Reflects transfer of CSS hedges to ELS
HL grass (B)	108	890.7	123	1158.8	+13.9	+30.1	Mainly P1 grass & arable reversion (both years), plus HK15-17 & H1 hay in 2009
HL arable summer (C)	81	98.2	82	99.1	+1.2	+0.9	Mainly 2m/6m arable margins (both years), plus cons. headlands, P&N mixes & fallow plots in 2009
HL arable winter (D)	10	2.7	16	50.8	+60.0	+1781.5	Wild bird seed mixes (mainly WM1, also HF2, HF12)
HL special barley (E)	42	183.3	42	159.5	0	-13.0	CSS Cirl bunting option
Total HL stubbles (E + CSS/HLS)	42	183.3	58	242.2	+38.1	+32.1	OS1/OS2 (CSS) plus HF15 (HLS)
HL summer (B+C)	121	988.9	136	1257.9	+12.4	+27.2	
HL winter (D+E+stubbles)	48	186.0	67	293.0	+39.6	+57.5	
EL hedge (F)	0	0	27	33.3	-	-	
EL grass (G)	0	0	100	823.0	-	-	Mainly low (EK2, 65%) & very low (EK3, 22%) input, some mixed stocking (EK5, 11%)
EL arable (H)	0	0	1	3.4	-	-	u/s spring cereals (EG1)
EL stubble (I)	0	0	4	25.1	-	-	EF6
AE hedge (A+F)	85	18.3	48	45.7	-43.5	+149.7	
AE summer (B+C+G+H)	121	988.9	168	2084.3	+38.8	+110.8	
AE winter (D+E+stubbles+I)	48	186.0	68	318.1	+41.7	+171.0	

The effect of the introduction of ELS management was considered by dividing land with no history of higher level AE management (i.e. category B in Table 3.2) into areas that subsequently entered, and did not enter, the ELS (Table 3.3). The area and bird count totals in Table 3.3 are higher than those in Table 3.2 (for category B) because they include the buffer areas added to all higher-level agreements in Table 3.2. For cirl bunting and yellowhammer, changes in densities were significantly more positive (cirl bunting) or less negative (yellowhammer) on land that entered ELS management between the two surveys (Table 3.3). Comparing land that subsequently entered the ELS with land that did not, cirl bunting densities were similar in 2003, but increased by 48% on land that entered ELS and remained unchanged on land that did not (Fig. 3.2a). Yellowhammer densities were higher in 2003, and remained unchanged in 2009 (-5%), on land that subsequently entered the ELS, but declined significantly (by 24%) on land that did not enter the ELS (Fig. 3.2b). Skylark densities were higher on land that subsequently entered the ELS but changes in counts did not differ between ELS and non-ELS status (Fig. 3.2c). Song thrush densities were initially higher, but subsequently declined, on land that did not enter the ELS (Fig. 3.2d).

Table 3.2. Aggregate raw counts and changes in observed densities of farmland birds across four higher-tier AE status categories (A = retained, B=absent, C=lost, D=gained). The significance of the Year\*A-E status terms indicates whether changes in density varied significantly between A-E status categories.

A-E status code	Area	Aggregate count 2003	Aggregate count 2009	Density 2003	Density 2009	Change in density (%)	Significance (P) of density change
Cirl bunting: P(Year*A-E status)=0.73							
A	11536	244	292	2.12	2.53	+19.3	*
B	28221	218	263	0.77	0.93	+20.8	*
C	1659	64	65	3.86	3.92	+1.6	ns
D	4174	56	55	1.34	1.32	-1.5	ns
Sum		582	675			+16.0	**
Yellowhammer: P(Year*A-E status)=0.41							
A	15138	558	457	3.69	3.02	-18.2	***
B	44927	833	719	1.85	1.60	-13.5	***
C	2178	84	41	3.86	1.88	-51.3	***
D	6717	193	154	2.87	2.29	-20.2	*
Sum		1668	1371			-17.8	***
Skylark: P(Year*A-E status)=0.47							
A	15768	670	587	4.25	3.72	-12.5	*
B	49220	1009	984	2.05	2.00	-2.4	ns
C	2410	80	68	3.32	2.82	-15.1	ns
D	7913	245	189	3.10	2.39	-22.9	**
Sum		2004	1828			-8.8	***
Song thrush: P(Year*A-E status)= 0.048							
A	14980	231	225	1.54	1.50	-2.6	ns
B	53445	913	782	1.71	1.46	-14.6	***
C	2452	45	44	1.84	1.79	-2.7	ns
D	8289	155	98	1.87	1.18	-36.9	***
Sum		1344	1149			-14.5	***

Figure 3.1 Changes in bird density across four AE status categories between 2003 (filled bars) & 2009 (speckled bars). Significance of year / AE-status / year\*AE-status terms are indicated after species names. Error bars are +/- 1 SE. Asterisks above bars indicate significant changes in density (\*\*\* P<0.005, \*\* P<0.01, \* P<0.05).

