


## RESEARCH ARTICLE

# The effects of a decade of agri-environment intervention in a lowland farm landscape on population trends of birds and butterflies

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## Abstract

1. Declines in farmland biodiversity remain evident despite over three decades of research and implementation of agri-environment schemes (AES). Although positive effects of AES are often demonstrated locally or in the short term, studies exploring longer term trends in biodiversity often show contradictory results. Evidence for the potential of AES to drive beneficial changes in populations remains sparse, especially for mobile taxa such as birds and butterflies.
2. We analysed the abundance of 12 widespread bird and 9 butterfly species from a 10-year study of AES intervention in a farmland landscape in southern England. We compared estimates of annual population growth rates from our study landscape with rates derived from large-scale national monitoring schemes in equivalent landscapes without substantial AES.
3. Species trends in our study landscape were frequently stable or increasing, in contrast to concurrent declining trends in equivalent landscapes without AES. These differences were significant for total abundance of granivorous species and for chaffinch *Fringilla coelebs*, blue tit *Cyanistes caeruleus* and great tit *Parus major* individually. For butterflies, differences in trends were significantly more positive for gatekeeper *Pyronia tithonus* and green-veined white *Pieris napi*, while small white *P. rapae* showed a trend that was significantly more negative in our study landscape.
4. *Synthesis and applications.* Our results demonstrate that, for some bird and butterfly species, the higher abundances associated with areas of AES uptake within a typical commercial farmland landscape can co-occur with positive or stable population trends over long time scales and that these trends can show

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significant differences from those in equivalent landscapes without substantial AES interventions. Our results suggest that previously observed inconsistencies in AES benefits may in part reflect a lack of long-term studies with accurate data on AES uptake and quality (i.e. successful implementation and management). Our results, thus, affirm the importance of delivering and monitoring high-quality AES options if the design and implementation of the next generation of AES is to achieve significant benefits for biodiversity.

#### KEYWORDS

abundance, AES, agriculture, arable, biodiversity, farmland, landscape, trends

## 1 | INTRODUCTION

The widespread expansion and intensification of farming practices over the twentieth century has brought about well-known declines in farmland biodiversity (Donald et al., 2006; Green et al., 2005; Kleijn et al., 2009; Warren et al., 2001). These declines have been observed both for those species which are particularly associated with agriculture and those dependent on non-agricultural habitats which have become fragmented and reduced in extent.

A key mechanism for attempting to reverse these declines has been agri-environment schemes (AES). These offer financial compensation for taking land out of agricultural production or changing farming practices to achieve environmental targets. AES have been established in many countries for over three decades, with a corresponding history of research into their effectiveness. However, there is still debate regarding their true potential to reverse population declines, especially for more widespread and mobile species of farmland birds and butterflies (Birrer et al., 2007; Kleijn et al., 2006; Kleijn et al., 2011; Kleijn & Sutherland, 2003; Vickery et al., 2004). In the United Kingdom, farm-scale experiments comparing different levels of AES uptake (e.g. Field et al., 2005; Field et al., 2007; McHugh et al., 2018; Pywell et al., 2004) or monitoring of single sites before and after AES uptake (e.g. Aebischer et al., 2016; Taylor & Morecroft, 2009) have shown positive associations with AES, and there have been species-specific success stories (Peach et al., 2001; Perkins et al., 2011). However, farmland birds and butterflies continue to show aggregate declines despite over 20 years of AES aiming to support their recovery (Eaton et al., 2015; Fox et al., 2015; Gross, 2016).

Previous studies have suggested that this failure to translate local AES successes into reversing national declines in widespread species may be due to insufficient areas of land under AES management (e.g. Baker et al., 2012), poor implementation (e.g. Loblely et al., 2013; McCracken et al., 2015; Waddington et al., 2014), neglect of spatial configuration (e.g. Concepción et al., 2008; Siriwardena, 2010) or combinations of these (Daskalova et al., 2019; Emery & Franks, 2012; Kleijn et al., 2006). The high mobility of birds and butterflies also makes it difficult to distinguish local population increases from relocation to abundant resources (Aebischer et al., 2016; Hinsley et al., 2010). Many studies exploring longer-term or larger-scale

effects have also suffered from biases inherent in experimental designs (Josefsson et al., 2020) and from a lack of accurate and precise information on the location and quality of AES implementation over large spatiotemporal extents. AES option quality is here defined as the extent to which AES implementation has led to the provision of the intended resources for wildlife (e.g. seed bearing plants in wild bird seed mixtures, pollen and nectar rich flowers in wildflower areas), which varies widely depending on farmer experience and environmental context (McCracken et al., 2015). Thus, studies that link long-term population data from national-scale monitoring schemes with estimated levels of AES uptake (e.g. Baker et al., 2012; Brereton et al., 2002; Carvell et al., 2015; Dadam & Siriwardena, 2019) must interpret their results with the caveat that option configuration and quality are unknown. Longer-term studies of population trends on farms with known levels of high-quality AES, compared with spatially separated controls, are rare, and limited almost entirely to birds (e.g. Bright et al., 2015; Colhoun et al., 2017; Walker et al., 2018).

