Economic incentives for woodland creation on farmland: modelling the impacts on biodiversity.

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Abstract

This paper models the effects of economic incentives on woodland planting on UK farmland, and the spatially-varying impacts on three avian species. The economic model uses an agent-based approach: "farmers" in each parcel compare economic returns from keeping their current agricultural land use with the economic incentive for woodland planting. An ecological model then predicts the effects of both parcel-level and local landscape-level woodland cover on species distributions. We compare results from two case study areas, which vary in terms of the spatial correlation of opportunity costs and ecological potential. As the per-hectare value of this subsidy is increased, the values of our biodiversity indicator increase, but at rates which vary by case study region and by species. Cost-effectiveness of the economic instrument varies according to the sign of the spatial correlation between opportunity costs and ecological potential.

Keywords: Ecological-economic modelling, biodiversity, economic incentives, forest biodiversity, agent-based modelling.

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Introduction

Globally, afforestation and reforestation have long been acknowledged to be important options for mitigating the effects of climate change (Austin et al, 2020). In the UK, mitigation from increasing woodland potential is constrained (Bradfer-Lawrence et al., 2021) by a range of existing land uses, notably that over 70% of the land area is currently agriculture (Westaway et al., 2023). Future planting of woodland will thus likely need to occur on farmland. The UK government and the devolved administrations (of Scotland and Wales) have set ambitious targets for woodland creation, with between 15,000 and 30,000 new hectares of planting per annum included as net-zero policy targets[2]. Central to the achievement of these policy objectives is the supply of land suitable for woodland creation, with low-productivity agricultural land identified as one possible area with relatively low opportunity costs (Flack et al, 2022). Sufficient economic incentives will need to be offered to private landholders to enrol enough land to ensure that woodland creation targets can be met (de Vries and Hanley, 2016). Moreover, the impacts of higher woodland planting on the suite of biodiversity targets which the UK has committed itself need to be considered (Finch et al, 2023): despite gradual increases in UK woodland cover over the past century, many species continue to decline, and the biodiversity value of newly planted woodlands remains largely unknown (Fuentes-Montemayor et al, 2015).

In this paper, we combine ecological and economic modelling approaches to analyse the effect on biodiversity of economic incentives for the conversion of agricultural land to woodland. We model the decision of landholders to enrol agricultural land parcels into a woodland planting scheme at varying incentive rates. We then apply an ecological occupancy model to predict the presence/absence of three exemplar bird species within this newly-created woodland. These species are Long-tailed Tits *Aegithalos caudatus*, Treecreepers *Certhia familiaris* and Yellowhammers *Emberiza citronella*. Based on their ecologies, we hypothesized that these bird species would display a range of responses to varying woodland planting levels, both at the level of the individual land parcel within which woodland is created, but also to varying landscape-level woodland cover and arable area (Bellamy et al, 2000; Kampfer et al, 2022; Petit and Landis, 2023). Our ecological model, reported later in the paper, fails to reject these hypotheses. These bird species may therefore represent trajectories for a wider set of species which vary in their response to higher woodland cover.

[2] Although these targets have all been missed so far: Forest Research, 2023.

As the subsidy payment increases from the baseline level, we demonstrate the extent of woodland planting incentivised by these payments, how the location of this planting changes with increasing subsidy rates and – crucially – the extent of ecological benefits in terms of increasing predicted species occupancy at both the parcel and landscape level. Ecological and economic impacts are compared across two case-study landscapes, one in Scotland and one in England, in which the spatial correlation between the foregone returns from agriculture (the opportunity costs of woodland planting) and ecological potential - the potential increase in species occupancy- differs between the case study regions from positive (Scottish case study) to negative (English case study). Since this spatial correlation indicates the alignment between the agricultural and conservation values of alternative land uses, we expect that the sign and size of this correlation will influence both the ecological and economic performance of any subsidy scheme, and thus its cost-effectiveness.

Our paper provides a series of contributions to the literature. First, we show the effects of economic incentives on biodiversity outcomes from woodland planting on farmland, where both parcel-level and landscape-level woodland variables help determine biodiversity outcomes. Second, we demonstrate how the ecological and economic effects of each subsidy rate vary across three bird species, and between two landscapes which differ in the degree to which higher ecological potential is spatially correlated with higher economic returns from farming. Previous literature on the importance of the sign of spatial correlation between conservation costs and ecological benefits includes Armsworth (2014), Babcock et al, (1997) and Simpson et al, (2022). In particular, Armsworth (2014) shows that whether conservation costs and benefits are negatively or positively correlated changes both the total benefits that can be obtained from a limited conservation budget, and the difference between alternative spatial prioritization rules in terms of total benefit generated. This important point is also made by Babcock et al, (1997), both conceptually and empirically in the context of the US Conservation Reserve Programme. Intuitively, if those sites which offer highest conservation benefits are also those with the lowest opportunity (conservation) costs, then greater net benefits can be derived from a fixed conservation budget than when the best sites in terms of ecological potential are also those, on the whole, which are most profitable for agriculture. Similar results were demonstrated by Simpson et al, (2021) in the design of UK biodiversity offset markets where the choice of the metric (habitat or species) resulted in significantly different ecological and economic outcomes due to the differences in predictable spatial relationships in observable variables (agricultural profits and development rents).

In what follows, we first describe the two case study landscapes, and the methods used in our ecological-economic modelling. We then present results in terms of both the ecological and economic impacts of gradual increases in an economic incentive for woodland planting.

2. Methods

Overall approach

We represent each of the two case study landscapes as a grid of 1km by 1km (100 hectare) parcels. Within these landscapes are a variety of land uses, broadly categorised as urban, agriculture and woodland. Our model focuses on changing land-use decisions in parcels currently being used (fully or partly) for agriculture. Each parcel is assumed to represent an agent (land manager) who decides how best to manage their parcel. We assume agents maximise profits, with the baseline land use for agricultural land parcels being either crop or livestock production according to survey data. Agents can choose to enrol land parcels into a woodland creation scheme and receive a subsidy payment for this. We model each agent as choosing the best use of their land by comparing the returns (profits) from maintaining current farming practices with those from accepting a subsidy for woodland planting. All land use decisions occur at the same point in time, and, by implication, predicted biodiversity responses occur with no time delay.

