England's green and pleasant land: comparing land sparing and sharing in the UK

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Summary

Globally, agriculture is the leading threat to biodiversity. For many regions of the world, empirical evidence suggests that most species would fare least badly if food demand was met through high-yield production linked to the sparing of non-farmed habitats ('land sparing'), rather than producing both food and wildlife in a larger area of wildlife-friendly farmland ('land sharing'). The UK, however, may be different: several priority species are associated with sympathetically managed farmland, whilst many sensitive, land-sparing-preferring species may already have been lost. We therefore explored the trade-offs and synergies between food production and wildlife conservation for two regions of lowland England. We quantified the relationship between agricultural production and abundance of individual bird species, in order to evaluate various landscape-level food production strategies, including land sparing and land sharing.

Key words: Agriculture, birds, conservation, ecosystem services, land sparing, landscape scale

Introduction

Agriculture is the leading threat to biodiversity: it catalyses the conversion of natural habitat (Geist & Lambin, 2002), and the increasingly intense management of agricultural landscapes drives the decline of farmland wildlife (Donald *et al*., 2006). As human population size and consumption rates increase, it seems likely (much-needed reductions in waste and over-consumption (Alexander *et al*., 2016) notwithstanding) that global food production will have to rise too (Tilman *et al*., 2002). How can we meet future food production needs – whether higher, equal to, or lower than current production – whilst minimising biodiversity loss?

Broadly, there are two opposing strategies we might employ to examine the (generally competing) interests of food production and wildlife conservation. '*Land sharing*' involves managing the food production landscape in a way which is sympathetic to wildlife, promoting on-farm biodiversity. However, to achieve the same overall level of food production, the lower yields which are typical of wildlife-friendly farming require a larger area of farmland, thus reducing the potential area of natural habitat. Alternatively, '*land sparing*' involves maximising the productivity of farmed land so that land elsewhere can be protected from agricultural expansion or restored to natural habitat (Balmford *et al*., 2015).

The shape of the relationship between site-level food production yield and the population density of a species (the 'density-yield curve') indicates whether its total population size would be maximised (given a landscape-wide food production target) through land sparing, land sharing or an intermediate strategy (Green *et al.*, 2005). With remarkable consistency, every empirical study to date suggests that a majority of species (from birds in Kazakhstan and Uganda to trees in Ghana and India to dung beetles in Mexico) would do least badly under land sparing (Dotta *et al.*, 2016; Williams *et al.*, 2017; Hulme *et al.*, 2013; Phalan *et al.*, 2011; Kamp *et al.*, 2015). That is, species abundance declines rapidly as soon as natural habitat is converted to farmland, such that even low-yielding, wildlife-friendly farming supports few individuals of most species compared to natural habitat. Because, for a given level of food production, land sparing maximises the extent of natural habitat, it is therefore favoured by a high proportion of species.

To date, however, no study has fitted density-yield curves for wildlife species in the UK. The long history of disturbance from agriculture and glaciation, and the small remnant stock of natural habitat, may have filtered out many of the land-sparing-preferring species most sensitive to agriculture (Balmford, 1996), leading to suggestions that land sharing might be a better strategy in temperate regions (Ramankutty & Rhemtulla, 2012). Furthermore, many priority species in the UK are associated with sympathetically managed farmed landscapes, and are threatened by the intensification of agriculture (Wilson *et al.*, 2009; Hayhow *et al.*, 2016).

Here, for two regions of lowland England, we estimate species population density and agricultural yield for a suite of *c*. 1 km² sites spanning a continuum of land-use intensity. We then fit density-yield curves for all assessable breeding bird species, and present some illustrative examples.

Materials and Methods

Study landscapes
We focus on two lowland agricultural landscapes (Fig. 1). *The Fens* National Character Area is a flat, low-lying landscape dominated by arable farmland and dependent on drainage. *Salisbury Plain and the West Wiltshire Downs* National Character Area is characterised by rolling chalk downland and mixed farmland.

Our focal landscapes initially included all 1 km × 1 km cells within the boundaries of each National Character Area. We used the *NATMAP Soilscapes* vector layer to exclude cells with less than 50% cover of 'peaty' soil in The Fens and 'limey' soil in Salisbury Plain, leaving a total area of 1228 km² in The Fens and 1026 km² in Salisbury Plain.