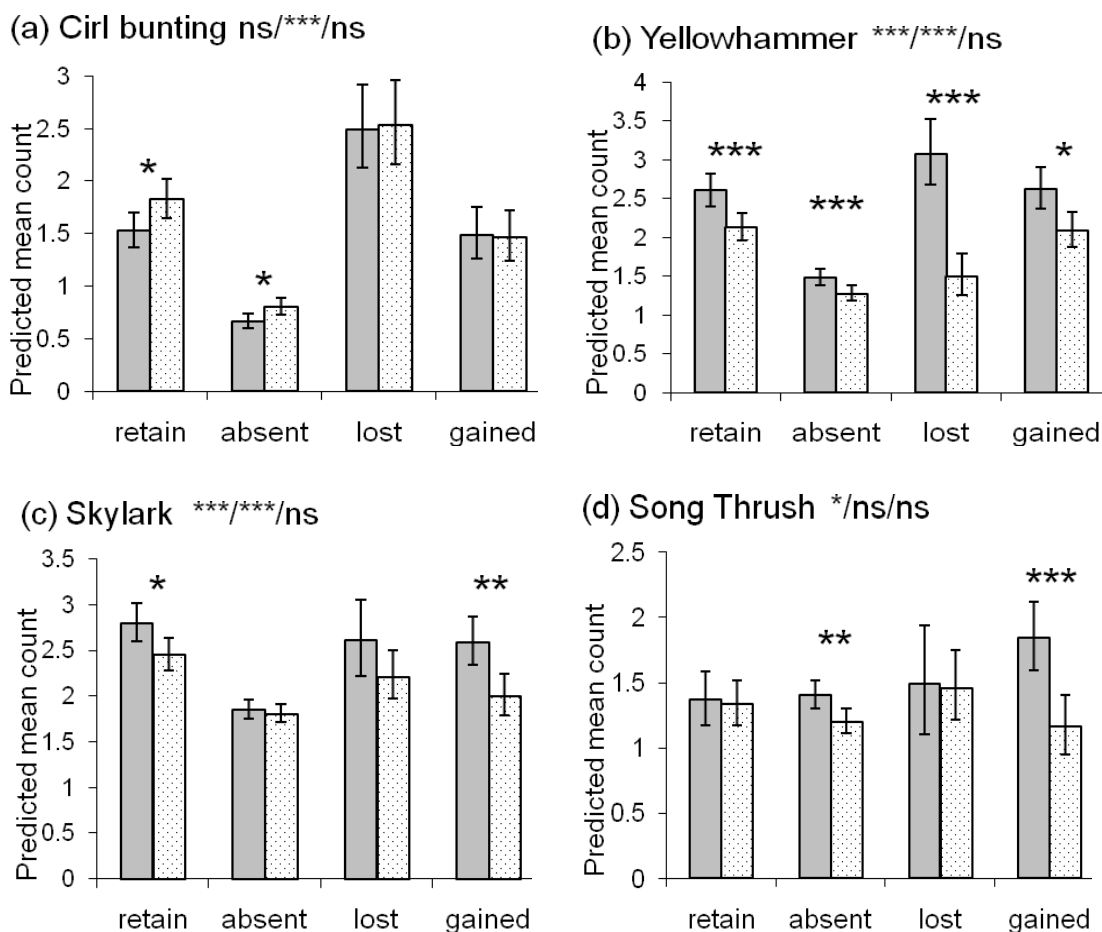


Figure 3.2. Changes in bird density on land entering ELS and not entering ELS between 2003 (filled bars) & 2009 (speckled bars). Significance of year / ELS-status / year\*ELS-status terms are indicated after species names. Error bars +/- 1 SE. Asterisks above bars show significant density changes (\*\*\* P<0.005, \*\* P<0.01, \* P<0.05).

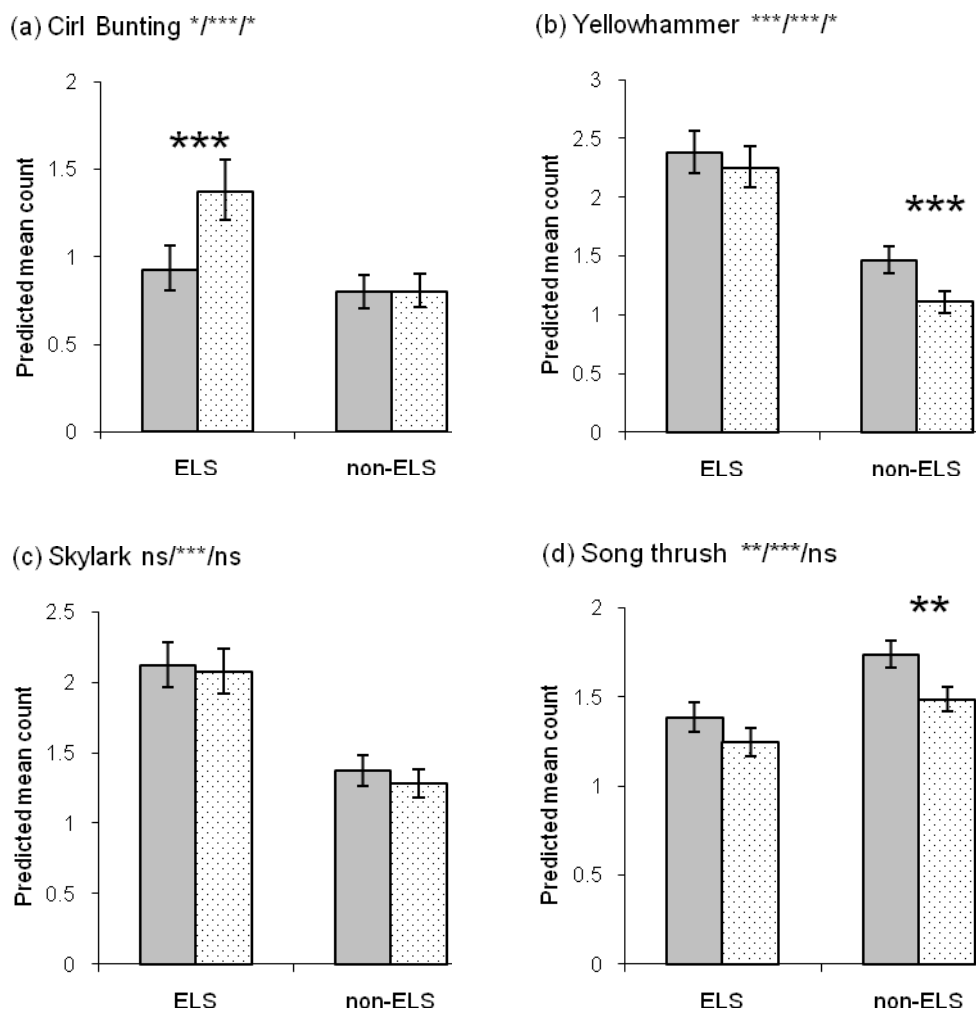


Table 3.3. Aggregate raw counts and changes in bird densities in two ELS status categories (ELS=no AE in 2003, entered ELS by 2009; Non-ELS=no AE in 2003 or 2009). The significance of the Year\*ELS-status terms indicates whether changes in density varied significantly between ELS-status categories.

Species/ ELS status	Area	Total count 2003	Total count 2009	Density 2003	Density 2009	Density Change (%)	Significance of change in density (P)
Cirl bunting: $P(\text{Year} \times \text{ELS status}) = 0.017$							
ELS	12003	106	157	0.88	1.31	+48.1	***
Non-ELS	18467	187	188	1.01	1.02	+0.5	Ns
Yellowhammer: $P(\text{Year} \times \text{ELS status}) = 0.019$							
ELS	20380	609	576	2.99	2.82	-5.4	Ns
Non-ELS	26151	409	309	1.56	1.18	-24.4	***
Skylark: $P(\text{Year} \times \text{ELS status}) = 0.580$							
ELS	22136	650	636	2.94	2.87	-2.2	Ns
Non-ELS	31234	458	427	1.47	1.37	-6.8	ns
Song thrush: $P(\text{Year} \times \text{ELS status}) = 0.581$							
ELS	22936	383	345	1.67	1.50	-9.9	ns
Non-ELS	37295	708	606	1.90	1.62	-14.4	**

### 3.3. Fine-scale habitat associations of territorial birds during the breeding season

Details of the compositional analyses are presented in Appendix 2, and a cross-species summary of habitat rankings in Table 3.4. Habitat selection was significantly non-random for all species in 2009, but only for yellowhammer in 2003 (Table 3.4). This may partly reflect the higher resolution, and more accurate habitat recording undertaken in 2009.

Given the differing ecological requirements of the four species, several of the most pronounced habitat preferences were surprisingly consistent across species. All four species showed a strong positive selection for hedges and extensively-managed grassland during both years (Table 3.4). Hedges are likely to provide nesting sites for three of the species (not skylark) and extensively-managed grassland is likely to provide invertebrate-rich foraging habitat for all four species and potentially nesting sites for skylark (Peach 2010). The positive association between skylark distribution and hedges seems counterintuitive as skylarks are in-field nesters that often avoid field boundaries (Wilson et al. 1997). However, the relatively high ranking of hedges for this species may reflect a preference for open grassland and cereal fields coupled with the relatively small field sizes and low hedges that prevail in South Devon. Ordinary hedges were significantly preferred to higher tier AE hedges by all species in both years (except song thrush in 2009), which may reflect generally lower suitability of hedges undergoing active conservation management.

