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1 Effects of agri-environmental habitat provision on winter and breeding season
2 abundance of farmland birds

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10

11 **Abstract**

12 Farmland bird populations continue to show declines in spite of over 20 years of research and
13 implementation of agri-environmental schemes (AES) intended to reverse this. Although it is well
14 known that provision of winter food resources can attract farmland birds, there is continuing
15 uncertainty over the ability of AES to provide tangible benefits for target species in terms of
16 increased abundance. Answering these questions is hampered by interannual fluctuations in bird
17 populations and the mobility and territoriality of farmland birds, which have complicated the
18 interpretation of previous studies.

19 We monitored birds for five years on a large arable estate in central England managed under varying
20 levels of AES uptake (low level uptake of simple and widely applicable AES options, more extensive
21 uptake of more complex AES options), and two control treatments (on-site and off-site). Bird
22 abundance in winter and both total abundance and number of territories in the breeding season
23 were calculated from monthly visits to 16 transects.

24 Several species showed significantly higher winter abundance on AES treatments, particularly
25 granivorous species (e.g. reed bunting, yellowhammer, linnet). Many other species (e.g. blackbird,
26 chaffinch, robin) also showed significant differences in winter abundance between treatments on
27 the estate and off-site controls. In the breeding season, linnet, reed bunting, goldfinch and
28 combined granivorous birds showed higher abundance or number of territories on AES treatments
29 compared to on-site controls. For most other species the differences were only significant between
30 treatments on the estate and off-site controls. Independently of AES treatment, a lower coverage of
31 cereals or greater Shannon diversity of crops in the local landscape also had a positive effect on the
32 abundance of many species.

33 Our results suggest that well-implemented AES can significantly enhance local populations of both
34 farmland specialists of conservation concern and generalist species. Our results also show that, in
35 many cases, these effects were only demonstrable at the farm scale, in comparison with off-site
36 controls. This is probably due to high levels of movement and dispersal of birds resulting in a farm-
37 scale spill-over of beneficial effects of agri-environment measures. Our results therefore highlight
38 the importance of thinking beyond the single-farm scale when designing schemes or studies for
39 monitoring the effectiveness of AES, and the importance of selecting appropriately located controls.

40 **Keywords:** Agri-environment, farmland birds, population, landscape, arable, supplementary feeding

41 1. Introduction

42 Agricultural intensification has led to widespread declines in farmland biodiversity over the last
43 century (Donald *et al.*, 2001; Newton, 2004; Kleijn *et al.*, 2011). Changes in farm management,
44 including removal of semi-natural habitats, increased pesticide and fertilizer input and more efficient
45 harvesting have all had deleterious impacts on farmland wildlife, including birds. Farmland birds
46 have undergone severe declines across the EU (Donald *et al.*, 2001; Donald *et al.*, 2006; EBCC, 2016),
47 and particularly in the UK (DEFRA, 2015). Whilst the exact mechanisms of negative impacts of
48 agriculture on populations vary between species, many share the loss of breeding and foraging
49 habitat due to removal of semi-natural features and increased management intensity, and the loss
50 of food resources in terms of invertebrates and seeds (Fuller, 2000). For many granivorous birds,
51 declines have been driven by the loss of overwinter food resources caused by increased herbicide
52 use and the dominance of autumn-sown cereals (Wilson *et al.*, 2009). For insectivorous species, loss
53 of foraging habitat and reduced invertebrate food resources in the breeding season have been
54 identified as important drivers of declines (Potts, 1986; Campbell *et al.*, 1997; Schaub *et al.*, 2010).

55 One of the key mechanisms for promoting population recovery of farmland birds are agri-
56 environment schemes (AES). The major AES intervention directly aimed at farmland birds has been
57 the provision of winter food resources by sowing areas of seed-bearing plants as an option for
58 participating farmers. Such resources are well known to be utilised by a wide range of farmland
59 birds, although the quality and quantity of the food provided can vary greatly with plant type and
60 subsequent management (Vickery *et al.*, 2009; Hinsley *et al.*, 2010; Field *et al.*, 2011) and across
61 space and time (Vickery *et al.*, 2009; Davey *et al.*, 2010a). There is also evidence that many sown
62 winter bird-food patches are largely depleted of seed by late winter, leaving birds with insufficient
63 resources (Perkins *et al.*, 2008; Siriwardena *et al.*, 2008; Hinsley *et al.*, 2010). Whilst some agri-
64 environmental management options have been introduced to address this issue (e.g. extended

65 overwinter stubbles, supplementary seed feeding), their efficacy is relatively unexplored (but see
66 Siriwardena *et al.*, 2007).

67 In addition to overwinter food, AES can also provide habitat for foraging and nesting in the breeding
68 season via creation, restoration or maintenance of hedgerows (Hinsley and Bellamy, 2000; Maudsley
69 *et al.*, 2000; Staley *et al.*, 2012), in-hedge trees (Redhead *et al.*, 2013), field margins (Vickery *et al.*,
70 2009; Pywell *et al.*, 2011) and other semi-natural habitat features. This can both increase local
71 populations of invertebrates (Vickery *et al.*, 2009; Woodcock *et al.*, 2010) and make them more
72 accessible to foraging birds (Perkins *et al.*, 2000; Benton *et al.*, 2003), as well as providing suitable
73 nesting habitat.

74 Many studies have demonstrated local and farm-scale successes of AES (Hinsley *et al.*, 2010; Baker
75 *et al.*, 2012; Aebischer *et al.*, 2016) and, recently, Bright *et al.* (2015) demonstrated that higher level
76 schemes enhanced breeding densities of some priority farmland bird species even in the absence of
77 ongoing advisory support. However, in spite of over 20 years of AES provision, and accompanying
78 research, declines in farmland birds have continued, both in terms of individual species (Eaton *et al.*,
79 2015; Harris *et al.*, 2016) and aggregate farmland bird indicators (DEFRA, 2015). The extent to which
80 AES have mitigated these declines remains largely unknown (Kleijn and Sutherland, 2003; Kleijn *et*
81 *al.*, 2006; Kleijn *et al.*, 2011). Because farmland birds are mobile, with many species being partially or
82 wholly migratory, the potential for movements of birds between winter feeding sites and breeding
83 areas even within the same study landscape has complicated the interpretation of several studies
84 (Hinsley *et al.*, 2010; Aebischer *et al.*, 2016). Therefore the overall effectiveness of AES remains
85 unclear, with the general consensus that the current level of uptake of beneficial options is
86 insufficient to promote a reversal of national-scale population declines (Davey *et al.*, 2010a; Davey *et*
87 *al.*, 2010b; Baker *et al.*, 2012). If new AES aim to be better equipped to achieve population increases,
88 it is important to improve the understanding of how AES management affects farmland birds at the
89 local scale, both in the immediate vicinity of AES interventions and the wider context of the farm or

90 holding. It is also important to understand whether the response to AES is consistent between
91 winter and breeding seasons, accounting for the year-round mobility of birds over farmland
92 landscapes (Siriwardena *et al.*, 2006; Siriwardena, 2010).