Study sites
Within each focal landscape, we conducted detailed surveys at sites covering the full spectrum of land-use intensity. We first selected all 1×1 km grid squares in each landscape surveyed as part of the national Breeding Bird Survey (Harris *et al.*, 2017; 25 and 36 sites in The Fens and Salisbury Plain, respectively). Equivalent bird survey data were available for an additional 148 1 × 1 km grid squares within the Salisbury Plain Site of Special Scientific Interest, as well as two National Nature Reserves in The Fens. Finally, we selected a further 18 sites in The Fens (four nature reserves, 10 'wildlife-friendly' farms and four intensive arable farms) and four 'wildlife-friendly' farms in Salisbury Plain for additional surveys in 2016–17. Sites were categorised as 'natural habitat', 'farmland', or 'urban' based on the dominant *CEH Land Cover Map 2015 (LCM)* (Rowland *et al.*, 2017) land cover type.

After excluding urban sites and those with bird surveys carried out in only 1 year, there were 42 and 172 sites in The Fens and Salisbury Plain, respectively. Most of our survey sites were 'standard' 1 km × 1 km BBS squares, but 10 were displaced from the national grid squares and nine were irregularly shaped sites, though all employed the same BBS line transect field survey method (see below).

Fig. 1. Study regions. Axis tick marks are 100 km apart.

Bird density data

Species selection
Using the *British Breeding Bird Atlas 2007–11* (Balmer *et al.*, 2014) we identified all species with apparent breeding populations in either The Fens or Salisbury Plain. We excluded aerial foragers and nocturnal species for which densities are difficult to estimate, as well as gulls and terns which have large foraging ranges but are unlikely to have breeding populations at any of our study sites (with the exception of Black-headed Gulls *Chroicocephalus ridibundus* and Common Terns *Sterna hirunda* which were retained for natural habitat sites in The Fens). We also excluded introduced species.

BBS data
In the UK, the Breeding Bird Survey is coordinated by the British Trust for Ornithology and conducted by volunteers (Harris *et al.*, 2017). Randomly selected 1 km × 1 km grid squares are surveyed along a 2 km transect once between early April and mid-May (early visit) and again between mid-May and late June (late visit). Broad habitat categories are recorded for each 200 m transect sections, and birds are assigned to one of two distance bands (0–25 m, 25–100 m). This standard methodology was followed at all sites, though at six irregular sites the transects were longer than 2 km.

Data analysis followed two steps: (1) estimating site-, visit- and species-specific detection probabilities and (2) estimating site- and species- specific densities. For step 1, analysis was conducted at the 200 m transect level, using the number of individuals detected in each distance band. We supplemented data from our study sites with BBS data from all squares south of 54°N to estimate the half-normal detection function for each species, with broad habitat type and visit

(early or late) as a co-variate. The integration of this function is the estimated probability of detecting a species within 100 m of the transect line. Site-specific species detection probabilities were calculated by weighting the habitat-specific detection probability by the proportion of each habitat type within a site.

For step 2, we used generalised linear mixed models with Poisson error structure and log link function. For each species and region, we modelled the maximum count (from the early and late visits) with an offset of the logarithm of the site- and visit-specific 'effective area' (the product of species detection probability, transect width [2 × 100 m = 0.2 km] and total transect length [usually 2 km]). We fitted the additive effects of site and year (both fixed factors), and model predictions for 2013–16 were averaged to estimate mean site-specific species densities. This approach thus accounts for varying survey years by controlling for inter-annual variation in density at the regional level. To increase the power of these regional 'year' effects, we supplemented data from our study sites with BBS data from all squares within the *East of England* region (for The Fens) and the *South West* region (for Salisbury Plain). Density estimates were available for 100 and 82 species in The Fens and Salisbury Plain, respectively.

Food production data

For each study site, we estimated total annual food production in two currencies; GJ food energy per ha per year and tonnes of edible protein ha yr^{-1}. Aggregate food production is a function of (1) the area and (2) the yield of agricultural land-uses, which were estimated using various methods.

Land-use areas

Land-use areas were derived remotely from two sources. We relied on the more specific but less comprehensive *CEH Land Cover plus: crops* (*crops*) data where possible, using the less specific *LCM* categories only for parcels for which no *crops* data were available. These 'raw' land-use areas were thus calculable for the entire study area.

These satellite-derived data sources do not account for narrow strips and corners of uncropped land, so are likely to over-estimate the total area of agricultural land. For a subset of sites, we manually digitised all fields using aerial photography, and clipped the 'raw' land-use areas to exclude any uncropped boundary features and corners ('clipped areas').

