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RESEARCH ARTICLE

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The costs of delivering environmental outcomes with land sharing and land sparing

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Abstract

1. The biodiversity and climate crises demand ambitious policies lowering the environmental impacts of farming. Most current interventions incentivise so-called land-sharing approaches to address the widespread trade-off between farm yields and on-farm environmental outcomes by compensating farmers who adopt yield-reducing interventions that encourage wildlife or reduce net emissions within farmed land.
2. Here, we present the first quantification of the likely costs to taxpayers of land sharing compared with land sparing, in which large areas are removed from production altogether because of high-yielding practices elsewhere in the landscape. Focusing on arable production in the United Kingdom, we used a choice experiment to explore farmer preferences and estimated the overall costs of contrasting agri-environment schemes that delivered increased populations of three well-studied farmland birds and reduced net carbon emissions in England. We included capital, administration and monitoring costs, and lost food production.
3. Sparing delivered our target biodiversity and carbon emission outcomes at 79% of the food production cost and 48% of the taxpayer cost of sharing. The difference in subsidy payments required by farmers roughly tracked lost food production but other costs favoured sparing even more strongly.
4. The cost-related merits of sparing would probably increase further in studies incorporating (1) the many species and ecosystem services not deliverable on farmland, (2) the costs of food imports to compensate domestic lost production and (3) countries without as long and extensive a history of agriculture as the United Kingdom.
5. Our results suggest that, for at least some conservation outcomes, continuing a land-sharing approach in countries such as the United Kingdom is not only an inefficient use of government funds but also undermines conservation and food security in food-exporting countries which bear the burden of compensating domestic production forgone in the name of conservation.

KEYWORDS
agri-environment schemes, biodiversity conservation, carbon emissions, choice experiment, environmental economics, land sharing, land sparing, land use policy

1 | INTRODUCTION

Globally, agriculture is the greatest threat to biodiversity (Tilman et al., 2017), accounts for an estimated 34% of annual anthropogenic carbon emissions (Crippa et al., 2021), and covers roughly 50% of all habitable land (Ritchie, 2019). The vast area under farming production offers huge opportunity for interventions that deliver biodiversity and carbon storage. To date, most policies for reconciling food production and environmental outcomes have promoted a land-sharing approach, where wildlife-friendly measures are implemented on farmed land, usually at the cost of yield (Green et al., 2005). However, 15 years of empirical data from >2500 species across five continents suggests that the same quantity of food could be produced at substantially lower cost to biodiversity, the climate and a suite of ecosystem services, if it was instead met through land sparing (Balmford, 2021; Dotta et al., 2016; Finch et al., 2019, 2020; Kamp et al., 2015; Phalan et al., 2011; Williams et al., 2017), with higher yields on already-cleared land freeing-up land elsewhere for the retention or restoration of natural habitats (Godfray et al., 2014; Williams et al., 2021). While some policies aim to protect larger areas for nature, under a more land-sparing approach, these have received far less funding than (predominantly land sharing) agricultural schemes; for example, in Europe, the Natura scheme which creates protected areas received just 1% of the funding paid to farmers through agri-environment schemes (Kettunen et al., 2011). This substantial investment into land sharing has continued despite there being, to date, no attempt to estimate and compare the costs to taxpayers of pursuing these alternative approaches to reducing the environmental footprint of farming.

Here we address this important gap using data for the United Kingdom. Agriculture constitutes only 0.58% of the UK's GDP (World Bank, 2021), yet covers 70% of its land surface (Defra, 2018a). Following the UK's exit from the European Union, the UK Government are presently devising an entirely new agricultural policy. Therefore, Brexit offers an opportunity to review current sharing-oriented environmental policies which have mostly failed, in the United Kingdom and the European Union, to reverse biodiversity declines (Batáry et al., 2015; Inger et al., 2015; Pe'er et al., 2020), despite some measures having positive effects for some species (Baker et al., 2012; Walker et al., 2018) and public expenditures of €3.2 bn/year across Europe (Batáry et al., 2015) and >£600 m/year in the United Kingdom (RSPB, 2020). Importantly, the EU is a net importer of calories and protein (Ruiz Mirazo et al., 2022); and, in the United Kingdom, 67% of the land used to grow food consumed in the country is located overseas (De Ruiter et al., 2016) so any conservation efforts that reduce domestic production risk increasing off-shored demand, potentially exacerbating, rather than alleviating, the global extinction and climate crises (De Ruiter et al., 2016; Lenzen et al., 2012; Smith et al., 2019).