We monitored bird and butterfly populations on a long-term (10-year) experiment in southern England, within a large, well-characterised farm landscape with high-quality AES. Our aims were as follows:

1. To explore whether the results of previous studies demonstrating higher abundances of widespread farmland bird and butterfly species in the parts of the study landscape with higher AES uptake (Heard et al., 2012; Hinsley et al., 2010; Redhead et al., 2018) were associated with positive population trends over the 10-year time series following establishment of AES.
2. To compare such trends with concurrent trends from national monitoring schemes in equivalent control landscapes (i.e. similar climate, soils, topography, land cover composition and landscape structure but without substantial AES).

Thus, we assess the potential for well-implemented AES to be associated with positive long-term impacts on farm-scale populations of widespread bird and butterfly species, using AES options and levels of uptake that are plausible targets for the majority of lowland English farms. Although not 'long-term' in comparison with other ecological datasets (Lindenmayer et al., 2012), 10 years is beyond the range of most studies involving experimental manipulations of

AES (Aebischer et al., 2016) and represents the length of a typical AES agreement in England.

## 2 | MATERIALS AND METHODS

### 2.1 | Study landscape – The Hillesden experiment

The Hillesden Estate is situated in southern, lowland England (51°57'N, 1°00'W, Figure 1). The Estate comprises ~1000 ha of predominantly arable farmland on seasonally wet clay soils. AES was first implemented in 2006, as a 5-year experiment to explore the effects of Environmental Stewardship (England's main AES over the study period, newly launched at the time) on biodiversity. The experiment was modified in 2011, to reflect changes in AES policy, and monitored for a further 5 years. For both 5-year phases of the experiment (2006–2010 and 2012–2017, see below) areas under AES had 1%–5% of land taken out of production and replaced with habitats (i.e. options) delivering a range of resources, including grass margins, perennial wildflowers, pollen- and nectar-producing flowers, and sown wild bird seed mixtures. AES implementation and management was informed by expert environmental and agronomic advice to ensure that AES option quality was high (Heard et al., 2012; Hinsley et al., 2010). Hillesden was selected as being typical in terms of landscape structure and farming system of much of lowland England. Extensive details on AES habitats and the Hillesden landscape can be found in Hinsley et al. (2010) and Redhead et al. (2018).

### 2.2 | Bird and butterfly transects at Hillesden

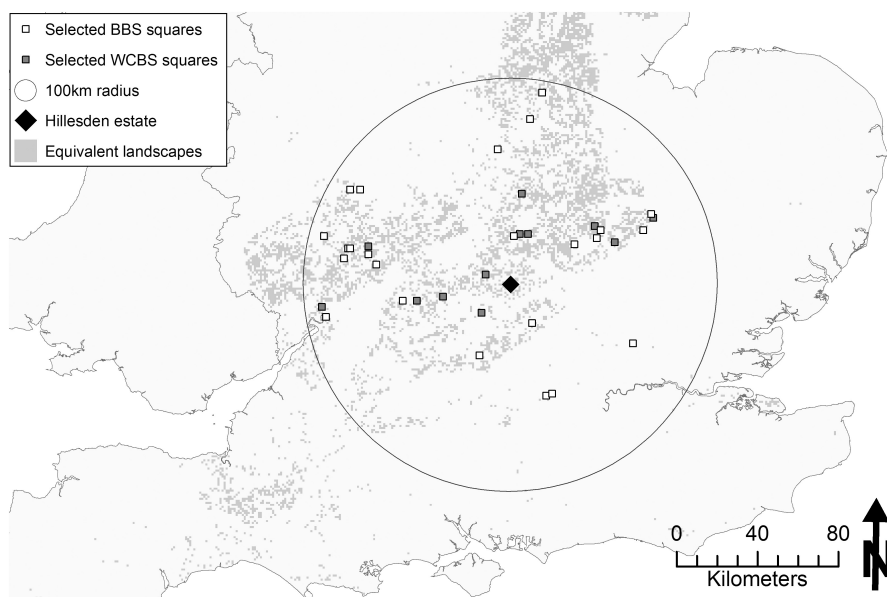
Birds were surveyed monthly in the breeding season (April–July) on ~1 km transects following field boundaries. Surveys took place within 4 hr after sunrise, with three transects visited per surveyor

per day. Surveyors and transect order were varied every month to avoid bias. All birds detected in or near the hedge and adjacent AES habitats (e.g. field margins) were recorded on ~1:2,000 scale maps, using standard methods to denote species and activity (Bibby et al., 1992). Transects were not surveyed during heavy rain or winds above Beaufort force 4. For further details of bird surveys see Hinsley et al. (2010) and Redhead et al. (2018). Data were filtered to adults recorded within 10 m of transects, excluding birds seen only in flight.