The satellite-derived data sources were only available for a single cropping season (2015) and do not provide exhaustive fine-scale land-use categories. For a subset of sites, we therefore conducted surveys with farmers to accurately map the land-use (e.g. winter wheat, potatoes, permanent pasture, etc.) of each field during 2013–16 ('surveyed areas').

Land-use yields

Each mapped land-use corresponds to one or more arable crops types. For example, 'winter wheat' in the *crops* data includes both winter wheat and winter oats, and 'arable and horticulture' in the *LCM* data includes all arable crops. The yield of every arable crop (tonnes per ha per year),) as well as the relative area of multiple crops belonging to the same land-use category, were estimated remotely using data from the Farm Business Survey (Duchy College. Rural Business School 2017). Farm Business Survey data were available for the period 2012–15 from 59 farms in Salisbury Plain and 55 in The Fens. We also collected data on crop yields from farmer surveys for a subset of sites, for which the relationship between crop and land-use category was generally one-to-one.

For grazed land-uses (improved grassland, calcareous grassland, neutral grassland, acid grassland, fen marsh and heather grassland) we estimated the typical yield of dry matter grass production (Tallowin & Jefferson, 1999), and converted this into edible animal products using standard fodder conversion ratios (Wilkinson, 2011).

We then converted each tonne of harvested product into energy and protein currencies, replicating the processing of harvested goods into multiple end products. Cereal grains, for example, were partitioned into edible grain products as well as meat via animal feed. We used the same conversion process for remote-estimated as well as survey-derived yields; we did not distinguish between cereals grown for animal feed *vs* human consumption, but instead applied a regional estimate of the ratio between these two end-uses. Energy and protein values of final edible products were taken from the USDA Food Composition Database (US Department of Agriculture, 2015).

Yield estimates were therefore available for a subset of sites based on surveyed areas and surveyed yields, a subset of sites based on clipped areas and remote yields, and the entire landscape based on raw areas and remote yields. In addition, we predicted clipped yields for the entire landscape based on the linear relationship between raw and clipped yields (see *Results*).

Density yield curves

We fitted density-yield curves following Phalan *et al.* (2011), testing two model structures (giving rise to a wide range of curve shapes) for each species:

(A) $d = \exp(b_0 + b_1(x^\alpha))$

(B) $d = \exp(b_0 + b_1(x^\alpha) + b_2(x^{2\alpha}))$

Where d is the density of a species in a site, x is the 'clipped' yield of a site, and b_0 (intercept), b_1, b_2 (coefficients) and α (exponent) are parameters estimated from the data. Parameters were estimated by Maximum-Likelihood using Nelder-Mead numerical optimisation. The value of α was constrained to be between 0 and 4.6, and all model parameters were constrained such that the maximum predicted density did not exceed $1.5 \times$ the maximum observed density. AICc values were calculated for models A and B, and the model with the lowest value was used.

Results

Data collection and analysis is still ongoing, so the following sections represent preliminary results based on partial data.

Yields

There was a strong correlation between energy and protein yields derived from 'raw' land-use areas in both The Fens (Pearson correlation, r = 0.968, d.f. = 41, $P>0.001$) and Salisbury Plain (r = 0.991, d.f. =174, $P>0.001$), so for the remainder of this paper we focus only on energy yields.

The correlation between energy yield estimates based on 'raw' and 'clipped' land-use areas was also very strong (Pearson correlation, The Fens r = 0.995, d.f. = 31, $P<0.001$; Salisbury Plain r = 0.995, d.f. = 24, $P<0.001$), but as expected 'clipped' yields were consistently lower (average difference = 19% in The Fens, 11% in Salisbury Plain). The slope of the linear relationship between clipped (dependent variable) and raw (independent variable) yields was less than one (The Fens β = 0.91 ± 0.02; Salisbury Plain β = 0.90 ± 0.02), such that difference between the two estimates increased as yield increased. This relationship allows us to predict clipped yields for sites at which field boundaries have not been digitised.

Additionally, there was good correspondence between clipped energy yields and those derived from farm surveys (Pearson correlation, The Fens r = 0.945, d.f. = 23, $P<0.001$; Salisbury Plain r = 0.863, d.f. = 15, $P<0.001$). Encouragingly, the slope of this relationship was not significantly different from 1, and the intercept was not significantly different from 0 (The Fens β = 1.00 ± 0.07, α = –0.83 ± 3.62; Salisbury Plain β = 1.03 ± 0.16, α = –0.64 ± 2.35; Fig. 2). We are therefore confident in using remotely calculated clipped yield as a reliable estimate of true food production yield at the 1 km^2 scale.