Table 3.4. Summary of compositional analyses of habitat selection by territorial birds. The table lists the overall significance of habitat selection (from randomisation tests), and the aggregate areas and selection rankings of each habitat category for each species in each year. Habitats are listed in aggregate rank order based on all four species at the broader scale (usually 250m) in 2003. Woodland was excluded as an unsuitable habitat for all species except song thrush.

Significance / Habitat	Area (%)	CB Rank 100m	CB Rank 250m	YH Rank 250m	SK Rank 250m	ST Rank 100m	Rank sum*	Area (%)	CB Rank 100m	CB Rank 250m	YH Rank 250m	SK Rank 250m	ST Rank 100m	Rank sum*
Year	2003							2009						
Significance (P) habitat selection	-	0.218	0.362	0.059	0.536	0.242	-	-	0.001	0.003	0.004	0.001	0.001	-
Extensive grass	37.4	2	1	2	3	1	<b>7</b>	28.0	2	4	3	5	3	<b>15</b>
Hedge	2.1	1	1	1	4	2	<b>8</b>	1.9	3	1	2	4	4	<b>11</b>
Cereals	19.0	7	3	3	1	10	<b>17</b>	16.5	4	2	1	1	9	<b>13</b>
HL grass	6.1	3	4	4	7	7	<b>22</b>	7.7	4	8	10	10	8	<b>36</b>
Intensive grass	14.9	10	7	6	2	8	<b>23</b>	19.2	7	5	6	3	5	<b>19</b>
Garden	5.7	5	9	8	5	3	<b>25</b>	5.5	8	6	7	7	2	<b>22</b>
AE arable summer	0.7	4	5	10	5	9	<b>29</b>	0.8	1	3	5	6	7	<b>21</b>
Scrub	4.0	5	6	12	9	6	<b>33</b>	4.0	6	6	8	8	6	<b>28</b>
AE hedge	0.1	7	8	9	11	5	<b>33</b>	0.3	10	13	9	13	9	<b>44</b>
HL arable winter	<0.1	10	9	4	12	12	<b>37</b>	0.4	13	9	15	15	9	<b>48</b>
Fallow	2.5	12	9	11	7	11	<b>38</b>	0.8	14	15	10	10	15	<b>50</b>
HL special barley	1.3	9	12	6	13	13	<b>44</b>	1.1	12	14	12	2	14	<b>42</b>
Other crops	6.2	13	13	13	10	14	<b>50</b>	6.7	11	10	4	9	13	<b>36</b>
AE stubbles	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.8	15	10	12	10	16	<b>48</b>
EL grass	n/a	n/a	n/a	n/a	n/a	n/a	n/a	6.2	9	10	12	13	12	<b>47</b>
Woodland	7.1*	n/a	n/a	n/a	n/a	4	n/a	8.1*	n/a	n/a	n/a	n/a	1	n/a

Grassland under higher-level AE management is likely to be similar in character to extensively managed grassland and was weakly positively selected by all four species particularly during 2003 (Table 3.4) although song thrushes (both years) and yellowhammers (2009 only) showed a significant preference for extensive grassland over higher level AE grass. Intensively managed grassland was positively selected by skylarks (significant preference compared to extensive grass and AE grass in 2009 only) which are known to nest at high densities in grass silage (Browne et al. 2000, Peach 2010). Extensively managed grass was significantly preferred to intensively managed grass by ciril buntings (both years & scales except 2009 at 250m) and by song thrushes (in 2003), but not by yellowhammers.

The distribution of both bunting species and skylarks was positively associated with cereal distribution, although skylark (in 2009) was the only species to show any strong association with the CSS special barley option (2009 only; Table 3.4). Cereals are known to provide attractive nesting habitats for skylarks (especially when spring-sown; Donald et al. 2000) and ripening cereal grain provides an important source of chick food for buntings during wet weather (Evans et al. 1997). The same three species showed positive selection of higher-level summer arable options (mainly grass margins around arable fields), especially ciril buntings and especially during 2009 (Table 3.4). Fallow (which will have included much rotational set-aside in 2003) was not strongly selected by any

species in either year. Confirming previous work elsewhere (Peach et al. 2004), song thrushes showed a strong preference for woodland and garden habitats.

Patterns of habitat selection were generally similar in 2003 and 2009, the most notable differences being a stronger positive association between bunting distribution (both species) and summer AE arable options in 2009, and between skylarks and the special barley AE option in 2009 (Table 3.4). More generally, the association between bird distribution and extensive grassland (AE and non-AE) was weaker in 2009.

### 3.4. Habitat and AE option correlates of bird population changes between 2003 and 2009

Changes in total abundance and occupancy of 1-km squares within the 56 tetrads for which detailed habitat data were collected are summarised in Table 3.5. Changes in abundance within this subset of tetrads show a similar direction to those from the full set of tetrads (Table 3.2), the changes being of smaller magnitude for all species except yellowhammer for which the decline was marginally greater (Table 3.5).

Table 3.5. Changes in aggregate abundance and occupancy of 1-km squares within the subset of 56 tetrads for which detailed habitat data were collected.

Species	Aggregate count in 2003	Number of occupied 1km squares in 2003	Aggregate count in 2009	Number of occupied 1km squares in 2009	Change in abundance (%)	Significance of the between-year change (P)
Cirl bunting	342	117	353	121	+3.2	0.087
Yellowhammer	470	113	369	102	-21.5	0.035
Skylark	592	147	558	139	-5.7	0.61
Song thrush	326	127	318	136	-2.5	0.25

GLMM's describing associations between changes in bird counts and changes in habitat and AE provision are summarised in Table 3.6 (a & b). After allowing for other habitat-related predictors of abundance, there was strong evidence of a large decline in yellowhammer abundance between 2003 and 2009 and a smaller but significant decline in skylark abundance (Table 3.6). Bird abundance in the surrounding landscape exhibited a weak positive relationship with cirl bunting counts, while for yellowhammers both counts and changes in counts were more positive in landscapes with relatively large numbers of yellowhammers in 2003 (Table 3.6a). Counts of cirl buntings and skylarks were higher close to the coast, and skylark population changes were more positive inland than they were close to the coast (significant distance-to-coast\*year interaction, Table 3.6b).

Area of woodland in 2003 had a positive influence on counts of song thrushes, and a negative influence on counts of the other three species (Table 3.6) especially skylark, which is known to have a strong preference for open habitats away from woodland edge (Wilson et al. 1997). Song thrush was the only species to show any association between abundance and the area of extensive grassland: a positive association with initial area and a negative association with change in area between 2003 and 2009 (i.e. changes in counts were more positive in areas that lost extensive grassland, much of which entered into ELS agreements; Table 3.1). Cirl buntings were less abundant in squares with large areas of intensive grass in 2003, while yellowhammers were more abundant, and changes in counts were more positive, in squares having large areas of intensive grass in 2003 (Table 3.6a). Changes in song thrush counts were more negative in squares where the area of intensive grass increased between 2003 and 2009 (Table 3.6b).