Butterflies were surveyed on a series of 2 × 50 m transects located within AES habitats, along with 2-min timed observations of a 2 × 2 m quadrat at either end of each transect. Surveys took place monthly May–August, when UK butterfly activity is greatest. Surveys took place between 10:00–16:00 GMT, with minimum temperatures of 13°C in sunny conditions (>60% clear sky), or 17°C if cloud cover was greater, with wind force Beaufort <4 and no precipitation. We analysed transects in all AES habitats (grass margins, wildflower patches, pollen and nectar margins, bird food patches), as all provide potential larval foodplants or floral resources from sown, incidental or weed species (Ouvrard & Jacquemart, 2018).

No licences, permits or specific permissions were required for fieldwork, other than landowner access permission. No ethical approval was required for the bird and butterfly observations at Hillesden.

For both groups, we filtered transects to identify those surveyed across all years (2006–2010 and 2012–2016 for birds, 2007–2010 and 2014–2017 for butterflies). We excluded transects with no AES intervention and those which underwent changes in level of AES uptake. This resulted in nine bird transects and seven butterfly transects for analyses. We analysed 12 bird and nine butterfly species (Table 1), selected on the basis of being present on the majority (>50%) of selected transects across all surveys. Species which were poorly detected by the field boundary transect method were excluded (e.g. carrion crow *Corvus corone*, common



**FIGURE 1** Location of the Hillesden Estate in southern United Kingdom. Also shown are 1 km × 1 km squares in equivalent landscapes, and squares in these landscapes covered by the Breeding Birds Survey (BBS) and Wider Countryside Butterfly Survey (WCBS), with sufficient data for analyses. The 100 km radius around the Hillesden Estate determined the limit of BBS and WCBS data used for analyses.

woodpigeon *Columba palumbus*). Individual species trends were combined (see below) to produce aggregate trends for: granivorous birds (directly targeted by AES habitats providing winter bird food, Table 1), all bird species combined, butterflies with grass-feeding larvae (particularly likely to benefit from increased resources under AES, Table 1) and all butterfly species combined.

### 2.3 | Bird and butterfly national survey data

Concurrent control data from equivalent landscapes to Hillesden but without AES were obtained from two national-scale monitoring schemes. For birds, we used the Breeding Birds Survey (BBS), coordinated by the British Trust for Ornithology (BTO), and funded by BTO, Joint Nature Conservation Committee (JNCC) and Royal Society for the Protection of Birds. For butterflies, we used the Wider Countryside Butterfly Survey (WCBS), organised and funded jointly by Butterfly Conservation, UK Centre for Ecology & Hydrology, BTO and JNCC. Both surveys are undertaken by volunteers ('citizen scientists'), visiting stratified random samples of 1 × 1 km Ordnance Survey grid squares ('squares' from here on), each containing two approximately parallel 1 km transects. BBS squares were visited twice each year, in the early (April–May) and late (May–June) breeding season. All birds detected by the observer were recorded (Field & Gregory, 1999). WCBS squares were visited at least twice, in July and August, with additional visits being permitted (Brereton et al., 2011). Transects on WCBS squares were surveyed at a continuous, steady pace, recording all butterflies seen in a 5 × 5 m box in front of the recorder (Pollard & Yates, 1993).

BBS data covered the full time span of Hillesden bird surveys, so analyses were undertaken across this range. However, WCBS was launched in 2009, so data were available for comparison with the latter part of the Hillesden butterfly surveys (2009–2017). We filtered BBS and WCBS squares to identify those which had been consistently surveyed (i.e. received the minimum two visits per year) from 2006 to 2016 (BBS) or 2009–2017 (WCBS). We further filtered BBS and WCBS squares to ensure that they represented equivalent landscapes to Hillesden but without substantial AES intervention (i.e. controls). Criteria used to determine equivalence were as follows:

1. Same land class as Hillesden (1e/11e, Bunce et al., 2007). Land classes define zones of similar environmental character based on multivariate analysis of environmental variables, including climatic, topographic and geological data.
2. Within 100 km of Hillesden to further ensure similar climate and species pools.
3. Not within 2 km of Hillesden, to reduce potential for movement between Hillesden and control squares.
4. Similar to Hillesden in land cover composition from CEH Land Cover Map 2007 (Morton et al., 2011). Criteria were >80% agricultural (arable+improved grassland), >50% arable, <5% woodland, <5% urban, as determined from 1 km squares within Hillesden.

5. No substantial AES intervention. Defined as <450 points under Environmental Stewardship, equivalent to 1 ha of seed mixture for birds or pollinators (Natural England, 2012, 2018) and thus below the minimum 1% land area allocated to AES at Hillesden. Although spatiotemporally precise data on the location of individual AES options are lacking, available data are sufficient for estimating relative uptake at the 1 km square level, especially for detecting squares where AES is absent (Staley et al., 2021).

Spatial analyses were undertaken in ArcMAP (v10.3 © ESRI). Applying selection criteria gave 24 BBS squares and 20 WCBS squares (Figure 1). Once squares had been selected, we compared landscape structure metrics (habitat diversity, number of patches of non-crop habitat and non-crop patch edge: area ratio) to ensure no systemic bias between Hillesden and control squares (Appendix SF1).

### 2.4 | Analysis and statistical methods for estimating trends

Total abundance per year was calculated as maximum count across surveys per transect/square. We excluded July Hillesden bird surveys, as these fall outside the date range permissible for BBS data. For butterflies, all Hillesden visits overlap the range of permissible WCBS visits (~30% of WCBS squares had at least one visit outside the core July–August range).

