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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 the subsidy for woodland planting is increased, the values of our biodiversity indicator increase, but at rates which vary by case study area and by species. The cost-effectiveness of the economic instrument varies according to the sign of the spatial correlation between opportunity costs and ecological potential.

1. 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.[1] 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 to, both domestically and internationally, 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 specific biodiversity indicators 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.[2] These

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

[2] We use bird species are our biodiversity indicator since birds are commonly used in biodiversity assessments, because there are well established survey methods, extensive spatial and historic data on populations (at least in large parts of the world) and their demographics are well known.

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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; Kämpfer 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.

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 areas, one in Scotland and one in England, in which the spatial correlation between the foregone returns from agriculture (which represent 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, a site-specific ecological model is integrated with a site-specific economic model of land manager decision-making in a manner which yields predictions of biodiversity outcomes. Second, we show the effects of economic incentives on biodiversity outcomes from woodland planting on farmland, where both parcel-level and local landscape-level woodland variables help determine biodiversity outcomes. Third, 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), Naidoo et al. (2006) and Simpson et al. (2023). 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 which, on the whole, 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 housing 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

2.1. Overall approach

We represent each of the two case study areas as a grid of 1 km by 1 km (100 ha) 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 local 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.

2.2. Case study locations and data development

We apply our agent-based model in two UK case study areas (Fig. 1). In Scotland, the case study is the watershed of the Forth Estuary, covering around 5400 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 case study landscape is then divided into 1 km by 1 km land parcels and georeferenced to Ordnance Survey British National Grid.

Using ArcGIS, for each 100 ha land parcel we extracted the proportional area (ha) of 33 landcover types using landcover data from the Land Cover Map (LCM2015) and the Land Cover Plus Crops map (Rowland et al., 2017). This includes 11 arable crops, and additional land cover types including improved grassland, coniferous woodland, broad leaved woodland and urban. The Land Cover data uses information from real production choices of land managers collected through the annual Agricultural Census.[3] To account for differences in yield across space, data on soil quality was derived from Soilscapes[4] at the 1 km by 1 km resolution. Thus, for each land parcel, we possess information regarding the proportion of arable crops cultivated and the corresponding soil fertility classification, categorised as low, medium, or high. Subsequently, we utilise information from the SRUC Farm Management Handbook to ascertain gross margin values for each crop and livestock use type. These values are provided per hectare across a productivity spectrum ranging from low to medium to high yield, which we

[3] June survey of agriculture and horticulture conducted by Defra (England) and their the devolved equivalent (Scotland) https://www.gov.uk/agricultural-survey.