Area of cereal production in 2003 had a positive influence on skylark counts and a negative influence on song thrush counts. Changes in yellowhammer counts were more positive in squares with a large area of cereal in 2003 (significant cereal-03\*year interaction) and more negative in squares that lost most cereal between 2003 and 2009 (positive cereal effect in Table 3.6). Cirl bunting and skylark counts showed a non-linear relationship with the proportion of arable cultivation in the square (Fig. 3.3a), both species being most abundant where the landscape was a mixture of arable cultivation and grass (i.e. abundance was maximised where the proportion of arable was approximately 26% and 40% respectively).

Skylark was the only species to show any association between abundance and fallow area. Skylark counts from both years showed significant positive relationships with fallow area in 2003 ( $P=0.0001$  in 2003 and  $0.025$  in 2009), although that relationship was significantly less positive in 2009 (Table 3.6) implying larger population declines in squares that contained relatively large areas of fallow in 2003 (Fig. 3.3b). As fallow area in 2003 was strongly correlated with change in fallow area between 2003 and 2009 ( $r=-0.91$ ,  $P<0.0001$ ), this suggests that change in fallow area is correlated with change in skylark counts although the raw correlation (which does not allow for other influential factors) was weak ( $r=0.10$ ,  $P=0.16$ ). The mean area of fallow (per 1km square) in 2003 was 1.85 ha which equates to a 20% reduction in predicted mean skylark count between 2003 and 2009 associated with loss of fallow (derived from the filled vertical line in Fig. 3.3b).

Table 3.6. Summary of factor affecting (changes in) the abundance of (a) ciril buntings & yellowhammers, and (b) skylark & song thrush. Model terms are significant predictors of counts in GLMMs conducted at the 1-km square level. F (all 1 degree of freedom) and P values are presented along with an indication of effect directions.

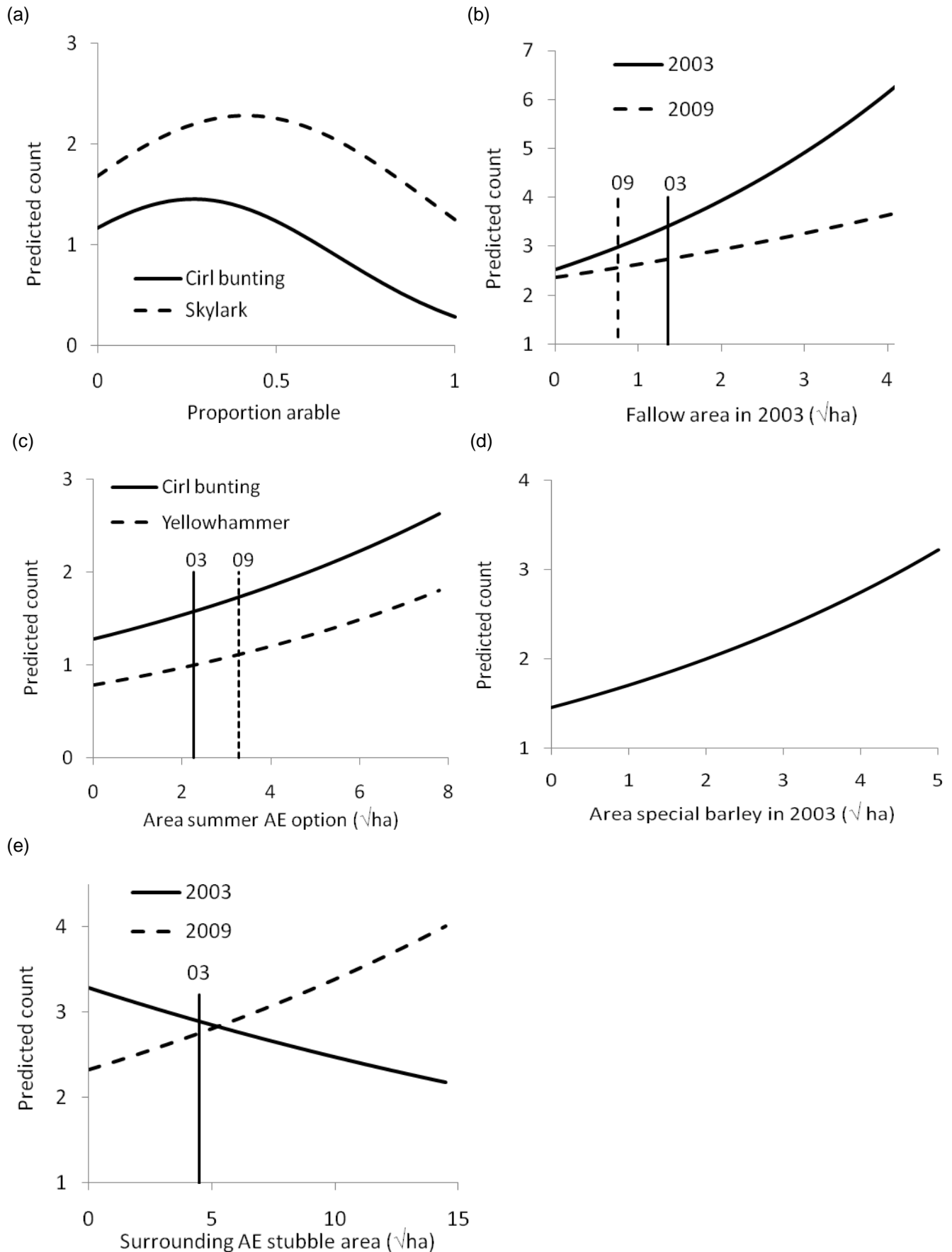
(a)

Categories	Model term	Ciril bunting		Yellowhammer	
		F/P values	Direction	F / P values	Direction (03 / 09)
Temporal	Year			12.5 / 0.0004	-
Geographic	Surrounding abundance	7.5 / 0.008	+	14.6 / 0.0002	+
	Surrounding abundance*year			4.9 / 0.027	+ / ++
	Distance to coast	17.3 / 0.0001	-		
Habitat	Wood-03	7.3 / 0.0074	-	6.2 / 0.014	-
	Intensive grass-03	15.7 / 0.0001	-	4.7 / 0.032	+
	Intensive grass-03*year			8.5 / 0.0038	+ / ++
	Cereal-03			0.8 / 0.38	
	Cereal-03*year			4.9 / 0.027	-- / -
	Cereal			8.5 / 0.0038	+
	Mixed	3.7 / 0.057	+		
	Mixed <sup>2</sup>	8.3 / 0.0041	-		
AE	AE summer-03	1.3 / 0.24		0.01 / 0.93	
	AE summer	7.7 / 0.006	+	8.5 / 0.0038	+
	HL special barley-03	4.2 / 0.043	+		
	Winter stubbles-03 @ 5km*5km	3.9 / 0.053	-		
	Wild bird seed mix-03 @ 5kmx5km			0.1 / 0.90	
	Wild bird seed mix @ 5kmx5km			4.3 / 0.038	-