A key component of the overall costs of current policy is the payment required by farmers to change their practices for the benefit of the environment. These environmental payments are expected to cover the opportunity costs of forgone profits, since otherwise many farmers will not participate in such agri-environment schemes. If biodiversity outcomes for a given level of food production are greater under land sparing (as found in the empirical studies of Dotta et al., 2016; Finch et al., 2019, 2020; Kamp et al., 2015; Phalan et al., 2011; Williams et al., 2017), such costs are anticipated to be lower under sparing than sharing interventions. However the payments farmers require also reflect attitudes towards the time, expense and effects of participating in such agri-environment schemes (AES) (Dessart et al., 2019). Farmer attitudes towards sharing and sparing interventions may differ; the larger scale of sparing may be attractive, given uncertainty over the future profitability of farming (Defra, 2018b), but sharing may be more familiar, which may reduce the payments farmers require to participate. Indeed, past criticisms of land sparing have included unquantified suggestions that farmers prefer wildlife-friendly farming (Jiren et al., 2018; Kremen, 2015; Quandt, 2016), among other concerns about the outcomes of sparing for farmers and for habitat heterogeneity (Kremen, 2015; von Wehrden et al., 2014). There are additional financial costs to consider as well: these include one-off capital costs of changing production methods, the administration costs of scheme delivery and the costs of monitoring schemes. All may differ between sharing and sparing, but so far none have been compared in a like-for-like manner. Last, in addition to these costs to taxpayers, the relative amount of food production lost in delivering environmental outcomes on currently farmed land is important. If any scheme leads to a reduction in farmed land, yields must increase or demand for imported food would rise with consequences for biodiversity, carbon emissions and people elsewhere (Lenzen et al., 2012; Smith et al., 2019). One might expect levels of food production forgone to co-vary with payments required by farmers (see above), but it is important to explore whether the same is true of the other costs to taxpayers.

Here, we present a novel comparison of the taxpayer and food production costs of sharing and sparing schemes that deliver equivalent environmental outcomes. We identified a series of outcomes—each deliverable by both sharing- and sparing-style interventions—that are broadly representative of outcomes targeted by existing arable subsidy schemes. We used a stated preference choice experiment to establish the minimum payments required by farmers to implement sharing (stubble/spring cropping, reduced fertiliser, winter bird cover, fallow plots and hedgerow creation) and sparing (scrub, woodland and wet grassland creation) interventions, and the variation in this minimum supply price across farmers. From this, we simulated fixed-price AES, where a uniform subsidy is paid to all farmers who participate, that delivered the target outcomes, and calculated the associated capital, administration and monitoring costs. Finally, we compared these taxpayer costs with the amount of food energy lost in delivering the same outcomes through sharing and sparing.

2 | METHODS

2.1 | Identification of sharing and sparing interventions

To compare the costs of delivering environmental outcomes via sharing or sparing, we selected a set of conservation outcomes which are targeted by current agri-environment schemes, which are deliverable on arable land by both sharing and sparing interventions, and for which the effects of such interventions are well characterised. Given these considerations, we assessed the costs of meeting hypothetical but plausible targets for conserving three bird species and delivering net reductions in carbon emissions. Our three focal species all occur on farmland but differ in their response to changes in farm yield (Finch et al., 2019). In order of decreasing abundance on farmland, our three study species were: Yellowhammer *Emberiza citrinella*, Northern Bullfinch *Pyrrhula pyrrhula* and Northern Lapwing *Vanellus vanellus*. Using existing literature, we identified sharing and sparing interventions which increase populations of these species by boosting a limiting life-history parameter (without necessarily meeting all of a species' needs year-round; Table 1). In identifying interventions, we assumed land sharing to involve practices that are implemented on the farmed area and that cause a production loss. By contrast, we assumed land-sparing interventions to be separate to the farmed area with production entirely forgone, except for any low-level grazing needed to prevent succession of the habitat, where such grazing is managed primarily to maximise the biodiversity value of the landscape. For broader discussion of the distinction between sparing and sharing interventions, see Sidemo-Holm et al. (2021).

We considered two different types of sharing intervention: in-field, which affects food-producing practices across the whole field, and field-edge, which involves addition of an intervention outside the area used to produce food, typically the field margin. For both bird and carbon outcomes we then calculated the associated per-area benefit delivered by the appropriate in-field sharing, field-edge sharing and sparing options (Table 1; Supporting Information). In line with evidence of the rapid recovery of birds on previously farmed land restored to natural habitat (Eglington et al., 2007; Marren, 2016; Vanhinsbergh et al., 2002), we assumed our estimated per-area benefits would emerge within the 20-year timeframe of the schemes. We could not incorporate the uncertainty associated with these estimates since many of the studies from which they were derived did not report their standard errors.

2.2 | Choice experiment set-up

We conducted a stated preference choice experiment to establish the payments required by farmers to implement these sharing and sparing interventions. The experiment was run via an online Qualtrics survey, although participants had the option to use paper, which eight did. Participants were asked to make 12 choices, each of which involved an in-field sharing, field-edge sharing and sparing option, plus the option not to select any of the contracts (see Figure 1 for a sample choice card). As well as varying in the type of intervention, these options differed in area, duration and payment rate, since a large number of other studies have shown farmers' willingness to participate to depend on these contract attributes (e.g. Barreiro-Hurlé et al., 2010; Christensen et al., 2011; Villanueva et al., 2016). These attributes were set at the following levels (summarised in Table S4):