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

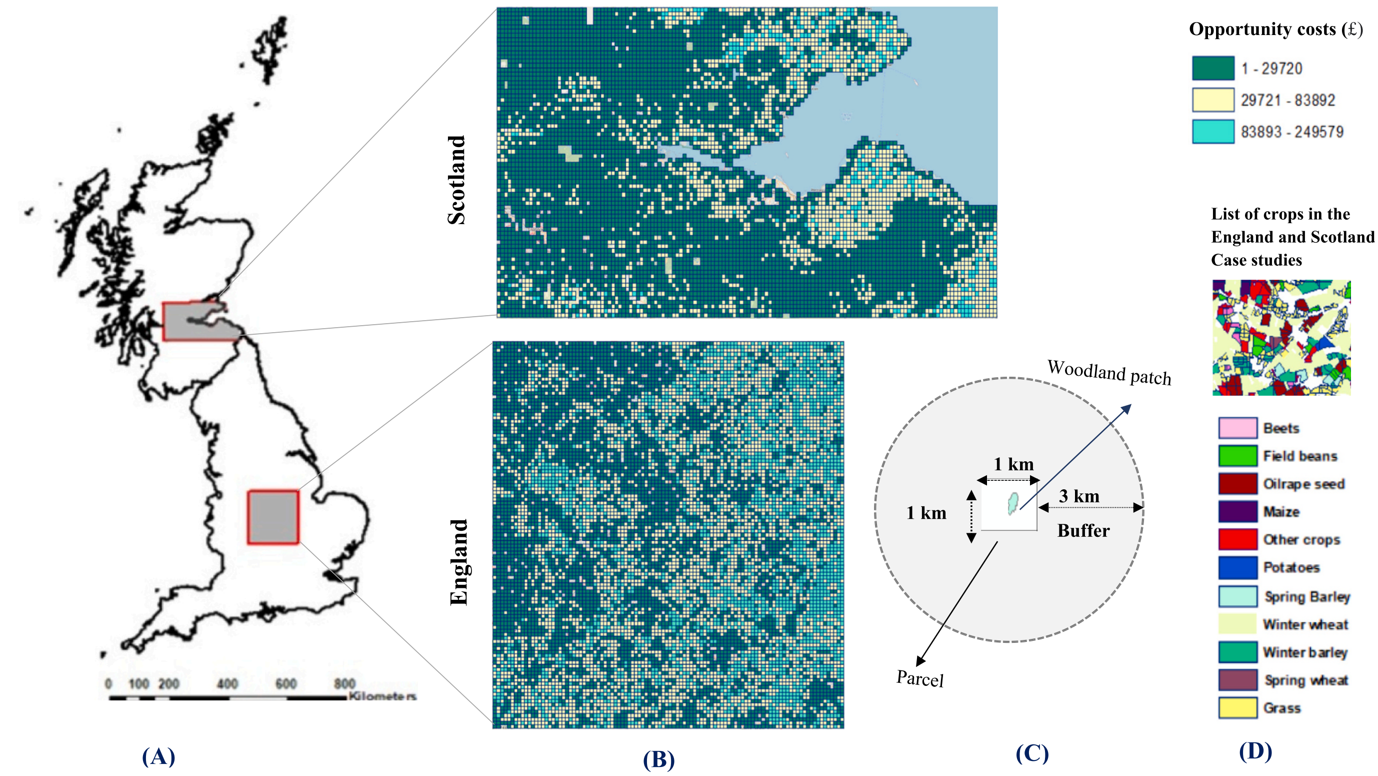

Fig. 1. Map of Great Britain (A) showing the geographical location of the two case study areas (B), the hypothesized local landscape (C) and the spatial distribution of opportunity costs and crops used to calculate them (D) in England and Scotland under current land use.

align with soil fertility categories derived from the Soilscapes dataset. Thus, for each land parcel, we compute the total gross margin for each crop by multiplying the area (in hectares) with the respective crop gross margin value sourced from the handbook (in GB pounds), with adjustments made based on the quality of soil fertility, namely low, medium, or high. For example, the opportunity cost associated with cultivating one hectare of winter wheat on high-productivity soil is anticipated to exceed that of the same crop cultivated on low-productivity soil. We thus incorporate heterogeneity in opportunity costs across the landscape based on current production choices of land managers, using real survey data. Fig. 1 shows the current distribution of opportunity costs (=farm gross margin) across the two case study areas.

2.3. 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. 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. The model is static, and considers only relative economic returns in year t. 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 in year t (their opportunity cost) plus the cost of tree planting, where all planting costs are incurred in year t. 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 over-compensation 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 (Hanley et al., 2012). 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. We did not specify an incentive payment which varies spatially with opportunity costs, since (i) this is not an approach which has been undertaken in the UK so far for agri-environment schemes or woodland planting and (ii) it is unrealistic to assume that the regulator could perfectly observe the spatial variation in opportunity costs, which is to some degree private information to the land manager. No account is taken of the estimated future returns on investment from timber sales, carbon credits or of potential tax benefits from woodland planting, or of future management costs for woodland plots, since no economic benefits or costs from alternative land uses in years $t + 1 \ldots T$ are considered. The average opportunity cost per hectare is £455 for the Scotland case study, and £445 per hectare for the England case study. We assume that agents also face 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 then simulate the agents' land-use choices at 6 payment levels: this base payment rate, plus a 20%, 40%, 60%, 80% and 100% increase in the subsidy.

A range of quantity constraints were also placed on agents considering woodland planting. 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 what are referred to as Policy Scenarios 1, 2 and 3 respectively. We also constrained the total woodland (existing woodland plus new woodland) within each enrolled parcel 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 entire case study landscapes using ArcGIS. Finally, we estimate the change in predicted occupancy of the three bird species in response to woodland planting using an ecological model.