Changing land management decisions at the parcel level are expected to affect biodiversity outcomes, both within the parcel and across the surrounding landscape. Moreover, we expect parcel-level biodiversity response to depend at least partially on landscape-level land use context (Bradfer-Lawrence et al, 2023). To explore this, we used an ecological occupancy model to predict the presence/absence of three bird species for each parcel across the full landscape within each case study. This allows us to study how species with differing ecological requirements respond to both parcel-level and landscape-level land cover decisions. Thus, the agent-based model represents the economic choices of land managers, whilst the ecological model converts these land management decisions into predicted biodiversity outcomes.

Case study locations and data development

We apply our agent-based model in two UK case study areas (Figure 1). In Scotland, the case study is the watershed of the Forth Estuary, covering around 5,400 km^2. In England we use part of the English Midlands, with an area of around 11,000 km^2. These case study areas were chosen to cover the original sampling sites of the Woodland Creation and Ecological Networks

(WrEN) project, from which our biodiversity data is derived (Watts et al, 2016). Each landscape is then divided into 1 km by 1 km land parcels and georeferenced to Ordnance Survey British National Grid. We extracted 33 land use types, including improved grassland, arable (by crop type), coniferous woodland, broadleaved woodland and urban from the Land Cover Map (LCM2015) and Land Cover Plus Crops map (Rowland et al, 2017). Existing agricultural returns (gross margins) for current crop types are taken from agricultural census data, whilst data on costs and benefits per crop/livestock activity are taken from (SRUC, 2021). We adjust gross margin value calculations using Soilscapes[3] data to account for the effect of soil quality on parcel profitability. We thus incorporate heterogeneity in opportunity costs across the landscape based on current production choices of land managers, using real survey data. Figure 1 shows the current distribution of opportunity costs (=farm gross margin) across the two case study areas.

[3] Soils Data © Cranfield University (NSRI) and for the Controller of HMSO 2023 used with permission.

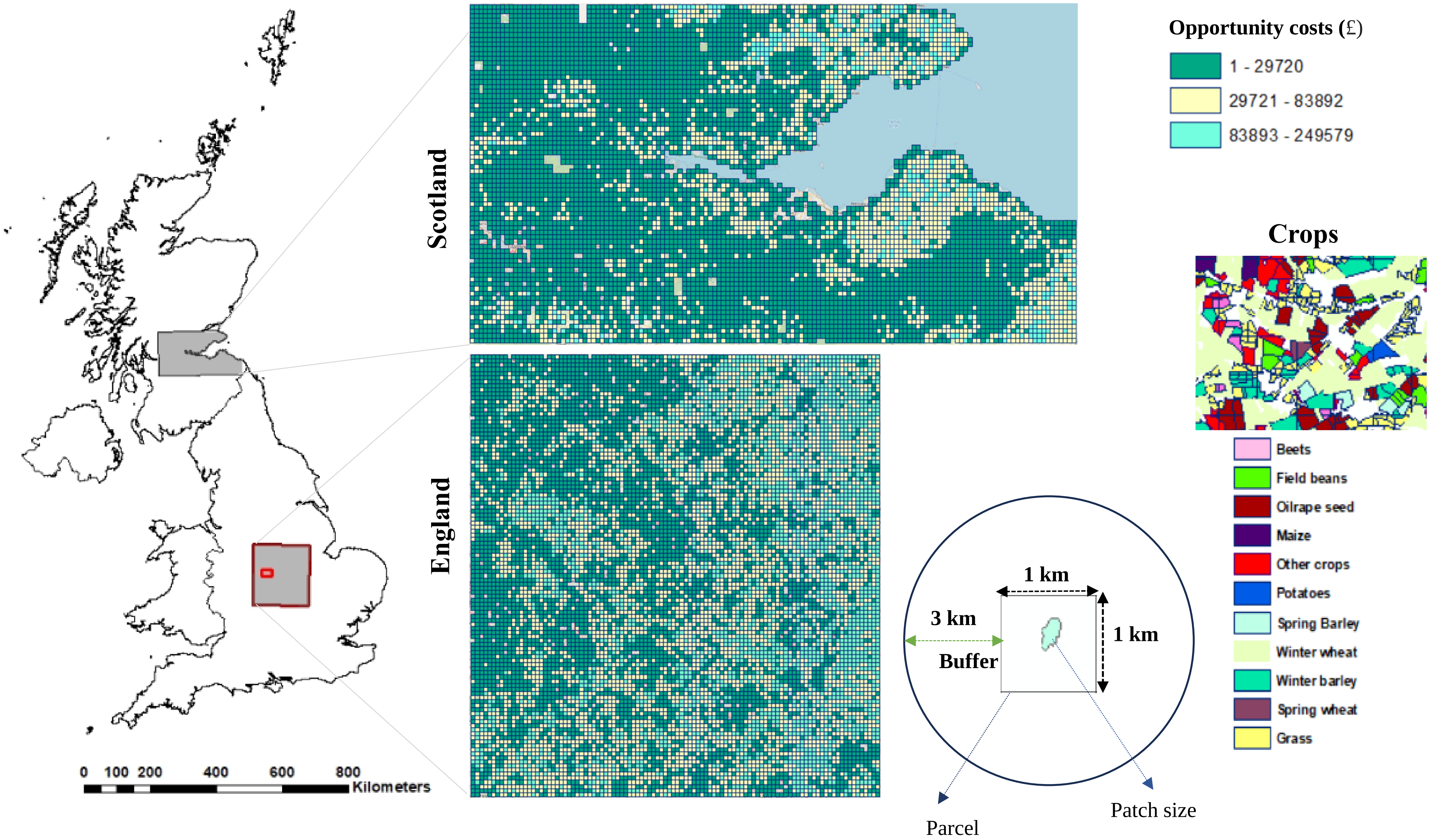

Figure 1: Map showing the geographical location and the spatial distribution of opportunity costs for the two-case study areas under current land use.