a. Areas were set to be achievable on most arable farms. In-field and sparing areas were set at 10, 20 and 50 ha (with 50 ha excluded for farms <100 ha), and all field-edge sharing options set at 5, 10 and 20 ha (except hedgerow creation, where we set smaller areas of 2, 4 and 8 ha which, for simplicity, were presented to participants as km lengths [assuming 6 m hedgerow width]).
b. Durations were set at 10, 20 and 50 years for all sparing options and (given their permanence) for creation of hedgerows; and 5, 10 and 20 years for all other sharing options. We did not explore 5-year timeframes for sparing (and hedgerow) schemes given this is likely inadequate time to create high-quality habitats.
c. Payment rates were set such that the compensation offered reflected the costs of implementing each intervention on an average English arable farm. Payment rates (in GBP/year) were set at approximately 0.33×, 0.67×, 1×, 1.33× and 2× the average participant's estimated lost gross margin from participating in the scheme (calculated using means from the Farm Business Survey (Farm Business Survey, 2020); Supporting Information). Where appropriate, capital costs were stated to be covered separately and in full.

Given this number of attributes and levels, a large number of combinations was possible. Using pilot data, we used Ngene (Metrics, 2018) to generate an efficient design. The resulting design consisted of 12 blocks each comprising 12 choices, with each participant randomly assigned to one block. The survey began by asking participants whether they preferred to answer in acres or hectares, followed by the area they farmed (to allow 50 ha interventions to be removed for those farming <100 ha). Participants then completed the 12 choices and some follow-up questions about their reasons for their choices (not explored here). Then, participants were asked to detail the crops/livestock they produced, and the associated areas, yields, selling prices and input costs, in order to allow calculation of each farmer's food energy and gross margin lost by implementing each of the studied options.

2.3 | Choice experiment data collection

We obtained ethics approval from the University of Cambridge Psychology Research Ethics Committee (HVS/2018/2582). Informed written consent was given by all participants before completing the study. We piloted the study with 11 participants in June/July 2019. We then launched the final version of the survey and obtained 118 responses from individuals in England and bordering areas in Wales between September 2019 and June 2020 who farmed a total of 76,072 ha, that is, 1.7% of lowland arable land in England (Defra, 2019). We recruited participants through a variety of means including farming newsletters, magazines, Twitter and online fora. Respondents were offered a summary of the findings, a personalised estimation of their costs of implementing the studied interventions, and the opportunity to win a subscription to Farmers Weekly. Our sample was over-representative of younger farmers and larger farms (Figure S4).

TABLE 1 The sharing and sparing interventions that deliver the environmental outcomes studied and the estimated per-area benefit, in terms of the number of birds or tonnes carbon per unit area, delivered by implementing each intervention instead of continuing to farm as before

Environmental outcome	Intervention type	Intervention	Benefit (birds/ha or, for reduced net carbon emissions, tC/ha/year)	Source
Yellowhammer	In-field sharing	Stubble, spring cropping on wheat, barley and/or oats	0.26	Hancock and Wilson (2003)
	Field-edge sharing	Winter bird cover	0.83	Henderson et al. (2012); Parish and Sotherton (2004); Stoate et al. (2003)
	Field-edge sharing	Hedgerow creation	4.67	Macdonald and Johnson (1995); Bradbury et al. (2001)
	Sparing	Scrub	0.59	Morgan (1975); Donovan (2013)
Bullfinch	Field-edge sharing	Hedgerows	0.92	Macdonald and Johnson (1995)
	Sparing	Scrub	0.20	Morgan (1975); Knepp Estate
	Sparing	Woodland	0.05	Lamb et al. (2019); Newson et al. (2005); Gregory and Baillie (1998)
Lapwing	In-field sharing	Stubble, spring cropping	0.05	Wilson et al. (2001); Shrubb et al. (1991)
	Field-edge sharing	Fallow	0.17	Chamberlain et al. (2009)
	Sparing	Wet grassland	0.49	Ausden and Hirons (2002); Eglington et al. (2007); RSPB Reserves data
Reduced net carbon emissions	In-field sharing	50% reduction of inorganic N fertiliser on wheat, barley, oil seed rape, sugar beet and/or potatoes	0.27[a]	Kindred et al. (2008)
	Field-edge sharing	Hedgerows	1.84	IPCC (2019)
	Sparing	Woodland	3.77	Falloon et al. (2004)

Abbreviations: ha, hectare; tC, tonnes carbon; y, year.

[a]Benefit shown here was estimated according to mean rates of fertiliser application (Farm Business Survey, 2020); our study estimated the benefit delivered based on participants' reported fertiliser application rates.

FIGURE 1 Sample choice card.