(b)

Categories	Model term	Skylark		Song thrush	
		F/P values	Direction (03 / 09)	F / P values	Direction (03 / 09)
Temporal	Year	4.1 / 0.045	-	0.50 / 0.48	
Geographic	Distance to coast	6.8 / 0.0104	-		
	Distance to coast*year	8.4 / 0.0040	-- / -		
Habitat	Wood-03	26.2 / 0.0001	-	17.8 / 0.0001	+
	Extensive grass-03			8.4 / 0.0039	+
	Extensive grass			6.9 / 0.009	-
	Intensive grass-03			0.01 / 0.98	
	Intensive grass			5.1 / 0.025	-
	Cereal-03	23.0 / 0.0001	+	8.6 / 0.0037	-
	Mixed	4.4 / 0.037	+		
	Mixed <sup>2</sup>	5.5 / 0.019	-		
	Fallow-03	16.8 / 0.0001	+		
	Fallow-03*year	5.2 / 0.024	+++ / +		
AE	AE summer-03			9.8 / 0.002	-
	HL arable winter-03	1.4 / 0.24			
	HL arable winter	8.5 / 0.0037	-		
	Winter stubble-03 @ 3kmx3km	0.07 / 0.79		0.4 / 0.53	
	Winter stubble-03 @ 3kmx3km*year	15.1 / 0.0001	- / +	6.9 / 0.009	- / +
	Wild bird seed mix-03 @ 3kmx3km			0.01 / 0.91	
	Wild bird seed mix-03 @ 3kmx3km*year			6.5 / 0.012	+ / -

Figure 3.3. Predicted relationships between bird abundance and (a) the proportion of arable cultivation in the farmed landscape, (b) fallow area in 2003 (skylark), (c) the extent of summer AE options, (d) the extent of the CSS special barley option (cirl bunting) and (e) the extent of AE stubbles in the surrounding landscape in 2003 (skylark). Vertical lines show the mean areas (across all 1-km squares contributing to each model) of the relevant habitat / option in each year (filled=2003, dashed=2009) and therefore provide a measure of average change in habitat extent between 2003 and 2009.



Change in the area of summer AE options had a positive influence on changes in counts of cirl bunting and yellowhammer (Fig. 3.3c). For cirl bunting, this was mainly associated with the introduction of ELS grassland, because the area (initial and change) of higher level grass and arable summer options (mainly 2m and 6m arable margins) had no detectable influence on counts (all  $P > 0.3$ ). Change in the area of higher level plus ELS grass had a significant positive influence on cirl bunting counts (substituting for AE summer in model summarised in Table 3.6a;  $F_{1,377} = 7.2$ ,  $P = 0.0075$ ), while change in the area of higher level AE grass ( $P = 0.35$ ) and in arable summer options ( $P = 0.94$ ) did not. A similar effect of introduction of ELS grass was evident for yellowhammer ( $F_{1,376} = 5.2$ ,  $P = 0.024$ ), although for this species there was also evidence of a positive influence of summer arable options (mainly 2m and 6m margins;  $F_{1,378} = 5.4$ ,  $P = 0.021$ ). The mean area of summer AE option (per 1km-square) increased from 5.1ha in 2003 to 10.7ha in 2009, and this equates to predicted increases in abundance of 9.9% for cirl bunting and 11.5% for yellowhammer (derived from Fig. 3.3c).

The area of higher-level summer options in 2003 (mainly CSS P1 grass & 6m arable margins) was a negative predictor of song thrush counts. Cirl bunting counts (in both years) were positively correlated with the area of CSS special barley option in each 1-km square in 2003 (Fig. 3.3d), although changes in counts were not (Table 3.6). This might reflect effective targeting of known breeding areas or perhaps positive population responses that pre-date this study. Counts of skylarks were significantly more negative in squares where the area of higher-level arable winter options (mainly wild bird seed mixes) increased between 2003 and 2009.

All other associations between bird counts and AE provision were evident at a wider geographic scale and involved higher-level AE measures providing winter food. Cirl bunting counts (but not changes in counts) were weakly negatively correlated with the area of higher-level winter stubble in the surrounding 5km\*5km area during 2003 (Table 3.6a). Skylark and song thrush counts both showed (non-significant) negative relationships with the extent of AE stubbles in surrounding squares in 2003 (entirely special barley option at that time), which changed to positive relationships (significant in the case of skylark but not song thrush:  $P = 0.048$  and  $0.39$  respectively) in 2009 (Fig. 3.3e). Changes in skylark and song thrush counts were therefore more positive in landscapes having relatively large areas of special barley stubbles in 2003. The mean area of AE winter stubbles in the surrounding landscape in 2003 was 20ha which equates to a 4.9% reduction in predicted mean skylark count between 2003 and 2009 (derived from the vertical line in Fig. 3.3e). Although AE stubble provision increased between 2003 and 2009 (by 46% including ELS stubble, Table 3.1) this increase in AE provision had no detectable impact on bird counts. Most of the new stubble was in areas that lacked AE stubbles in 2003 (correlation between AE stubble area in 2003 and change in AE stubble area:  $r = -0.89$ ,  $P < 0.0001$ ).

Yellowhammer counts were (weakly) negatively related to changes in the extent of wild bird seed mixes in the surrounding landscape (Table 3.6). This result is difficult to interpret as yellowhammers often use wild bird seed mixes during winter (Field et al. 2010) and the area of wild bird cover increased substantially between 2003 and 2009 (Table 3.1). Perhaps there was a tendency for yellowhammers to settle to breed close to new winter feeding areas provided by the new wild bird cover, although if this were the case we would have expected a positive relationship with change in wild bird cover area within the square. Song thrush counts in 2003 were positively related to the area of wild bird seed mix in the surrounding 3km\*3km landscape in 2003, which changed to a negative relationship in 2009 (Table 3.6). This relationship is difficult to interpret because the change in wild bird cover area was uncorrelated with the area in 2003 ( $r = 0.11$ ), and the total area in 2003 was small (2.7ha spread across ten 1-km squares).