93 The present study seeks to understand the impact of the provision of winter food and summer
94 breeding habitat on the local populations of a range of farmland bird species, over a long-term
95 experiment (5 years) within a single, large-scale farmland landscape. We monitored bird numbers in
96 both winter and the spring/summer breeding season, and utilised two levels of AES (low level uptake
97 of simple and widely applicable AES options, more extensive uptake of more complex AES options
98 tailored to local circumstances), a control, and a spatially separated control to allow investigation of
99 'spill-over' of birds from AES to non-AES treatments. The aims were to:

- 100 1. Determine the impact of AES management on winter bird numbers.
- 101 2. Investigate whether increased winter bird abundance due to AES seed provision resulted in
102 increased numbers of breeding birds or territories.
- 103 3. Examine whether such increases were detectable at the treatment scale and/or farm scale.

104 **2. Methods**

105 2.1. STUDY SITE

106 The study took place on the Hillesden Estate, which comprises approximately 1000 ha of
107 predominantly arable farmland close to Buckingham, central England (51°57'N, 1°00'W, Figure 1).

108 The estate lies on seasonally wet clay soils with crop rotations dominated by winter wheat *Triticum*
109 *aestivum*, winter oilseed rape *Brassica napus*, field beans *Vicia faba* and spring barley *Hordeum*
110 *vulgare*.

111 The experimental layout of the Hillesden Estate was initially established in 2005/2006 in order to
112 monitor the effects of Environmental Stewardship (ES, the then recently introduced UK AES) on
113 farmland biodiversity, including farmland birds, and productivity (e.g. Hinsley *et al.*, 2010; Woodcock

114 *et al.*, 2010; Redhead *et al.*, 2013; Broughton *et al.*, 2014; Pywell *et al.*, 2015). In 2011 the
115 experimental design was altered to improve the ability to compare the effects of management under
116 i) Entry Level (ELS) and ii) Higher Level Stewardship (HLS) schemes, which involved the relocation of
117 treatments and AES options to the current experimental design (Figure 1). A high level of spill-over
118 effects between adjacent treatments (i.e. effects resulting from movement or dispersal of organisms
119 between different treatments) had been detected after the first five years of the study (Hinsley *et*
120 *al.*, 2010; Broughton *et al.*, 2014) so the 2011 revision of the experimental design increased the
121 spacing between treatments and controls (Figure 1, and see Hinsley *et al.* 2010, Figure 1). In
122 addition, four off-site control areas, located 2-7 km from the estate, were added to further
123 investigate the extent of spill-over between treatments and controls on the estate (Figure 1). The
124 updated experimental design of the estate and off-site controls was then monitored for birds for five
125 years, from spring 2012 to spring 2016.

126 2.2. AGRI-ENVIRONMENT TREATMENTS AND CONTROLS

127 Under the two AES treatments (ELS and HLS), a percentage of land was removed from production
128 and replaced with habitat creation options under ES (see Table 1 for details). In accordance with
129 common agricultural practice, options were placed to minimise impact on farm productivity, mostly
130 being situated in field corners which were difficult for farm machinery to access or along field
131 margins. For further detail on the composition and creation of habitats at Hillesden under ES see
132 Hinsley *et al.* (2010); Redhead *et al.* (2013); Broughton *et al.* (2014) and the ES handbooks (Natural
133 England, 2012a, b). The control treatment, termed Cross Compliance (CC), represented the minimum
134 level of environmental management required of farms receiving the Common Agricultural Policy
135 single farm payment and was thus typical of lowland arable landscapes in central England. Under CC,
136 fields had uncultivated, annually cut margins measuring 2 m in width from the centre of a bordering
137 hedgerow, or 1 m from the top of a ditch. Such margins were also located on the agri-environmental
138 treatments (ELS and HLS) wherever fields were not bordered by specific agri-environmental options.

139 Off-site controls were chosen to have the same level of management as Cross Compliance controls,
140 with similar soil types, landscape composition and cropping regimes as at Hillesden. In practice, on
141 some off-site controls farmers did add a small number of game cover strips (mostly maize) and
142 gamebird feeders, neither of these being present on the Hillesden estate, but these formed much
143 lower proportions (0.3-0.5 %) of the farmed landscape than ELS options at Hillesden.

144 2.3. WINTER BIRD SURVEYS

145 Winter bird numbers were surveyed on transects following a stretch of hedgerow internal to each
146 treatment, approximately 1km in length (range 659m – 1450m, Figure 1). The landscape around
147 Hillesden is typified by hedgerows with large, mature, emergent trees (mostly English oak, *Quercus*
148 *robur*) so hedgerows were chosen to be relatively uniform in this regard, with no lengthy sections of
149 continuous tree canopy ('treelines'). Transects were walked monthly in December, January and
150 February in the winters of 2012-2013, 2013-2014 and 2015-2016, with each surveyor visiting three
151 transects per site visit, and each visit completed by 13:00 GMT. On each visit, all birds seen or heard
152 in the hedge and in any adjacent bird food patches or field margins were recorded at their observed
153 location on a 1:10000 scale Ordnance Survey map (zoomed to ~1:2000 scale), using standard
154 methods to denote species and activity (Bibby *et al.*, 1992; BTO, 2016). All surveyors were
155 professional ornithologists or ecologists with high levels of experience in ornithological field survey.
156 The selection of transects assigned to each surveyor and the order in which they were visited was
157 varied every month to avoid surveyor bias of likelihood of detection and effects of the time of day on
158 the activity levels of the birds. Transects were not visited when heavy rain or strong winds were
159 present (i.e. favourable conditions for a transect visit were approximately wind force of Beaufort 0-4
160 and precipitation absent or light and intermittent).

161 2.4. BREEDING BIRD SURVEYS

162 Breeding birds were surveyed in spring/summer ('breeding season') on the same transects as the
163 winter surveys. Transects were walked monthly in April to July from 2012 to 2016, using identical

164 methods to winter surveys, except that breeding season visits began shortly after sunrise and were
165 completed within 4 hours. Particular attention was paid to recording activities that assisted with
166 defining territories (e.g. song, aggression, nest building or provisioning etc.).

167 2.5. PROCESSING BIRD SURVEY DATA

168 All mapped records, for both winter and breeding season visits, were digitised using GIS software
169 (ArcMAP v10.1-10.3 © ESRI, Redlands, CA). Annotated field survey maps were scanned, and the
170 digital copy georeferenced using a digital version of the 1:10000 Ordnance Survey data. Bird
171 observations were then digitized by placing digital points at locations indicated by the annotated
172 map. Because potential minor errors in the spatial placement of bird records derive from the
173 accuracy of field annotations, georeferencing the scanned maps and the placement of digital points,
174 exact accuracy is hard to quantify, but with experienced field surveyors and GIS staff it is likely to be
175 less than $\pm 10\text{m}$. This is highly likely to be sufficient to accurately determine numbers of birds and
176 territories per transect, which were the response variables for this study. We filtered the digitized
177 data to remove birds located more than 10m from the transect hedgerows and adjacent AES
178 habitats. This filtered out bird records associated with habitats (e.g. small woodlands, copses, small
179 ponds) other than those the transects were intended to survey and birds seen only within crop
180 fields, which were poorly detected by our survey method of walking along hedgerows, especially
181 when crops were well grown (Atkinson *et al.*, 2006). We also removed birds seen only in flight.
182 For breeding season data, we used GIS to overlay bird registrations across visits within each year,
183 and then assigned birds to territories based on location and recorded bird behaviour (Bibby *et al.*,
184 1992). Total winter abundance, breeding season abundance and numbers of territories (for
185 territories where at least 50% of the territory was deemed to overlay the transect hedgerow) for
186 each species were then calculated.