Population trends over the 10 years were calculated for each dataset by estimating annual growth rates from a generalised linear model (GLM). By focusing on interannual change, differences in the survey methodology between Hillesden and control squares which would affect estimates of absolute abundance (e.g. length of transects, number of visits) become irrelevant provided they are consistent over time. This method also reduces the impact of any difference in starting conditions (i.e. levels of biodiversity) between Hillesden and control squares in equivalent landscapes which would bias direct analyses of abundance in the absence of baseline monitoring prior to AES implementation (Josefsson et al., 2020).

We used the loglinear method developed by Freeman and Newson (2008), previously applied to BBS data by Baker et al. (2012) and WCBS data by Roy et al. (2015) and Dennis et al. (2017). This assumes that annual proportional change in abundance is constant across sites but varies over time, such that the annual rate of growth,  $R_t$  is given by:

$$R_t = \log\left(\frac{u_{i,t+1}}{u_{i,t}}\right),$$

where  $u_{i,t}$  is the expected total count of a species in year  $t$  at site  $i$ . This equation is intractable to fit directly as the ratios of observed counts are potentially non-integer and undefined whenever the associated count is zero. However, following algebraic rearrangement (Freeman & Newson, 2008), this can be expressed for any time  $t$  as:

$$\log(u_{i,t}) = \sum_{j=1}^{t-1} R_j + \log(u_{i,1}). \quad (1)$$

Group	Species	Two-letter code	Scientific name	Hillesden abundance
Birds	Chaffinch <sup>G</sup>	CH	<i>Fringilla coelebs</i>	879
	Blackbird	B.	<i>Turdus merula</i>	462
	Yellowhammer <sup>G</sup>	Y.	<i>Emberiza citrinella</i>	462
	Whitethroat	WH	<i>Sylvia communis</i>	423
	Blue tit	BT	<i>Cyanistes caeruleus</i>	354
	Dunnock	D.	<i>Prunella modularis</i>	323
	Great tit	GT	<i>Parus major</i>	302
	Linnet <sup>G</sup>	LI	<i>Carduelis cannabina</i>	299
	Wren	WR	<i>Troglodytes troglodytes</i>	228
	Robin	R.	<i>Erithacus rubecula</i>	189
	Reed bunting <sup>G</sup>	RB	<i>Emberiza schoeniclus</i>	120
	Song thrush	ST	<i>Turdus philomelos</i>	100
	Butterflies	Large white	LW	<i>Pieris brassicae</i>
Meadow brown <sup>G</sup>		MB	<i>Maniola jurtina</i>	167
Small white		SW	<i>Pieris rapae</i>	141
Green-veined white		GW	<i>Pieris napi</i>	66
Gatekeeper <sup>G</sup>		GK	<i>Pyronia tithonus</i>	58
Common blue		CB	<i>Polyommatus icarus</i>	53
Marbled white <sup>G</sup>		MW	<i>Melanargia galathea</i>	45
Small tortoiseshell		ST	<i>Aglais urticae</i>	42
Ringlet <sup>G</sup>		RI	<i>Aphantopus hyperantus</i>	38

**TABLE 1** Bird and butterfly species for which trends in abundance were assessed from Hillesden and national survey datasets. Species codes are given for use in subsequent plots. Granivorous birds and butterflies with grass-feeding larvae are denoted by <sup>G</sup>. Rows are ordered by decreasing abundance within groups, abundance being calculated as the sum of maximum annual count across all years for the selected transects

We can, thus, estimate the log annual growth rate by fitting this model to observed counts via standard GLM methods. GLMs used a Poisson distribution for birds, and a quasi-Poisson adjustment for overdispersion for butterflies because butterfly counts showed a higher frequency of zero values. Missing and zero counts can be included, and standard errors can be used to assess the significance of growth rates by comparing overlap of 95% confidence intervals with zero. We summed growth rates  $R_t$  across the entire time series ( $T$ ) to obtain an estimate of net change ( $N$ ) and associated confidence intervals:

$$N = \sum_{t=1}^{T-1} R_t; \text{var}(N) = \sum_{t=1}^{T-1} \text{var}(R_t) + 2 \sum_{t=1}^{T-1} \sum_{k=1}^{t-1} [\text{cov}(R_t, R_k)]. \quad (2)$$

To compare between trends we calculated the difference as  $d = N_{\text{Hillesden}} - N_{\text{controls}}$ , along with 95% confidence derived from  $SE_d = \sqrt{(SE_{\text{Hillesden}}^2 + SE_{\text{controls}}^2)}$ . Where confidence limits included zero, we assume no significant difference between population trends.

Annual growth for missing years in Hillesden data, caused by the gap between the two phases of the experiment (2011 for birds and 2012 for butterflies), cannot be estimated separately (i.e. models produce a single estimate of aggregate change across periods of non-surveyed years). To obtain comparable figures from BBS and

WCBS data we calculated net change across 2010–2012 (birds) or 2011–2013 (butterflies) using Equation 2. All trends were estimated in R (v3.4.0 R Core Team, 2017).

