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# Biodiversity-food trade-offs when agricultural land is spared from production

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## Abstract

The Kunming-Montreal Global Biodiversity Framework proposes to address biodiversity decline by expanding areas under conservation. Biodiversity conservation in agricultural landscapes, the world's predominant land use, could involve sparing, or setting aside, agricultural land from production, implying biodiversity-food trade-offs. Employing bird species and agricultural data, we undertake a novel empirical analysis of such trade-offs on a set-aside scheme implemented in England between 1992-2007. Expanding set-aside increases bird species abundance and richness by, respectively, 1.2-2.1% and 0.7-0.9%, but has no impact on diversity (Shannon-Wiener index). These effects are discontinuous, subject to thresholds in set-aside areas. A minimum 3% of agricultural land set aside is required for a positive effect on biodiversity while 13% of agricultural land generates a 15-25% and 30-35% increase in abundance and richness, respectively. Estimates of short- and long-run effects show that impacts are larger in the long-run. Expanding set-aside is also associated with a 10-17% decline in cereal output, with weak evidence of an attenuating land-sparing effect on yields. Our results suggest that although biodiversity-food trade-offs are likely in high-yield agricultural landscapes, such as those in England, the risk of a reduction in food supply could be minimised in settings where there is still scope for intensification.

*Keywords:* Agriculture, Biodiversity, Food Production, Land Sparing, Set-aside  
*JEL:* Q18, Q57, R52

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## 1. Introduction

Globally, over one million recorded species are threatened with extinction (Brondizio *et al.*, 2019). The decline of terrestrial biodiversity is mainly driven by the conversion, degradation and fragmentation of natural habitats (Haddad *et al.*, 2015; Newbold *et al.*, 2015), to meet our growing – and changing – food demands (Angelsen, 2010; Crist *et al.*, 2017; Dasgupta, 2021). In response, member states of the Convention on Biological Diversity committed to designating 17% of the earth’s land area as protected areas by 2020. Although this target was not met (Xu *et al.*, 2021; IPCC, 2022), conservationists and policymakers have proposed more ambitious, post-2020 protection targets. In particular, a goal of the Kunming-Montreal Global Biodiversity Framework proposes an expansion of area-based conservation measures to at least 30% of the earth’s surface by 2030 (Waldron *et al.*, 2020; UNEP, 2022). An even more ambitious proposal, known as ‘Half Earth’, advocates for the conservation of half of the earth’s land (Wilson, 2016; Schleicher *et al.*, 2019). These proposals to safeguard biodiversity and ecosystems have been endorsed by the Intergovernmental Panel on Climate Change in its Sixth Assessment Report, which views them as fundamental for climate resilient development (IPCC, 2022).

Agriculture presently accounts for 50% of the earth’s habitable land (Ritchie and Roser, 2020). Expanding the amount of earth’s land under conservation could involve sparing, or setting aside, agricultural land from production thus trading biodiversity benefits for a reduced food supply (Mehrabi *et al.*, 2018; Usubiaga-Liaño *et al.*, 2019). A biodiversity-food trade-off is most likely in settings, such as those found in Europe, where agriculture is the predominant land use and agricultural land remains home to a substantial share of the continent’s remaining biodiversity, including 50% of its bird species (Pain and Pienkowski, 1997). One way to attenuate a potential biodiversity-food trade-off is through a land-sparing effect that intensifies production and raises yields on land remaining under production (Waggoner, 1996; Borlaug, 2007; Rudel *et al.*, 2009; Kremen, 2015). Yet, there is an evidence gap regarding the impacts on both biodiversity and food output when agricultural land is set aside from production, and the extent to which any reductions in food output might be attenuated by land-sparing effects.

With a focus on bird species and cereals, we construct two panel datasets and empirically evaluate the biodiversity-food trade-offs from an agricultural land set-aside scheme implemented in 1988 by the European Economic Community (EEC), predecessor to the European Union. The set-aside scheme was implemented to address the over-production of cereals and stabilise low cereal prices in Europe, as well as reduce the costs of the Common Agricultural Policy (CAP) (Bourgeon *et al.*, 1995). We evaluate the extent to which cereal production and biodiversity were impacted by set-aside in England, where agriculture constitutes around 70% of its land, over half of which is utilised for crop cultivation (FAO, 2020). Knowledge of these impacts could help inform the UK’s post-Brexit agricultural policy, including a possible future

role for agricultural land set-aside (Bateman and Balmford, 2018). More generally, it contributes to our understanding of potential biodiversity-food trade-offs when area-based conservation measures are implemented at scale in agricultural landscapes.

To reduce cereal output, the EEC paid farmers a fixed payment per hectare in exchange for retiring land from cereal production. What began as a voluntary scheme in 1988 became compulsory, that is, a legal requirement, in 1992. Until 2007, farmers were mandated to retire a minimum proportion of land cultivated with cereals but were free to decide which land to set-aside; they could also, voluntarily, set aside additional land. The mandatory set-aside requirement changed over time, varying between 5% and 15% of cereal area, in accordance with the demands of the CAP. As this requirement did not change in response to the demands of conservation, it provides a useful source of exogenous variation to help identify set-aside's impacts on biodiversity.

Although the EEC set-aside scheme lacked explicit environmental goals, it reportedly played a role in sustaining biodiversity as well as reducing agricultural inputs, buffering watercourses and other habitats, and protecting soils (DEFRA, 2007; Silcock and Lovegrove, 2007). A lack of robust evidence demonstrating a causal effect of the EEC scheme on biodiversity is mirrored in studies, typically found in the conservation biology literature, which use empirical methods with limited capacity for causal inference to examine the biodiversity impacts of agri-environment schemes (AES). Such schemes do have explicit environmental aims that sometimes incorporate an element of set-aside. For example, in a Swiss AES a higher proportion of 'biodiversity promotion areas' on farmland is found to be associated with greater bird and butterfly abundance and richness (Zingg *et al.*, 2018; Zingg *et al.*, 2019). More rigorous empirical studies, on the other hand, tend to focus more on identifying the land-use and economic impacts of AES, (e.g. Chabé-Ferret and Subervie, 2013; Bertoni *et al.*, 2020; Wuepper and Huber, 2022), neglecting biodiversity outcomes. While not focused on evaluating the impacts of AES, recent research using bird data has applied econometric methods to the evaluation of relationships between bird diversity and, respectively, insecticides (Li *et al.*, 2020), crop diversity (Strobl, 2022), and farm size (Noack *et al.*, 2022).

Our paper applies similar, rigorous methods to bird data in our first panel dataset to empirically evaluate set-aside's impacts on bird diversity before evaluating impacts on agricultural outcomes in our second dataset. In doing so, we are able to estimate biodiversity-food trade-offs from a policy which, if implemented at scale, has the potential to contribute to the Kunming-Montreal Global Biodiversity Framework targets (Waldron *et al.*, 2020; UNEP, 2022). That said, our estimates of trade-offs are anticipated to underestimate the potential for biodiversity benefits from set-aside, not only because of the absence of biodiversity policy goals but also because of how the scheme was designed. Specifically, farmers were permitted to rotate set-aside areas, with up to 50% of aggregate set-aside areas rotated after a single year (Silcock and Lovegrove, 2007). As such, the EEC scheme was not conceived as a conservation set-aside in the manner of, for example, China's Sloping Land Conversion Program

in which environmental-food trade-offs have been analysed if not subject to empirical evaluation (e.g. Xu *et al.*, 2006; Groom *et al.*, 2010).

We first identify set-aside's impacts on biodiversity using three measures applied to bird species data from the UK's annual Breeding Bird Survey (BBS): abundance, richness and the Shannon-Wiener diversity measure. Generally considered good indicators of the general state of wildlife, bird populations in England and the rest of the UK fell between 1970 and 2014 (Figure A1) Much of this decline was due to the loss of suitable nesting and feeding habitats and a reduction in food availability, in turn driven by increased pesticide use and other changes in farmland management associated with intensification (Chamberlain *et al.*, 2000; RSPB, 2013; DEFRA, 2014; DEFRA, 2021). With the implementation of mandatory set-aside, 50-80% of set-aside land in England each year naturally regenerated, which along with restrictions on the use of herbicides and pesticides on set-aside land created biodiversity-friendly habitats (DEFRA, 2007). Although the practice of rotation likely prevented the development of ecological communities on some set-aside land, England's bird populations appeared to stabilize after 1992. Indeed, previous research indicates a positive association between set-aside and bird abundance in England (e.g. Henderson *et al.*, 2000; Firbank *et al.*, 2003; Gillings *et al.*, 2010).

Causal inference in previous work focusing on bird abundance is, however, limited due to a failure to control for unobserved determinants of abundance and a reliance on cross-sectional data. We identify set-aside's impacts on biodiversity between 1994 and 2007 by applying fixed effects models to our first panel dataset. Annual data on geographically-referenced land uses in 10km-by-10km grid cells are integrated with species-level bird counts collected in 1km-by-1km transects. We consider five groups of bird species: (1) All Species, which is sub-divided into (2) Non-farmland and (3) Farmland; the latter is further sub-divided into (4) Farmland Generalists and (5) Farmland Specialists. An instrumental variables approach is adopted to provide robustness.

Going further beyond previous work, we next unpack our results in two ways to better understand the response of biodiversity to set-aside. We augment our fixed effects models with other econometric specifications. First, our fixed effects models assume a continuous linear or log-linear relationship between set-aside and biodiversity. It is well-known, however, that many taxa respond in non-linear, non-smooth ways to habitat loss and fragmentation. Often there are discontinuities or thresholds in ecological responses to land-use changes, below which species abundance and richness become significantly more sensitive to habitat loss (Andr n, 1994; Muradian, 2001; Yin *et al.*, 2017). Less understood is whether thresholds exist for habitat gain. Previous research, (e.g. Banks-Leite *et al.*, 2014), assumes habitat 'restoration scenarios' based on observed thresholds of habitat loss rather than habitat gain. Yet, evidence of hysteresis in restored habitats (Curran *et al.*, 2014) suggests little support for this assumption. To test for a discontinuous relationship between set-aside and biodiversity, we extend our fixed effects models by including binary indicators for a range of set-aside 'bins' that estimate impacts at different set-aside area thresholds

in a non-parametric way.

Second, the results of our fixed effects models describe the average effect of set-aside on biodiversity over the duration of the policy. Yet, these models do not control for the possibility that growth in the stock of biodiversity (however measured) in a given period, particularly with respect to abundance, might be dependent on the stock of biodiversity in the previous period (Dasgupta, 2021; Li *et al.*, 2020; Perman *et al.*, 2003). As such, the models fail to separate out short-run contemporaneous effects from long-run effects that build over time due to ecological stock dynamics. This separation is important for understanding how policy interventions might sustain biodiversity over time. To help disentangle short- from long-run effects and capture the ecological dynamics of abundance and species relationships, a lagged dependent variable that controls for the historical stock of biodiversity, is included in our models. We apply the dynamic panel methods of Moral-Benito *et al.*, 2019 to address heterogeneity and dynamic Nickell biases (Nickell, 1981).