187 Aggregate total abundance and number of territories were also calculated for all species combined,
188 granivorous species (i.e. those most likely to benefit from provision of winter seed, see Table 2),

189 granivores excluding chaffinch (which accounted for around 40% of total granivore records), resident
190 insectivorous species (i.e. those most likely to respond to changes in local invertebrates populations,
191 see Table 2) and species on the UK farmland bird indicator (FBI, Gibbons *et al.* (1996); Gregory *et al.*
192 (2005); DEFRA (2015), see Table 2). Species for which there were less than 10 records over the five
193 years were excluded from all analyses (these were mostly migrants on passage or species from other
194 habitats) and species without at least one territory or recorded adult for each treatment within each
195 year were excluded from individual analyses, although they still contributed to aggregate totals (see
196 Table 2). Species excluded by the latter threshold were either those with insufficient data for robust
197 analysis (e.g. lesser whitethroat, bullfinch) or those where we had less confidence in the suitability of
198 our hedgerow transect method for accurate estimation of local abundance, i.e. species associated
199 with habitats other than hedgerows (e.g. open fields for grey partridge, woodland for woodpeckers).

200 For similar reasons, woodpigeons and carrion crows were omitted from analyses, despite being
201 relatively abundant on transects. Woodpigeons were observed in large flocks within fields and/or
202 woodland patches off the hedgerow transects, and crows were recorded in small, highly mobile
203 groups observed to cover large distances. We therefore concluded that our hedgerow transects
204 were not suitable for recording highly mobile, wide-ranging and gregarious species and that transect
205 counts for such species were unlikely to provide accurate estimates of local abundance.

206 We here present territory results only for species groupings (i.e. all species, insectivores, granivores,
207 granivores excluding chaffinches and farmland bird index species), with full species results available
208 in Supplementary Material (Tables S1 and S2). This is because territory data showed very similar
209 results to abundance for most individual species and because our method of assigning territories
210 from the four breeding-season visits may be less informative than total abundance for species with
211 semi-colonial breeding behaviour (e.g. linnets (Drachmann *et al.*, 2000)), species which largely forage
212 outside the core breeding territory (e.g. goldfinches (Conder, 1948)) or species with complex mating
213 systems (e.g. dunnock (Birkhead, 1981; Bishton, 2001))

214 2.6. HABITAT COVARIATES

215 Whilst most habitat variables were broadly similar across treatments, being within the same
216 farmland landscape, some transects were closer to habitats which were not directly manipulated by
217 experimental treatments but which may influence bird presence and numbers (Fuller *et al.*, 2004).
218 These included improved grassland (present in small proportions of the landscape along the edge of
219 some transects) and areas of woody vegetation (small copses and woodlands), as well as woody
220 vegetation on the transect itself arising from in-hedge trees. We therefore quantified improved
221 grassland area (using mapped farm management data) and extent of woody vegetation (i.e. hedges
222 and trees, using airborne LiDAR data) as potential covariates (Table 3). LiDAR data were gathered on
223 28th August 2007, under conditions of full leaf canopy, from a mean flight altitude of 1190 m
224 (Optech 3033 Airborne Laser Terrain Mapper, scan half angle of 20°, ground sampling rate of 1 pulse
225 per square metre, each pulse supplying a first and last return elevation measurement). The use of
226 LiDAR data for determining woody vegetation cover within the study landscape is described in
227 Redhead *et al.* (2013). Although the LiDAR data were collected in 2007, the structure of woody
228 vegetation in the study landscape has remained relatively constant over the past decade, with any
229 changes consisting of annual growth and minor changes in hedgerow dimensions due to cutting
230 regimes, rather than significant changes in the location or extent of woody vegetation within
231 treatment areas.

232 Other habitat variables changed between years due to crop rotation, so that cropping patterns were
233 not always consistently balanced between treatments. We therefore calculated Shannon diversity
234 indices of crops and the coverage of cereals (the dominant crop in the landscape) per transect per
235 year from mapped farm management data, as further potential covariates (Table 3).

236 It should be noted that potential habitat covariates were intended to capture variation beyond that
237 expected from the different agri-environmental treatments and so do not account for habitats
238 created as part of such (e.g. grass margins). Habitat covariates were originally extracted, using GIS,

239 within three buffers around the transects to explore potential effects of varying spatial scale (10m,
240 50m and 100m) but preliminary analyses showed the results from different distances to be strongly
241 collinear, so the 100m buffer was selected as containing most information on the landscape beyond
242 the relatively consistent structure of the transect hedgerows.

243 2.7. STATISTICAL ANALYSIS

244 Generalized linear models were constructed to determine differences between treatments and
245 years, and to analyse potential impact of habitat covariates. A negative binomial distribution was
246 assumed for abundance and territory counts, with transect length as an offset term to account for
247 variation in the length of transects (longer transects being expected to have more birds/territories).
248 Modelling was performed using the *nb.glm* function of R (v3.2.2, R Core Team, 2015). For each
249 species/grouping, we constructed all possible independent models of the form:

$$250 \quad Y \sim \text{offset} + \text{Treatment} + \text{Year} + (\text{Treatment} * \text{Year}) + (\text{habitat variable})$$

251 Where Y = bird abundance or territory count, offset = length of transect, and terms in parentheses
252 are optional. Candidate models were constrained to include treatment, year and the offset term,
253 giving a total of 16 candidate models per species analysed. We did not fit any models containing
254 more than one habitat variable in order to avoid overfitting or attempting to include collinear
255 variables in the same model.

256 We used the MuMIn package (Barton, 2015) in R to generate candidate models. Candidate models
257 were ranked according to the corrected Akaike information criterion (AICc, (Burnham and Anderson,
258 2003) and the 'best' model (i.e. with lowest AICc) examined in detail. Because the 'best' model as
259 determined by AICc may still be non-significant, overall significance of the 'best' model was
260 determined via a likelihood ratio test against a null model consisting of only the intercept and offset
261 terms. Where a significant effect of treatment was observed, Tukey post-hoc tests, implemented in

262 the multcomp package (Hothorn *et al.*, 2008), were applied to determine which treatments showed
263 significant differences from one another.

264 3. Results

265 3.1. WINTER BIRD SURVEYS

266 All species, except song thrush, and all species groupings (Fig. 2) showed a significant effect of
267 treatment on the abundance of birds recorded in winter (Table 4). For the majority of species, there
268 were significantly higher numbers of birds on the Hillesden estate (i.e. Cross Compliance controls,
269 ELS and/or HLS treatments) than on off-site controls in the winter (Tukey *post hoc* tests, Table 5). In
270 some cases (granivores excluding chaffinch, farmland bird index species, dunnock, reed bunting and
271 yellowhammer) there was no significant difference between off-site and Cross Compliance controls,
272 but significantly higher numbers on ELS and/or HLS treatments than on Cross Compliance. Linnets
273 showed significantly higher numbers on ELS and HLS than Cross Compliance controls but, uniquely,
274 also showed significantly higher numbers on off-site controls than Cross Compliance controls.

275 Only a few individual species and species groups showed significant differences between the two
276 AES treatments, with greater numbers on HLS than ELS treatments for only reed bunting, dunnock,
277 insectivores and all species combined.