## 3 | RESULTS

### 3.1 | Trends in bird populations

For all bird species which showed a significant net change at Hillesden, these trends were positive (Table 2), with those for blue tit, linnet, all species combined and granivorous species being statistically significant (i.e. 95% confidence intervals did not overlap zero). Comparing trends between Hillesden and equivalent BBS control squares, the trend at Hillesden was more strongly positive or less strongly negative than the trend in equivalent BBS squares for all bird species except dunnock, robin and song thrush (Figure 2a). Although several species showed significant positive trends in BBS control squares (blackbird, linnet, robin, song thrush, wren and all species combined), there were also two significant negative trends (chaffinch and yellowhammer) where no significant trend was observed at Hillesden. Calculating the difference between the trends

at Hillesden and those in equivalent BBS control squares showed significant differences for chaffinch, blue tit, great tit and granivorous species combined, all in favour of Hillesden.

Analysing interannual growth rates allowed us to explore temporal patterns of change, to examine whether net change occurred

**TABLE 2** Net change  $N (\pm SE)$  2006–2016 for birds, from Hillesden data, equivalent BBS squares and the differences between the two (Hillesden – BBS, such that positive values indicate greater growth at Hillesden). Statistically significant trends or differences (i.e. 95% confidence intervals do not overlap zero) are indicated by \*

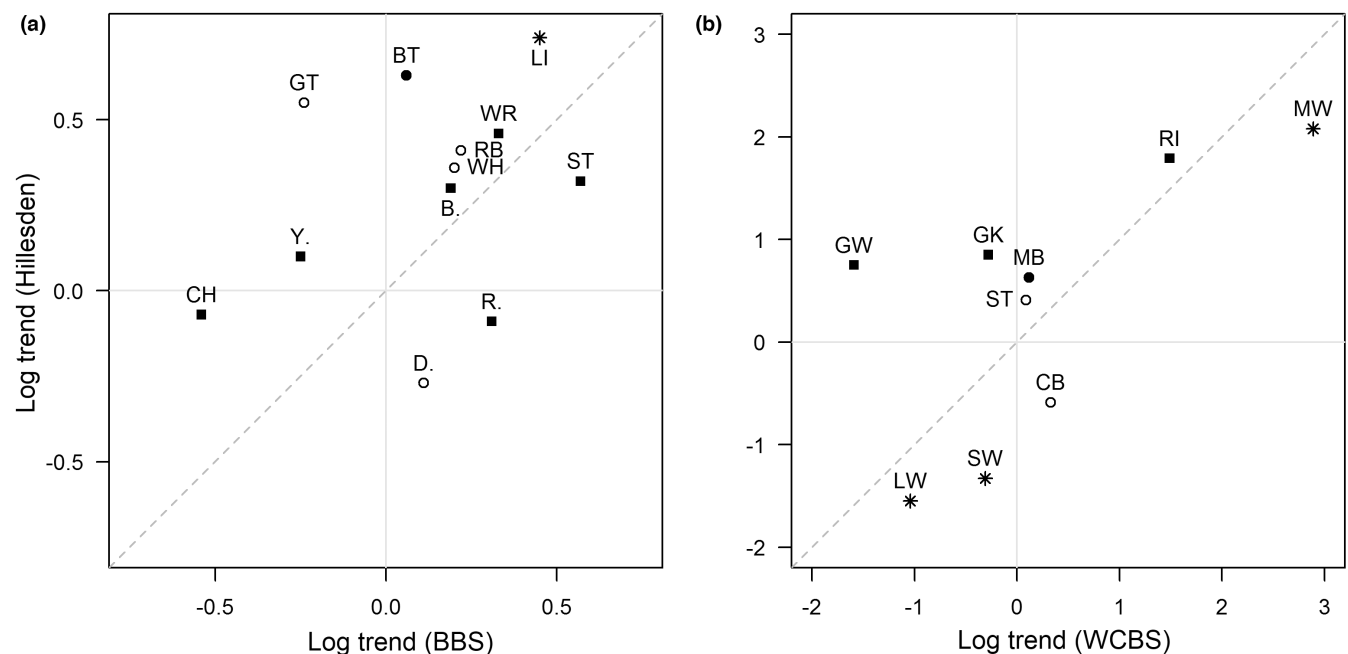
Species	Hillesden net change ( $\pm SE$ )	BBS net change ( $\pm SE$ )	Difference ( $\pm SE$ )
Blackbird	0.30 (0.24)	0.19 (0.09)*	0.11 (0.25)
Blue tit	0.63 (0.25)*	0.06 (0.11)	0.57 (0.28)*
Chaffinch	-0.07 (0.16)	-0.54 (0.10)*	0.48 (0.19)*
Dunnock	-0.27 (0.26)	0.11 (0.14)	-0.38 (0.29)
Great tit	0.55 (0.29)	-0.24 (0.13)	0.79 (0.32)*
Linnet	0.74 (0.28)*	0.45 (0.15)*	0.30 (0.32)
Reed bunting	0.41 (0.37)	0.22 (0.25)	0.18 (0.45)
Robin	-0.09 (0.30)	0.31 (0.12)*	-0.40 (0.32)
Song thrush	0.32 (0.46)	0.57 (0.19)*	-0.25 (0.50)
Whitethroat	0.36 (0.24)	0.20 (0.14)	0.16 (0.28)
Wren	0.46 (0.27)	0.33 (0.10)*	0.14 (0.29)
Yellowhammer	0.10 (0.22)	-0.25 (0.12)*	0.35 (0.25)
All species	0.29 (0.08)*	0.12 (0.04)*	0.17 (0.09)
Granivores	0.29 (0.13)*	-0.03 (0.08)	0.33 (0.16)*