We used the choice experiment data to simulate fixed-price schemes which enrol only the most-willing participants, so we were interested in the distribution of preferences across our sampled farmers. Therefore, we used a mixed logit model which assumes that preferences vary within the population according to a specified distribution. We assumed preferences towards all parameters were normally distributed in the population except the payment parameter for which we assumed a u-shifted negative log-normal distribution (Crastes dit Sourd, 2021) to ensure that no participant disliked greater payments while also avoiding so-called exploding ratio issues when computing welfare estimates(see Table S5 for variations, all of which worsened model fit). Under mixed logit, the probability of individual n choosing alternative j is:

$$P_n = \int \frac{e^{V(\beta, X_{nj})}}{\sum_j^J e^{V(\beta, X_{nj})}} f(\beta | \theta) d\beta, \tag{1}$$

where X_{ni} is the vector of explanatory variables for alternative j faced by participant n, and β is the vector of taste coefficients, and the function $V(\beta, X_{nj})$ gives the observed utility of alternative j (Train, 2009). For mixed logit, the vector β is distributed randomly across participants, with density $f(\beta | \theta)$ where θ is a vector of parameters to be estimated that represent the mean and variance of preferences in the population. Modelling then seeks to find the parameters that maximise the log-likelihood, LL, of the model across all N participants who complete T choice situations, that is:

$$\mathrm{LL} = \sum_{n=1}^{N} \ln \prod_{t=1}^{T_n} P_{nj}. \tag{2}$$

2.4 | Choice experiment analysis

We calculated participants' minimum willingness to accept (WTA) compensation for implementing a scheme with specific attribute values first for the sample mean, and then for each individual using the posterior sensitivities produced by Apollo (an R package for choice experiment analysis; Hess & Palma, 2019). These individual-level estimates of each participant's mean WTA (rather than the whole survey sample) were obtained by conditioning the model estimates on survey choices for each respondent, as further detailed by Train (2009). To do so, we assumed WTA payment for a nonmonetary parameter ($\mathrm{WTA_{NM}}$) was given by the ratio of nonmonetary parameters (β_{NM}) to the payment parameter (β_M), that is:

$$\mathrm{WTA_{NM}} = -\frac{\beta_{NM}}{\beta_M}. \tag{3}$$

Based on individual-level estimates of participants' WTA and the benefit delivered by each intervention, we next simulated the cost of delivering different amounts of our target outcomes with fixed-price schemes of 20 years' duration in 2019 GBP and using a 3.5% discount rate (as advocated by HM Treasury, 2018) to reflect society's tendency to perceive future payoffs as lower in value. For sharing, we costed the combination of in-field and field-edge sharing interventions that achieved the target outcomes at least expense to the taxpayer. Similarly, because bullfinches could be delivered by two sparing interventions, we allowed both to contribute to the outcome, based on what was least expensive. Across all sharing and sparing interventions, we assumed farms could implement multiple interventions where the area enrolled in any one intervention was not extrapolated beyond the areas presented in the choice experiment.

2.5 | Simulating the costs of delivering the target outcomes

We set the target for the three bird species as increasing the adult population size by 300 in the area farmed by our participants. This was set to be ambitious but also, according to the choice experiment output, deliverable within our sampled group with payments below

£2000/ha/year. We then set the net carbon emissions reductions target so that, under sharing interventions, the same amount was spent on carbon as on our three biodiversity outcomes combined. We treated the small number of negative WTA values derived from the choice experiment analysis as zeros (negative values imply that a farmer would be willing to pay to enrol in the scheme); they mostly arose for stubble/spring cropping which is commonly practised for weed/pest control and was often found to require no additional compensation. We then found the 95% confidence intervals of our estimates of delivering all the targets with sharing and sparing by bootstrapping. We produced 1000 bootstrap samples of our choice experiment data by selecting results from respondents at random, with replacement. We fitted the model to the data from each bootstrap sample and calculated the cost of sharing and sparing schemes, and the difference between sharing and sparing schemes, from the parameters of the fitted model for each sample. We took the lower and upper 95% confidence limits of these modelled outcomes to be the 2.5th and 97.5th percentiles of the 1000 bootstrap values of each outcome.

In setting the compensation payment rates required to deliver our targets within the sample, we also need to consider noncompliance; this reduces the benefit delivered by scheme participants, such that the target may not be delivered in full. Increased monitoring deters noncompliance but is costly. The financially optimal monitoring rate depends on the trade-off between increased spend on monitoring and the cost of paying additional participants to enrol in the scheme to make up the benefit lost to noncompliance (Ozanne et al., 2001). In summary, our approach to estimating noncompliance, and the cost of delivering targets in spite of it (detailed in Supporting Information), used utility theory to assess the noncompliance arising at given compensation payment and monitoring rates for each intervention. Based on this, we found the payment and monitoring rates that delivered the target outcomes at least cost despite noncompliance and found the cost of delivering these monitoring rates using cost estimates from current schemes.

Knowing the area enrolled by each participant in each intervention, we then estimated the associated capital and administration costs. Capital costs were estimated for hedgerows, scrub, wet grassland and woodland creation based on per-ha cost estimates published in the grey and white literature (Supporting Information). The per-agreement administration costs were set at £458/year, estimated from the reported £6.48 m spent on administering 19,118 agreements in 2009 (Natural England, 2009), and adjusting for inflation through to 2019 (Bank of England, 2021).