278 3.2. BREEDING BIRD SURVEYS

279 The majority of species and species groupings (Fig. 2) showed a significant effect of treatment on
280 abundance during the breeding season, with the two exceptions being wren and yellowhammer
281 (Table 6). However, in contrast to winter abundance, this effect was largely attributable to
282 differences between off-site controls and the three treatments on the Hillesden estate (Tukey *post-*
283 *hoc* tests, Table 5). Only reed bunting showed significantly greater breeding season abundance on
284 ELS or HLS treatments in comparison to Cross Compliance controls, whilst goldfinch and blue tit
285 showed the opposite trend, with numbers in Cross Compliance controls being significantly greater

286 than in ELS and HLS, respectively. Some species (blackbird, goldfinch, reed bunting) did show
287 significantly higher numbers on HLS than ELS.

288 All five species groupings (all species, granivores, granivores excluding chaffinch, resident
289 insectivores, and farmland bird index species) showed a significant effect of treatment and year on
290 territory numbers (Fig. 2, see Supplementary Material, Table S1 for full results). As for winter and
291 spring abundance, there were significant differences between off-site controls and treatments on
292 the Hillesden estate. However, unlike the abundance results, numbers of territories also showed
293 significantly greater numbers on HLS than ELS for all species groupings except insectivores. For the
294 latter, the significant effect of treatment appeared to be due to a significant difference between the
295 two most contrasting treatments, off-site controls and HLS (see Supplementary Material, Table S2).
296 Territory level results for individual species can be found in supplementary material (Tables S1 and
297 S2).

298 3.3. INTERANNUAL VARIATION

299 The majority of species showed significant inter-annual variation in winter abundance (Table 4).
300 Tukey *post hoc* tests showed that all significant differences in winter abundance across years were
301 attributable to higher numbers in winter 2013-2014 and/or 2015-2016 than in 2012-2013
302 (Supplementary material, Table S3). For many species, numbers in the breeding season appeared
303 less variable, with no significant year effect on breeding season abundance (Table 6). Where a
304 significant effect of year on breeding season abundance did occur, pairwise differences between
305 years varied across species. Robin and wren (and therefore the resident insectivores grouping)
306 showed lower numbers in 2012 and 2013 compared to other years of the study, whilst chaffinch
307 showed significantly lower numbers in 2016 than other years and dunnock was lower in 2013.
308 Generally, species showing no significant year effect in winter (blackbird, goldfinch, great tit, reed
309 bunting, song thrush) did not show a year effect in the breeding season either. Although the
310 treatment*year interaction term was not retained in any of the 'best' models, it can be clearly seen

311 from Figure 2 (and Supplementary Material, Figs S1 - S3) that the effect of treatment did vary to
312 some extent with year.

313 3.4. HABITAT COVARIATES

314 Nearly 70% of 'best' models contained a habitat covariate (Tables 4 and 6). $\Delta AICc$ and Akaike weights
315 were frequently relatively low for the 'best' models. This was mostly because the top few candidate
316 models per species/grouping often involved habitat covariates which were to some extent
317 correlated (e.g. total area of trees vs. total area of woody cover) and therefore showed only minor
318 differences in model fit. This can be seen from the much higher Akaike weights achieved by summing
319 the top ranked three models (Tables 4 and 6). The habitat covariates which appeared most
320 frequently in best-fitted models across species and groupings were a positive effect of crop diversity
321 (3 cases) or a negative effect of proportional cover of cereals (i.e. proportion of the transect
322 surroundings covered in cereal crop).

323 4. Discussion

324 4.1. IMPACT OF AES MANAGEMENT ON WINTER BIRD NUMBERS

325 Provision of winter food resources on the Hillesden estate via ELS and HLS treatments co-occurred
326 with increased winter abundance for a wide variety of species, including nationally-declining
327 farmland granivores (e.g. reed bunting, yellowhammer) and other generalist species of hedgerows,
328 woodlands and gardens (e.g. blackbird, dunnock, wren, robin). This result is unsurprising, as the
329 provision of sown bird food is well known to attract foraging birds (Perkins *et al.*, 2008; Hinsley *et al.*,
330 2010; Field *et al.*, 2011; Aebischer *et al.*, 2016). Even for species which are primarily insectivorous,
331 agri-environmental field margins are likely to provide increased winter resources in comparison to
332 crops due to an increased range of host plants and a lack of direct application of insecticides (Wilson
333 *et al.*, 1999; Vickery *et al.*, 2009). The exact mechanism behind increased winter abundance of
334 insectivores in our results is unclear, as many species which are primarily insectivores are to some

335 extent omnivorous in winter (e.g. robin, dunnoek) and so likely to benefit from both increased seed
336 provision via sown bird food and any coincident increase in invertebrates in AES habitats.

337 It was apparent that the increases in bird numbers were not necessarily in direct proportion to the
338 quantity of food provision – for many species there was no apparent difference between ELS and
339 HLS treatments, despite the area of sown winter bird food being approximately three times greater
340 in HLS. Birds do not necessarily occur in higher abundances at the most concentrated resources
341 (Siriwardena *et al.*, 2006), especially over the relatively short separation distances between ELS and
342 HLS treatments at Hillesden (mean distance from ELS sown bird food patch to closest HLS patch =
343 574m). Previous studies have also established that that many farmland bird species readily move
344 between patches where resources are less than 1 km apart (Siriwardena *et al.*, 2006; Siriwardena,
345 2010), so many species will effectively treat nearby ELS and HLS treatments as part of the same
346 foraging landscape.

347 For the more generalist species (e.g. blackbird, blue tit, chaffinch, robin) which showed differences
348 only between off-site controls and transects on the Hillesden estate, detecting the exact mechanism
349 is difficult without intensive studies using tracking or marking of individual birds (e.g. Siriwardena *et*
350 *al.*, 2006; Siriwardena, 2010). However, the results suggest that the movement of birds within the
351 local landscape of the Hillesden estate, driven by spatial and temporal variation in food availability
352 and weather conditions (Siriwardena *et al.*, 2008), created a spill-over effect, such that the influence
353 of AES treatments on winter abundance is more widely distributed than their immediate
354 surroundings.

355 The unusual result for linnets, with greater winter abundance on off-site controls, ELS and HLS than
356 on Cross Compliance controls may result from the behaviour of this species in forming particularly
357 large, mobile winter feeding flocks. A single flock locating a viable resource, for example a game
358 cover strip, in the otherwise resource poor local landscape of an off-site control may have a large
359 effect on total abundance.

360 4.2. IMPACT OF AES MANAGEMENT ON BREEDING BIRDS

361 Of potentially greater interest is the effect of winter food provision and associated habitat
362 improvements on breeding populations, since this is far more likely to signal the status of local
363 populations. Our results suggest that AES management under ELS and HLS may have significant,
364 positive impacts on local breeding populations. However, in nearly all cases these impacts were only
365 detectable when comparing off-site controls to controls and AES treatments on the Hillesden estate,
366 suggesting that benefits were largely at a farm scale. This is likely to be due to similar spill-over
367 effects as for winter abundance, but there are several reasons why these might be expected to be
368 more pronounced in the breeding season. Firstly, birds may prioritise different resources over the
369 course of the year, so that those areas that are best for winter feeding are not necessarily the most
370 suitable for nesting or feeding offspring (Vickery *et al.*, 2009). Secondly, territoriality in the breeding
371 season may set an upper limit to the numbers of birds in a single treatment, forcing dispersal to
372 other parts of the estate (Newton, 1992). Finally, birds utilising sown bird food patches in winter
373 may have migrated at local, national or international scales (Wernham, 2002; Siriwardena *et al.*,
374 2008) and so breed at sites far distant from where they spend the winter.