2.4. 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 predicted probabilities are derived from a model reported in Bradfer-Lawrence et al. (2023) and are functions of (i) species-specific intercepts, (ii) parcel-level woodland cover, and (iii) existing woodland and arable cover within a 3-km "local landscape" buffer around each parcel. As agents change land use within the economic component of the model, the predicted probability of occupancy for each species responds according to whether woodland is created in a specific parcel, and to the amount of existing woodland and arable area within the 3 km local landscape.[6] Three indicator bird species that 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 concern), although their recent population trend differs between Scotland (slight increase) and England (marked decrease).[7]

To determine how woodland planting affects parcel-scale occupancy probabilities for the three focal species, we ran a Bayesian hierarchical occupancy model that accounts for imperfect detection (Kery and Royle, 2021) 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). We use a logit function to estimate probability of occupancy P for the j^{th} species in the i^{th} parcel of land (MacKenzie et al., 2006; De Wan et al., 2009):

$$P_{i,j} = \beta_{0j} + \beta_{1j}x_{1i} + \beta_{2j}x_{2i} + \beta_{3j}x_{3i} \tag{1}$$

where β_{0j} is the species-specific intercept, and β_{1j}, β_{2j} and β_{3j} are species-specific parameters. Variable x_{1i} is the area in hectares of new woodland planted in land parcel i, x_{2i} is the amount of existing woodland as a proportion of the surrounding local landscape, x_{3i} is the proportion of agricultural land which is arable in the local landscape (Table 1). We transformed predicted probability of occupancy (P derived from Eq. (1)) to *predicted presence/absence* using separate Bernoulli trials for each species in each parcel, with the likelihood of 'success' (i.e., species presence) in each trial determined by the parcel-specific probability of occupancy. This yielded a value of 1 for species predicted to present and 0 for absent. We focus on this predicted occupancy outcome when reporting results for cost-effectiveness and other incentive outcomes.

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	Area of woodland in the *local* landscape (3 km radius) divided by the local landscape area.
Landscape-level arable cover	Arable area divided by total agricultural cover in the *local* landscape (i.e. arable / (arable + improved grassland)

3. Results

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

We conducted a pairwise correlation analysis between the current agricultural gross margins for each land parcel (prior to woodland creation) and the ecological potential of all eligible land parcels. Ecological potential is defined as the predicted probability of occupancy for the three species if all eligible land parcels planted 2 ha, 5 ha, or 10 ha of woodland. Eligible land parcels are those containing 5 ha or more of arable land, with the total woodland (existing plus new) not exceeding 30 ha within the parcel. This pairwise correlation was completed for both case study areas (see Table 2). In Scotland, we found a positive correlation between the predicted probability of occupancy for each species and the opportunity costs for each land parcel. In contrast, for the England case study, the predicted probability of occupancy for all three species showed a negative correlation with the opportunity costs for each land parcel. This implies that in the Scottish case study, the most valuable land parcels for agricultural production are also those likely to deliver the greatest predicted increases in occupancies for the three bird species if woodland were created there. Conversely, in England, the agricultural parcels with the lowest agricultural value are also those predicted to offer the greatest increase in occupancy. These contrasting findings suggest that differences in the cost-effectiveness of a given incentive policy can be expected between the two case study areas.

3.2. Economic and ecological impacts at the parcel level

As expected, the increase in woodland cover results in an increase in the predicted probability of occupancy by the focal species in the 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 probability of occupancies for all bird species in the Scottish case study. For the English case study, the increase in parcel-level probability of occupancy remains roughly constant across the three bird species for the three policy scenarios and across payment rates. The amount of woodland created is also different in the two case studies: more woodland is created in England than in Scotland at each payment rate. This itself does not necessarily lead to a predicted increase in occupancy, since occupancy for each species also depends on *where* the new woodland is planted, and the local 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 of occupancy). 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.

Output from the individual Bernoulli trials described above is shown in Fig. 2 for each parcel after woodland creation. Recall that these predicted occupancy values for a given parcel take either the value of 1

[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 3 km buffer around each site, used to predict parcel-level occupancy, from the larger landscape of the case study area.

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

Table 2
Pairwise correlation between opportunity costs of forgone agricultural production (gross margins) under current land use and predicted occupancies (number of predicted occupied grid squares) of Long-tailed Tit, Treecreeper and Yellowhammer if all eligible landholders enrolled in the scheme and created either 2 ha, 5 ha or 10 ha of woodland (ecological potential of the land parcel).