Finally, we estimated the food lost in delivering our outcomes through the interventions assessed, based on participants' reported yields (Supporting Information). In doing so, we took account of the fact that yields vary across farms; and that yields vary within fields, with field-edge sharing options probably being implemented on the least productive parts of the field. We assumed spared land would come from all crop/livestock types produced by the farmer, in proportion to their relative areas, to allow for rotation. In this way, we likely overestimated the food production lost to sparing since, in reality, farmers may be able to disproportionately allocate land from less profitable aspects of the rotation to agri-environment schemes. Given these assumptions, we estimated the tonnes of each crop/livestock type lost given the area enrolled in each intervention. We converted from tonnes to food energy given, for each crop/livestock type, the proportion consumed by humans versus livestock, the edible proportion and the per-weight energy content (as per Finch et al., 2019; Supporting Information).

3 | RESULTS

Mixed logit analysis of our choice experiment data revealed preferences for contracts varying in the intervention required and the area and duration over which it was implemented (Table 2). To eliminate the effects of protest votes (Adamowicz et al., 1998) we excluded six participants who opted out of every choice as this improved model fit (Table S5). On average, these participants were less likely to be participating in current schemes (17% vs. 43%) and were more confident of their future profitability (3.2 vs. 2.4 on a 5-point scale where higher numbers indicate greater confidence).

Aside from the price offered, the resulting mean parameter estimates reflecting average farmers' preferences towards each contract attribute were negative. This indicates, as expected, that farmers require monetary compensation to implement any AES option, with greater compensation required for contracts with larger areas and longer durations. The sparing contract attribute parameters were more negative than the sharing parameters (except for hedgerow creation), indicating that, for a given size and duration of intervention, more compensation was required for the average participant to participate in a sparing scheme than a sharing scheme. Participants demonstrated significant preference heterogeneity for all contract attributes, as reflected by the sizeable standard deviations of our parameter estimates. This heterogeneity is important, since those farmers with the lowest minimum WTA are those which are more willing to participate in fixed-price AES, with the number of participants required for each option to achieve a given outcome driven by the area required to deliver that outcome (Supporting Information).

Figure 2 shows our estimates of the cost of fixed-price AES, including payments to farmers, capital costs, compliance monitoring costs and administration costs, that delivered varying proportions of the target outcomes. The combined target outcomes of 300 Northern Bullfinches *Pyrrhula pyrrhula*, 300 Northern Lapwings *Vanellus vanellus*, 300 Yellowhammers *Emberiza citrinella* and a reduction in net greenhouse gas emissions of 1557 tC/year are shown as being delivered when the 'Proportion of Target' equals 1. We present costs for outcomes smaller than our targets since the government may opt for actions less ambitious that ours, as indeed is the case in current schemes (Figure S5).

Our calculations revealed that sparing interventions were less expensive than sharing in terms of each component of taxpayer costs, regardless of the proportion of the targets delivered (Figure 2). Although the average farmer was willing to accept lower

TABLE 2 Mixed logit model excluding participants that opted out of every choice and assuming all parameters were normally distributed besides the payment parameter which is presented here back-transformed from its negative log-normal specification (see Table S5 for other distributional assumptions). Standard errors for mean WTA calculated via bootstrapping

	Contract attribute	Mean	SE	Standard deviation	SE	Mean WTA/£	SE/£
Sharing	Stubble/spring cropping	−0.357	0.273	1.235*	0.250	75.58	74.58
	Reduced fertiliser	−1.616*	0.373	1.851*	0.405	370.11*	83.72
	Winter bird cover	−1.686*	0.342	1.560*	0.358	405.59*	71.49
	Fallow plots	−1.968*	0.341	1.223*	0.431	447.43*	84.30
	Hedge	−6.687*	1.001	4.750*	0.810	1498.49*	279.50
Sparing	Scrub	−5.190*	0.825	2.574*	0.624	1190.45*	156.04
	Woodland	−6.014*	0.866	3.122*	0.870	1445.48*	254.61
	Wet grass	−8.128*	1.565	−6.082*	1.141	2007.44*	488.14
	Area	−0.020*	0.008	−0.047*	0.011	4.88*	1.96
	Duration	−0.047*	0.011	0.058*	0.010	11.85*	3.47
	Payment	0.004*	0.001	0.006*	0.001		
	Log-likelihood	−1109					
	R^2	0.29					
	AIC	2264					
	BIC	2374					

*Significant at 5% level.

compensation payments per hectare for sharing interventions (Table 2), the overall costs of the compensation payments to farmers needed to deliver our target outcomes were substantially lower for sparing because of the greater environmental benefits delivered per unit area. Capital costs, which are paid to farmers at the start of a contract, were greater for sharing because hedgerow creation, the only sharing intervention that involved capital costs, was far less efficient at sequestering carbon than woodland, the equivalent sparing option (Figure 2b). Administration and compliance monitoring costs were also both substantially cheaper for sparing interventions because the greater benefit delivered per unit area meant our target outcomes could be delivered with far fewer scheme participants compared to those needed to meet the same outcomes through sharing interventions (Figure 2c,d).