375 It is important to note that we do not attempt to distinguish between, on the one hand, a true spill-
376 over of surplus birds resulting from local population increases in HLS and ELS treatments and, on the
377 other, the simple movement or dispersal of birds between treatments that results in the benefit of
378 treatments being more widely distributed than their immediate surroundings. However, either or
379 both mechanisms may explain the apparent lack of response in breeding populations to winter food
380 provision when birds are monitored within a single farm or on sites without spatially separated
381 controls (Hinsley *et al.*, 2010; Aebischer *et al.*, 2016), and emphasise the importance, and difficulties,
382 of considering spatial scale when designing monitoring studies.

383 Although much of the potential effect of HLS over ELS is likely to be masked by the effects described
384 above, some species did show significantly greater numbers in the breeding season on HLS

385 treatments (blackbird, goldfinch and reed bunting abundance; number of territories for all species
386 combined, granivores and farmland bird index species). This suggests that there are benefits from
387 the more extensive uptake of more complex HLS options over the low level uptake of simple and
388 widely applicable ELS ones, at least for some species.

389 The lack of significant differences in yellowhammer breeding season abundance between
390 treatments on the Hillesden estate and the off-site controls, despite the clear differences seen in
391 winter, warrants particular attention. As a declining granivore, yellowhammers would be expected
392 to benefit from provision of resources under AES. Previous studies have presented somewhat
393 contradictory findings regarding the association between numbers of wintering yellowhammers and
394 those of breeding birds (Robinson *et al.*, 2001; Whittingham *et al.*, 2005). Our results suggest that
395 this may be because any such association is highly likely to depend on the spatial scale at which the
396 study is performed, and the extent to which the survey methodology samples the resources and
397 likely locations of yellowhammers at different times of the year. Yellowhammers are relatively
398 mobile over winter, travelling to locate food resources within the local landscape (Siriwardena *et al.*,
399 2006; Siriwardena, 2010) and then dispersing to find suitable territories in early spring (Andrew,
400 1956). Yellowhammers also have comparatively large territories, which they actively defend against
401 conspecifics (Andrew, 1956). This may help to drive dispersal from winter feeding areas to the
402 surrounding landscape, as might the known reduction in seed availability on bird food patches in late
403 winter (Hinsley *et al.*, 2010).

404 In contrast to yellowhammers, reed bunting abundance responded strongly to AES treatment, at
405 both the farm scale (off-site controls vs. all other treatments) and treatment scale (ELS vs. HLS) in
406 winter and the breeding season. On farmland, reed buntings nest and forage preferentially in tall,
407 non-woody vegetation, avoiding hedges (Brickle and Peach, 2004; Surmacki, 2004), with the latter
408 being apparent in our results (Table 6). Nesting and feeding opportunities for this species may,
409 therefore, be much more closely associated with the more diverse field margins of HLS than is the

410 case for yellowhammers. Increased breeding season abundance on Cross Compliance controls over
411 off-site controls suggests that some spill-over does occur, with breeding populations on Cross
412 Compliance controls potentially enhanced by winter food provision in ELS and HLS bird food patches
413 in the winter.

414 It should be borne in mind that provision of increased food resources under AES may increase the
415 fitness of individuals and thus the ability to breed successfully or increase productivity, even if there
416 is no apparent effect on total abundance because of other limiting factors (Robb *et al.*, 2008). For
417 example, whilst the area and proximity of tree canopy within nesting territories was previously
418 demonstrated to be strongly predictive of breeding productivity in both blue tits and great tits at the
419 Hillesden site (Redhead *et al.*, 2013), only blue tits showed a positive response in abundance to the
420 area of woody vegetation around the transect.

421 4.3. EFFECTS OF INTERANNUAL VARIATION AND HABITAT CONTEXT

422 Interannual variation was evident for many species, and therefore important to account for in the
423 models. Few patterns were consistent across species, beyond the generally lower abundance for
424 most species in winter 2012-13 and lower breeding season abundance of insectivores in 2012 and
425 2013. Some of this may be due to the fact that in the first year (2012) of surveys, patches sown in
426 the autumn of 2011 had yet to fully establish in terms of overwinter seed provision and reduced
427 hedgerow cutting regimes would not have had time to have an impact on winter berry yield (Croxton
428 and Sparks, 2002; Staley *et al.*, 2012) or invertebrate numbers (Maudsley *et al.*, 2000; Amy *et al.*,
429 2015). However, the impact of weather on bird populations is also likely to be a major contributor to
430 interannual variation (Robinson *et al.*, 2007). Whilst winter temperatures were consistently mild
431 across the study years, 2012-2013 was the coldest of the three surveyed (Supplementary material,
432 Figure S4A). The breeding season in 2012 had very high rainfall, (Supplementary Material, Figure
433 S4B) including the wettest April on record for southern England (Met Office, 2016) whilst 2013 had
434 the coldest breeding season (mean April-July temperature) for the UK since 1962 (Met Office, 2016).

435 All of these may have combined to reduce numbers in 2012 and 2013, especially for small-bodied
436 insectivores (Robinson *et al.*, 2007).

437 Landscape covariates were frequently included in the 'best' models, suggesting that even in an
438 experimental design intended to make the level of AES intervention the major difference across the
439 study landscape, bird numbers were still significantly affected by other landscape factors. The
440 frequent negative effect of cereal cover in the local landscape is unsurprising. Cereals are the
441 dominant agricultural vegetation in the study landscape, so in effect coverage of cereals is the
442 inverse of total cover of semi-natural habitat, non-cropped areas and crops other than cereals.
443 Although cereal stubbles can be beneficial for overwintering birds (Perkins *et al.*, 2008; Field *et al.*,
444 2011), the majority of Hillesden's cereals are autumn-sown and thus do not provide overwinter
445 stubbles. In the breeding season, cereal crops are likely to be of least value compared to other land
446 uses in the study area in terms of invertebrates and seed food supplies (Wilson *et al.*, 1999;
447 Woodcock *et al.*, 2010), although not necessarily in comparison to other arable crops (Holland *et al.*,
448 2012). Cereals, however, are less structurally diverse than other crops and so are likely to provide
449 fewer opportunities for nesting and cover than oilseed rape or field beans. This is also reflected in
450 the fact that crop diversity had a positive effect for some species. Because grass crops and fallows
451 were included in this metric, this suggests that mixed landscapes, as well as more diverse crop
452 rotations, can be beneficial (Holland *et al.*, 2012; Santana *et al.*, 2017). With current shifts in UK
453 policy potentially moving away from EU crop diversification greening rules, simpler rotations and a
454 greater predominance of cereals are likely outcomes, with potentially detrimental results for
455 farmland birds.

456 **5. Conclusions**

457 Our results show that AES management, including provision of winter food resources, coincided with
458 significantly increased winter numbers of farmland birds, especially linnets, yellowhammers and
459 reed buntings. More importantly, they suggest that for several species this resulted in increased

460 numbers of breeding birds and territories, both for farmland specialists of conservation concern (e.g.
461 reed bunting, linnet) and more widespread and generalist species (e.g. blackbird, chaffinch, robin).
462 Our results therefore suggest that provision of winter and breeding-season resources under agri-
463 environmental schemes can be of real benefit to a wide range of farmland bird species. This effect
464 was, however, often only demonstrable when AES treatments were compared with off-site controls
465 rather than with controls on the same farmland estate, likely due to spill-over of birds between
466 treatments. This reinforces the importance of considering effects beyond the single-farm scale when
467 designing schemes or studies for monitoring the effectiveness of AES, and the importance of
468 selecting appropriately located controls (Kleijn and Sutherland, 2003). The mobility of farmland birds
469 means that effects are not always predictable or confined to the areas of highest resource provision
470 (Siriwardena *et al.*, 2006; Siriwardena, 2010), suggesting that planned management of farmland
471 landscapes at a wider scale than single farms is likely to bring increased benefits. There was some
472 evidence for an increased benefit from the “low uptake, high intensity” HLS options over the “high
473 uptake, low intensity” ELS ones. However, it should also be noted that whilst the levels of AES
474 uptake and the combinations of options selected for this study were realistic, efforts were made to
475 ensure that options were well managed and successful in achieving their desired outcomes (e.g.
476 sown winter bird food producing good coverage of seed bearing plants). Such a situation is by no
477 means guaranteed across the wider farmed landscape.