	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

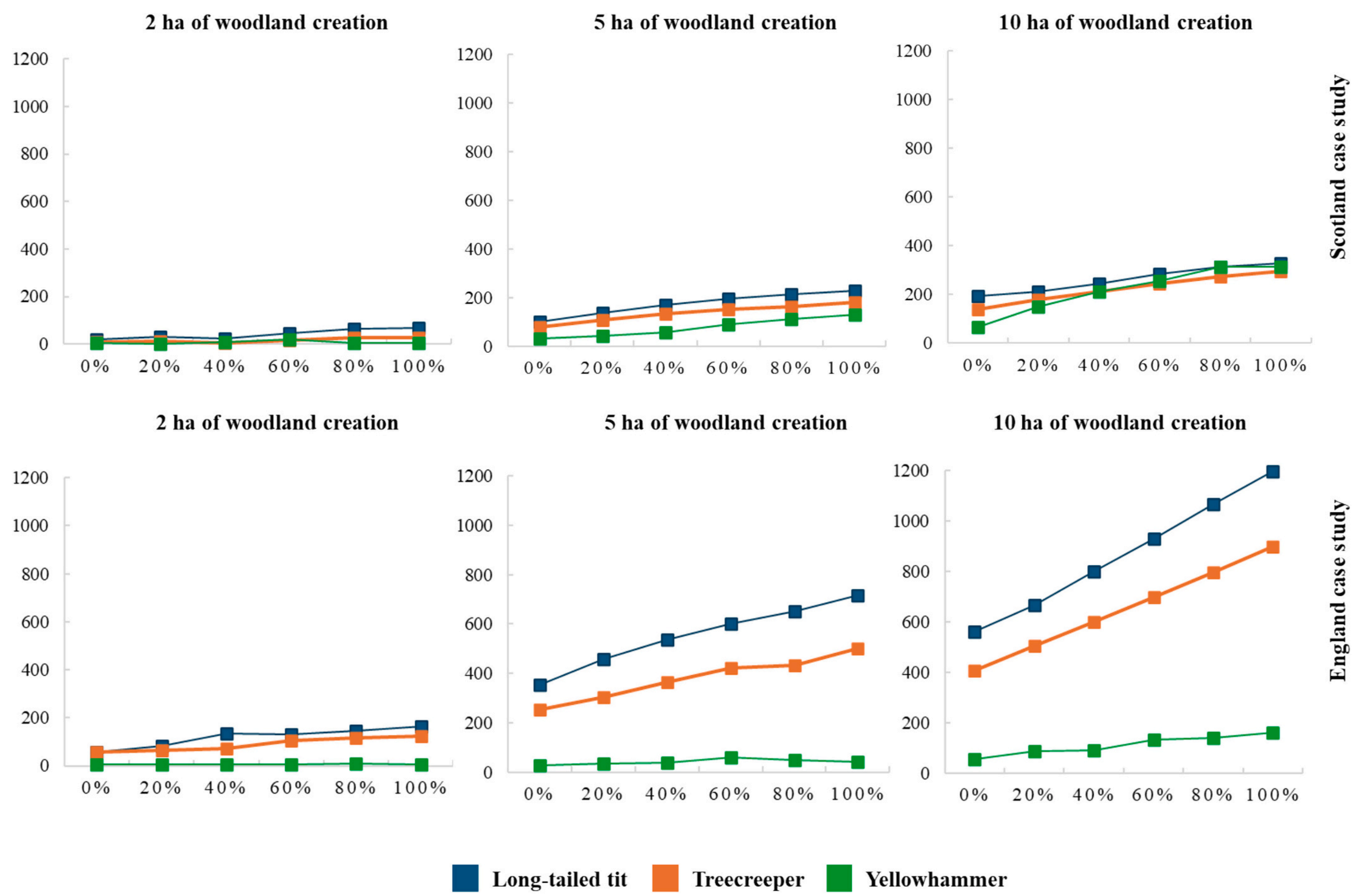

Fig. 2. A comparison of the number of parcels predicted to be occupied by Long-tailed Tit, Treecreeper and Yellowhammer under 2 ha, 5 ha and 10 ha of woodland creation for each case study region as the payment for woodland subsidy increases relative to the baseline (0%–100%: horizontal axis).

(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 the 5 ha and 10 ha policy scenarios, 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 the 5 ha policy scenario fluctuated more in the English case study area than the Scottish case study area. The number of parcels occupied by Yellowhammers was greater under the 10 ha policy scenario in Scotland than in England. 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 than in Scotland (compare, for example, Long-tailed Tits in the 10 ha. policy scenario). This is not only because of the higher enrolment in England, but also because of the overall size of the case study area, the spatial distribution of opportunity costs, and their relationship with ecological potential.