Density-yield curves

A range of curve-shapes described the relationship between species population density and agricultural yield, as illustrated in Fig. 2 for four species in The Fens and Salisbury Plain. Species such as Blackcap *Sylvia atricapilla* in The Fens and Skylark *Alauda arvensis* in Salisbury Plain show a negative-decreasing curve, reaching peak density in natural habitat (fenland or chalk grassland, respectively) and declining in abundance with increasing agricultural yield. They will achieve maximum population size under a land sparing strategy, which maximise the area of natural habitat.
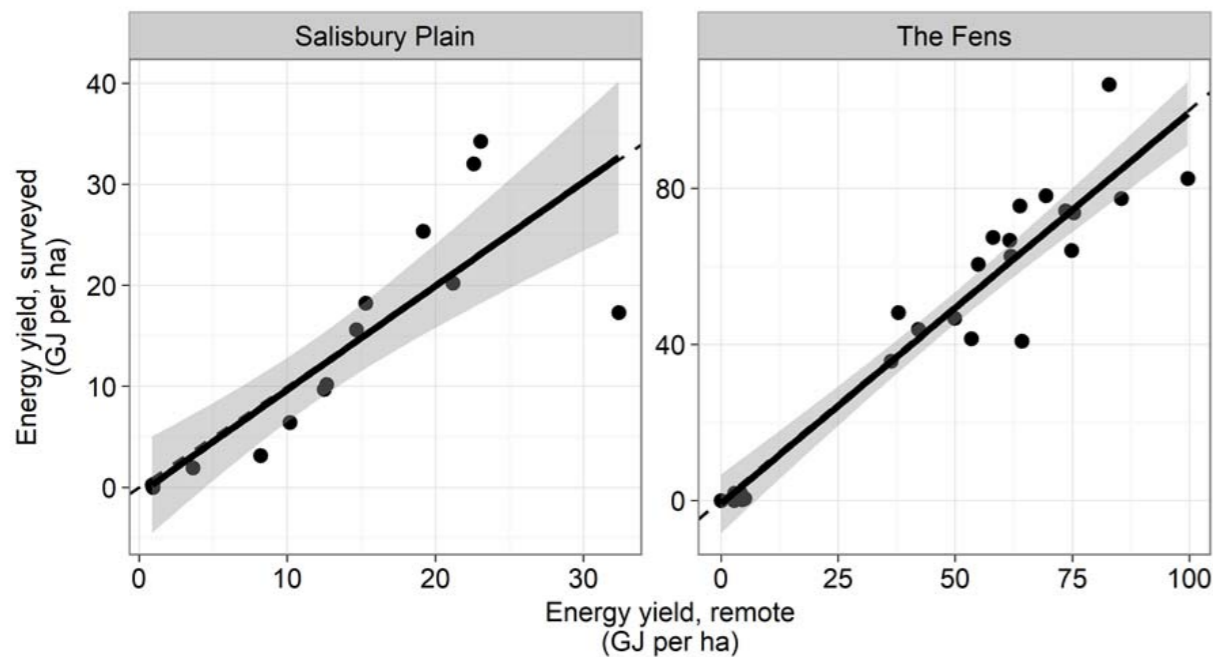
Fig. 2. Relationship between energy yield estimated remotely and derived from landowner surveys.
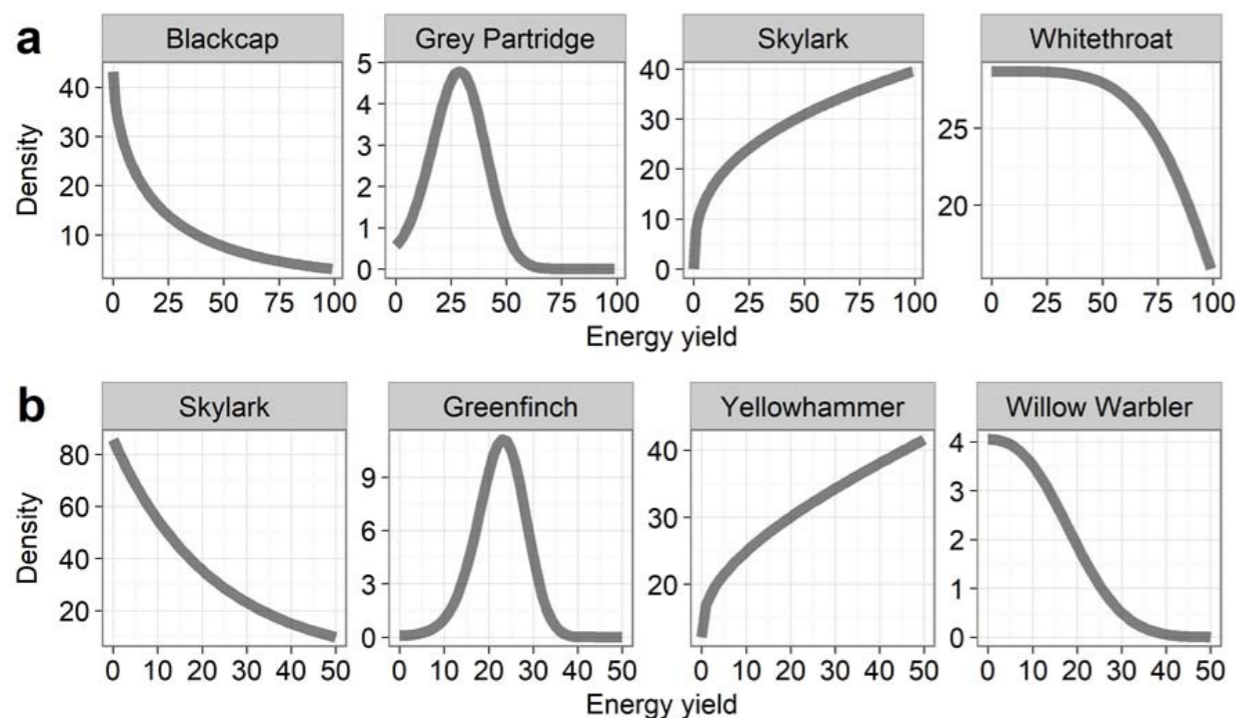