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656 **Table 1** Agri-environment scheme (AES) interventions on Entry Level (ELS) and Higher Level
 657 Stewardship (HLS) treatments. Option codes refer to those in the Environmental Stewardship
 658 handbooks (Natural England, 2012a, b)

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Option Description	Option Code	per ELS treatment	per HLS treatment	Detail
Total area of land removed from agricultural production	-	Approx. 1%	Approx. 5%	Land which would otherwise be used for arable crops, replaced with AES options
Permanent tussocky grass margins	EE3, EJ9	2 margins (6m wide)	1 margin (12m wide)	Uncut except in first year of establishment (to suppress weeds)
Pollen- and nectar-producing flowers	EF4, HE10	1 patch	3 patches 1 margin	Contained a variety of flowering species including legumes (e.g. <i>Trifolium</i> spp.) [‡]
Spring sown wild bird seed mixture	EF2, HF2, HF12	1 patch	2 patches	See Hinsley <i>et al.</i> (2010) for details of seed mixes [‡]
Autumn sown wild bird seed mixture	HF12		1 patch	Autumn sown to provide seed resources earlier in the season ^{† ‡}
Perennial wildflowers	EF1, EF4 HE10	-	3 patches	Contained a wide variety of 'meadow' flowers
Enhanced permanent grass margins	HE10	-	1 margin (12m wide)	Contained a mixture of grass and wildflower species
Extended overwintered stubble	EF22	-	1 area	Provided late-winter seed resources. Relocated within treatments every year
Supplementary feeding for wild birds	EF23	1 site (100m track)	-	12.5 kg seed* spread twice per week from beginning January - end April

* Seed mix comprised 68% wheat, 10% white millet, 10% red millet, 6% canary and 6% oil seed rape. Supplied by Vine House Farm Ltd., Lincolnshire

[†] Seed mix comprised 1% fodder radish, 5% kale, 1% stubble turnip, 38% winter linseed, 5% gold of pleasure, 25% winter barley, 25% winter triticale. Supplied by Cotswold Seeds Ltd., Gloucestershire

[‡]Rotational – these options exchanged location within treatments after 3 years

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670 **Table 2** Total number of records within 10m of transects and/or adjacent patches over spring and
 671 winter visits, across study years, and mean (\pm standard error) numbers per year. Only species with a
 672 total of at least ten records in either winter or the breeding season are shown. Migratory (or largely
 673 so) species are indicated by S = summer or W = winter. Species groupings are indicated by columns
 674 GV (Granivorous), IV (resident insectivorous) and FBI (farmland bird indicator). Note that total winter
 675 numbers appear lower due to fewer visits per year (three vs. four) and fewer censuses overall (three
 676 vs five). Bold type indicates species with at least one record/territory per treatment per year.

Common name	Scientific name	Migrants	GV	IV	FBI	Winter numbers		Breeding Season		Territories	
						Total	Mean (\pm SE)	Total	Mean (\pm SE)	Total	Mean (\pm SE)
Blackbird	<i>Turdus merula</i>					569	190 (\pm11)	1067	213 (\pm10)	276	55 (\pm3)
Blackcap	<i>Sylvia atricapilla</i>	S				0	0	43	9 (\pm 2)	6	3 (\pm 2)
Blue tit	<i>Cyanistes caeruleus</i>					466	155 (\pm20)	763	153 (\pm5)	186	37 (\pm1)
Bullfinch	<i>Pyrrhula pyrrhula</i>		X			41	14 (\pm 2)	58	12 (\pm 2)	8	4 (\pm 3)
Buzzard	<i>Buteo buteo</i>					12	6 (\pm 4)	6	1 (\pm 0)	-	-
Carrion crow	<i>Corvus corone</i>					88	29 (\pm 4)	273	55 (\pm 9)	-	-
Chaffinch	<i>Fringilla coelebs</i>		X			1157	386 (\pm65)	1777	355 (\pm31)	450	90 (\pm7)
Chiffchaff	<i>Phylloscopus collybita</i>	S				0	0	67	13 (\pm 3)	16	5 (\pm 1)
Dunnock	<i>Prunella modularis</i>			X		418	139 (\pm40)	645	129 (\pm12)	224	45 (\pm4)
Fieldfare	<i>Turdus pilaris</i>	W				1015	338 (\pm 116)	0	0	-	-
Goldfinch	<i>Carduelis carduelis</i>		X			46	23 (\pm6)	273	55 (\pm8)	4	4 (\pm 0)
Greenfinch	<i>Chloris chloris</i>		X		X	51	17 (\pm 5)	56	11 (\pm 2)	5	3 (\pm 2)
Grey partridge	<i>Perdix perdix</i>				X	9	3 (\pm 1)	20	5 (\pm 3)	-	-
Great tit	<i>Parus major</i>					187	62 (\pm5)	427	85 (\pm9)	120	24 (\pm3)
Green woodpecker	<i>Picus viridis</i>					7	2 (\pm 0)	19	4 (\pm 1)	-	-
Great spotted woodpecker	<i>Dendrocopos major</i>					6	2 (\pm 1)	29	6 (\pm 1)	-	-
House sparrow	<i>Passer domesticus</i>					5	5 (\pm 0)	37	9 (\pm 3)	-	-
Jackdaw	<i>Corvus monedula</i>				X	111	37 (\pm 7)	129	26 (\pm 6)	-	-
Kestrel	<i>Falco tinnunculus</i>				X	7	4 (\pm 1)	12	4 (\pm 1)	-	-
Lesser whitethroat	<i>Sylvia curruca</i>	S				0	0	60	12 (\pm 2)	9	5 (\pm 2)
Linnet	<i>Carduelis cannabina</i>		X		X	800	267 (\pm136)	728	146 (\pm16)	127	25 (\pm3)
Long tailed tit	<i>Aegithalos caudatus</i>					30	10 (\pm 3)	71	14 (\pm 2)	-	-
Magpie	<i>Pica pica</i>					19	10 (\pm 2)	50	10 (\pm 2)	-	-
Meadow pipit	<i>Anthus pratensis</i>					27	14 (\pm 11)	21	5 (\pm 2)	-	-
Pheasant	<i>Phasianus colchicus</i>					66	22 (\pm 7)	59	12 (\pm 2)	-	-
Redwing	<i>Turdus iliacus</i>	W				559	186 (\pm 98)	0	0	-	-
Reed bunting	<i>Emberiza schoeniclus</i>		X		X	257	86 (\pm36)	131	26 (\pm5)	51	10 (\pm1)
Red legged partridge	<i>Alectoris rufa</i>					39	13 (\pm 3)	16	3 (\pm 1)	-	-
Robin	<i>Erithacus rubecula</i>			X		267	89 (\pm12)	356	71 (\pm10)	126	25 (\pm4)
Song thrush	<i>Turdus philomelos</i>					223	74 (\pm3)	141	28 (\pm4)	54	11 (\pm2)
Starling	<i>Sturnus vulgaris</i>				X	193	64 (\pm 36)	43	9 (\pm 5)	-	-