3.3. 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 for 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 demonstrated 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 increases as we move from 2 ha to 5 ha 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 the change in predicted occupancy for each species divided by the total number of occupied parcels under current (baseline) land use. All the ecological benefits are positive, although much higher percentage increases are observed in England than in Scotland. Cost-effectiveness for a given species and specific payment rate is also better in England than in Scotland. Since the impact on a given species depends on the existing local-landscape woodland cover, these values will depend both on new planting incentivised by the subsidy and the baseline land use. We further observe that the ecological benefits and costs differ among species and by payment rate. *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 than Treecreepers or Yellowhammers in both case study areas (although of course planting new woodlands delivers increases in all 3 species); whilst cost-effectiveness levels are better in the English case study across all scenarios considered.

3.4. Spatial distribution of enrolled parcels

Fig. 3 demonstrates how the enrolled parcels differ in space across each case study region. 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.

4. Discussion and conclusions

We construct 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: (i) to evaluate the effect of financial incentives for woodland

Table 3
Numbers of farmers signing up, area of woodland created, total subsidy, and producer surplus per policy to switch arable area to woodland for participating farmers in the Scotland case study.

			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	0	129	258	246,444	91,071	280	1400	1337,295	352,225	470	4700	4,489,487	1,132,895
2	1146	20	168	336	385,140	148,785	378	1890	2,166,417	665,119	652	6520	7,473,566	2,198,889
3	1337	40	226	452	604,457	223,154	488	2440	3,262,999	1,081,605	799	7990	10,684,984	3,595,079
4	1528	60	265	530	810,018	317,828	592	2960	4,523,876	1,598,834	937	9370	14,320,514	5,248,976
5	1719	80	318	636	1,093,525	430,822	705	3525	6,060,810	2,218,577	1066	10,660	18,328,577	7,170,424
6	1910	100	362	724	1,383,145	559,426	796	3980	7,603,475	2,935,049	1165	11,650	22,256,402	9,305,269

Table 4
Numbers of farmers signing up, area of woodland created, total subsidy, and producer surplus per policy to switch arable area to woodland for participating farmers in the England case study.

	Subsidy (£) per ha	% increase per ha	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			
			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.	945	0	221	442	417,548	109,502	512	2560	2,418,376	606,963	766	7660	7,236,233	2,052,788
2.	1133	20	312	624	707,375	212,183	637	3185	3,610,561	1157,066	946	9460	10,723,988	3,670,345
3.	1322	40	398	796	1,052,749	343,867	741	3705	4,900,044	1807,220	1131	11,310	14,958,029	5,630,557
4.	1511	60	467	934	1,411,727	508,325	826	4130	6,242,433	2,546,160	1322	13,220	19,981,832	7,949,906
5.	1700	80	548	1096	1,863,660	700,787	925	4625	7,864,443	3,368,906	1516	15,160	25,778,367	10,636,521
6	1889	100	598	1196	2,259,670	915,382	1000	5000	9,446,780	4,276,377	1699	16,990	32,100,158	13,659,137

creation on the predicted probabilities and occupancies of birds at the parcel-level, accounting for spillovers at the local landscape scale; (ii) to determine the spatial correlation between predicted probabilities of bird occupancies and opportunity costs; and (iii) to compare the cost-effectiveness of uniform subsidy payments between the two case study regions 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 study areas 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 in a given parcel to a specific biodiversity target depends partly on local landscape-level woodland and arable cover, and not just on the land-use characteristics of that 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 case study. 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, whilst this effect was also found to vary across species.

Woodland planting at the parcel level increases the predicted probability of occupancy for the bird species modelled here at both the parcel and landscape scale. Moreover, we incorporate feedback effects from local landscape-level woodland and arable cover to patch-level biodiversity outcomes. Previous studies have also associated the increase in woodland proportion with increases in woodland bird populations (Petit and Landis, 2023). Kleijn and Frank van (2006) further argue that parcel-level biodiversity measures and what happens at landscape level are interconnected.