Fig. 3. Density-yield curves for four illustrative species in a. The Fens and b. Salisbury Plain.

Whitethroat *Sylvia communis* in The Fens shows a negative-increasing curve, with high densities in natural habitat and moderate-yield farmland. Species with this shape of curve achieve maximum population size under land sharing, which maximises the area of moderate-yield farmland.

Skylark in The Fens and Yellowhammer *Emberiza citronella* in Salisbury Plain show a positive-decreasing curve. These species reach highest densities in moderate- to high-yield farmland, and will usually achieve maximum population size under land sharing.

Grey Partridge *Perdix perdix* in The Fens and Greenfinch *Carduelis chloris* in Salisbury Plain have more complex non-monotonic curves, with densities peaking at intermediate yields. Willow Warbler *Phylloscopus trochilus* in Salisbury Plain also shows a hump-shaped curve, closely resembling the negative-increasing curve of Whitethroat in The Fens. For these species, the best strategy depends

on the total food production target, which determines the yield of farmed land under land sharing and the area of farmed land under land sparing. In some cases, an intermediate strategy – with farmland yields in between the two extremes of land sharing and sparing – will be best.

Discussion

Density-yield curves

We have presented density-yield curves for four illustrative species in each landscape. The shape of these curves indicates whether a species' total population size would be maximised under land sparing, land sharing or an intermediate strategy; in some cases, this will depend on the landscape-wide food production target (Green *et al.*, 2005). Clearly, different species have different resource requirements, so are likely to do better or worse under different food production scenarios (as illustrated in Fig. 3). A simple 'vote-counting' exercise could demonstrate which single strategy would benefit the most species, with species weighted according to UK conservation status, or the relative importance of each region in supporting the national breeding population. It's likely, however, that mixed strategies which combine elements of land sharing and land sparing will benefit more species than either extreme sparing or sharing alone (Law *et al.*, 2016). Additionally, we also plan to examine the *magnitude* of predicted changes in population size between scenarios.

Our preliminary results also suggest that the same species might respond differently in different landscape contexts (Law & Wilson, 2015). The Skylark has a strikingly different curve-shape in the two regions, which is probably attributable to the different baseline habitats; chalk grassland supports very high densities, whereas the species is virtually absent from fenland.

Estimating yields

The strong correlations between remote-estimated and survey-derived yields are encouraging, and suggest that food production yields can be estimated without the need for intensive field work. Even yield estimates derived from unclipped satellite derived land cover maps correlated strongly with estimates based on farmer surveys. This opens up the possibility of repeating similar exercises for other parts of the UK, drawing on the existing biodiversity monitoring data.

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