Stock dove	<i>Columba oenas</i>		X	21	7 (±2)	47	9 (±1)	-	-
Whitethroat	<i>Sylvia communis</i>	S	X	0	0	815	163 (±8)	249	50 (±3)
Woodpigeon	<i>Columba palumbus</i>		X	493	164 (±56)	232	46 (±9)	-	-
Wren	<i>Troglodytes troglodytes</i>		X	222	74 (±10)	533	107 (±22)	191	38 (±7)
Yellowhammer	<i>Emberiza citrinella</i>		X	766	255 (±80)	1238	248 (±10)	297	59 (±2)
Yellow wagtail	<i>Motacilla flava</i>	S	X	0	0	40	8 (±2.9)	8	4 (±2.0)

- Insufficient data/not suitable for territory assignment

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678 **Table 3** Landscape covariates calculated within the surroundings of each transect.

Variable group	Data source	Variable name	Description	Mean (±SD)	Range (Min – Max)
Land use	Mapped farm management records	Div. Crop	Shannon diversity of crops	0.18 (±0.24)	0 - 0.67
		Div. Habitat	Shannon diversity of broad habitats (arable crop, grass crop, woodland)	0.38 (±0.22)	0 - 0.75
		% Cereal	Percentage cover of cereal cropped fields	48.43 (±38.63)	0 - 99.85
		% Grass	Percentage cover of grass-cropped fields	6.55 (±8.53)	0 - 26.6
Woody cover	LiDAR data	% Woody	Percentage cover of all woody vegetation (hedges, trees, copses)	7.00 (±1.96)	4.23 - 11.13
		% Tree	Percentage cover of trees over 3m high	4.66 (±1.78)	2.09 - 7.92
		% Hedge	Percentage cover of hedgerows >1m and < 3m high	2.34 (±0.58)	1.07 - 3.61

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680 **Table 4.** Results of generalized linear models for winter abundance (i.e. total number of adult bird
681 observations), using multi-model comparisons to select the ‘best’ fitting model for each
682 species/group. Results are: Δ AICc (difference in AICc from the top ranked model to the second
683 ranked model), Akaike weights (for the top model only and summed over the top three models),
684 likelihood ratio test of the top ranked model against a null model, p value of Moran’s I test of spatial
685 autocorrelation. Covariates are named as in Table 3. The significance of factors and covariates are
686 denoted as follows: * p <0.05, + p <0.05 and positive coefficient, - p <0.05 and negative coefficient

Species	Δ AICc	Akaike Weight (1)	Akaike Weight (3)	Likelihood ratio test		Factors		Habitat covariates		
				χ^2	p value	Moran's I	Treatment	Year	Covariate	Effect
All species	1.08	0.28	0.59	33.78	<0.001	0.013	*	*	Div. Habitat	+
Granivores	0.19	0.21	0.55	44.58	<0.001	0.009	*	*	Div. Crop	+
Granivores excl. chaffinch	1.02	0.25	0.53	40.48	<0.001	0.780	*	*		
Insectivores	1.92	0.30	0.51	46.67	<0.001	0.866	*	*		
Farmland bird index	2.61	0.33	0.51	33.29	<0.001	0.717	*	*		
Blackbird	1.33	0.28	0.53	23.11	<0.001	0.722	*		Div. Crop	+
Blue tit	6.01	0.84	0.91	39.10	<0.001	0.082	*	*	% Cereal	-
Chaffinch	1.40	0.54	0.87	32.45	<0.001	0.080	*	*	Div. Habitat	+
Dunnock	2.39	0.30	0.47	40.80	<0.001	0.531	*	*		
Goldfinch	0.86	0.25	0.55	14.60	0.012	0.793	*			
Great tit	1.56	0.37	0.67	15.72	0.015	0.322	*		% Cereal	-
Linnet	1.61	0.46	0.76	43.13	<0.001	0.951	*	*	% Woody	+
Reed bunting	4.78	0.81	0.94	35.35	<0.001	0.491	*		% Hedge	-
Robin	5.60	0.80	0.89	33.99	<0.001	0.380	*	*	% Cereal	-
Song thrush	1.29	0.25	0.49	5.71	0.335‡	0.792				
Wren	0.77	0.26	0.60	16.64	0.005	0.696	*	*		
Yellowhammer	0.14	0.28	0.62	32.14	<0.001	0.435	*	*		

‡ No significant difference from null model

688 **Table 5.** Results of Tukey *post hoc* tests between treatments for adult bird winter and breeding
689 season abundance (total number of adult birds). All pairwise comparisons between treatments are
690 represented (OSC = off-site control, CC = Cross Compliance, ELS = Entry Level Stewardship, HLS =
691 Higher Level Stewardship). +/- indicates the direction of a significant difference between treatments,
692 e.g. OSC relative to CC, etc.

Species	Winter abundance						Breeding season abundance					
	OSC / CC	OSC / ELS	OSC / HLS	CC / ELS	CC / HLS	ELS / HLS	OSC / CC	OSC / ELS	OSC / HLS	CC / ELS	CC / HLS	ELS / HLS
All species	-	-	-	-	-	-	-	-	-	-	-	-
Granivores	-	-	-	-	-	-	-	-	-	-	-	-
Granivores excl. chaffinch	-	-	-	-	-	-	-	-	-	-	-	-
Insectivores	-	-	-	-	-	-	-	-	-	-	-	-
Farmland bird index	-	-	-	-	-	-	-	-	-	-	-	-
Blackbird	-	-	-	-	-	-	-	-	-	-	-	-
Blue tit	-	-	-	+	+	-	-	-	-	-	+	+
Chaffinch	-	-	-	-	-	-	-	-	-	-	-	-
Dunnock	-	-	-	-	-	-	-	-	-	-	-	-
Goldfinch	-	-	-	-	-	-	-	-	-	+	-	-
Great tit	-	-	-	-	-	-	-	-	-	-	-	-
Linnet	+	-	-	-	-	-	-	-	-	-	-	-
Reed bunting	-	-	-	-	-	-	-	-	-	-	-	-
Robin	-	-	-	-	-	-	-	-	-	-	-	-
Song thrush	-	-	-	-	-	-	-	-	-	-	-	-
Whitethroat	-	Not present in winter					-	-	-	-	-	-
Wren	-	-	-	-	-	-	-	-	-	-	-	-
Yellowhammer	-	-	-	-	-	-	-	-	-	-	-	-

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701 **Table 6.** Results of generalized linear models for breeding season abundance (i.e. total number of
702 adult bird observations), using multi-model comparisons to select the ‘best’ fitting model for each
703 species/group. Results are: Δ AICc (difference in AICc from the top ranked model to the second
704 ranked model), Akaike weights (for the top model only and summed over the top three models),
705 likelihood ratio test of the top ranked model against a null model, p value of Moran’s I test of spatial
706 autocorrelation Covariates are named as in Table 3. The significance of factors and covariates are
707 denoted as follows: * p <0.05, + p <0.05 and positive coefficient, - p <0.05 and negative coefficient