Agent-based model of land use choice

Our agent-based model is developed in Stata (Version 16) to represent farmers' land management choices based on the relative economic returns for switching from agricultural production to woodland creation under a uniform subsidy scheme[4]. By "uniform", we mean that each agent is offered the same fixed payment rate for converting agricultural land into woodland – payments are not differentiated according to location or opportunity cost. For a given financial value of this subsidy, the model determines whether an agent will enrol their land parcel in the woodland creation scheme, or else retain its current agricultural land use. We assume that the agent will choose to enrol their parcel where the subsidy offered per hectare of woodland planting exceeds the sum of the foregone agricultural returns per hectare (their opportunity cost) plus the cost of tree planting. We explore how woodland creation rates change as the uniform subsidy is increased for all landowners, tracing out a supply response for woodland creation; and how responses vary according to different upper limits being placed on the area of new planting allowed for each agent.

Under uniform subsidies, the least profitable land parcels are enrolled first (Iho et al, 2014). If opportunity costs vary across agents, such payments are not cost-effective and typically lead to overcompensation of all but the marginal farmer (the land manager for whom the subsidy just exceeds their opportunity cost plus planting cost) (Connor et al, 2008; Jack et al. 2008, Armsworth et al, 2012). However, the implementation of a differentiated payment scheme – where different subsidy rates are offered to different land managers for undertaking the same action based on their true opportunity costs – is often viewed as both unrealistic and politically difficult. In particular, the agency offering the payments is unlikely to have accurate information on how marginal costs vary across landowners (given that opportunity costs are private information); whilst policy makers may not like farmers to be seen to be offered different payments for taking the same actions (Armsworth et al, 2012).

The subsidy was initially set to be equal to an estimate of the mean opportunity cost per hectare across each case study area, plus an estimate of planting costs. No account is taken of the estimated future returns on investment from timber sales, or of potential tax benefits from woodland planting, or of future management costs for woodland plots. The average opportunity cost per hectare is £455 for the Scotland case study, and £445 per hectare for England. We

[4] Here we use the phrases "woodland planting" and "woodland creation" interchangeably. The incentive we model, strictly speaking, relates to the planting of trees on agricultural land. Woodland creation can also involve natural regeneration of previously-wooded areas, eg by removing stock and fencing sites.

assume that agents are also offered a planting cost of £500 per hectare, giving a total baseline subsidy amount of £955 per hectare for Scotland, and £945 per hectare for England. We simulate the agents' land use choices at 6 payment levels: the base payment rate, plus a 20%, 40%, 60%, 80% and 100% increase in the subsidy.

No woodland sites in the WrEN project[5] exceed 32 ha limiting the variable space over which the ecological model is estimated. We therefore constrain the amount of new planting that each agent can undertake in their parcel. Maximum planting rates per parcel are set at values of 2 ha, 5 ha and 10 ha in Policy Scenarios 1, 2 and 3 respectively. We also constrained the total woodland (existing woodland plus new woodland) for the parcels enrolled to be below or equal to 30 ha. For each policy scenario and each subsidy payment rate, the model identifies which land parcels are enrolled in the woodland planting scheme and which remain in agricultural production. We then map this change in woodland and agricultural covers across the landscape using ArcGIS. Within a parcel, woodland of either 2, 5 or 10 ha is created on the agricultural land cover within the parcel. Finally, we estimate the change in predicted occupancy of the three bird species in response to woodland planting using an ecological model.

Ecological modelling and biodiversity outcomes

Biodiversity outcomes are defined here in terms of probabilistic presence or absence of the three bird species within each $1km^2$ parcel. These probabilities are derived from a model reported in Bradfer-Lawrence et al (2023) and are functions of species-specific (i) intercepts, (ii) parcel-level woodland cover, and (iii) woodland and arable cover within a 3-km landscape buffer around each parcel. As agents change land use within the economic component of the model, predicted occupancy for each species responds according to whether woodland is created in a specific parcel, and to the amount of woodland and arable area within a 3 km local landscape[6]. Three indicator bird species which are woodland and hedgerow-affiliated are the focus of analysis: Long-tailed Tit (*Aegithalos caudatus*), Treecreeper (*Certhia familiaris*) and Yellowhammer (*Emberiza citrinella*). These species show differing responses to both the amount of woodland and hedgerows in each parcel and the total amount of woodland in the surrounding local landscape. Treecreeper and Long-tailed tits are green-listed for their UK conservation status (species of least concern). Yellowhammers are red-listed (species of most

[5] Sites in the WREN data set range in size from 0.5 ha to 32 ha.
[6] The term "local landscape" is used to distinguish the 3km buffer around each site, used to predict parcel-level occupancy, and the larger landscape of the case study area.

concern), although their recent population trend differs between Scotland (slight increase) and England (marked decrease)[7].

A Bayesian hierarchical occupancy model that accounts for imperfect detection was constructed using the package "jagsUI" (ver 1.5.2; Kellner 2022) in R (R Core Team, 2022) (for further details see Bradfer-Lawrence et al, 2023). Three regression parameters were extracted (Table 1) to integrate the ecological and economic components, and these were used to determine how woodland planting affects occupancy probabilities for the three focus species at the parcel and the local landscape scale. These parameters relate to the area of new woodland planted by an agent in any land parcel, the amount of current woodland in the surrounding landscape, and the amount of arable land in the surrounding landscape. We use a logit function to estimate probability of occupancy P for the j^{th} species for the i^{th} parcel of land (MacKenzie et al. 2006; De Wan et al, 2009):

$$P_{i,j} = \beta_{0j} + \beta_{1j}x_{1i} + \beta_{2j}x_{2i} + \beta_{3j}x_{3i} + \varepsilon \qquad (1)$$

where β_{0j} is the species-specific intercept, x_{1i} is the area of woodland in hectares, x_{2i} is the amount of woodland in the landscape as proportion of total landscape size, x_{3i} is the proportion of agricultural land which is arable in the landscape (Table 1). β_{1j}, β_{2j} and β_{3j} are species-specific terms derived from the Bayesian occupancy model. We transformed predicted occupancy probabilities from equation (1) to predict *actual occupancies* using individual Bernoulli trials for each parcel, yielding a value of 1 for species presence and 0 for absence. We focus on this predicted occupancy outcome when reporting results for cost-effectiveness and other incentive outcomes.