uniformly or in specific portions of the time series. Variation in interannual growth rates was wide (Appendix ST1 and ST2, see also plots in Appendix SF2), but the greatest number of species showing a significant, positive difference in growth rates between Hillesden and national survey data occurred between 2006 and 2007 for birds (7 species), followed by 2008–2009 (4 species) while no significant, positive differences were observed in the final time step (2015–2016), potentially suggesting a levelling off of growth (Table 4).

### 3.2 | Trends in butterfly populations

The majority of butterfly species showed positive net change at Hillesden (Table 3; Figure 2b), with statistically significant increases for marbled white and meadow brown, and for species with grass-feeding larvae combined. Five of the nine species showed trends at Hillesden that were more strongly positive or less strongly negative than the trend in equivalent WCBS squares. However, the only species for which this difference was significant were gatekeeper and green-veined white, both of which declined significantly on WCBS sites but had positive trends at Hillesden. Two species (large white and small white), which are pests associated with brassica crops, decreased significantly at Hillesden. Significant decreases in these two species were also evident in WCBS squares, but in the case of small white the decrease at Hillesden was significantly stronger.

There was no obvious pattern in interannual growth rates for butterflies (Table 4), suggesting that where the overall growth rate across years was significantly positive, this was a result of incremental growth across the time series rather than single years of high growth.



**FIGURE 2** Scatterplots of overall change across all years from Hillesden data (y-axes) and equivalent national survey data (x-axes), for (a) birds and (b) butterflies (points labelled with species codes in Table 1). For both plots, the dashed line indicates equality. Filled circles indicate statistical significance in the trend from Hillesden data only, filled squares indicate statistical significance from national survey data only, asterisks indicate significance from both datasets. Open circles indicate that neither trend was statistically significant.

## 4 | DISCUSSION

### 4.1 | Trends in bird populations

The result for combined granivorous species (chaffinch, linnnet, reed bunting, yellowhammer) offers support to the hypothesis that

**TABLE 3** Net change  $N$  ( $\pm$ SE) 2009–2017 for butterflies, from Hillesden data, equivalent WCBS squares and the differences between the two (Hillesden – WCBS, such that positive values indicate greater growth at Hillesden). Statistically significant trends or differences (i.e. 95% confidence intervals do not overlap zero) are indicated by \*

Species	Hillesden net change ( $\pm$ SE)	WCBS net change ( $\pm$ SE)	Difference ( $\pm$ SE)
Common blue	-0.59 (0.56)	0.33 (0.19)	-0.92 (0.59)
Gatekeeper	0.85 (0.49)	-0.28 (0.10)*	1.12 (0.50)*
Green-veined white	0.75 (0.43)	-1.59 (0.16)*	2.34 (0.46)*
Large white	-1.55 (0.35)*	-1.04 (0.12)*	-0.50 (0.37)
Marbled white	2.08 (1.06)*	2.89 (0.30)*	-0.81 (1.10)
Meadow brown	0.63 (0.31)*	0.12 (0.07)	0.50 (0.32)
Ringlet	1.79 (1.08)	1.49 (0.11)*	0.30 (1.09)
Small tortoiseshell	0.41 (0.65)	0.09 (0.19)	0.31 (0.67)
Small white	-1.33 (0.37)*	-0.31 (0.09)*	-1.02 (0.38)*
Grass feeders	1.34 (0.41)*	1.06 (0.08)*	0.28 (0.41)
All species	0.34 (0.22)	0.19 (0.05)*	0.15 (0.22)

**TABLE 4** Numbers of species/groups showing significant positive and significant negative growth rates over each time step, in the Hillesden data and national survey data from equivalent control landscapes. The number of significant differences between the two trends are also shown, indicating whether the trend at Hillesden was higher (i.e. more positive) or lower (i.e. more negative)

Group	Time step	Hillesden		Equivalent control landscapes		Difference	
		Positive	Negative	Positive	Negative	Higher	Lower
Birds	2006–2007	3	0	0	5	7	0
	2007–2008	0	6	2	1	1	4
	2008–2009	4	0	3	2	4	0
	2009–2010	0	5	1	0	0	4
	2010–2012	0	1	1	2	3	0
	2012–2013	3	0	5	1	3	1
	2013–2014	0	5	1	1	0	1
	2014–2015	4	0	0	0	2	0
	2015–2016	0	1	0	0	0	0
	All years	4	0	6	2	4	0
Butterflies	2009–2010	0	1	1	2	0	1
	2010–2014	0	0	8	2	1	0
	2014–2015	1	0	1	1	0	0
	2015–2016	0	4	0	6	1	1
	2016–2017	4	0	5	0	1	0
	All years	3	2	4	4	2	1

positive responses in abundance to local levels of AES, observed by previous studies within the Hillesden landscape (Hinsley et al., 2010; Redhead et al., 2018), indicate long-term population increases, and that these are significantly different from population trends in equivalent control landscapes without AES. However, although individual granivorous species generally showed positive or stable trends at Hillesden, the extent to which these differed from trends in control landscapes was variable.