Combining all of the component taxpayer costs presented in Figure 2, we found that sparing delivered the target outcomes at 48% of the cost of sharing (Figure 3). These taxpayer costs were dominated by compensation payments to farmers, which represent the minimum annual financial compensation they would demand to participate under a fixed-price scheme (Figure 4; orange area). Capital costs were a sizeable component, particularly for sharing, where substantial hedgerow creation was needed to deliver the carbon emissions reduction target. Administration costs were a relatively small component, although they reflect only the processing costs associated with each agreement; other running costs were not explored since they were not thought to differ substantially between sharing and sparing schemes. Compliance monitoring was a small, but very important, component of scheme costs. With inadequate monitoring scheme costs would increase dramatically since many more participants must be paid to enrol to make up the benefit lost to noncompliance.

Turning to lost food production, we found sparing delivered the target outcomes with loss of <3% of the total food produced by the sampled farmers; this is 79% of the food lost in delivering the same outcomes with sharing (Figure 5a). This difference is approximately in line with the relative difference in compensation payments to farmers (Figure 5b, orange vs. black line). The relative difference, between sharing and sparing schemes, was greater for other costs (capital, administration and compliance monitoring; Figure 5b, grey, green and lilac lines). As a result, the overall difference in taxpayer costs between sharing and sparing schemes was greater than the difference in the energy value of lost food production (Figure 5b, red vs. black lines).

4 | DISCUSSION

We found that sparing interventions delivered our target environmental outcomes at less than half the overall cost to the taxpayer of sharing interventions. The difference in compensation payments to farmers between sharing and sparing was roughly in line with the energy costs of lost food production. However, although payments to farmers comprise the majority of taxpayer cost, other types of cost favoured sparing even more strongly; thus, the savings to the taxpayer offered by sparing, relative to sharing were greater than the difference in lost food production (48% vs. 79%). To our knowledge this is the first evidence that sparing schemes cost the taxpayer less than sharing schemes which deliver the same environmental

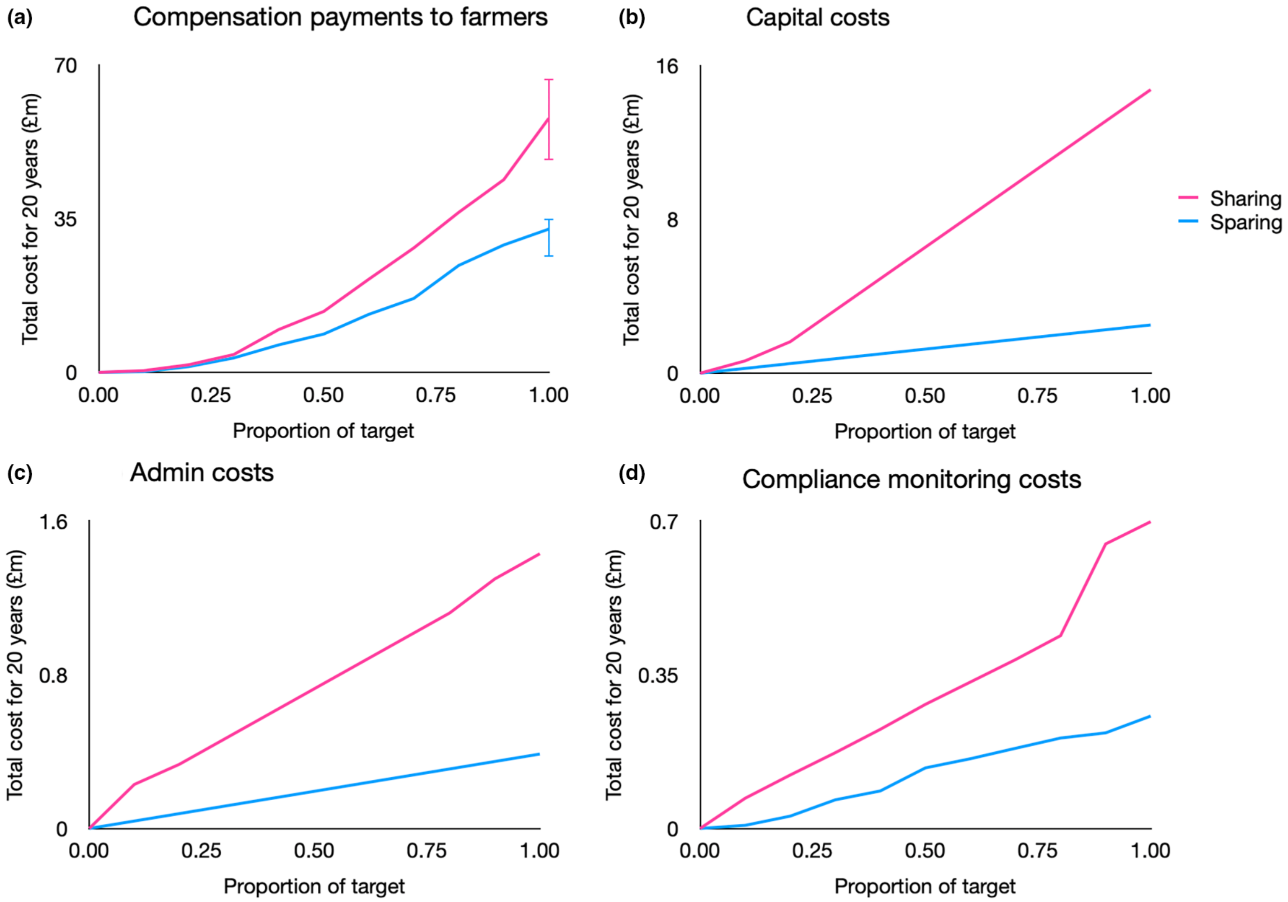
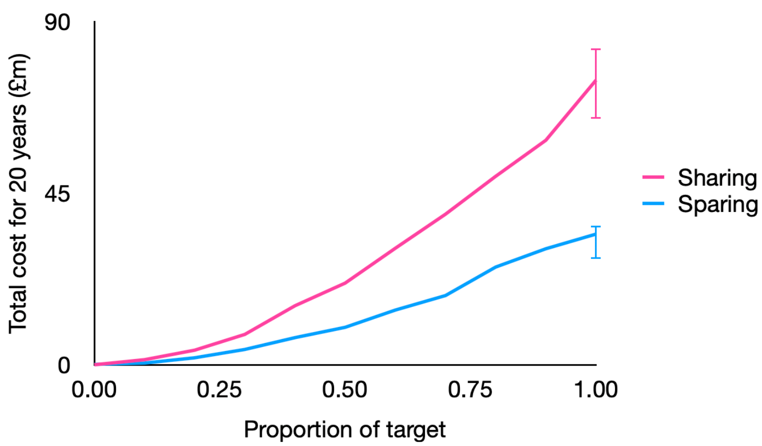