[7] Conservation status downloaded from www.bto.org on 29/11/23

Table 1: Summary of the variables from the ecological occupancy model used in our analysis.

Variable	**Description**
Parcel-level woodland	Area of woodland in each parcel (the sum of coniferous and broadleaved woodland in ha).
Landscape-level woodland	Total area of woodland in a landscape divided by the total landscape area.
Landscape-level arable cover	Arable area divided by total agricultural cover (i.e. arable / (arable + improved grassland)

4. Results

4.1 Spatial correlation between opportunity costs of agricultural returns for woodland creation and ecological benefits

We conducted a pairwise correlation analysis between predicted occupancy probabilities of birds post-woodland creation with the current agricultural gross margins for the two case study regions. Predicted occupancy probability of each species was positively correlated with opportunity costs for the Scottish case study and negatively correlated for the English case study (Table 2). This implies that for the Scottish case study, the most valuable land parcels for agricultural production are those which would also deliver the greatest increase in occupancies for the three bird species if woodland was created there. In contrast, for England, the agricultural parcels with the lowest agricultural value are also those which offer the greatest increase in occupancy. These contrasting findings mean one would expect to find differences in the cost-effectiveness of a given incentive policy between the case study areas.

Table 2: Pairwise correlation between opportunity costs of forgone agricultural production and ecological benefits

	Case study: Scotland				**Cast study: England**			
	Policy Scenario 1: 2 Hectares of woodland creation				**Policy Scenario 1: 2 Hectares of woodland creation**			
	Long-tailed tit	Treecreeper	Yellowhammer	Gross Margin	Long-tailed tit	Treecreeper	Yellowhammer	Gross Margin
Long-tailed tit	1				1			
Treecreeper	0.9723***	1			0.8761***	1		
Yellowhammer	0.5947***	0.6805***	1		0.6821***	0.8707***	1	
Gross Margin	0.0544***	0.0580***	0.0493***	1	-0.0225*	-0.0288**	-0.006	1
	Policy Scenario 2: 5 Hectares of woodland creation				**Policy Scenario 2: 5 Hectares of woodland creation**			
Long-tailed tit	1				1			
Treecreeper	0.9792***	1			0.9739***	1		
Yellowhammer	0.5522***	0.6398***	1		0.3231***	0.4479***	1	
Gross Margin	0.0440***	0.0433***	0.0453***	1	-0.0424***	-0.0413***	-0.0102	1
Policy Scenario 3: 10 Hectares of woodland creation					**Policy Scenario 3: 10 Hectares of woodland creation**			
Long-tailed tit	1				1			
Treecreeper	0.9871***	1			0.9981***	1		
Yellowhammer	0.6273***	0.6747***	1		0.4943***	0.5204***	1	
Gross Margin	0.0356**	0.0308**	0.0370***	1	-0.0429***	-0.0426***	-0.0200*	1

4.2 Economic and Ecological Impacts at the parcel level

As expected, the increase in woodland cover prompts a positive change in the predicted probability of occupancy for enrolled parcels. However, the increase in woodland created as the payment rate is raised does not lead to a consistent marginal increase in the predicted probabilities for all bird species in the Scottish case study. Indeed, as a higher payment rate is offered, the marginal predicted probabilities of Long-tailed Tit and Treecreeper declines, while that of the Yellowhammer increases. For the English case study, the case is different as the increase in probability of presence at the parcel level remains constant across the three bird species for the three policy scenarios and across payment rates. The amount of woodland created is also different under the two case studies: more woodland is created in England than Scotland at each payment rate. This itself does not necessarily lead to an increase in predicted occupancy, since occupancy for each species also depends on *where* the new woodland is planted, and the landscape characteristics around this parcel. The negative correlation between gross margin and predicted occupancy for the English case study means that the cheaper land parcels that enrol in the scheme are more likely to be those most beneficial for the bird species (hence the increased probabilities). For Scotland, the farm gross margins and predicted probability are positively correlated, therefore, the land parcels that enrol in the scheme in Scotland are unlikely to be those which are most beneficial to the three species.

Predicted probability of occupancy does not necessarily imply presence of a species at the parcel level. Therefore, using individual Bernoulli trials as described above, we estimated the actual occupancies (Figure 2) of each parcel after woodland creation. These occupancy values for a given parcel take either the value of 1 (occupied) or 0 (not occupied) for each species. We found that occupancy trends were similar in both case study areas, although they fluctuated among policy scenarios. Under policy scenarios 2 and 3, occupancy of Long-tailed Tits increased faster as payment rates increased compared to Treecreepers and Yellowhammers for both case studies. The number of parcels occupied by Treecreepers under policy scenario 2 fluctuates more in the Scottish case study than the English one. The number of parcels occupied by Yellowhammers was greater under policy scenario 3 in the Scottish site and lower in the English case study. Overall, changes in the number of parcels occupied by species were lowest when planting 2 ha, and highest when planting 10 ha in both Scotland and England. However, more parcels were predicted occupied in England compared to Scotland. This is not only because of the higher enrolment in England, but also because of the overall size of the case study and the spatial distribution of opportunity costs.

Budgetary cost, Producer Surplus and Cost-effectiveness analyses

We estimated the total budgetary costs of woodland created and total producer surplus for each payment rate and each policy scenario (Tables 3 and 4). Budgetary costs are equal to the subsidy rate multiplied by number of farmers enrolling: these increase as the payment rate increases. The resulting woodland created is sensitive to the payment rate in both Scotland and England, although as discussed above, increased investment in woodland planting does not necessarily translate to linear increases in biodiversity outcomes. Increased payment rates lead to higher total spending on the scheme, with total subsidy costs typically higher in the English case study. Producer surplus is defined as the difference between payment and opportunity cost for each participant in the scheme. Since farmers' opportunity costs do not change across scenarios, higher payment rates also translate into higher producer surplus or rents. Therefore, as payments increase the incremental producer surplus that an agent receives acts as an incentive to participate in the scheme. Moreover, the total producer surplus accrued by agents increased as we move from 2ha, 5ha to 10 ha limits on planting in each of the case study.