The set-aside scheme aimed to reduce cereal output by restricting the amount of agricultural land farmers could put into production. Farmers could, however, have attenuated a reduction in output depending on how they responded to set-aside (Wu, 2000). For example, farmers might have chosen to set aside their least-productive land so that the reduction in land did not lead to a one-for-one reduction in output. Yet, the mandatory nature of the scheme implies that highly-productive land was likely to have been set-aside too (Bourgeon *et al.*, 1995), and indeed this was the case in England (Crabb *et al.*, 1998). Earlier, unpublished research found that a reduction in land led to an approximately one-for-one decline in output in England between 1993 and 1998 (Rural Economic Research (CRER), 2001). Productivity remained unchanged and farms did not initially adjust fixed or variable inputs, e.g. machinery and labour, in line with the one-for-one fall in output. Farmers eventually adjusted and reallocated inputs, which attenuated set-aside’s impact on output.

We estimate set-aside’s impacts on cereal output, area and yield in our second panel dataset, of farm-level data on land uses and outputs, collected between 1981 and 2007. To identify set-aside’s impacts, we conduct a type of event study in which all of the farms included in our dataset were sampled both before and after the start of mandatory set-aside. The average impact of set-aside is estimated by the parameter on an indicator variable for the pre- and post-setaside periods (pre- and post-1992). Even after accounting for farmers’ behavioural responses to set-aside, we find clear evidence of biodiversity-food trade-offs and little evidence of yield improvements when land in England’s agricultural landscapes is spared from production.

In the remainder of the paper, we first provide some background to the EEC set-aside scheme in England, in Section 2, before describing the data and methods used in our study, in Section 3. Section 4 presents our results, beginning with set-aside’s impacts on biodiversity followed by those on agricultural outcomes. Finally, Section 5 discusses the implications of our results and concludes.

## 2. Background to set-aside in England

In England, under what was known as the Arable Area Payment Scheme (AAPS), farmers who were not defined as ‘small producers’, that is, with fewer than 15.6ha of land, were mandated to set aside a certain percentage of the area of their AAPS claim (MAFF, 2000). Between 1993 and 1997, land could be entered either into rotational and/or non-rotational set-aside (Rural Economic Research (CRES), 2001). Set-aside could be left in the same place or rotated on the farm. Around 35% of the set-asides were non-rotational for three years or more (Silcock and Lovegrove, 2007). A combination of rotational and non-rotational set-aside was more likely to occur on farm holdings with over 50ha of set-aside than holdings with under 20ha of set-aside. After 1997, there was a single set-aside rate, replacing the previous rotational and non-rotational set-aside rates. The June Survey (see Section 3) does not distinguish between rotational and non-rotational set-aside and hence, we are unable to examine the differences between these two types of set-aside in our empirical analysis.

Farmers were able to enter additional land into voluntary set-aside, up to a maximum of 50% of the area on which AAPS payments are claimed (Crabb *et al.*, 1998). The voluntary option was also open to small producers. Compulsory rates of set-aside between 1993 and 2006 are shown in Figure 1, alongside the actual rates, which included voluntary set-aside. Figure 2 shows how the actual rates translated into land areas thus illustrating the scale of the EEC set-aside scheme in England.

Each individual set-aside plot was supposed to cover an area of at least 0.3ha (MAFF, 2000). Farmers were encouraged to set aside strips of land along permanent water courses and lakes, as well as field margins and headlands. These strips had a minimum width requirement of 20 metres and were subject to the 0.3ha rule. More common than setting aside field strips and margins, however, was the set-aside of entire fields, particularly when farmers engaged in rotational set-aside. Indeed, rotational set-aside fields were found to have had a higher average field size (6.2ha) than non-rotational set-aside fields (3.5ha) (Silcock and Lovegrove, 2007). Whether in the form of fields or not, rotational set-aside reportedly provided more suitable habitats and feeding grounds for breeding birds than non-rotated set-aside, which tended to be concentrated on the least-productive (e.g. poor soils) and most-difficult-to-cultivate (e.g. remote) land (Firbank *et al.*, 2003).

From 15 January until 31 August, set-aside land could not be used for any form of agricultural production other than the production of non-food and non-feed crops, for example, short rotation coppice. These rules, although strict, were relatively simple. No contracts were required and farmers had to use the crop for one of the approved non-food purposes. In addition, farmers could not receive any other EU aid for non-food crops grown on set-aside land. Farmers were allowed to set aside up to 100% of the area of their AAPS claim provided all of the set-aside land was used for the production of multi-annual biomass crops. Set-aside land could also be used for the temporary storage of products such as straw bales. Farmers were prohibited from damaging, destroying or removing certain features, including stone



walls, hedges, trees, watercourses and ditches, situated in or adjacent to set-aside land (MAFF, 2000).

Although the EEC set-aside scheme had no explicit environmental objectives, European policymakers recognised the potential of set-aside to generate environmental benefits from agricultural land (DEFRA, 2007). Set-aside payments were made conditional on farmers following certain conditions. For example, there were restrictions on the application of fertilisers, manures and wastes to set-aside land, and the retention of traditional field boundaries adjacent to set-aside land (MAFF, 2000). Farmers were also obliged to establish a ‘green cover’ on set-aside land by the start of the set-aside period to minimise nitrate leaching. This was undertaken through natural regeneration, or the sowing of suitable cover or, in the case of land in set-aside the previous year, by maintaining the existing cover (Rural Economic Research (CRER), 2001). Around 85% and 52% of, respectively, rotational and non-rotational set-asides were in natural regeneration, with most of the remainder sown with grass cover (Silcock and Lovegrove, 2007).

The cover had to be maintained until 1 July. Selective, non-residual herbicides could be used at any time to control weeds (Silcock and Lovegrove, 2007). The green cover could not be put to any commercial use, which meant that it either had to be cut short between 15 July and 15 August, with the cuttings left on the ground, or destroyed completely by 31 August. From 15 July, crops could be sown on land that was not returned to set-aside in the following year. For land that is returned to set-aside, there were restrictions on the use of the cover after the end of the set-aside period, between 1 September and 14 January (MAFF, 2000).

Farmers could apply for exemptions from the rules, for example, to follow their own management plan. Organic farmers or those in conversion were eligible for some exemptions from the general set-aside management rules.

### 3. Data and Methods

#### 3.1. Data

We construct two panel datasets for our empirical analysis of set-aside’s impacts on, first, bird species abundance, richness and diversity and, second, cereal output, land and yield. For our biodiversity analysis, annual land-use data (ha) at the 10km-by-10km grid cell scale are integrated with annual data on species-specific bird counts at the 1km-by-1km (transect) scale for the period 1994-2007 while our analysis of agricultural impacts relies on annual farm-level data on land uses (ha) and outputs (tonnes), spanning the period 1981-2007.

The annual June Survey of Agriculture and Horticulture (‘June’) is our source of land-use data (ha) utilised for our biodiversity analysis (Table 1). A full Census was undertaken annually until 1995, surveying around 80% of England’s farms (DEFRA,

2012), before becoming a sample survey. Although a full Census is still held every 10 years, the annual sample size of farms declined after 2000, by 20-40%. Farms are sampled via stratified random sampling on the basis of their economic size, with higher sampling rates used in the larger strata (DEFRA, 2014). Information is collected on arable and horticultural cropping activities, land usage, livestock populations and the agricultural labour force. We utilise data for all land categories defined as part of the Utilised Agricultural Area (UAA): cereal crops (wheat, maize, winter barley, spring barley), oilseed rape (osr), pulses (peas and beans), sugarbeet, linseed, stockfeed, grasslands, set-aside, fallow, industrial set-aside and woodland.

June land-use data are geo-referenced but only use a single point reference for each farm, that is, not at parcel scale. Thus, the data for each grid cell includes all land for holdings with a point location within the grid. For non-responding and non-sampled farms, data are imputed, which involves adjusting its previous response by national trends. A number of grid cells are excluded due to rules on information disclosure. In practice, this mainly excludes grids around the coast, plus those close to major urban centres. The June data are, however, only available in grids at the 10km-by-10km scale, based on the Ordnance Survey National Grid, a standard map reference system for the UK. Due to data restrictions, to ensure that farms' identities are not revealed, information on the spatial location of farms and the spatial distribution of individual land uses within each grid are unavailable. Data for the number of farms located in each landscape are also unavailable.

Our bird count data are from the annual Breeding Bird Survey (BBS) organised by the British Trust for Ornithology (BTO), which began in 1994 (Table A1-A8). In general, birds are one of the better monitored taxonomic groups globally yet there is large geographic variation in monitoring effort (Meyer *et al.*, 2015; Meyer *et al.*, 2016; Amano *et al.*, 2016), with the UK's BBS regarded as one of the world's most rigorous structured breeding bird monitoring schemes (Gregory *et al.*, 2000).

Within each BTO region, 1km-by-1km transects were selected randomly from a list of all transects in the Ordnance Survey National Grid reference system, and allocated to volunteers via a stratified random sampling process. Initially, the number of BBS transects assigned to each of the BTO's regions was a fixed proportion of the number of potential volunteers in the region. In regions with relatively few potential volunteers, a minimum level of coverage was set. The aim was to obtain as many of the same transects to be surveyed every year in order to gather a run of comparable data from each transect. Around 2,000 to 3,000 transects are surveyed annually. Data for 2001 are excluded due to low coverage brought about by foot and mouth disease access restrictions.

Fieldwork involves three visits to each BBS transect per year, between April and June: a reconnaissance visit and two bird recording visits. Volunteers report a count of birds observed per species. Given that the data reflect the number of birds seen, more visible birds will have higher counts. However, detection is not a major issue because of our interest in explaining within-transect changes over time. So, the

absolute numbers are less critical for our analysis.

Geo-referenced bird data are assigned to our 10km-by-10km grid cells for which we have land-use data (Figure A2-A3). Thus, the BBS transects are not randomly sampled within a given grid and instead there is variation in number of transects across grids and (in some instances) over time for a given grid. The number of transects contributing to each grid is known. Where there is more than one transect for each grid, we calculate the average bird count across all transects in the grid. Measures of abundance are scaled up to the 10km-by-10km grid level using the sample of 1km-by-1km bird counts associated with each grid, correcting for the recorded UAA in each 10km-by-10km grid.

For 195 grid cells no data are available, which leaves 1,158 grids remaining. Transects are excluded from the dataset for grids where more than half of the grid area was classified as urban, coastal or upland, in order to help focus on agricultural landscapes. For the remaining grids, total bird abundance is averaged across transects per year within each grid. Using these estimates, we test for evidence of spatial autocorrelation with respect to any of the bird groups, with the results reported in Section 4. Estimates of species richness, a measure of alpha diversity at the transect scale, are obtained by counting a species as present in a grid if, on average (across transects in a particular grid), the abundance of a particular species is greater than zero. Diversity is estimated by applying the Shannon-Wiener index, which reflects the number of different bird species while simultaneously taking into account how evenly the numbers of birds are distributed among these species:

$$H = - \sum_{i=1}^n p_i \ln p_i$$

Where  $H$  is the Shannon-Wiener expression for entropy (Shannon, 1948; Wiener, 1948), and  $p_i$  is the relative abundance (proportion of total population) of birds belonging to the  $i$ th species. This index is based on the weighted geometric mean of the relative abundances of bird species over the  $n$  species observed in the grid square. With increasing inequality in the abundances of bird species, the larger the weighted geometric mean of the  $p_i$  values, and thus the smaller the corresponding Shannon-Wiener entropy. When almost all abundance is concentrated in a single species, Shannon-Wiener entropy approaches zero.