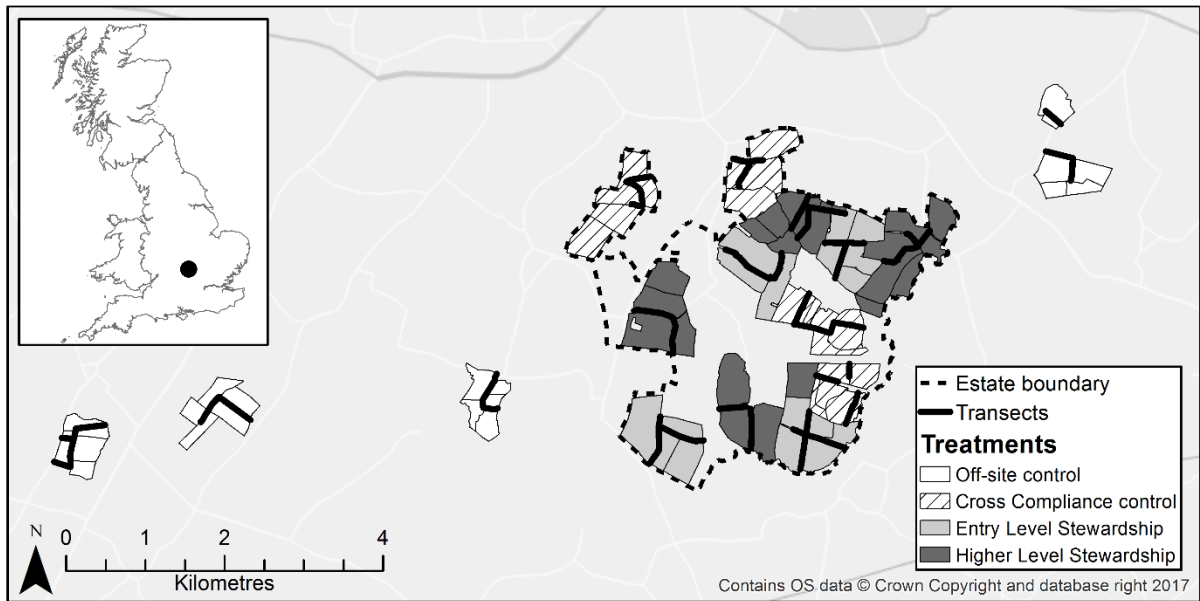
Species	Δ AICc	Akaike Weight (1)	Akaike Weight (3)	Likelihood ratio test			Factors		Habitat covariates	
				χ^2	p value	Moran's I	Treatment	Year	Covariate	Effect
All species	8.61	0.97	1.00	57.36	<0.001	<0.001	*	*	% Cereal	-
Granivores	8.43	0.98	1.00	47.26	<0.001	<0.001	*	*	% Cereal	-
Granivores excl. chaffinch	5.66	0.93	0.99	23.30	<0.001	<0.001	*		% Cereal	-
Insectivores	3.17	0.77	0.98	47.63	<0.001	<0.001	*	*	% Cereal	-
Farmland bird index	9.53	0.99	1.00	38.30	<0.001	0.781	*	*	% Cereal	-
Blackbird	0.34	0.21	0.54	33.06	<0.001	0.083	*		Div. Crop	+
Blue tit	0.29	0.20	0.53	39.63	<0.001	0.994	*		% Woody	+
Chaffinch	0.14	0.22	0.59	72.51	<0.001	0.047	*	*		
Dunnock	7.02	0.89	0.94	31.49	<0.001	0.420	*	*	% Cereal	-
Goldfinch	1.59	0.69	1.00	28.21	<0.001	0.575	*		% Woody	-
Great tit	0.48	0.23	0.53	22.10	0.002	0.181	*	*		
Linnet	9.28	0.98	1.00	23.84	0.002	0.273	*		% Cereal	-
Reed bunting	13.54	1.00	1.00	44.21	<0.001	0.726	*		% Hedge	-
Robin	1.75	0.41	0.70	39.25	<0.001	0.819	*	*	Div. Habitat	+
Song thrush	1.45	0.39	0.68	30.43	<0.001	0.124	*		% Hedge	-
Whitethroat	6.33	0.93	0.99	39.95	<0.001	0.253	*		% Cereal	-
Wren	3.91	0.76	0.91	60.82	<0.001	0.589		*	% Cereal	-
Yellowhammer	0.99	0.24	0.53	9.19	0.327 [‡]	0.144				

‡ No significant difference from null model

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710 **Figures**

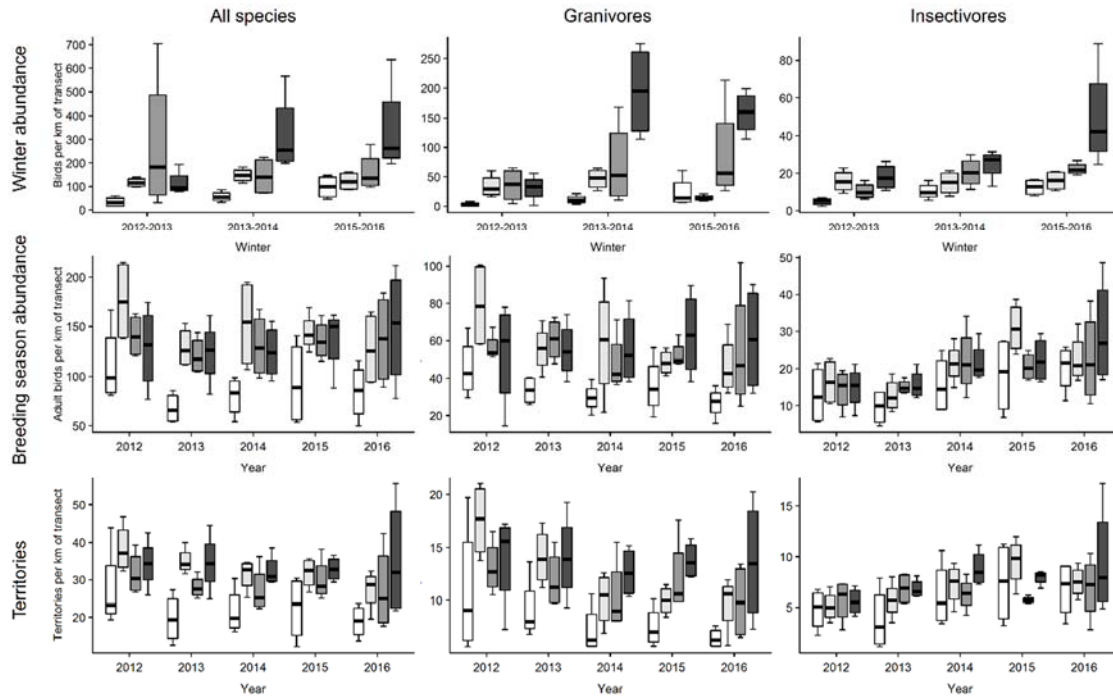


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712 **Figure 1. Map of the study area showing experimental treatments on the Hillesden Estate and the**
713 **locations of off-site controls in surrounding farmland. Inset map shows location of Hillesden within**
714 **central England.**

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718 **Figure 2. Boxplots of winter, breeding season and territory abundance per kilometre of transect by**

719 **treatment and year, for three groupings of farmland birds (all species, granivores and resident**

720 **insectivores). Treatment is indicated by colour fill of boxes: white = off-site control, light grey = Cross**

721 **Compliance control, mid grey = Entry Level Stewardship, dark grey = Higher Level Stewardship.**

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