We estimated the cost-effectiveness of increasing payment rates to determine the cost of a 1% increase in "ecological benefits" for each of the three bird species as payment rates are increased (Table 5). This provides an idea of the variation in benefit-cost ratios across payments, species and case study areas. We express ecological benefits as a percentage given by change in predicted occupancy for each species divided by total number of occupied parcels under current (baseline) land use. All the ecological benefits are positive –although higher percentages are observed in England compared to Scotland. Cost-effectiveness for a given species and specific payment rate is also higher in England than Scotland. We further observe that the ecological benefits and costs differ among species and by payment rate, and that a 1% increase in occupancy costs the least for long-tailed tits followed by treecreepers and finally yellowhammers. Overall, we can say that for a fixed budget, it is more cost-effective to conserve Long-tailed Tits compared to Treecreepers and Yellowhammers in both case study areas (although of course woodland planting benefits *all* of our three bird species, as evidenced by the parameters in Table S1); whilst cost-effectiveness levels are higher in the English case study across all scenarios considered.

Table 3: Showing how many farmers sign up, area of woodland created subsidy per ha and total subsidy, producer surplus per policy to switch arable area to woodland for participating landowners in Scotland.

			Policy scenario 1: Offer uniform payments to switch 2 ha of arable land to woodland				(b) Policy scenario 2: Offer uniform payments to switch 5 ha of arable land to woodland				(c) Policy scenario 3: Offer uniform payments to switch 10 ha of arable land to woodland			
	Subsidy (£) per ha	% increase per ha	Number of participants	Total area converted (ha)	Total subsidy (£)	Producer surplus (£)	Number of participants	Total area converted (ha)	Total subsidy (£)	Producer surplus (£)	Number of participants	Total area converted (ha)	Total subsidy (£)	Producer surplus (£)
1	955.21	0	129	258	246444	91,071	280	1400	1337295	352,225	470	4700	4489487	1,132,895
2	1146.25	20	168	336	385140	148,785	378	1890	2166417	665,119	652	6520	7473566	2,198,889
3	1337.29	40	226	452	604457	223,154	488	2440	3262999	1,081,605	799	7990	10684984	3,595,079
4	1528.34	60	265	530	810018	317,828	592	2960	4523876	1,598,834	937	9370	14320514	5,248,976
5	1719.38	80	318	636	1093525	430,822	705	3525	6060810	2,218,577	1066	10660	18328577	7,170,424
6	1910.42	100	362	724	1383145	559,426	796	3980	7603475	2,935,049	1165	11650	22256402	9,305,269

Table 4: Showing how many farmers sign up, area of woodland created subsidy per ha and total subsidy, producer surplus per policy to switch arable area to woodland for participating landowners in England.

		Policy scenario 1: Offer uniform payments to switch 2 ha of arable land to woodland				**(b) Policy scenario 2: Offer uniform payments to switch 5 ha of arable land to woodland**				**(c) Policy scenario 3: Offer uniform payments to switch 10 ha of arable land to woodland**			
Subsidy (£) per ha	**% increase per ha**	**Number of participants**	**Total area converted (ha)**	**Total subsidy (£)**	**Producer surplus (£)**	**Number of participants**	**Total area converted (ha)**	**Total subsidy (£)**	**Producer surplus (£)**	**Number of participants**	**Total area converted (ha)**	**Total subsidy (£)**	**Producer surplus (£)**
1. 944.68	0	221	442	417548	109502	512	2560	2418376	606963	766	7660	7236233	2052788
2. 1133.61	20	312	624	707375	212183	637	3185	3610561	1157066	946	9460	10723988	3670345
3. 1322.55	40	398	796	1052749	343867	741	3705	4900044	1807220	1131	11310	14958029	5630557
4. 1511.49	60	467	934	1411727	508325	826	4130	6242433	2546160	1322	13220	19981832	7949906
5. 1700.42	80	548	1096	1863660	700787	925	4625	7864443	3368906	1516	15160	25778367	10636521
6 1889.36	100	598	1196	2259670	915382	1000	5000	9446780	4276377	1699	16990	32100158	13659137

Table 5: Cost-effectiveness analyses index for 5ha Policy scenario in Scotland and England

		Scotland			England		
Payment increase	**Species**	**% change in number of parcels occupied**	**Policy Cost (£)**	**Cost-effectiveness £ per 1% increase in occupancy**	**% change in number of parcels occupied**	**Policy Cost (£)**	**Cost-effectiveness £ per 1% increase in occupancy**
0%	Long-tailed tit	1.74	1,337,295	768,746	33.15	2,418,376	72,962
	Treecreeper	1.23		1,084,251	6.43		376,000
	Yellowhammer	1.20		1,117,654	3.49		692,990
20%	Long-tailed tit	2.34	2,166,417	924,869	43.10	3,610,561	83,774
	Treecreeper	1.73		1,252,309	7.78		463,809
	Yellowhammer	1.60		1,357,950	4.09		882,463
40%	Long-tailed tit	2.93	3,262,999	1,114,410	50.33	4,900,044	97,361
	Treecreeper	2.16		1,508,956	9.29		527,428
	Yellowhammer	2.07		1,578,833	4.69		1,044,086
60%	Long-tailed tit	3.38	4,523,876	1,340,083	56.62	6,242,433	110,252
	Treecreeper	2.45		1,845,919	10.80		578,200
	Yellowhammer	3.19		1,417,824	6.98		894,390
80%	Long-tailed tit	3.65	6,060,810	1,659,861	61.31	7,864,443	128,264
	Treecreeper	2.59		2,335,657	11.05		711,614
	Yellowhammer	4.06		1,492,474	6.02		1,307,070
100%	Long-tailed tit	3.93	7,603,475	1,936,218	67.14	9,446,780	140,711
	Treecreeper	2.88		2,637,139	12.79		738,772
	Yellowhammer	4.75		1,600,793	5.05		1,869,113