Our three biodiversity measures are calculated for five groups of species (Table 2): (1) all bird species (All Species – 130 species), which is sub-divided into (2) bird species classified as non-farmland (Non-farmland – 111 species) and (3) those typically found in agricultural areas (Farmland - 19 species); the latter group is further sub-divided into (4) Farmland Generalists (7 species) and (5) Farmland Specialists (12 species). Farmland Specialists are those species that are restricted to, or highly dependent on, farmland habitats while Generalists include species found in a wider range of habitats (DEFRA, 2021).

We create a balanced land use and biodiversity panel dataset, which includes only

those grid cells that appear in the data every year between 1994 and 2007, excluding 2001. Land uses do not change rapidly, so a long time period for the panel tries to maximise the within-grid variation. This generates a total of 351 grids. The average (median) grid contains approximately 15,000 (14,000) birds based on an average of over 200 birds (All Species) per transect (see Table 2). The UAA comprises, on average, just over 70% of land in a grid (Table 3). We test the bird and land use data for statistical stationarity using the Harris-Tzavalis panel unit root test, which is suitable for large numbers of panels (grid cells) and a fixed time period. For all variables, we reject the null hypothesis that the data are non-stationary and possess a unit root (Table A9).

Farm-level data utilised in our second panel dataset include cereal output, areas cultivated and set-aside (Table 3). These are sourced from the Farm Business Survey (FBS). Beginning in 1981, the FBS is an annual financial survey based on a stratified, representative sample that covers more than 90% of agricultural production. Almost 3,000 farms are recruited annually by farm type and stratification is undertaken based on the June Survey. Larger areas, likely associated with larger farms, are mostly found in the east and south of England. Since farms enter and leave the sample each year, our farm-level panel dataset is unbalanced. To conduct an event study, our final dataset is determined by the criteria that farms must be sampled both before and after the start of mandatory set-aside, in 1992, and that there are no missing years of data for any given farm within the period in which the farm was recruited. Also, the dataset only includes farms with a yield of less than 15 tonnes, and a planted cereals area greater than 15ha because farms with less than 15.6ha of land were exempt from mandatory set-aside rates (MAFF, 2000). Due to data restrictions, to ensure that farms' identities are not revealed, information on the spatial location of farms and associated land uses is unavailable.

### 3.2. Empirical approach

Obtaining estimates of biodiversity-food trade-offs, in particular with respect to obtaining a better understanding of how biodiversity responds to set-aside, necessitates the application of different econometric specifications to our data. In this sub-section, we describe these specifications and explain our rationale for choosing them. First, to isolate the impact of set-aside on biodiversity, we control for potentially correlated time-invariant confounders that are particular to the 10km-by-10km grid cells. The fixed effects strategy identifies the marginal effects of setaside hectares on measures of biodiversity using within-grid variation in the data:

$$Y_{cit} = \alpha + \beta \mathbf{A}_{it} + \sum_{l=1}^n \beta_l \mathbf{X}_{lit} + \sum_{r=1}^R \sum_{t=1}^T \delta_{rt} D_t D_r + \alpha_i + \lambda_t + \epsilon_{it} \quad (1)$$

where for each bird group  $c$ , the dependent variable is  $Y_{cit}$ . For abundance,  $Y_{cit}$  is the natural logarithm of grid-level abundance. This transformation deals with the skewness of the abundance data, making  $\ln Y_{cit}$  approximately normally distributed.

For species richness,  $Y_{cit}$  represents the raw number of species, which, being a mean over several BBS transects, is already approximately normally distributed. For the diversity measure, the outcome is the Shannon-Wiener Index, which is also approximately normally distributed. Count data models were considered for species richness but we opted for the linear model because it is consistent with the non-parametric analysis that follows (see below).

Each biodiversity outcome is measured at the level of the grid  $i$  at time  $t$ . In each case, the outcome is modelled as a function of reported agricultural area  $A$  (ha) under the set-aside scheme in the grid  $i$  at time  $t$ :  $A_{it}$ . Landscape fixed effects,  $\alpha_i$ , control for time-invariant, grid-scale characteristics such as soil quality, topography and field configurations. These unobserved factors are specific to individual grids and are likely to be correlated with the policies and/or our biodiversity measure thus biasing our estimates of impact. For example, low-quality, marginal or sloped land might be fallowed even in the absence of the set-aside scheme. Year fixed effects,  $\lambda_t$ , control for spatially-invariant macroeconomic and climatic factors that affect all grids in year  $t$ . Region ( $r$ )-year ( $t$ ) interactions are included,  $D_t D_r$ . These interactions control for unobserved trends that differ by region due to the heterogeneity of the underlying grids, for example, agricultural production trends, weather and climate trends (see e.g. [Ay et al., 2014](#)) and price trends. Regions are defined as 10-by-10 grids, that is, 100km-by-100km land areas. This generates 24 regions in our sample based on the first two letters of the Ordnance Survey National Grid reference system. Standard errors are calculated using a two-way clustering approach at the region-by-year level, of which there are 309 clusters ([Arellano, 1987](#); [Sarafidis and Weber, 2015](#)).

We include a set of time-varying, grid-specific land-use area controls (in ha) for each land use,  $X_{lit}$ , including all cereal and non-cereal agricultural land uses, and an index of fractionalisation of land uses to account for the diversity of crops grown within each grid cell. These controls are included because they are also determinants of our biodiversity outcomes. In addition, they proxy for input levels associated with agricultural practices. We run specifications with and without these controls to allay potential concerns that other land uses might be bad controls in the sense of being affected by set-aside. To control for biases arising due to variation in the number of BBS transects over time, and more broadly the voluntary nature of the BBS, we include both linear and quadratic terms for sample number. Finally, we include the error term,  $\epsilon_{it}$ . We also consider the possibility that testing for multiple null hypotheses (one for each bird group) using a single underlying dataset may lead to over-rejection. The Romano-Wolf correction is implemented to control for the probability of rejecting one true null from multiple hypotheses in a testing framework ([Romano and Wolf, 2005](#); [Romano and Wolf, 2016](#)). We provide Romano-Wolf p-values calculated with 1000 bootstrap runs, estimated in Stata using the `rwolf2` package, to control for the familywise error rate (FWER) together with traditional clustered standard errors at the region-year level.

Another concern relates to possibility that farmers voluntarily put more of their land into set-aside, above and beyond the mandatory set-aside rate, where the land is less

profitable or of lower quality, and that this decision is related to existing biodiversity levels. This would introduce a positive bias in the relationship estimated between set-aside and biodiversity in Eq. 1. Such behavioural responses could be interpreted as a threat to the identification of the policy effect using a fixed effects approach. There are several arguments against this threat. First, it has been documented that the farmer’s decision over *where* to locate set-aside on agricultural land was primarily based on supporting production rather than conservation (e.g. Crabb *et al.*, 1998; Walford, 2002). In this sense, regarding the decision of *how much* land to set-aside voluntarily, the correlation with biodiversity could work in either direction, with an overall null effect, which is the case as shown by the small correlation coefficients between the difference in actual and mandated set-aside rates and biodiversity outcomes, reported in Table A10. Second, our data are at grid cell scale rather than at farm scale, hence at a higher geographical level than the decision-making unit. This potential source of endogeneity at the farm level is likely to be diluted at the grid level, especially when considering the potentially offsetting effects of over-provision among farms within a grid. Ultimately though, an exogenous intervention like the set-aside policy will induce many different types of response, each determining the overall effect on biodiversity. It is this overall policy effect that we want to estimate, that is, regardless of how it transpired. The key threat to identification in this case stems not from farmers’ choice of implementation, but from spatial differences and trends, which are largely controlled for by our fixed effects approach.

Nonetheless, potential endogeneity concerns are addressed in two ways. First, we produce a quantile-quantile plot of recorded set-aside against mandated set-aside in Figure 3, which highlights how farms in grid cells, on average, over- or under-provided with respect to mandatory set-aside rates. Importantly, the extent of over-, or indeed under-provision is limited, with the majority of observations located close to the 45° line. Second, we devise an instrumental variables specification similar to a Bartik-type shift-share interaction (Goldsmith-Pinkham *et al.*, 2020). We construct our instrument by interacting the 1-year lag of recorded total cereal production area (the share) with the contemporaneous mandated set-aside value (the shift). By taking into account total cereal production at time  $t - 1$ , the instrument captures farmers’ decision-making processes at the aggregate (grid) level, since the exogenous mandated (not chosen) level of set-aside at time  $t$  is dependent on grid-level production in the previous year and on the exogenous shift given by the mandatory set-aside rate. Our instrument is thus constructed as:

$$IV_{it} = \left( \sum_{k=1}^{N_k} C_{kit-1} \right) SA_{it}$$

Where  $\left( \sum_{k=1}^{N_k} C_{kit-1} \right)$  is the sum of total ha cereal production over each cereal type  $k$ , in grid cell  $i$  at time  $t - 1$ , and  $SA_{it}$  is the exogenous set-aside rate mandated by the policy in year  $t$ . We employ this instrument in a standard two-stage least square extension of the regression described in Eq. 1, estimated in Stata using the `ivreghdfe` package. Again, to dispel the probability of rejecting the null hypotheses when they are true in a multiple-hypotheses testing framework, we calculate Romano-Wolf

p-values with 1000 bootstrap runs (Romano and Wolf, 2005; Romano and Wolf, 2016).

Next, we relax the continuous linear or log-linear relationship between set-aside and our biodiversity measures by estimating a flexible, non-parametric model that allows the data to express any thresholds or discontinuities in the response of birds to set-aside. Specifically, we replace the set-aside variable in Eq. 1 with one that is non-parametric in set-aside. Dummy-indicator variables are included to reflect whether a grid lies within a particular set-aside bin. Year fixed effects and time-region interactions are applied as before:

$$\ln Y_{cit} = \alpha + \sum_{h=1}^H \beta_h D A_{hit} + \sum_{l=1}^n \beta_l X_{lit} + \sum_{r=1}^R \sum_{t=1}^T \delta_{rt} D_t D_r + \alpha_i + \lambda_t + \epsilon_{it} \quad (2)$$

In Eq. 2, dummy variables,  $D$ , denote different threshold bins of set-aside area,  $h$ , starting with the smallest amounts (1-25ha, 26-50ha, 51-100ha) before moving up in increments of 100ha (200ha, 300ha, 400ha, 500ha, 600ha, 700ha, 800ha, 900ha, > 900ha). Individual grid cells can and do move in and out of different set-aside bins over time. The model compares the effect of each set-aside bin to the baseline category. In other words, the model estimates whether or not the impact of each set-aside bin on the biodiversity measure is significantly different than that of the baseline category. The baseline category, 1-200ha, is larger than 100ha partly due to sample size but also because the estimated coefficients on the smaller-sized set-aside bins, including the 1-100ha and 100-200ha bins, are zero.<sup>1</sup> For each bin, we run a Wald test to assess whether the estimated parameters of the bin are statistically different from smaller-sized bins. In Section 4, we test the robustness of our results using a baseline of 1-100ha.