FIGURE 2 The component taxpayer costs of sharing (pink; stubble/spring cropping, 50% reduction in N fertiliser, winter bird seed plots, fallow plots and hedgerow creation) and sparing (blue; creation of scrub, wet grassland and woodland) schemes that delivered varying proportions of the combined target outcomes of yellowhammers, lapwings, bullfinches and net carbon emissions. Ninety-five per cent bootstrapped confidence intervals reflect uncertainty in compensation payments to farmers only. Costs expressed in 2019 GBP and with a 3.5% discount rate, following HM Treasury (2018).

FIGURE 3 The overall costs to the taxpayer (compensation payments, capital, administration and compliance monitoring) of 20-year sharing (pink) and sparing (blue) schemes that delivered a range of proportions of the combined target outcomes of biodiversity and net carbon emissions. Ninety-five per cent bootstrapped confidence intervals reflect uncertainty in compensation payments to farmers; other sources of error exist but were not quantified (see Section 4).



outcome, and importantly that the extent to which sparing is cheaper is greater than the difference in lost food production. That we found this conclusion in a country with a history of agriculture as long as the UK suggests that even greater cost efficiencies may be afforded by land sparing rather than sharing in countries where many farmland-sensitive species are not already extinct (see below).

Inevitably our study has several important limitations. First, while the difference between the cost of sharing and sparing scheme is substantial, not all sources of uncertainty were incorporated. In particular, we could not incorporate the uncertainty in estimates of the environmental benefits delivered per unit area of each intervention type since these estimates were derived from existing studies, many

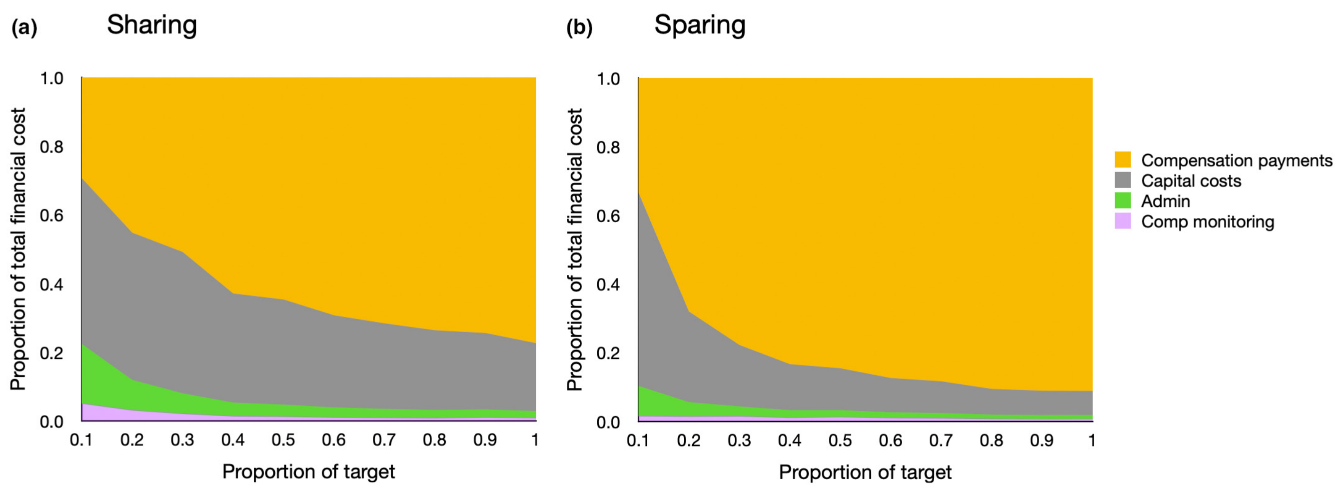


FIGURE 4 The proportion of taxpayer costs of (a) sharing and (b) sparing schemes that delivered varying proportions of the combined target outcomes that were compensation payments to farmers (orange), capital costs (grey) administration costs (green) and compliance monitoring (pink).