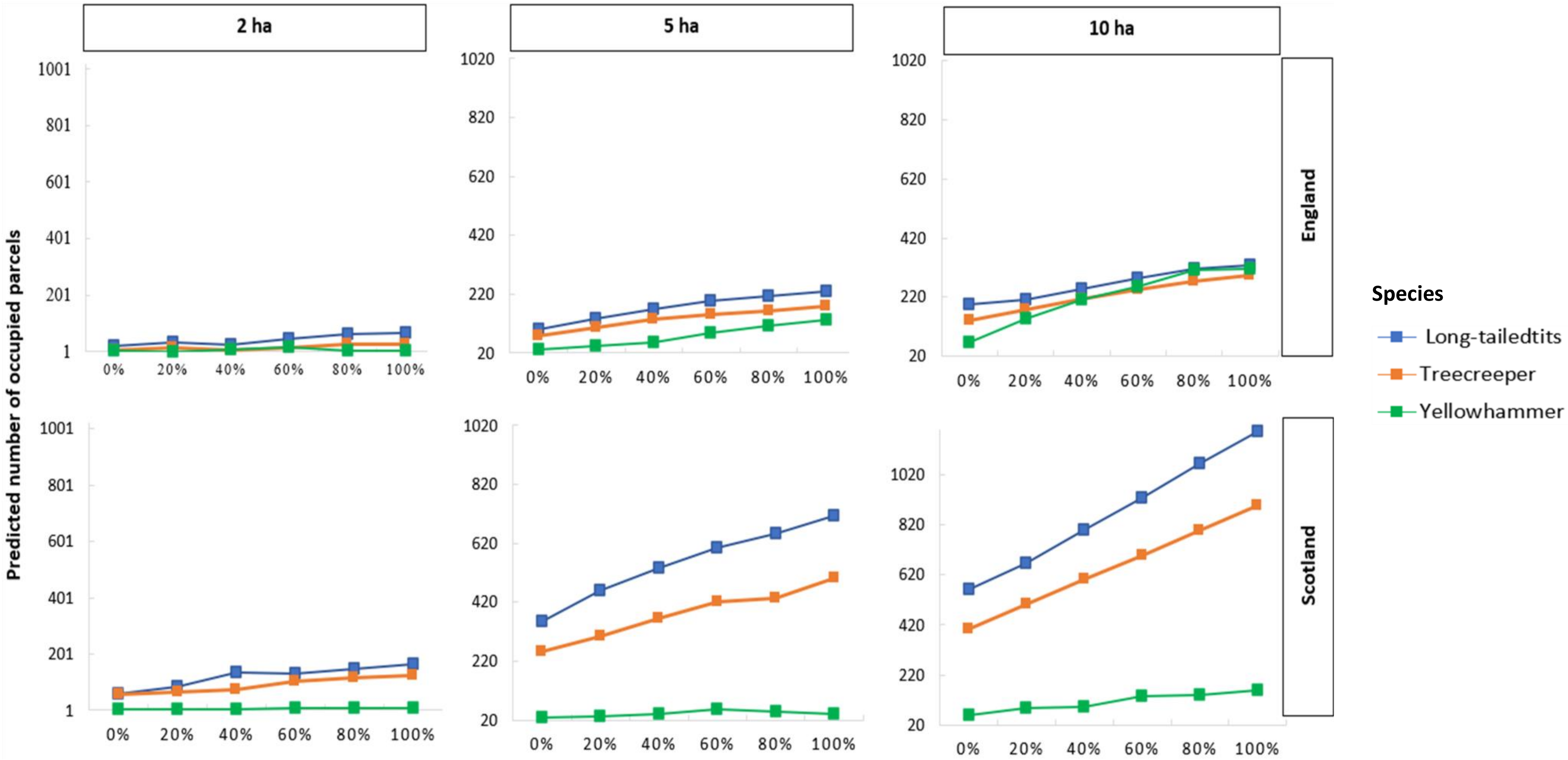

Figure 2: Graph showing changes in predicted bird occupancy for Long-tailedtit, Treecreeper and Yellowhammer as payment increase at parcel level in Scotland and England

Spatial distribution of enrolled parcels

Figure 3 demonstrates how the enrolled parcels differ in space across each case study area. The density of enrolled parcels increases as we move from 2 ha to 5 ha to 10 ha planting per parcel, reflecting the spatial pattern of opportunity costs. Given the intensity of clusters across space we can further deduce that if an agency were to offer differential compensation payments to agents based on connectivity or location, it is possible that highly fragmented habitats could form spatially interconnected habitat networks. The cost-effectiveness and ecological effectiveness of such clusters may be limited given that the agri-environment scheme we design in this study targets only cheaper parcels. It is also important to recall that clustering of enrolled parcels (clusters of new woodland) under uniform payments depend on the spatial auto-correlation of opportunity costs.

Spatial spillovers of bird species in Scotland and England case studies

Parcel-level biodiversity outcomes (see Figure S3) are directly related to landscape level outcomes due to ecological spatial spillovers, and both differ across the two case studies and the three agri-environment scheme policy scenarios analysed. Based on these findings, there is strong evidence that tree planting as a conservation measure for our target bird species increases the probability of occurrence and number of occupied parcels by each of the three species across the two landscapes. The total number of parcels occupied by Treecreepers are the highest followed by Long-tailed Tits and finally Yellowhammers in the Scottish case study. We observe similar performance of Treecreepers in England although under policy scenario 1, the number of patches occupied by Yellowhammers exceed that of Long-tailed tits. Under policy scenarios 2 and 3, the total number of parcels occupied by Long-tailed Tits increase by the greatest amount, followed by Yellowhammers. It is possible that since woodland planting raises the probability of birds occupying a specific parcel, it may result in a spillover into neighbouring parcels and (possibly over time) into the wider farmland landscape. Further, based on these spatial clusters (see spatial distribution in Figure 3) that form as parcels are enrolled into the agri-environment scheme - due to the spatial correlation of opportunity costs - it is possible that as agents engage in tree planting measures, the positive parcel-level biodiversity effects could result in positive spillovers to adjacent parcels.