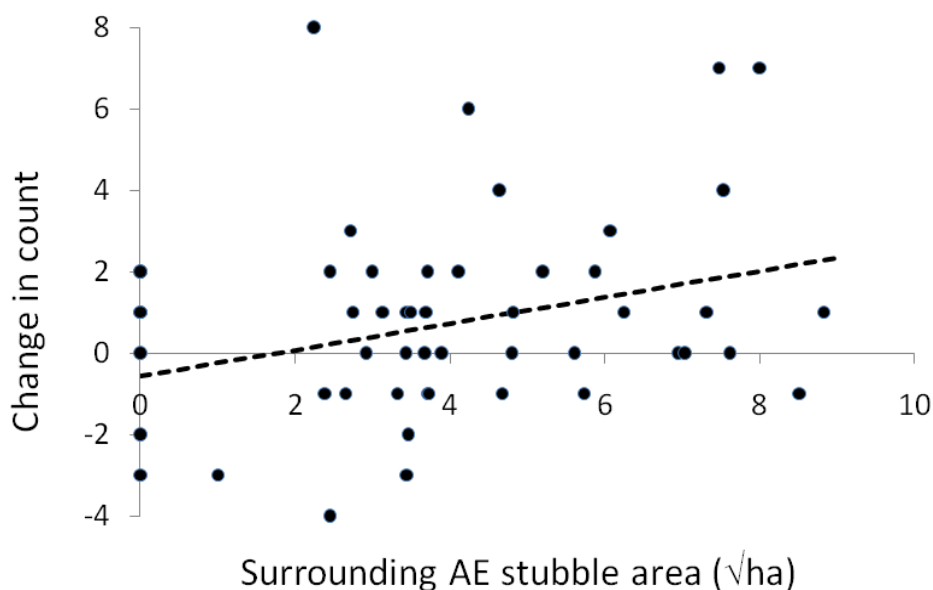
### 3.5. Correlates of bird population changes across individual CSS agreements

At the CSS agreement scale, only one correlate of change in bird count was identified: the area of CSS stubble (mainly the CSS special barley option) in the surrounding 5km\*5km area was a (weak) positive correlate of change in cirl bunting count (Fig. 3.4). Although the area of CSS stubble in the surrounding 24 1-km squares was a slightly stronger correlate of change in count ( $P = 0.0714$  vs.  $P = 0.090$  for total stubble area including that within the agreement), we plot total stubble area in Fig. 3.4 as this reflects true availability to wintering birds.

The ordinal relationship was stronger (surrounding stubble:  $P = 0.034$ ; total stubble:  $P = 0.052$ ) when one outlier agreement (where the change in count was +8, Fig. 3.4) was excluded from the analysis. The predicted relationship suggests that average population changes were positive above a total CSS stubble area of 4ha (note the x-axis in Fig. 3.4 is on a square root scale) within the 5km\*5km landscape (i.e. 0.16% of the land area), and most observed changes were positive above a total stubble area of 16ha (0.64% of the landscape). For the other three species, there were no significant associations between change in count and CSS option area.



Figure 3.4. Relationship between change in cirl bunting count on individual CSS agreements between 2003 and 2009, and total area of CSS stubble within a 5km\*5km grid centred on the individual CSS agreement. The predicted line is based on a simple regression model of stubble area against change in count that excluded one outlier (where change in count=+8). The area of stubble was square root transformed prior to analysis.



#### 4. Discussion

This study represents one of the most detailed studies to date of the effects of AE provision on changes in the abundance of farmland birds. None of the previous studies of AE provision that have considered changes in farmland bird abundance (e.g. Peach et al. 2001, Stevens & Bradbury 2006, Davey et al. 2010) have attempted to allow for the potentially confounding effects of changes in wider land use outside of AE agreements. This was particularly important during the current study period as it included the widespread loss of set-aside, which will have removed large areas of potentially important nesting and foraging habitats for a wide range of species (Gillings et al. 2010).

An important caveat to the current study relates to the rather untypical character of the farmed landscape (a mixed farming area with relatively large areas of semi-natural grassland), the maritime climate (characterised by damp summers and mild winters) and the relatively generous extent of AE provision (since 1995 this area has been the focus of intensive conservation action targeted at the cirl bunting with a special low-input spring barley option and additional resources for on-farm advice for participating farmers). The extent to which our conclusions can be extended to other parts of Britain is therefore debateable. This study does, however, provide a valuable regional assessment of the utility of targeted AE provision on priority farmland bird species in an area where AE provision was historically high, and where the intensity of farming is lower than that in many intensive arable or livestock dominated areas of Britain. Much of the higher level AE provision in the study area has been targeted on cirl buntings. Although several of the more widespread AE options (Table 3.1) are likely to have been beneficial for other farmland species (e.g. Bradbury et al. 2008), these measures were not intended to provide key resources for species like song thrushes. Thus, we might have predicted positive impacts of AE provision on both bunting species and skylark, but perhaps not for song thrushes. Having acknowledged this, options such as CSS P1 grassland and ELS low-input grassland might provide suitable foraging habitat for song thrushes.

##### 4.1. Effects of land use in the wider countryside

Several of the bird density-land use relationships confirm previous habitat association studies. For example, previous studies have highlighted positive associations between cirl bunting breeding distribution and extensive grassland (Stevens et al. 2002), between yellowhammer distribution and cereals (Kyrkos et al. 1998) and between song thrush distribution and woodland and garden habitats (Peach et al. 2004). A striking finding from this study, is the consistent selection by all four species of hedges and extensively managed grassland, and (with the exception of song thrush) cereals. Hedges provide nesting sites for buntings and thrushes, while extensive grassland is likely to provide abundant invertebrate resources for a wide range of foraging birds (Buckingham & Peach 2005). A detailed study of yellowhammers in South Devon showed that 50% of all summer foraging occurred on grassland, and 68% of this on grazed pasture (Buckingham 2005). Unlike many areas of England, extensively managed grassland is still common on the sloping fields of South Devon and this study suggests this may be a key habitat providing important summer invertebrate resources for a range of priority farmland bird

species. Intensively managed grassland was avoided by circl buntings and song thrushes, but was selected by skylarks, which are known to use grass silage as a nesting habitat (Wilson et al. 1997, Browne et al. 2000).

Cereals are a preferred nesting habitat for skylarks (Wilson et al. 1997, Donald et al. 2001), and provide important ripening grain as food for chicks especially during wet weather (Evans et al. 1997). Although previous studies have suggested that farmland birds select landscapes with a mixture of grass and arable cropping (e.g. Evans 1997) this is the first study to identify and quantify this relationship in Britain (Fig. 3.3a). A preference for nesting in cereals probably accounts for the more even mixture of arable-grassland preferred by skylarks (abundance peaked at 40% arable). Circl bunting abundance peaked at a lower proportion of arable (26%), possibly reflecting this species' reliance on cereals primarily as a 'backup' foraging habitat when (grassland) invertebrate availability is limited by wet weather (Evans et al. 1997).

The only species to show any relationship between abundance and fallow area was skylark (Table 3.6). Skylarks are known to nest at relatively high density in set-aside fields (Wilson et al 1997, Donald et al. 2001). In 2003, the relationship between skylark density and fallow area was relatively strong, with density doubling (from 2.5 to 5 territories) as fallow area increased from 0 to 10 ha per 1-km square (Fig. 3.3b). The shallower (but still significant:  $P=0.025$ ) relationship between skylark counts in 2009 and fallow area in 2003 (Fig. 3.3b) implies relatively large reductions in skylark numbers in squares that contained relatively large areas of fallow in 2003. The predicted reduction in skylark abundance associated with the mean area of fallow in 2003 was 20%. Further consideration of the sensitivity of skylark numbers to changes in fallow area are considered below.