The model described in Eq. 1 does not distinguish between long- and short-run effects of set-aside on biodiversity. The estimated co-efficients rather reflect the effect of a marginal (100ha) change of set-aside on biodiversity on average for the period of observation (1994-2007). One way to disentangle contemporaneous short-run effects and long-run effects that build over time is to use a dynamic model in which the dynamic properties of biodiversity are captured by a lagged-dependent variable. Understanding the dynamics of set-aside’s impacts on biodiversity is critical from the perspective of sustainable policy design. As is well known, however, the inclusion of a lagged-dependent variable in a fixed effects model can bias the estimate of both short- and long-run effects of set-aside via so-called Nickell bias (Nickell, 1981). Fortunately, estimators exist that can provide consistent (asymptotically unbiased) estimates of the parameters of dynamic models. After a process of model selection, in which

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<sup>1</sup>A number of grid cells have no record of set-aside; these tend to be located in hilly regions with poorer soils dominated by livestock rather than arable farming. Such grids recorded zero set-aside but also report little cereal production. Plus, as noted farms with less than 15.6ha of land were exempt from mandatory set-aside rates (MAFF, 2000). Thus, if our baseline category is zero then this either implies a comparison of bird abundance in cereals vs non-cereals areas, or with cereals areas that contained lots of small farms.

we include up to three time-lagged dependent variables, we drop the statistically-insignificant lags and specify a model that includes a single time-lagged dependent variable,  $\ln Y_{ci,t-1}$ :

$$\ln Y_{cit} = \alpha + \gamma \ln Y_{ci,t-1} + \beta \mathbf{A}_{it} + \sum_{l=1}^n \beta_l \mathbf{X}_{lit} + \sum_{r=1}^R \sum_{t=1}^T \delta_{rt} D_t D_r + \alpha_i + \lambda_t + \epsilon_{it} \quad (3)$$

where the short-run effect is given by  $\beta$  and the long-run effect =  $\frac{\beta}{1-\gamma}$

A typical dynamic estimator for the model described in Eq. 3 is the Arellano-Bond estimator (Arellano and Bond, 1991), which uses more lagged-dependent variables as instrumental variables to overcome Nickell bias (Nickell, 1981). In recent years, maximum likelihood approaches to dynamic panel models have become available. These approaches are asymptotically equivalent to the Arellano-Bond estimator and yet have a number of advantages. We use the maximum likelihood approach proposed by Moral-Benito *et al.* (2019), implemented by Stata’s `xtdpdml` routine (Williams *et al.*, 2018). The maximum likelihood approach tends to have better small sample properties: less biased and more efficient. There are some practical advantages too in that the structural modelling approach taken in the `xtdpdml` routine, which models each time period separately, implies that there is no need to choose among several different potential combinations of instrumental variables among the lagged variables in the time series, a choice for which the Arellano-Bond estimator is sometimes, as in our case, very sensitive.

We estimate the short- and long-run effects in a dynamic model that is as close to our fixed effects model described in Eq. 1 as possible. In addition to set-aside, we control for other important land uses, time-invariant fixed effects and year fixed effects. All explanatory variables are assumed to be strictly exogenous. Parameter values are assumed constant across the yearly structural equations, hence a single set of results is shown in Section 4. Convergence issues limit the extent to which year-by-region fixed effects and a full set of land-use controls could be used. In addition to the fractionalisation index, the land uses included as controls in the maximum likelihood estimation include the main land uses: wheat, maize, spring barley and winter barley. Again, to control for variation in the BBS sampling strategy, we include the number of sampled transects included in each sampled grid cell (and its square). Robust standard errors are used for inference to account for heteroskedasticity and autocorrelation.

The addition of the time-lagged dependent variable,  $\ln Y_{ci,t-1}$ , in Eq. 3 captures in a parsimonious way the broad dynamics of biodiversity in short-run ( $\beta$ ) and long-run ( $\beta/(1-\gamma)$ ) effects of set-aside. This is undertaken via an estimate of the dependence of current-period biodiversity stocks on stocks in the previous period, captured by the autocorrelation coefficient  $\gamma$ . Any dependence (autocorrelation) implies that the effects of set-aside are potentially sustained until another long-run equilibrium is



reached, defined by:  $Y_{it} = Y_{it-1}$ . The coefficient on the lagged-dependent variable,  $\gamma$ , captures the extent of dependence of the stock of biodiversity at time  $t$  on its stock at time  $t - 1$ . When  $\gamma$  is close to 1, the biodiversity stock in the current period is highly dependent on the stock in the previous period. The extent of dependence reflects dynamic ecological processes, which are likely to vary depending on the biodiversity measure adopted, but could be related to, for example, intra- and inter-species competition for resources. A short-run impact due to increased set-aside will likely have additional marginal effects in the future via the change in the previous period's stock of biodiversity. Since  $0 < \gamma < 1$ , the marginal change in biodiversity at  $t + 1$  is only a fraction of the instantaneous effect at  $t$ , and again at time  $t + 2$  until arriving at a new long-run biodiversity equilibrium that is  $\frac{\beta}{1-\gamma}$  larger. Hence, these effects persist but dissipate over time until a new equilibrium biodiversity stock emerges. The long-run effect measures the difference between current and future equilibria. For abundance, for example, if set-aside has a contemporaneous effect on populations, this effect could cause population change in future periods too, until a new equilibrium that reflects the new level of set-aside is reached.

For comparison with the results of the maximum likelihood (xtdpdml) approach, we run fixed effects regressions with a lagged-dependent variable for all bird groups and for all outcome measures of biodiversity. The results are reported in [Section 4](#).

To investigate the effects of set aside on production, we apply a linear fixed effects model similar to Eq. 1 to a farm-level panel dataset. The existence of a strong correlation between cereal land and set-aside at the farm level implies that set-aside's impact on output – expected to be negative – cannot be empirically disentangled from the effect of cereal land on output, which is expected to be positive. Instead of estimating the impact of set-aside in levels on output, we use a more reduced-form indicator variable to signal the implementation of the set-aside scheme in 1993, when farms first reported a positive value of set-aside. We apply the following specification:

$$\ln Y_{it} = \alpha + \beta \mathbf{A}_{it} + \sum_{r=1}^R \sum_{t=1}^T \delta_{rt} D_t D_r + \beta_U U A A_{it} + \alpha_i + \lambda_t + \epsilon_{it} \quad (4)$$

where  $\ln Y_{it}$  is the natural logarithm of cereal output for farm  $i$  at time  $t$ , modelled as a function of set-aside, a dummy variable,  $A_{it}$ . Farm fixed effects,  $\alpha_i$ , control for time-invariant, farm-scale characteristics such as soil quality, topography and field configurations. These unobserved factors are thus specific to individual farms. Year fixed effects,  $\lambda_t$ , control for spatially-invariant macroeconomic and climatic factors that affect all farms in year  $t$ . Region ( $r$ )-year ( $t$ ) interactions are included,  $D_t D_r$ , as is the farm's UAA,  $U A A_{it}$ . Regions are again defined as 100km-by-100km land areas. Standard errors are clustered at the farm level to account for heteroskedasticity and autocorrelation.

The average impact of set-aside, between 1992 and 2007, is compared to a pre-set-aside baseline, the start year of which varies between 1981 and 1990. The latest year,

1990, allows for a minimum of two years of data, 1990 and 1991, in the baseline. The earliest year, on the other hand, is 1981, which includes data from the first FBS survey, in 1981, and all subsequent years until 1991. The estimates of impact reflect the difference between the average of the outcome variable over the duration of the mandatory set-aside scheme and the average of the outcome variable for the baseline period, that is, the period prior to the introduction of the set-aside scheme.

We examine set-aside’s impacts on area of land under cereal production and cereal yield by simply treating these, individually, as an outcome variable (replacing cereal output), on the left-hand side of Eq. 4.

## 4. Results

### 4.1. Impacts of set-aside on biodiversity

Intensive farming practices on land that was not set aside, and instead remained under production, were not prohibited. Hence, there was potential for negative spillover effects on biodiversity, for example, due to the application of pesticides (Li *et al.*, 2020). Our empirical estimates of set-aside’s impacts on biodiversity therefore capture the net impact of two countervailing effects: the positive effect from the creation of new habitats and the negative effect from potential spillover effects.

The results from our fixed effects models, estimated using the specification described in Eq. 1, suggest evidence that bird diversity benefited from the set-aside scheme. Table 4 summarises the impacts of set-aside on bird species abundance, species richness and diversity. These results are shown in full in Table A11-A13.

Our results for abundance are suggestive of a Malthusian response to the creation of new habitats that provide additional food sources for birds. Specifically, we find a positive and statistically significant (at the 1% level) impact of set-aside on species abundance for All Species. Populations of Non-Farmland and Farmland species are both positively affected by set-aside (respectively, 5% and 1% level). Within the latter group, Farmland Generalists are the species most affected (5% level). Across these bird groups, an additional 100ha of set-aside is associated with a 1.2-2.1% increase in abundance. Assuming bird numbers accumulate spatially across the landscape, this implies a mean (median) impact of between 195–270 (180-250) birds per 100ha of set-aside in a landscape, an impact of 0.1 standard deviations in birds per standard deviation of set-aside.

Set-aside has a positive impact on All Species richness (1% level), an effect equivalent to a 0.9% increase that appears to primarily originate among Non-farmland species (1% level), which increase by 0.7%. A 100ha increase in set-aside is associated with an increase of approximately 0.3 species, translating into an increase of one species per 300ha of set-aside, which is the median set-aside area, equivalent to a 0.1 standard deviation increase. Thus, set-aside appears to have enabled an expansion of species’

ranges for species that are not typically found in agricultural ecosystems. Given that the species richness data reflect sample estimates of the average number of species in the population, these marginal effects do not accumulate spatially in the same way as abundance. Therefore, our estimates can be interpreted as the marginal effects in a grid cell.

The marginal impact of a 100ha increase in set-aside on diversity is negative for four out of five bird groups yet is only statistically significant for Farmland species (10% level) (Table 4). The relatively weak effect on diversity, coupled with the positive results on abundance and species richness, suggests either that the latter effects were small or that changes in abundance occurred for already common birds, hence increasing unevenness in relative abundance with the end result that the Shannon-Wiener index is declining or unaffected.

Our results in Table 4 are not affected by the omission of the land use controls other than becoming less precise. This is in part because our data are at the grid cell level rather than at the individual farm level. The statistical significance of the results is not affected when we take into account the probability of rejecting the null when it is true in a multiple-hypotheses testing framework. All significant results are indeed confirmed by Romano-Wolf p-values calculated with 1000 bootstrap runs (Romano and Wolf, 2005; Romano and Wolf, 2016), and the positive effect found for the species richness of Farmland Generalists turns statistically significant at the 1% level (Table 4).

We note further observations in relation to the results summarised in Table 4. First, we find relatively little evidence of spatial autocorrelation with respect to any of the bird groups, which obviates the need for more explicit spatial modelling (Figure A4-A5). Second, expanding the area under maize and winter barley is associated with a decline in biodiversity (Table A11-A13). This result could reflect the loss of habitat and increased use of inputs utilized in the expansion of cultivated areas. Third, although we do not observe the spatial distribution of land uses within grid cells our measure of land fractionalisation controls for the diversity of land uses. We find that increasingly fractionalised farmland has a positive effect on biodiversity (Table A11-A13).