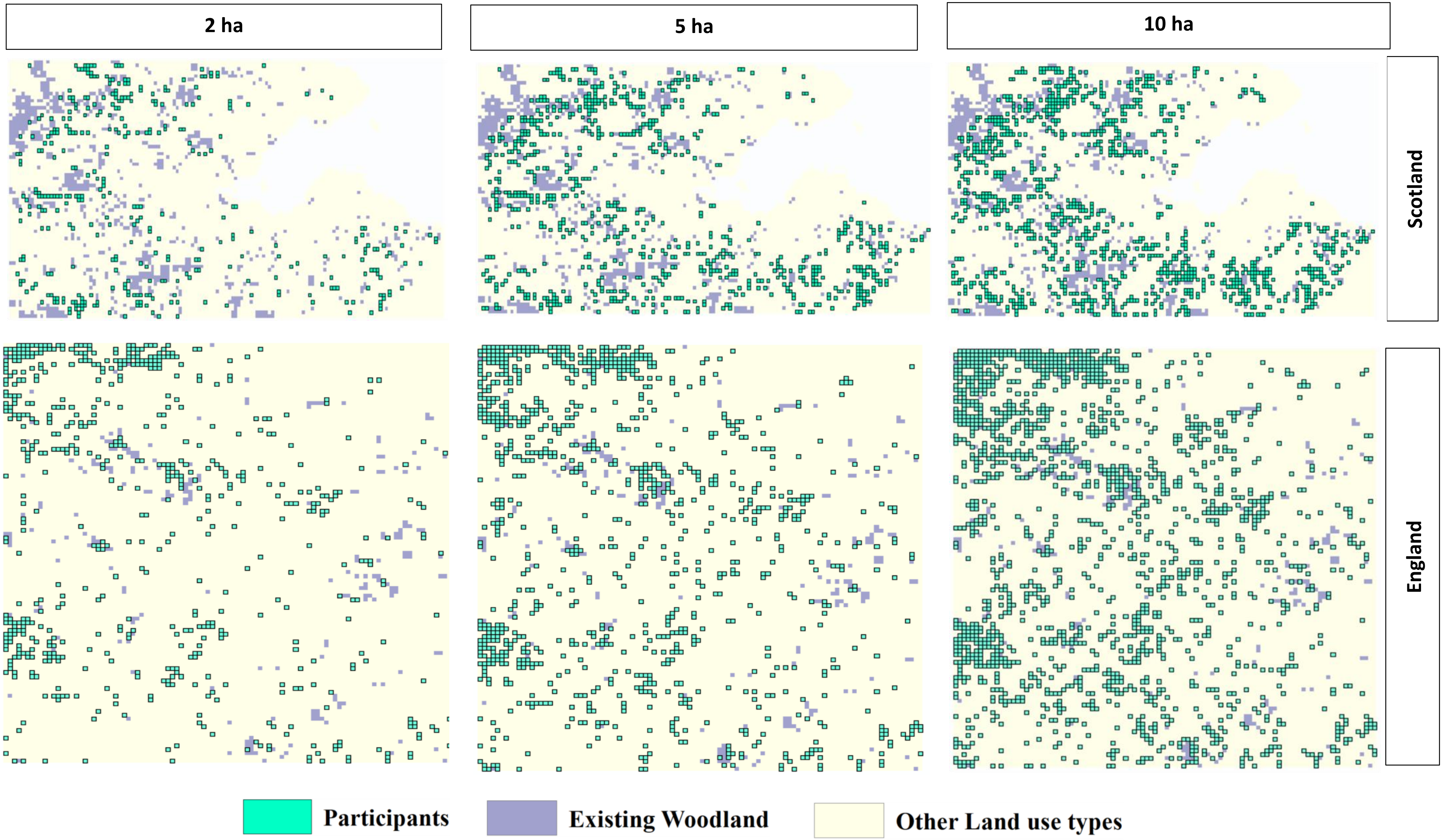

Figure 3: Maps showing the distribution new of woodland created for 2 ha, 5 ha and 10 ha policy scenarios in Scotland and England case studies.

5. Discussion and Conclusions

We construct and then estimate a combined ecological-economic model to assess the effect of economic incentives for woodland planting on biodiversity outcomes in two UK case study areas. The aims of this study were three-fold to: (i) determine the spatial correlation between predicted probabilities of bird occupancies and opportunity costs, (ii) evaluate the effect of financial incentives for woodland creation on the predicted probabilities and occupancies of birds at parcel-level, accounting for spillovers at the landscape scale (iii) to compare the cost-effectiveness of uniform subsidy payments between the two case study areas with positive and negative spatial correlations. Our study is unique in that we demonstrate how the performance of agri-environment schemes varies between two case studies whose spatial correlation between forgone agricultural returns and ecological benefits differ in sign and size. Our study also introduces case study specific spatial spillover effects, whereby the contribution of woodland planted at a given parcel to a specific biodiversity target depends partly on landscape-level woodland and arable cover, and not just on the land use characteristics of a specific parcel.

Findings from the spatial correlation analysis help predict the relative cost-effectiveness of an economic incentive to increase biodiversity by encouraging more woodland planting. For example, the positive correlation coefficients in the Scottish case study signify that the trade-offs between ecological benefits and agricultural returns are higher than those in the English site. This means that if the goals of tree planting incentives are to contribute to woodland biodiversity outcomes[8] in Scotland (Finch et al, 2023), it is expected that higher predicted occupancy probabilities of bird species will be found in more expensive parcels, compared to England (given how we defined the correlation scores, in terms of potential ecological benefit in each parcel relative to opportunity cost). Moreover, ecological outcomes were found to exhibit non-constant marginal returns as payment rate increase (although this effect varies among species).

Tree planting at the parcel level in our study also influences biodiversity outcomes at landscape scale. Woodland planting at the parcel level increases predicted probability of occupancy for the bird species modelled here at both the parcel and landscape scale. Previous studies have also associated the increase in woodland proportion with increases in woodland bird

[8] Although, or course, the policy goal might relate to carbon sequestration, in which case biodiversity impacts need to be considered as off-target effects of potential social relevance.

populations (Petit and Landis, 2023). Kleijn et al. (2006) further argues that parcel level biodiversity measures and what happens at landscape level are interconnected.

The cost-effectiveness index shows that the cost-effectiveness of an economic incentive varies across species, and a specific subsidy rate does not guarantee maximum ecological benefits at minimum costs for all species. In our study, the most cost-effective woodland planting can be considered as the one that gives the lowest cost per 1% increase in ecological benefits. As the subsidy rate is increased, cost-effectiveness declines, implying declining marginal returns to conservation actions in this instance. Broadly speaking, we see a stable ranking of cost-effectiveness between the two case study sites according to which species outcome is used to construct the index: increasing the distribution of Long-tailed Tits is always the lowest cost option; increasing Treecreepers is always more cost-effective than increasing distribution of Yellowhammers. To a degree, this variation in cost-effectiveness across species was to be expected, since we specifically chose case study species which varied in their responsiveness to woodland planting.