#### 4.2. Effects of AE provision

The evidence from this study of impacts of AE options on farmland bird abundance can be summarised as follows:

- (a) a positive association between AE summer option area (particularly ELS grassland) and changes in the abundance of both circl bunting and yellowhammer (sections 3.2 & 3.4);
- (b) a positive association between CSS special barley area and circl bunting density (section 3.4), and weaker associations (both positive and negative) at the landscape scale (sections 3.4 & 3.5);
- (c) positive associations between area of CSS special barley in the surrounding landscape and changes in skylark and song thrush counts (section 3.4);
- (d) negative associations between the introduction of wild bird seed mixtures and skylark counts (in the same 1-km squares), and yellowhammer and song thrush counts (in the surrounding landscape) (section 3.4);
- (e) a negative association between the area of CSS grassland in 2003 and song thrush density (section 3.4).

The sensitivity of bird abundance to changes in AE provision is compared for the potentially beneficial AE measures (and fallow for skylark) in Table 4.1. To do this we used the relationships from the multivariate models (Fig. 3.3) to predict the amount of habitat (assuming a starting point of no provision) needed to increase bird abundance by 10%, 20%, 50% and 100%. Note the relationships in Fig. 3.3 are based on square root transformations of the (skewed) habitat/option area data, and the predictions in Table 4.1 highlight the non-linear nature of the bird abundance – habitat relationships (i.e. declining rates of bird delivery with increasing area of habitat). An important caveat to the circl bunting and CSS special barley prediction is that this relationship did not involve any change in bird abundance between 2003 and 2009. Consequently, while it is plausible that the extent of this option (which provides biologically important winter foraging habitat, Bradbury et al, 2008) is maintaining and limiting circl bunting density, it is also possible that the relationship simply reflects effective geographic targeting of this option in areas with pre-existing groups of circl buntings and has no direct influence on circl bunting demography or abundance.

Table 4.1 highlights the relatively sensitivity of both bunting species to changes in the extent of AE summer options, mainly ELS grassland (e.g. a 10% increase in abundance could be achieved by providing just 1.1 ha and 0.8 ha for circl buntings and yellowhammers respectively; Table 4.1). This is a new and rather unexpected finding of this study, as ELS low input grassland only restricts applications of inorganic fertilizers to 50 kg N /ha /year, and until recently (i.e. new ELS agreements from 2009) imposed no restrictions on sward height. Optimal sward heights for foraging buntings probably occur at average sward heights of 9-12cm (Peach 2010) but typical ELS management would be expected to produce swards shorter than this, and therefore lacking invertebrate resources (Peach 2010). Even the recently introduced restrictions on ELS grass sward heights (at least 20% of the sward to be less than 7cm, and at least 20% greater than 7cm; Natural England 2008) may be satisfied by swards subjected to normal cattle grazing practice on many lowland beef farms (unpublished RSPB data), and may therefore offer little benefit for foraging buntings. Thus, the apparent benefits of ELS grassland for buntings are rather surprising and require further research. Maybe farmers are grazing ELS grass fields more lightly or

intermittently than the option rules require. Further research would need to assess the prevailing sward heights and associated grazing management of ELS grasslands, and could assess the usage of ELS grassland by buntings relative to other habitats. We note that a recent England-wide assessment of ELS impacts on farmland bird abundance found some evidence of positive associations between ELS grassland area and population changes for corn bunting and linnet (in pastoral areas), and for turtle doves (in arable areas), but not for yellowhammer (Davey, et al. 2010). If the benefits to farmland birds of ELS grassland management can be confirmed then this option would constitute a low-cost and widely applicable conservation measure.

Table 4.1. Predicted provision (ha / km<sup>2</sup>) of AE options and fallow needed to increase bird abundance by between 10-100%. Predictions are based on the relationships plotted in Fig. 3.3, and assume a baseline AE provision of zero. \*In the case of CSS special barley and skylarks, the required level of provision is predicted using two baseline levels of provision: the level where net benefits across the two years became positive (i.e.  $\sqrt{\text{area}}=5.25$  in Fig. 3.3e, = 27.6 ha/8km<sup>2</sup>, or 3.4 ha/km<sup>2</sup>) and (in parentheses) a baseline level of zero (assumes all provision has positive benefits). The skylark relationships are based on the positive relationships involving CSS special barley in 2009 (Fig. 3.3e) and fallow in 2003 (Fig. 3.3b). Predicted land area is quoted in standard units (ha / km<sup>2</sup>). Not applicable (na) means required provision is beyond the range observed in this study.

	Option / Habitat				
Abundance increase (%)	AE summer	AE summer	CSS special barley	CSS special barley - surrounding landscape*	Fallow
Species	Cirl bunting	Yellowhammer	Cirl bunting	Skylark	Skylark
+10	1.1	0.8	0.4	7.6 (0.9)	0.2
+20	3.9	2.9	1.3	12.8 (3.0)	0.7
+50	19.4	14.4	6.5	32.5 (14.9)	3.4
+100	56.3	42.3	18.9	na	9.9

This study provides further supporting evidence of beneficial impacts of the CSS special barley option on cirl buntings, to add to that from previous studies (Peach et al 2001, Bradbury et al 2008). However, the relationship highlighted here (Fig. 3.3d) relates to abundance and not change in abundance, and could therefore reflect efficient geographic targeting of the option by NE/RSPB project staff, rather than any genuine impact on of cirl bunting abundance. Assuming the relationship does reflect a genuine impact on cirl bunting numbers, the sensitivity of cirl bunting numbers to relatively small areas of this option (Table 4.1) suggests this AE option is also relatively cost-effective. Stronger evidence relating to the main AE drivers of the cirl bunting population recovery should be provided by a similar analysis covering a longer time period (e.g. between surveys conducted in 1992 and 2009). The statistical method employed here (section 2.2.3) can be readily applied across multiple surveys conducted over a longer period, and this should be another priority for further research.

The positive relationship between skylark abundance in 2009 and area of CSS special barley in the surrounding landscape in 2003 (Fig. 3.3e) probably reflects an impact of weedy winter stubble availability on overwinter survival. There was also a much stronger association between the breeding distribution of skylarks and CSS special barley in 2009 than in 2003 implying a shift in breeding distribution towards areas with CSS special barley (Table 3.4). It is unclear why the area of CSS special barley in 2003 might have had a negative association with skylark counts in that year (Fig. 3.3e), although we stress this relationship was not statistically significant (P=0.14). Perhaps the relative abundance of suitable nesting habitat in the landscape in 2003 (e.g. more fallow and cereals, Table 3.1), combined with a concentration of CSS special barley into smaller fields (Donald et al. 2001), accounted for this weak relationship. Taken together, these results suggest little influence of the CSS special option on skylark abundance in 2003, but a relatively strong positive influence in 2009. The latter probably involved the provision of both nesting habitat and winter foraging habitat. Interpretation of the amount of habitat that might be needed to ensure skylark population growth is complicated by the change in the direction of the relationship between 2003 and 2009 (Fig. 3.3e). Given that the negative relationship between CSS special barley area and skylark abundance in 2003 was not statistically significant (P=0.137), it might be reasonable to base any predictions entirely on the positive relationship in 2009. In this case, less than 1ha of special barley per km<sup>2</sup> might be needed to ensure population growth (Table 4.1). However, a more conservative estimate based on all the data from this study would be that population level benefits for skylark only begin to accrue above special barley areas of 3.4 ha / km<sup>2</sup> (the point where the lines cross in Fig. 3.3e) or 7.6 ha / km<sup>2</sup> for a 10% increase in abundance (Table 4.1). Interestingly, this is quite similar to the area of AE stubble needed to ensure population growth (5-10 ha / km<sup>2</sup>) estimated by Winspear et al. (2010) and based on the observed relationships between area of non-AE winter stubble and population growth in yellowhammers and skylarks (Gillings et al. 2005). Skylark counts in 2003 were relatively sensitive to variation in fallow area, with only 0.2 ha / km<sup>2</sup> being needed to increase abundance by 10% (Table 4.1).