In Table 5, we report the results for the instrumental variable regressions described in Section 3. Recall that our IV is the total cereals area in the previous period multiplied by the contemporaneous mandatory set-aside rate. Notably, all coefficients obtained through IV estimation are larger than our fixed effects estimates in Table 4. One potential explanation for this is that the provision of randomly-selected additional hectares of set-aside by farmers could be more beneficial in terms of biodiversity compared to the mandatory set-aside rates alone. The latter are reflected by the fixed effects results, which could contain some selection within the scope of the farm's UAA. Nonetheless, we test for the difference between the fixed effects and IV coefficients using paired t-tests with unequal variances, and are unable to reject the null of coefficients' equality in all but two instances (the positive impact on non-Farmland birds'

abundance and the negative effect on Farmland Specialists’ species richness). All other IV coefficients are not statistically different to the fixed effects coefficients: they are larger but estimated more imprecisely. In conclusion, without strong evidence of selection effects, we adopt the more conservative fixed effects estimates of set-aside’s impacts on biodiversity as our main results. Again, bootstrap resampling to test for familywise error rates in a framework of multiple hypotheses being tested does not alter the significance of the results, reassuring us about the robustness of our findings.

#### 4.2. Non-linear and threshold effects in the responses of biodiversity to set-aside

The results from the inclusion of a variable that is non-parametric in set-aside, estimated using Eq. 2, suggests evidence of up to two set-aside area thresholds. Figure 4 and 5 show the estimated parameters on each of the set-aside bins, which express a measure of change in species abundance (Figure 4) and richness (Figure 5) for each bird group compared to the baseline category (1-200ha). For abundance and richness, there is a minimum threshold of 200ha below which the effect of set-aside is zero. Positive effects are evidenced for bins greater than 200ha compared to the 1-200ha baseline. Furthermore, Figure 4 and 5 show patterns of non-linear impacts when moving from the lowest (300ha) to highest (900ha) threshold.

For abundance, non-linear impacts are evident across all bird groups, although Farmland populations (Figure 4c) experience higher rates of increase compared to Non-farmland populations (Figure 4b). The 300ha bin, and all bins up until 900ha, generate positive impacts on abundance of around 5-15% depending on bird group, which represents an increase of 120-350 birds in a grid cell. The estimated parameters of set-aside bins in the 300-900ha range are not (statistically) significantly different from one another (Table A14). Beyond 900ha, the last bin parameterised, a larger impact of 15-25% is estimated. This estimate is significantly different from the parameter estimates on the smaller-sized bins (Table A14), implying evidence of a non-linear, possibly threshold-type effect.

In contrast to abundance, and consistent with our results in Table 4, the patterns of impact for species richness (Figure 5a) are generated solely from additional Non-farmland species (Figure 5b). We also find that, when moving from the 300ha to the 900ha bin, the positive effect of set-aside on species richness trends upwards, translating into one to two additional species observed in the landscape. The positive effects for all bins in the 300-900ha range are not significantly different from one another (Table A14). When set-aside exceeds 900ha, species richness rises to four additional species, an increase of around 30-35%. This estimate is significantly different from smaller-sized bins (Table A14). These results are suggestive of a threshold effect: additional set-aside does not lead to additional species once the minimum threshold has been passed until set-aside exceeds 900ha. Such large scale areas of set-aside existed mostly in the centre and east of England (Figure A6).

Consistent with our linear models (Table 4), we find null results for diversity (Fig-

ure A7). Results are qualitatively similar for all biodiversity measures when we use a 1-100ha baseline category instead of 1-200ha (Figure A8-A10).

### 4.3. Short- and long-run biodiversity impacts

Results from our dynamic panel analysis, estimated using Eq. 3, emphasise the importance of a sustained, long-run policy response to biodiversity loss. Table 6 summarises the results of our analysis in which the stock of biodiversity in the previous period is used as an explanatory variable for the current period stock. The marginal effects of set-aside on biodiversity estimated in Table 4 are disentangled into short- and long-run equilibrium effects. Compared to the Table 4 estimates, the short-run effects of set-aside on abundance in Table 6 are smaller and not statistically significant. These effects increase in magnitude in the long run, and are close to the Table 4 impact estimates, but again are not statistically significant.

The short-run effects on species richness in Table 6, particularly those for All Species, Non-farmland species and Farmland Generalists, are statistically significant at the 5% level. These effects are also smaller than the corresponding Table 4 estimates yet are relatively strong compared to those for abundance. Set-aside's impacts on species richness again build in the long-run, with estimates approximating to the Table 4 estimates, to an equilibrium 24%, 26%, and 17% (all statistically significant at the 5% level) larger than the short-run effects for All Species, Non-farmland and Farmland Generalist species, respectively. For All Species, these results translate into an average of two additional species in the short run, rising to almost three species in the long run. Table 6 suggests that the results for diversity are in line with those in Table 4.

Results for the lagged dependent variable in Table 6 show that current biodiversity stocks are determined by those in the previous period, for all biodiversity measures and bird groups (all statistically significant at the 1% level). These results imply that changes to biodiversity today would affect biodiversity in subsequent years as well. Levels of abundance report the strongest dependence on levels in the previous period: 44% of current abundance for All Species down to 28% of current abundance for Farmland Specialists. For species richness and diversity the dynamic dependence is smaller, with ranges of 15-21% and 19-23%, respectively. Thus, levels of richness and diversity in the current period are less dependent on levels in the previous period and hence, less persistent than abundance.

To illustrate the need for the maximum likelihood approach to remove Nickell bias (Nickell, 1981), we run fixed effects regressions with a lagged-dependent variable. Several models are presented in Table A15-A17 with different controls for panel and year fixed effects and year-region interactions. The results indicate Nickell bias in the expected direction: downward bias for the autocorrelation coefficient ( $\gamma$ ) and upward bias for the contemporaneous effect ( $\beta$ ). Qualitatively, the results are quite similar, however.

#### 4.4. Set-aside’s agricultural impacts

Estimated using the specification described in Eq. 4, our results for set-aside’s agricultural impacts suggest evidence that set-aside is, as intended, responsible for a steep decline in cereal production. Figure 6 shows the results for set-aside’s impacts on land under cereal production, cereal output and yield. The magnitude of the effect is shown on the y-axis. The x-axis indicates the baseline period against which the average impact of set-aside over the duration of the scheme (1992-2007) is estimated. For instance, the impact for 1981 is the average impact of set-aside compared to the baseline period 1981-1991. Figure 6 shows that the estimated impact depends on the baseline period against which it is measured reflecting changes in agricultural production, land-use and yield over time. For 1981, we observe a mean impact of a 17% decline in output when the baseline period begins in 1981, falling to 10% when it begins in 1990 (Figure 6a).

The estimated effect is smaller when the baseline period contains fewer and more recent pre-set-aside years, reflecting changes in cereal production since 1981, due to technological change and increased allocations of land to cereals. For both output (Figure 6a) and land (Figure 6b), and for all years, the impacts of set-aside are significantly different from zero. The impact on land slightly exceeds the decline in output leading to a positive effect on yield (Figure 6c). This indicates that farmers were mostly, but not always, setting aside marginal land. Our finding is reflected in the 1% to 3% increase in yield on cereal land remaining under production during the set-aside period, a result that is, however, not statistically different from zero (Figure 6c). The wide confidence intervals are suggestive of a wide variety of responses to the set-aside scheme, with some farmers setting aside highly productive land, and others more marginal, low-productivity land. On average however, the results suggest a clear trade-off between the (albeit sought after) reduction in aggregate cereal production and the gains in biodiversity.

## 5. Discussion and Conclusion

Both in, and outside of, the context of the Kunming-Montreal Global Biodiversity Framework, the implementation of proposals to expand area-based conservation measures and conserve biodiversity in agricultural landscapes is likely to involve biodiversity-food trade-offs (Mehrabi *et al.*, 2018; Usubiaga-Liaño *et al.*, 2019). Robust evidence on the extent of such trade-offs when agricultural land is taken out of production remains scarce. To fill the evidence gap, we estimated the ecological and economic responses to the 1992-2007 EEC set-aside scheme using panel data on land uses, bird species and cereal production in England. Although it aimed to reduce cereal output and was not explicitly designed to conserve biodiversity, this scheme in effect introduced an uncoordinated network of small habitat reserves within privately-owned agricultural land across the country.

The impacts of set-aside on species abundance in our fixed effects models (Table 4)

were positive and consistent with results from previous studies at the plot level (Henderson *et al.*, 2000; Firbank *et al.*, 2003; Gillings *et al.*, 2010). We also found positive impacts of set-aside on species richness, particularly Non-farmland species. Given that our results for species richness represent averages over the set-aside period across England, we were unable to identify which species benefited from set-aside.

In contrast to our instrumental variables results (Table 5), the fixed effects results should be interpreted as conservative estimates of set-aside’s impacts on biodiversity. The potential selection issue concerned how much more (or less) land a farmer opted to set aside than was mandated, and whether this decision was correlated with biodiversity. The larger IV results suggest that there might have been a negative bias arising from this selection effect, with farmers setting aside land which was on average less suitable for biodiversity. The implication is that had farmers provided additional set-aside randomly on their farms, the marginal effects on biodiversity would have been larger. This result could be a signal that the targeting of set-aside land for biodiversity would lead to better biodiversity outcomes. Yet ultimately, the IV effects were, for the most part, non-significantly different to the fixed effects results, so the evidence for a systematic bias was not compelling.

Positive impacts of set-aside on species abundance and richness first materialised at certain thresholds of set-aside, specifically when cumulative set-aside land accumulated beyond 200ha, approximating to 3% of mean UAA. Thus, for a set-aside policy to be effective, a sizable minimum scale is required. As more land was set-aside within landscapes, set-aside areas were more likely to be both closer to one another and comprise whole fields in addition to field margins. Yet, between the 200ha the 900ha thresholds, we did not observe statistically significant changes in abundance and richness compared to the effect at 200ha. Above 900ha, we found evidence of a jump in abundance and richness, suggestive of a scale effect.

The 900ha+ set-aside threshold comprised almost 13% of average UAA over the study period, just under half of the 30% target in the Kunming-Montreal Global Biodiversity Framework (Waldron *et al.*, 2020; UNEP, 2022). It is also lower than thresholds of species abundance and richness estimated in studies of habitat loss, which suggests convergence towards thresholds of around 30-40% (of original habitat) for many species, including birds (Andr n, 1994; Muradian, 2001; Banks-Leite *et al.*, 2014; Yin *et al.*, 2017). Although 900ha is still a considerable amount of land to remove from production in an agricultural landscape, our relatively low estimated threshold of 13% is possibly due to differences in habitats, ecosystems and species under study. Previous research that analysed habitat loss to predict restoration thresholds examined tropical ecosystems with far larger numbers of species, which are potentially more sensitive to land-use change, in contrast to England’s highly-modified agricultural ecosystems with its more adaptable species. Moreover, land set-aside in England was not, from an ecological perspective, actively restored.

The results from our dynamic models (Table 6) suggest that set-aside had neither a (statistically significant) short- nor a long-run effect on abundance despite the

relatively strong dependence of current populations on previous populations. This implies that the positive impact estimated in [Table 4](#) resulted from a slow response of bird populations to the expansion of set-aside. By contrast, species responded relatively quickly to changes in set-aside, with a strong effect observed in the short-run. However, species richness was only weakly dependent on the level of species richness in the previous period and hence, this strong short-run effect only led to a slightly larger long-run effect. Indeed, the magnitudes of effect estimated in [Table 4](#) for species richness are very close to the long-run effects estimated in [Table 6](#). These results highlight the importance of policies with the capacity for sustainable impacts on biodiversity over longer periods of time.