For chaffinch, the trend at Hillesden was near zero, but this was differed significantly from BBS squares which showed significant declines. National-level declines in chaffinches over the study period have been associated with an emerging infectious disease (Lawson et al., 2012). It is plausible that AES food resources might reduce use of garden feeders, a key route of infection, or provide sufficient benefit to offset disease impacts and stabilise local populations. Chaffinches were highly abundant at Hillesden, so this species may have had limited capacity to increase its local population further, with birds feeding on AES interventions dispersing to breed (Redhead et al., 2018).

Trends for yellowhammer were positive at Hillesden and significantly negative in equivalent BBS squares. Yellowhammers showed higher interannual growth rates in early years of the study, suggesting initial population growth associated with the introduction of AES, followed by levelling off, as also observed by Colhoun et al. (2017) and Walker et al. (2018). Yellowhammers are highly mobile and have large territory sizes (Andrew, 1956), which may result in rapid saturation of available local breeding habitat, perhaps contributing to the difference between the two trends being non-significant.

Trends for linnet and reed bunting at Hillesden were more strongly positive than those in equivalent BBS squares, but not significantly so, despite high use of AES resources in previous studies (Hinsley et al., 2010; Redhead et al., 2018). Linnets are partially migratory and gregarious throughout the year (Drachmann et al., 2000), potentially allowing them to benefit from scattered resources (Redhead et al., 2018) and increase local populations with less requirement for additional breeding habitat. Thus, significant positive trends in both datasets may signal that this species has benefited from wider-scale AES uptake (Baker et al., 2012). Reed buntings were particularly variable in interannual growth rates in both datasets. This may be because this species nests in taller crops (Gruar et al., 2006), the amount of which in a given landscape varies year-to-year, affecting abundance and detectability.

This variability for the individual granivore species suggests that while the well-known positive responses of local winter abundance of widespread granivores to AES (Hinsley et al., 2010; Perkins et al., 2008; Redhead et al., 2018) may indicate a capacity to stabilise local populations, actually reversing declines may be contingent on a wider range of factors including provision of suitable breeding habitat (Bradbury et al., 2000; Whittingham et al., 2005).

Although the widespread granivorous species are those targeted by AES wild bird seed mixtures, other species of the wider countryside may also benefit from increased resource provision under AES. Blue tits and great tits showed significantly more positive trends at Hillesden than equivalent BBS squares. Although not the most obvious beneficiaries of AES, being insectivores associated with woodland, these species previously showed increased breeding success at Hillesden where AES increased local foraging habitat (Redhead et al., 2013).

## 4.2 | Trends in butterfly populations

As with birds, our results suggested that, for a few species, the significantly increased butterfly abundance in AES habitats observed at the field or farm scale (Field et al., 2005; Field et al., 2007; Meek et al., 2002; Pywell et al., 2004) may contribute to positive trends in local populations over longer timescales, which differ from the trends seen in equivalent landscapes without AES. Consistent AES impacts on butterfly populations over longer temporal scales are hard to detect (Brereton et al., 2002; Roth et al., 2008; Taylor & Morecroft, 2009), perhaps even more so than birds since butterfly abundance, detectability and phenology are even more strongly influenced by interannual variations in weather.

Gatekeeper and green-veined white showed positive trends at Hillesden which were significantly different from the declines in equivalent WCBS squares (Table 3). Gatekeepers have grass-feeding larvae, are strongly associated with hedgerows and field margins and can respond rapidly to AES implementation (Field et al., 2005; Field et al., 2007; Field & Mason, 2005), showing continued increases even after several years of AES management (Field et al., 2007). Other

butterflies with grass-feeding larvae (meadow brown, marbled white and ringlet) also showed significant positive trends at Hillesden, although not significantly more so than in WCBS data. Green-veined whites are also likely to benefit from increased larval foodplants, including incidental species which establish rapidly in AES habitats (e.g. charlock *Sinapis arvensis*) and those sown in bird food mixes (e.g. fodder radish *Raphanus raphanistrum*). This is in marked contrast to large and small white, which both prefer cultivated brassicas (including oilseed rape *Brassica napus*) as larval foodplants, and which declined significantly in both datasets. The proportion of Hillesden transects adjacent to oilseed rape varied between 17% and 56% (mean 36%), but there was no significant trend within the study period, nor did inter-annual trends in small and large white correspond to inter-annual differences in the amount of oilseed rape. National scale declines in these species have previously been attributed to pesticide exposure from feeding on crop species (Braak et al., 2018; Gilburn et al., 2015). However, AES interventions at Hillesden did not alter the conventional pesticide regime and both species have been observed to decline after introduction of AES even when this coincided with reduced pesticide usage (Taylor & Morecroft, 2009). Mechanisms may include increases in birds, invertebrate predators and parasitoids (Holland et al., 2012; Holland et al., 2014; Woodcock et al., 2010), without the compensatory effect of increased larval foodplants that benefit other, non-pest butterfly species.