Further exploration of the skylark GLMM model highlights how the effects of fallow area and surrounding winter stubble interact with each other. With all terms in the model, there was a significant 31% decline in abundance that was unexplained by any of the covariates. Removing the fallow term from the model, transfers the positive influence of fallow area on the 2003 skylark count to the year term and allows its influence to be quantified (Table 4.2). Thus, fallow area increased predicted counts in 2003 by approximately 9% (2.25/2.06), but its subsequent loss had no overall impact on abundance in 2009 (1.42 vs. 1.43). Removing surrounding winter stubble from the model reduced predicted counts in 2003 (by 11%), and increased predicted counts in 2009 (by 13%). The net effect of fallow and CSS winter stubble on skylark counts can be seen when both are removed from the model (Table 4.2); the positive effect of fallow on 2003 counts is almost exactly offset by the negative effect of stubble (2.04/2.06 = -1%), while the net positive impact of surrounding stubble on 2009 counts is clear (1.65/1.43 = +15%).

Table 4.2. Effects on the 'Year' term of dropping the fallow (fallow-03) and stubble in the surrounding landscape (winter stubble-@3km\*3km) terms from the skylark GLMM (in Table 3.6b) on the predicted mean counts. P(year) is the significance of the 'Year' term, predicted counts are derived from the 'Year' term estimates, and % change is derived from the predicted mean counts.

Model	P(year)	Predicted 2003	Predicted 2009
Full model	0.045	2.06	1.43
Drop Fallow-03	0.0024	2.25	1.42
Drop Winter stubble-03	0.95	1.83	1.61
Drop both	0.29	2.04	1.65

Despite a substantial increase in the provision of wild bird seed mixtures in our study area (from 3ha in 2003 to 51ha in 2009; Table 3.1), there was no evidence of any positive impacts on farmland bird abundance. The negative associations between the introduction of wild bird seed mixtures and skylark counts is not surprising because skylarks prefer to forage on relatively open ground (like stubbles) and make limited usage of wild bird covers (Donald et al 2001, Henderson et al. 2004). The negative relationship between yellowhammer counts and wild bird seed mixtures in the wider landscape is more surprising given that wild bird covers should provide an attractive food-rich habitat for this species (Henderson et al. 2004). While it is possible that the weak negative association reflects a local re-distribution of breeding yellowhammers in areas close to winter feeding habitat, we would have expected (but did not see) positive impacts of wild bird seed mixtures in the same square. There has been concern about variable seed yield and mid-winter seed depletion in wild bird seed mixtures (Stoate et al. 2003, Henderson et al. 2004) and it is possible that yellowhammers feeding on wild bird covers during early winter subsequently experienced food shortages later in the winter.

The general lack of positive association between AE options and song thrush abundance is perhaps not surprising given that none of the options were specifically designed to benefit this species. However, the declining abundance of song thrushes on land entering higher-level agreements (Fig. 3.1d) and the negative association between song thrush abundance and AE summer options (mainly CSS P1 grassland) is surprising given the positive association with extensive grassland (Table 3.6b) and general reliance on grazed grassland by this species (Peach et al. 2004). Perhaps grazing intensity was too low on these P1 fields to produce sufficiently short sward heights for this ground-feeding species. Wild bird seed mixtures can be attractive to wintering song thrushes when they contain kale (Henderson et al. 2004) but there was no evidence here of any positive impacts on breeding season abundance. Further research should aim to understand the utility of higher-level AE grassland for soil-invertebrate feeding birds like song thrush.

#### 4.3. Conclusions

This study has highlighted the importance of hedges, extensively-managed grassland and cereals as key habitats determining the distribution of farmland birds in South Devon. A mixed arable-grass landscape was associated with enhanced densities of two priority species (cirl bunting, skylark).

Cirl buntings increased in abundance (+16%) between 2003 and 2009, while yellowhammer (-18%), skylark (-9%) and song thrush (-15%) all declined. Bird population changes were not generally associated with higher-level AE status although song thrushes declined most in areas where higher-level agreements were gained. Population changes of both bunting species were more positive in areas where ELS grassland management was introduced suggesting a positive influence of this widespread AE measure.

The major reduction in fallow area (set-aside) had a large negative impact on the abundance of skylarks but there was no evidence of impacts on other species. There was strong evidence that CSS special barley (followed by stubbles) became more important as a nesting and wintering habitat for skylarks following the loss of set-aside.

Cirl buntings may have been buffered from any impacts of set-aside loss through their association with CSS special barley, which is known to provide suitable wintering habitat (Bradbury et al. 2008).

Despite a substantial increase in the regional provision of wild bird seed mixtures there was no evidence of any beneficial impacts on birds. In the case of yellowhammer and song thrush there was some evidence of negative impacts of wild bird seed provision although this might have reflected some local re-distribution of breeding birds. This study has not identified potential causes of the large declines in yellowhammer and song thrush abundance, although loss of set-aside (as a wintering habitat) and cereals (as a summer foraging habitat) is likely to have removed key yellowhammer habitats that may not have been replaced by alternative AE measures.

#### 4.4. Priorities for Further Research

This project has highlighted two key areas that merit further research.

First, the positive apparent impact of ELS low-input grassland on cirl buntings and yellowhammers is unexpected and merits further investigation. Our findings suggest that either the ELS option rules are producing unexpected benefits for breeding buntings or actual management of ELS grassland differs from the maximum intensity of management permitted under ELS rules (e.g. perhaps stocking rates are lower than on non-ELS pastures, despite the absence of grazing intensity constraints). Further work could assess actual ELS grassland management including the measurement of sward height and structure, and compare this to nearby fields not under ELS management. It might also be useful to assess the usage by foraging buntings of, and potentially invertebrate densities in, ELS grassland relative to other types of grassland.

Second, an analysis of the cirl bunting data covering a longer time-period (bird survey and AE provision data are available for the following years: 1992-1998-2003-2009) would provide more statistical power in identifying the main AE option drivers of population growth, and quantifying relationships between population change and AE option area. These relationships are fundamental to the assessment of how much AE habitat provision might be needed to reverse population declines of priority farmland bird species.

A third potential area for further work would be to assess the utility of current AE grassland for soil-invertebrate feeding species like song thrush. This should consider both the availability (i.e. sward structure) and abundance of key prey invertebrate groups. Impacts of other AE options like hedgerow management, scrub control and wild bird seed mixtures should also be considered.

## References to published material

9. This section should be used to record links (hypertext links where possible) or references to other published material generated by, or relating to this project.

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