Agricultural land set-aside has implications for species other than birds, with previous research suggesting that the population density and species richness of insects, spiders and plants increased with set-aside ([Van Buskirk and Willi, 2004](#)). Such benefits for biodiversity could be enhanced with greater coordination over the spatial distribution of set-aside leading to contiguous habitats ([Kremen and Merenlender, 2018](#)). A lack of data on the spatial distribution of set-aside areas in our study unfortunately limits further analysis with respect to habitat contiguity. Irrespective of habitat contiguity, we found evidence that greater diversity of land uses, including set-aside, is positively associated with our biodiversity measures, a result that is in line with recent research on food crop diversity and bird diversity undertaken in the US ([Strobl, 2022](#)).

A new agricultural land set-aside scheme with biodiversity conservation aims could target areas of agricultural land to be set-aside according to the spatial distribution of biodiversity. In the context of maintaining global food supplies such a scheme is highly unlikely have a goal to reduce food output. Yet, biodiversity targeting to meet the scheme's conservation aims could lead to the setting aside of both low- and high-yielding arable lands. Targeting high-yield lands, in particular, carries the risk of negative impacts on food supply. To minimise the risk of food output decline, a new set-aside scheme could include an element of rotation, to help sustain soil fertility and hence, manage this risk. Given that set-aside land was rotated and returned to production in the duration of the EEC scheme, our analysis shows the extent of food output decline. Compared to the pre-1992 baselines, or counterfactuals, the set-aside scheme led to a 10% to 17% decline in output between 1992 and 2007. That this decline was exceeded by the amount of land set-aside implies that the scheme primarily affected low-yield lands.

Our study focused on arable land, which constitutes over half of England's agricultural land. Set-aside could also target the other half of the country's agricultural land, which is utilised for livestock production and has a potentially smaller food impact in terms of calories. That said, more than one third of all crop calories produced are currently fed to animals, with only 12% of these feed calories coming back as human food ([Cassidy \*et al.\*, 2013](#)). Also, the biodiversity benefits of setting aside livestock areas are likely to differ, and any set-aside policy needs to consider how biodiversity is distributed across the landscape. Unless livestock areas, which in England tend to be uplands unsuitable for crop cultivation, are correlated with biodiversity benefits



then it is likely that arable areas also need to be targeted for removal from production.

The UK has a highly-productive agricultural sector, particularly with respect to crop cultivation (Thirtle and Bottomley, 1992; Amadi *et al.*, 2004). In other settings, with greater scope for increasing agricultural productivity, mandating farmers to set-aside land from production could be a useful tool for incentivising higher yields on land remaining under production, potentially attenuating the loss of output. Higher yields are likely more achievable - and have the potential to generate fewer negative externalities than lower-yield systems (Balmford *et al.*, 2018) - in settings where the agricultural landscape is still dominated by low yields. For example, in the Brazilian Amazon where research suggests evidence of higher-productivity outcomes caused by environmental regulations, (e.g. Koch *et al.*, 2019; Moffette *et al.*, 2021). Specifically, Brazil's Forest Code mandates that private landowners retain a minimum of 80% of their land in forest, with options to restore previously deforested land. Yet, compliance with this requirement is, in places, patchy at best (e.g. Azevedo *et al.*, 2017; Fontes and Palmer, 2018). Also, farmers are not compensated for production losses unlike in the EEC scheme, although incomplete data precludes a consideration of the extent to which farmers' production losses were adequately compensated (and more broadly, the economic costs of the scheme).

After the end of mandatory set-aside in 2007, some farmers continued to set-aside land on a voluntary basis in exchange for compensation when they participated in follow-up AES under the CAP. Although set-aside did not completely disappear it never operated at the scale observed during the period of mandatory set-aside. Given the observed scale effect, it is unlikely to have had the impacts that we observed.

Post-Brexit, there is scope for re-designing the UK's agricultural policy to arrest the continuing decline in biodiversity while minimising risks to the country's food security. Beyond 2022, the UK government's 25-year Environment Plan suggests that financial support to farmers will be made conditional on their environmental efforts (Bateman and Balmford, 2018). A mandatory set-aside scheme could target biodiversity hotspots, where species are in decline and where there is an abundance of marginal land. Set-aside could also be targeted at those crops, e.g. maize, winter barley, with the most negative impacts on bird populations and other taxa in order to generate a potential 'double-dividend' being mindful of potential production losses too.

More broadly, our study has demonstrated that conserving biodiversity in agricultural landscapes involves clear trade-offs. A loss of agricultural output in exchange for conserving biodiversity endemic to a particular area might be worthwhile if the loss could be attenuated through intensification or food imports, although we note the latter has the potential to induce trade-related spillovers to exporting nations. Additional costs, transaction costs, will be incurred depending on the complexity of conservation measures, which in turn, will depend on the conservation aims. Our study has shown how a simple set-aside policy, with no explicit conservation aims and where farmers are free to choose which land to set-aside, can have positive impacts

on biodiversity. The implementation of new area-based conservation measures in agricultural landscapes needs to go hand-in-hand with clear conservation aims, in particular, with respect to different measures of biodiversity. Not all measures of biodiversity were responsive to set-aside in our study, so agreement on the appropriate policy target is critical, particularly when there is potential for trade-offs. Whatever the measure of biodiversity applied though, a more sophisticated, targeted policy could generate even greater biodiversity benefits, albeit at the risk of higher transaction costs.

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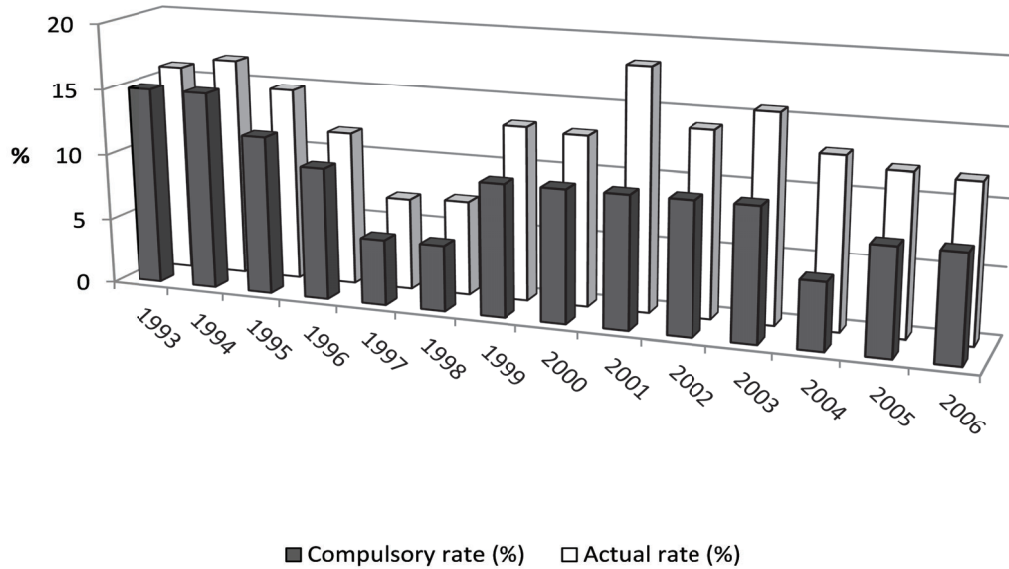
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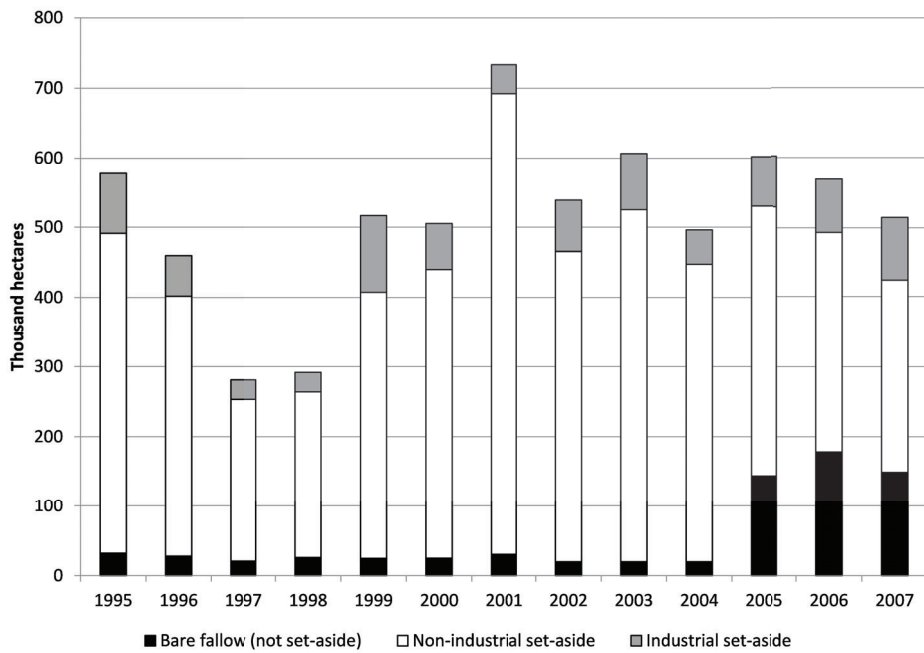
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## Figures

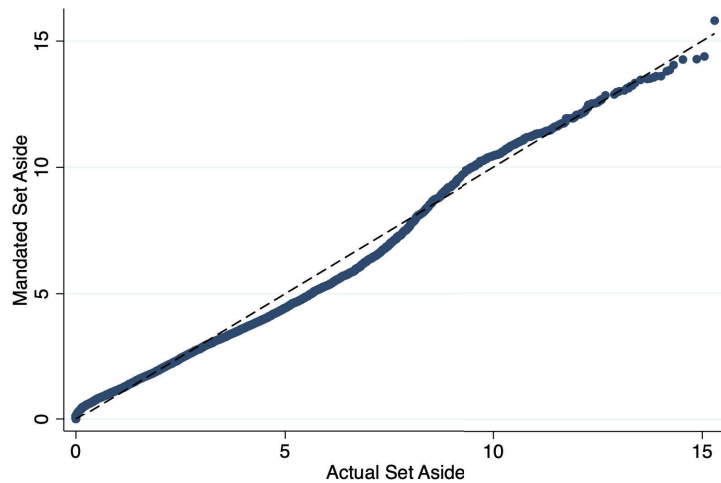


**Figure 1: Compulsory and actual rates of EEC set-aside in the UK, 1993-2006.**  
*Note:* The compulsory set-aside rate in 2007 is the same as in 2006; the actual rate in 2007 is unknown. Source: Department for Environment, Food and Rural Affairs (DEFRA) (DEFRA, 2007)

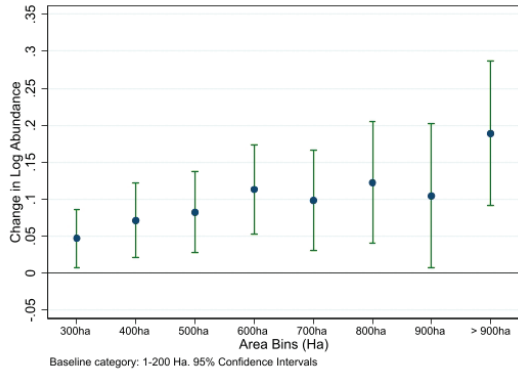




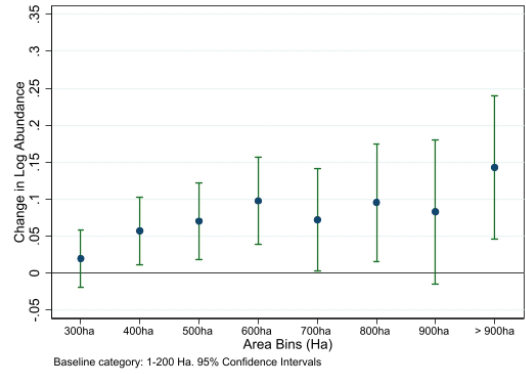
**Figure 2: Areas of set aside and bare fallow in England ('000 ha), 1995-2007.** *Note:* Set-aside/fallow split in 2005-7 is arbitrary. Data are unavailable for 1993-94. Industrial set-aside includes biofuel feedstocks. *Source:* June Survey, except for industrial crops (from Single Payment Scheme/Integrated Administration and Control System).