Variations in absolute cost-effectiveness scores between case study sites can be attributed to the differences in the sign of spatial correlation of opportunity costs and ecological benefits, and as conservation benefits differ across space. Previous literature has associated spatial variation in ecological benefits with variation in cost-effectiveness (Kimball et al., 2015). The cost-effectiveness index can be used as a tool for decision-making to select the species and location where conservation should be targeted to yield maximum benefits at minimum costs.

We acknowledge several limitations of this study. First, our payment for actions incentive scheme is almost certainly likely to be less cost-effective compared to a payment for results scheme, because it does not take into account the spatial heterogeneity of opportunity costs or the heterogeneity in ecological potential (Jack et al. 2008; Simpson et al., 2023). The current scheme does not also give us the desired flexibility to precisely select the minimum budget that is appropriate for maximum ecological benefits because our uniform payments target cheaper parcels which do not guarantee maximum ecological benefits across case studies, payment rates and species. Further, the uniform pricing mechanism used here is less economically efficient and is likely to lead to overcompensation compared to differentiated payments (Connor et al. 2008; Jack et al. 2008). The high producer surplus values generated for farmers as the incentive rate is increased to improve predicted biodiversity outcomes is some indication of this.

Second, landowners in the real world respond to multiple factors in deciding whether to create woodland on their farmland (for a recent review, see Staddon et al, 2021). In contrast, our model uses a very simple comparison of alternative returns to greatly simplify the decision criterion. An obvious avenue for future work is to broaden the concept of "returns" which our agents consider.

Finally, this paper prioritises specific bird species as the biodiversity indicators of interest. Yet increasing woodland cover on arable and grassland will likely come at a cost to other species. For example, higher woodland cover increases predation risks for bird species that nest in open habitats (Wilson et al 2014; Roos et al 2018). Moreover, losses of arable land to woodland planting can have negative impacts on some species, including Yellowhammer which we have not modelled. This implies trade-off exists between alternative biodiversity outcome indicators, and we have not considered this in the current paper.

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Supplementary material (SM)

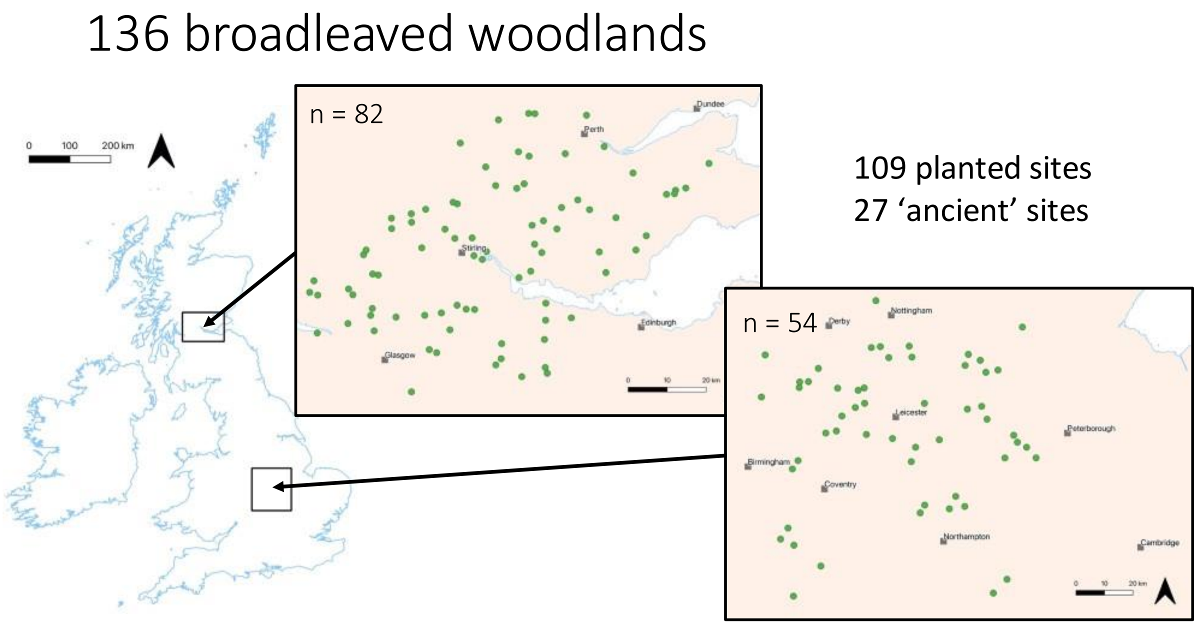

Figure S1: Overview of the case study regions showing the 136 woodlands studied in the WrEN project

Table S1: Parameters used to predict probability of occupancy of Long-tailed tit, treecreeper and yellowhammer in case study sites

Variable	Species	Parameter	SD
Intercept	*Aegithalos.caudatus*	0.6232	0.7439
Intercept	*Certhia.familiaris*	0.788	0.7354
Intercept	*Emberiza.citrinella*	-2.642	0.9362
Area planted (ha)	*Aegithalos.caudatus*	3.484	1.105
Area planted	*Certhia.familiaris*	1.849	0.7587
Area planted	*Emberiza.citrinella*	0.9959	0.7817
Current_Woodland	*Aegithalos.caudatus*	0.2363	0.23
Current_Woodland	*Certhia.familiaris*	0.1234	0.2305
Current_Woodland	*Emberiza.citrinella*	0.1848	0.2258
Arable_prop	*Aegithalos.caudatus*	-0.3237	0.4306
Arable_prop	*Certhia.familiaris*	0.1455	0.4389
Arable_prop	*Emberiza.citrinella*	0.9152	0.4802

Table S2: Payment schedule per ha

Case study	Base payment	20%	40%	60%	80%	100%
Scotland	955	1146	1337	1146	1337	1528
England	945	1134	1323	1512	1701	1890

Statement of conflicts of interest:

The authors have no conflicts of interest to declare.

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