Analysis shows that the cost-effectiveness of an economic incentive varies across species, and that a specific subsidy rate does not guarantee maximum ecological benefits at minimum costs. 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, where ecological benefits are defined in terms of increases in predicted occupancy. 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 regions according to which species outcome is used to construct the index: increasing the distribution of Long-tailed Tits is almost always the lowest cost option; and increasing Treecreepers is typically 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 species which varied in their responsiveness to woodland planting. It should be pointed out, however, that new woodland planting increases the distribution of all three of our exemplar species: the regulator cannot target just one of the three species using the incentive we have modelled here.

Variations in absolute cost-effectiveness scores between case study regions can be attributed to the differences in the sign of spatial correlation of opportunity costs and ecological benefits, and to conservation benefits differing across space. Previous literature has associated spatial variation in ecological benefits with variation in cost-effectiveness

[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.

Table 5
Cost-effectiveness analyses index for the 5 ha 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
	Long-tailed Tit	1.74		768,746	33.15		72,962
0%	Treecreeper	1.23	1337,295	1,084,251	6.43	2,418,376	376,000
	Yellowhammer	1.20		1,117,654	3.49		692,990
	Long-tailed Tit	2.34		924,869	43.10		83,774
20%	Treecreeper	1.73	2,166,417	1,252,309	7.78	3,610,561	463,809
	Yellowhammer	1.60		1,357,950	4.09		882,463
	Long-tailed Tit	2.93		1,114,410	50.33		97,361
40%	Treecreeper	2.16	3,262,999	1,508,956	9.29	4,900,044	527,428
	Yellowhammer	2.07		1,578,833	4.69		1,044,086
	Long-tailed Tit	3.38		1,340,083	56.62		110,252
60%	Treecreeper	2.45	4,523,876	1,845,919	10.80	6,242,433	578,200
	Yellowhammer	3.19		1,417,824	6.98		894,390
	Long-tailed Tit	3.65		1,659,861	61.31		128,264
80%	Treecreeper	2.59	6,060,810	2,335,657	11.05	7,864,443	711,614
	Yellowhammer	4.06		1,492,474	6.02		1,307,070
	Long-tailed Tit	3.93		1,936,218	67.14		140,711
100%	Treecreeper	2.88	7,603,475	2,637,139	12.79	9,446,780	738,772
	Yellowhammer	4.75		1,600,793	5.05		1,869,113