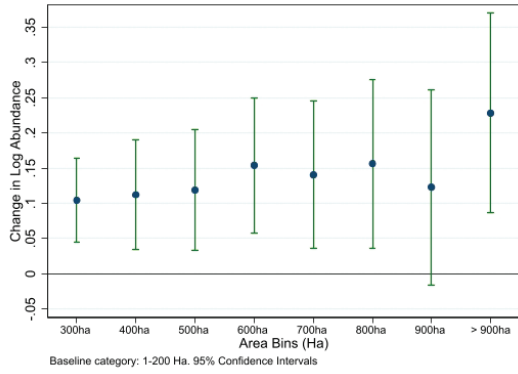
**Figure 3: Quantile-quantile plot of actual vs. mandated set aside (hectares)**



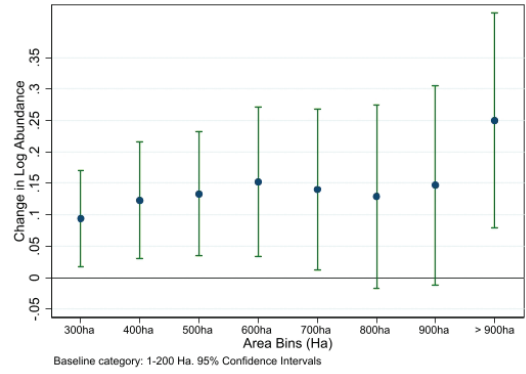
(a) All Species.



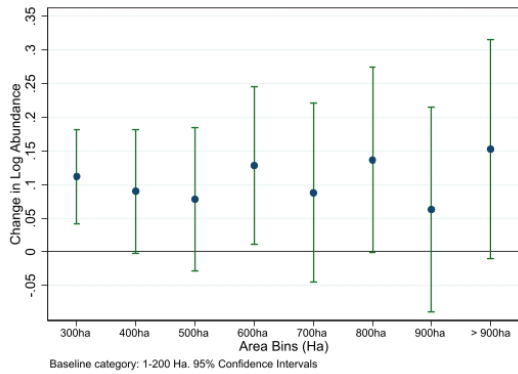
(b) Non-farmland



(c) Farmland

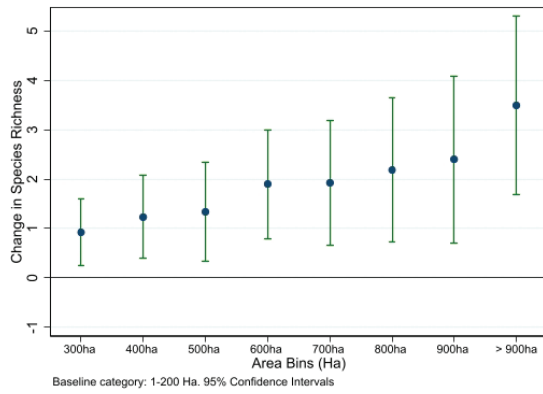


(d) Farmland Generalists

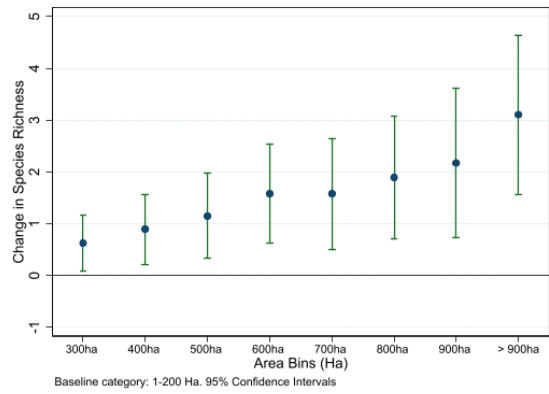


(e) Farmland Specialists

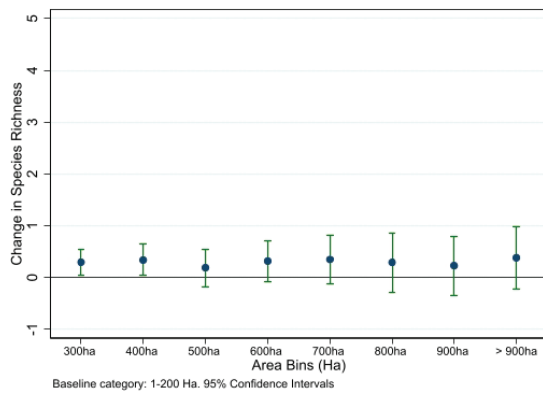
**Figure 4: The impact of set-aside on species abundance: Non-parametric estimation results.** *Note:* 300ha area bin denotes set-aside range of 201ha to 300ha; 400ha denotes 301-400ha; 500ha denotes 401-500ha; 600ha denotes 501-600ha; 700ha denotes 601-700ha; 800ha denotes 701-800ha; and, 900ha denotes 801-900ha.



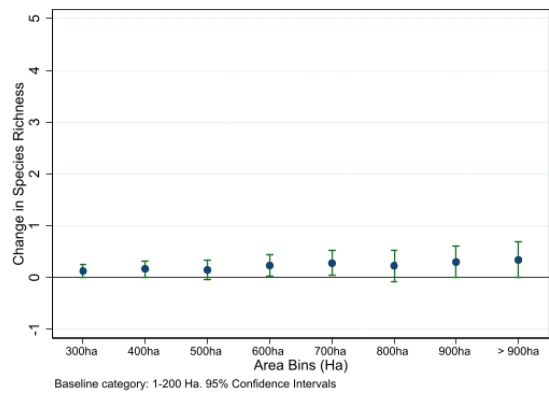
(a) All Species



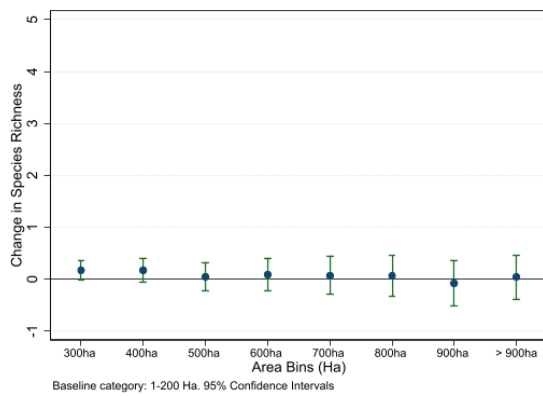
(b) Non-farmland



(c) Farmland

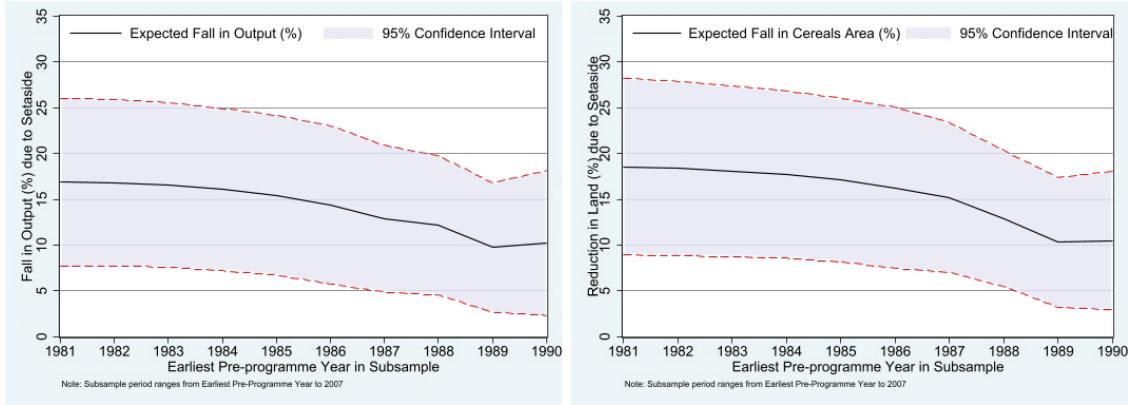


(d) Farmland Generalists



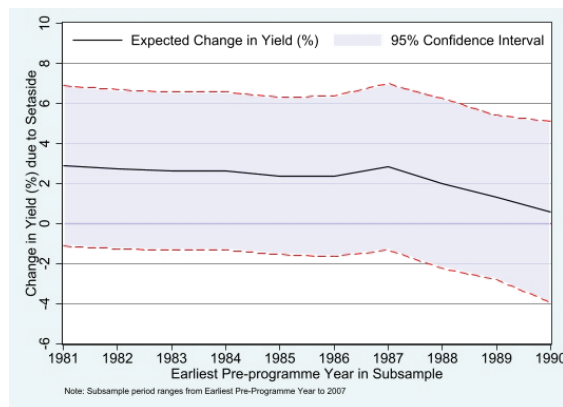
(e) Farmland Specialists

**Figure 5: The impact of set-aside on species richness: Non-parametric estimation results.** *Note:* 300ha area bin denotes set-aside range of 201ha to 300ha; 400ha denotes 301-400ha; 500ha denotes 401-500ha; 600ha denotes 501-600ha; 700ha denotes 601-700ha; 800ha denotes 701-800ha; and, 900ha denotes 801-900ha.



(a) Cereal output

(b) Cereal area



(c) Cereal yield

**Figure 6: Impact (%) of set-aside on cereals output, land under cereals, and yield by starting year.** *Note:* The solid line shows the coefficient estimates of the average impact (y-axis) for the set-aside period (1992-2007) compared to the period before, that is, the starting year until 1990 (x-axis). The latest starting year is 1990, allowing for a minimum of two years of data in the baseline.