## 4.3 | Limitations of the study

In common with many assessments of AES impacts, our study design falls short of the experimental ideal, in its comparison of a single AES landscape with controls (albeit one representative of conditions common across lowland England) and in a lack of a full before/after comparison (Josefsson et al., 2020). Our focus on trends rather than absolute differences in abundance reduces the impact of any bias arising from differences in initial populations between Hillesden and controls, while our selection of control landscapes that are as closely equivalent to Hillesden as possible should help to reduce the potential for such biases occurring. However, we cannot say whether our findings are representative outside the Hillesden landscape nor can we test directly the mechanism between AES and population changes.

Ideally, experiments would use replicated landscapes with accurately mapped, well-implemented AES, compared with paired controls and monitored over timespans allowing establishment of a robust pre-implementation baseline and stabilisation of the post-implementation response. However, the logistical challenges involved in doing so means that such studies are seldom feasible to fund and implement. Even the current study was only possible because a limited number of transects remained consistent across two 5-year experiments fortunate enough to run consecutively in the same study landscape. Although attempts have been made to construct robust monitoring frameworks across gradients of AES uptake (Staley et al., 2021), these remain limited in their ability to



explore temporal change. The fact that few other studies present data over similar timespans, and that there has been no apparent general improvement in the standard of AES study design in the last decade (Josefsson et al., 2020), suggests this trade-off between feasibility and experimental rigour has yet to be resolved.

By analysing only species present on the majority of transects across all visits (a measure to ensure we had sufficient data for robust analyses) we effectively limited the focus of our analyses to 'core' species (Supp et al., 2015). Although these include the widespread farmland species at which most widely applicable AES options are targeted, they exclude others of conservation concern, which occurred at low numbers precluding analysis (e.g. tree sparrow *Passer montanus*). They also exclude 'transient' species (Supp et al., 2015), which are widespread but occur at a given site only infrequently, perhaps in association with particular crops or conditions (e.g. yellow wagtails *Motacilla flava* were generally present at Hillesden, but not sufficiently consistently on any given transect to calculate robust trends). Even for species which are routinely present in the breeding season, migratory habits will affect population responses to AES, with factors in other parts of their range exacerbating or masking the effect of agricultural land management. The only fully migratory species in our study was whitethroat, which showed positive changes in both datasets, in keeping with the species' long-term recovery from mid-twentieth century declines driven by droughts on its wintering grounds (Baillie & Peach, 1992).

#### 4.4 | Applications for agri-environment implementation and policy

With these limitations in mind, the most conservative interpretation of our results is as a demonstration of the potential for well-implemented examples of widely applicable AES options within a typical farmland landscape to be associated with beneficial longer-term differences in the population trends of a range of widespread farmland species. Thus, positive impacts of AES at local scales may be indicative of the ability of widely applicable AES options to at least buffer against declines in local populations, when applied at the levels and consistently high quality achieved at Hillesden. Achieving this at a national scale would require improvements in the training and tools available to farmers (McCracken et al., 2015), and in subsequent monitoring. The constraints on interpretation imposed by our study design also demonstrate the importance of in-depth, long-term monitoring experiments for drawing robust conclusions on the effects of AES, and the critical need for accurate and accessible data on AES uptake and quality if we are to be able to use the valuable data gathered by national biodiversity recording schemes in the context of AES. New AES in England propose to emphasise delivery over mere uptake (Defra, 2018), and novel technologies are being developed which may help to monitor AES implementation and outcomes, so it is to be hoped that these challenges may yet be overcome.

#### AUTHOR CONTRIBUTIONS

Richard F. Pywell and Matthew S. Heard conceived the experimental setup and monitoring; Marek Nowakowski advised on experimental design and management; Shelley A. Hinsley, Richard K. Broughton and Paul E. Bellamy and Marc S. Botham coordinated collection of field data; Gavin Siriwardena and Zoë Randle provided national survey data; Stephen N. Freeman and John W. Redhead analysed the data and John W. Redhead led the writing of the manuscript. All authors contributed critically to drafts and gave final approval for publication.

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#### CONFLICT OF INTEREST

The authors have no conflict of interest to declare.

#### DATA AVAILABILITY STATEMENT

BBS and WCBS data are held by the British Trust for Ornithology and Butterfly Conservation, respectively and are available on request. Data used for spatial analysis and site selection are available online (see references in text). Hillesden bird and butterfly data from the consistently surveyed transects are available via the NERC Environmental Data Information Centre (EIDC) <https://doi.org/10.5285/c8d0ac67-6d47-42b1-af29-2339be8a5d82> (Redhead et al., 2022).

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## SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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