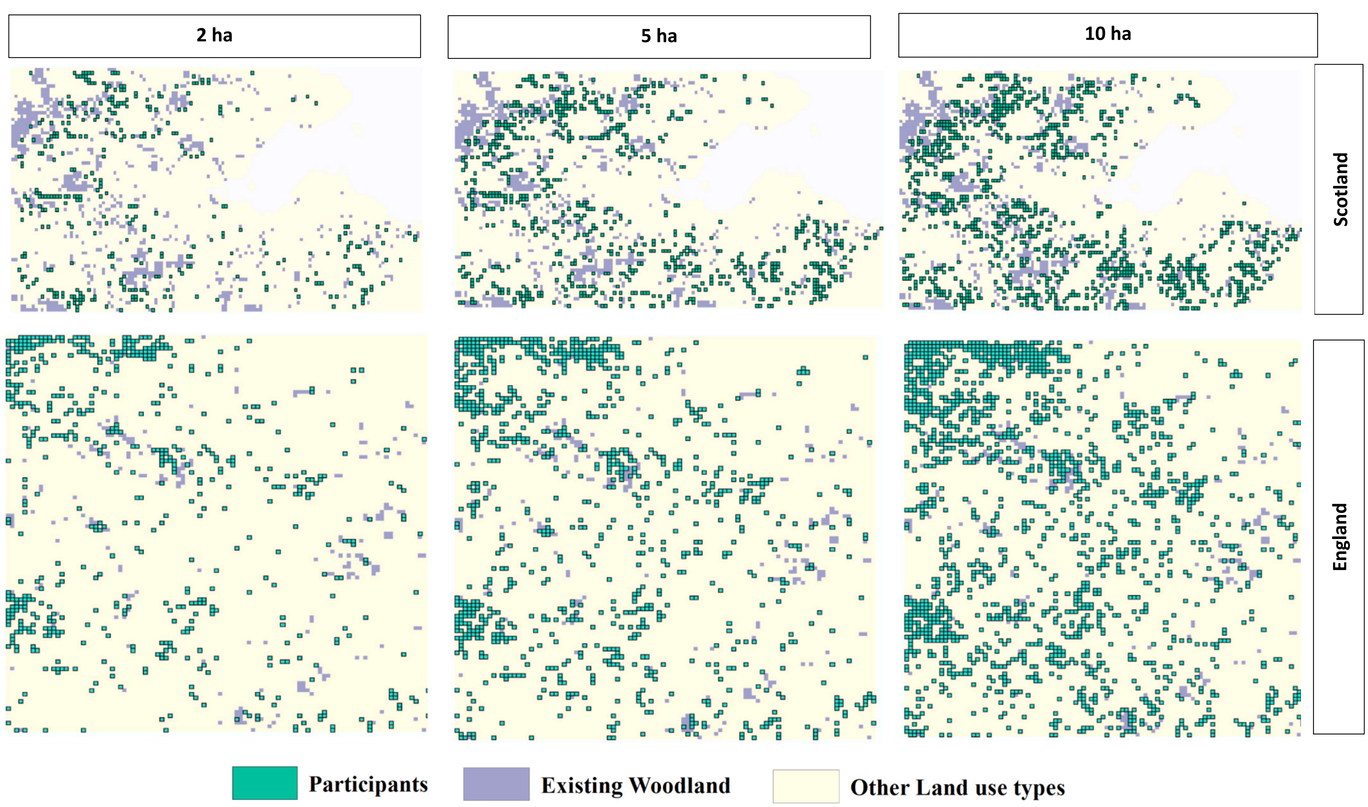

Fig. 3. Maps showing the distribution new of woodland created for 2 ha, 5 ha and 10 ha policy scenarios in Scotland and England case studies.

(Kimball et al., 2015). The cost-effectiveness index can be used as a tool for decision-making to select the species and locations where conservation should be targeted to yield ecological benefits at minimum costs.

We acknowledge several limitations of this study. First, we assume that a uniform payment rate is offered to all farmers in either the Scottish or English case study regions to encourage woodland planting. But such a simple payment for actions incentive scheme is almost certainly likely to be less cost-effective than 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 modelled also scheme does not give us the 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 than a differentiated payment or a conservation auction, and is likely to lead to overcompensation compared to either of these policy alternatives (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. Finally, we have not included any kind of additional reward for farmers who create new woodland adjacent to existing woodland – an agglomeration bonus – or higher payments for new planting in local landscapes with currently higher levels of woodland cover – a threshold bonus. Again, this is a task for future work.

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 financial returns to greatly simplify the decision criterion. That is, we assume that farmers as economic agents only compare the monetary rewards of alternative, mutually-exclusive land uses. An obvious avenue for future work is to broaden the concept of "returns" which our agents consider, for example to include non-pecuniary payoffs from participating in agri-environment schemes (Kuhfuss et al., 2021). Moreover, our static, one-period model treats land managers as myopic: they consider only the benefits and costs of alternative land uses in year t when they make a decision about whether or not to create woodland in year t. No economic costs or benefits in years ($t + 1, t + 2 \ldots T$) are considered. Moreover, the predicted ecological changes used to model biodiversity outcomes are treated as if they emerge instantaneously when new woodland is created. In reality, meta populations across a landscape will take many years to respond to this change in land cover.

Finally, this paper prioritises specific woodland-associated 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 farmland-associated species (e.g., Finch et al., 2023), which we have not modelled. This implies that a trade-off exists between alternative biodiversity outcome indicators, and we have not considered this in the current paper. There will also be impacts of incentivised woodland creation on the supply of a wide range of ecosystem services associated with woodland on farmland, such as changes to carbon storage and recreation opportunities, which we do not include in our model. Lastly, the response of birds to land-use change is not necessarily representative of other taxonomic groups, so we caution against extrapolating our results to biodiversity in general.

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CRediT authorship contribution statement

Mary Nthambi: Writing – original draft, Formal analysis. **Katherine Simpson:** Writing – original draft, Methodology, Investigation, Data curation. **Tom Bradfer-Lawrence:** Writing – original draft, Formal analysis, Data curation. **Andrew Dobson:** Writing – original draft, Formal analysis. **Tom Finch:** Writing – original draft. **Elisa Fuentes-Montemayor:** Writing – original draft. **Kirsty Park:** Writing – original draft, Funding acquisition, Conceptualization. **Kevin Watts:** Writing – review & editing, Funding acquisition, Conceptualization. **Nick Hanley:** Writing – original draft, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors have no conflicts of interest to declare.

Data availability

The authors do not have permission to share data.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolecon.2024.108265.

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