## Tables

**Table 1: Descriptive statistics for June Survey land-use data**

	<i>Pooled sample (1994-2007)</i>				
	Mean	Std.Dev.	Max	Min	Obs
<b><i>Set-aside (per 100ha)</i></b>					
Set-aside	3.74	2.67	15.06	0.01	4294
<b><i>Fractionalisation of land</i></b>					
Fractionalisation	0.70	0.16	0.92	0.04	4294
<b><i>Land uses (per 100ha)</i></b>					
Wheat	15.20	12.26	54.91	0.00	4294
Maize	0.90	1.14	8.01	0.00	4294
Winter barley	3.87	3.31	26.13	0.00	4294
Spring barley	1.94	2.08	18.89	0.00	4294
Oats	0.67	0.77	5.98	0.00	4294
Other cereal	0.09	0.18	2.28	0.00	4294
Temporary grass	4.78	3.75	22.80	0.00	4294
Other grass	22.56	17.26	128.09	0.79	4294
Woodland	2.29	1.91	14.35	0.04	4294
Other land	1.54	0.90	15.79	0.03	4294
Oilseed rape (Osr)	3.26	3.29	23.64	0.00	4294
Osr set-aside	0.24	0.55	6.07	0.00	4294
Peas and beans	1.91	1.91	13.78	0.00	4294
Stockfeed	0.23	0.31	6.71	0.00	4294
Sugarbeet	1.28	2.87	18.83	0.00	4294
Fallow	0.40	0.62	6.69	0.00	4294
Linseed set-aside	0.01	0.06	0.80	0.00	4294
Linseed	0.55	0.87	13.53	0.00	4294
<b><i>Total Area</i></b>					
Utilised Agricultural Area (UAA)	70.40	19.75	129.50	15.74	4,294
<b><i>Sampling (No. of 1km-by-1km transects in 10-by-10km landscapes)</i></b>					
Count of BBS transects (n)	2.43	1.56	14.00	1.00	4294

*Note:* The summary statistics are for the balanced panel dataset used in the empirical analysis. Data are described at the 10km-by-10km landscape scale. The variable *Count of BBS transects (n)* represents the number of 1km-by-1km transects from the BBS that are contained in each 10km-by-10km landscape. The averages are taken over the entire dataset. *Fractionalisation* varies between 0 and 1, with 1 being the most fractionalised, and 0 being the least, e.g. a mono-culture.

**Table 2: Descriptive statistics for the Breeding Bird Survey bird count data**

	<i>Pooled sample (1994-2007)</i>				
	Mean	Std.Dev.	Max	Min	Obs
<i>Abundance</i>					
All	235.20	97.60	1216.00	30.00	4294
Non-Farmland	149.24	70.96	1095.00	7.00	4294
Farmland	90.72	50.86	773.00	2.50	4294
Farmland Gen.	53.74	38.87	735.00	0.00	4294
Farmland Spec.	36.98	26.93	408.00	0.00	4294
<i>Species richness</i>					
All	44.71	10.97	82.00	10.00	4294
Non-Farmland	33.20	9.25	65.00	5.00	4294
Farmland	11.51	2.97	19.00	1.00	4294
Farmland Gen.	4.54	1.25	7.00	0.00	4294
Farmland Spec.	6.97	2.12	12.00	0.00	4294
<i>Diversity (Shannon-Wiener Index)</i>					
All	3.15	0.30	3.88	0.97	4294
Non-Farmland	3.00	0.30	3.69	1.11	4294
Farmland	1.79	0.34	2.58	0.00	4294
Farmland Gen.	0.98	0.32	1.76	0.00	4294
Farmland Spec.	1.47	0.43	2.30	0.00	4294

*Notes:* The summary statistics are for the balanced panel dataset used in the empirical analysis. The bird data are collected in 1km-by-1km transects. The June Survey land-use data are only available at the 10km-by-10km landscape scale and hence the BBS data are extrapolated to enable empirical analysis at the 10km-by-10km landscape scale. Where more than one 1km-by-1km transects have been sampled in a landscape, the average of abundance and species richness is estimated. These averages are assumed to be representative of the landscape and are used to calculate estimates of the Shannon-Wiener index.

**Table 3: Descriptive statistics for the Farm Business Survey cereals data**

	<i>Pooled sample (1981-2007)</i>				
	Mean	Std.Dev.	Max	Min	Obs
Sum of all cereal tonnes	591.51	521.86	3164.40	10.00	13960
Sum of all cereal hectares	86.49	69.22	392.04	15.04	13960
Yield (tonnes per hectare)	6.56	1.38	10.27	1.78	13960

*Note:* The summary statistics are for the panel dataset of farms that is observed both before and after the onset of the mandatory set-aside scheme in 1992-1993. The data are winsorised (95%).

Table 4: The impact of set-aside on three measures of biodiversity

	All	Non-farm	Farm	Farm Gen.	Farm Spec.
Abundance	0.0174*** (0.00540)	0.0124** (0.00575)	0.0192*** (0.00730)	0.0205** (0.00845)	0.00921 (0.00846)
Species richness	0.302*** (0.110)	0.295*** (0.0973)	0.00693 (0.0333)	0.0287 (0.0188)	-0.0217 (0.0233)
Diversity (Shannon-Wiener)	-0.00540 (0.00473)	0.00654 (0.00433)	-0.00810* (0.00480)	-0.00224 (0.00465)	-0.00455 (0.00518)
Grid FE	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes	Yes	Yes	Yes
Region by Year FE	Yes	Yes	Yes	Yes	Yes
N (Abundance)	4255	4255	4255	4254	4239
R-squared (Abundance)	0.78	0.77	0.77	0.73	0.73
Romano-Wolf p-value (Abundance)	0.001***	0.014**	0.004***	0.007***	0.179
N (Richness)	4255	4255	4255	4255	4255
R-squared (Richness)	0.84	0.84	0.77	0.64	0.74
Romano-Wolf p-value (Richness)	0.002***	0.001***	0.7692	0.0799*	0.2907
N (Diversity)	4255	4255	4255	4255	4255
R-squared (Diversity)	0.64	0.67	0.60	0.58	0.66
Romano-Wolf p-value (Diversity)	0.3327	0.1469	0.1049	0.5554	0.4436

Note: P-values in parentheses. Estimates from fixed effects models estimated using Stata's `reghdfe` routine. The panel is balanced and has 13 years of data. Using the Harris-Tzavalis panel unit root test, the null hypothesis that the bird species and set-aside data are non-stationary and possess a unit root is rejected (Table A9). Two-way clustering at the region and year level. There are 24 regions and 309 clusters in total. Results are robust to clustering by region alone, 10km-by-10km landscape, BTO region, and two-way clustering with year.

\*  $p < 0.10$ , \*\*  $p < 0.05$ , \*\*\*  $p < 0.01$



Table 5: IV regressions, total cereals at  $t - 1$

	(1)	(2)	(3)	(4)	(5)
	All	Non-farm	Farm	Farm Gen.	Farm spec.
Abundance	0.0354** (0.0166)	0.0499*** (0.0185)	0.00702 (0.0223)	0.0284 (0.0291)	-0.00541 (0.0285)
Species richness	0.429 (0.344)	0.492* (0.298)	-0.0632 (0.0991)	0.0628 (0.0564)	-0.126* (0.0707)
Diversity (Shannon-Wiener)	0.00950 (0.0143)	0.0115 (0.0139)	-0.00145 (0.0141)	-0.00560 (0.0153)	0.00249 (0.0160)
Controls	Yes	Yes	Yes	Yes	Yes
Grid FE	Yes	Yes	Yes	Yes	Yes
Region-Year FE	Yes	Yes	Yes	Yes	Yes
p-value: $\beta_{IV} = \beta_{FE}$ (Abundance)	0.3025	0.0530	0.6037	0.7943	0.6229
p-value: $\beta_{IV} = \beta_{FE}$ (Richness)	0.7251	0.5298	0.5024	0.5663	0.0473
p-value: $\beta_{IV} = \beta_{FE}$ (Diversity)	0.3226	0.7334	0.6553	0.8336	0.6755
N (Abundance)	4229	4229	4229	4228	4213
KP F-stat (Abundance)	85.95	85.95	85.95	85.93	81.82
Romano-Wolf p-value (Abundance)	0.017**	0.003***	0.8821	0.4036	0.8821
N (Richness)	4229	4229	4229	4229	4229
KP F-stat (Richness)	85.95	85.95	85.95	85.95	85.95
Romano-Wolf p-value (Richness)	0.2208	0.0849*	0.3726	0.2298	0.0589*
N (Diversity)	4229	4229	4229	4229	4229
KP F-stat (Diversity)	85.95	85.95	85.95	85.95	85.95
Romano-Wolf p-value (Diversity)	0.3327	0.1469	0.1049	0.5554	0.4436

Note: Estimated in Stata 17 using the `ivreghdfe` routine. Standard errors clustered at the Region-Year level in parentheses. The Kleibergen-Paap F-statistic for weak IV is reported.

\*  $p < 0.10$ , \*\*  $p < 0.05$ , \*\*\*  $p < 0.01$

**Table 6: Dynamic impacts of set-aside on three measures of biodiversity**

	All	Non-farm	Farm	Farm Gen.	Farm Spec.
<b>Abundance</b>					
Set-aside (short-run)	0.00533 (0.208)	0.0059 (0.218)	0.00664 (0.249)	0.00687 (0.363)	0.00475 (0.489)
Lagged effect	0.443*** (0.000)	0.410*** (0.000)	0.343*** (0.000)	0.305*** (0.000)	0.277*** (0.000)
Set-aside (long-run)	0.0096 (0.204)	0.01 (0.213)	0.0101 (0.246)	0.0099 (0.361)	0.0066 (0.488)
<b>Species richness</b>					
Set-aside (short-run)	0.217** (0.019)	0.181** (0.026)	0.036 (0.159)	0.0304** (0.029)	0.00331 (0.862)
Lagged effect	0.196*** (0.000)	0.205*** (0.000)	0.182*** (0.000)	0.148*** (0.000)	0.194*** (0.000)
Set-aside (long-run)	0.2694** (0.02)	0.2275** (0.027)	0.044 (0.159)	0.0357** (0.029)	0.0041 (0.862)
<b>Diversity (Shannon-Wiener)</b>					
Set-aside (short-run)	0.000143 (0.967)	0.00508* (0.099)	0.000985 (0.805)	0.00258 (0.454)	0.000249 (0.951)
Lagged effect	0.187*** (0.000)	0.221*** (0.000)	0.23*** (0.000)	0.207*** (0.000)	0.217*** (0.000)
Set-aside (long-run)	0.0002 (0.967)	0.0065 (0.102)	0.0013 (0.805)	0.0033 (0.454)	0.0003 (0.951)

*Note:* The number of observations in each bird group is the same as reported in [Table 4](#). Short-run denotes the instantaneous effect of set-aside. Long-run denotes the difference between the pre- and post-intervention equilibrium of a mean reverting process. The results are generated by the maximum likelihood estimator of [Moral-Benito et al., 2019](#) with one lagged-dependent variable and all other variables assumed exogenous. This estimator is asymptotically equivalent to the Arellano-Bond estimator. We use the maximum likelihood routine in Stata (`xtgls`). Each model contains year and panel fixed effects. All standard errors and associated p-values are robust. The control variables (not shown) are: Fractionalisation, wheat, maize, winter barley, summer barley, and the level and square of the number of BBS sample grids in the landscape. The extent of Nickell bias ([Nickell, 1981](#)) when a lagged-dependent variable is included in our fixed effects models is shown in [Table A15-A17](#), hence justifying a maximum likelihood approach.