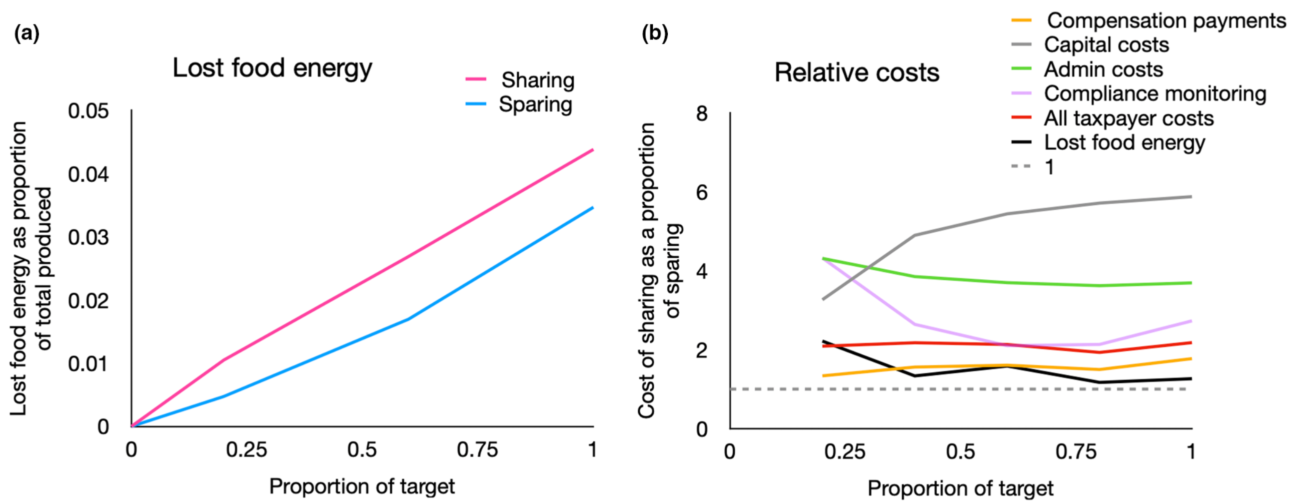


FIGURE 5 (a) The food energy lost, as a proportion of the total produced by the sampled farmers, in delivering the target environmental outcomes with sharing (pink) and sparing (blue). (b) The costs of sharing as a proportion of sparing.

of which did not report standard errors of effect sizes (Supporting Information). We did, however, explore the extent to which the relative benefits estimated to be delivered by sparing would need to be reduced before conclusions changed: we found sharing became the less expensive strategy when the benefit delivered by sparing was >33% lower than our original estimates (Figure S10). Furthermore, choice experiments rely on stated intentions, which may not align with actual behaviour, such that farmers may accept more or less compensation than found here. We did, however, compare the participation predicted by our model at the payment rates of current schemes and found good alignment (Figure S5). Second, our assessment of costs is incomplete. In particular, our combined total did not include the costs of monitoring schemes to assess intervention effectiveness. This is challenging because existing studies have not sought to compare the costs of monitoring the effectiveness of sharing and sparing schemes in a like-for-like way. Third, we were limited in the areal extent of the interventions considered, given what is feasible for the 'typical' English arable farmer. A comprehensive exploration of the relative costs of contrasting approaches would ideally involve the cost of implementing interventions over larger areas across multiple adjacent farms, particularly for sparing interventions, whose conservation benefits are likely to increase disproportionately in larger, and better connected, patches (Lamb et al., 2016); however, such an analysis would also have to consider the financial incentives needed to encourage spatial coordination (Banerjee et al., 2021; Liu et al., 2019). Finally, some stakeholders might only be interested in either delivering biodiversity or carbon emission outcomes (which here we have presented together). However, we did explore the relative costs of delivering each in turn; again we found sparing cheaper, although for biodiversity it was 77%

the cost of sparing, compared to 11% when only carbon was considered (Figure S8). This underscores the huge efficiency gains generated by using sparing rather than sharing interventions to reduce net carbon emissions, particularly at higher targets (Figure S9).

Although much research has explored the factors driving the adoption of different farming practices (reviewed in Dessart et al., 2019), we had little prior knowledge of farmers' willingness to implement the less familiar and larger-scale sparing interventions relative to sharing. Indeed, on average, farmers did require less compensation to implement sharing options. That the difference in compensation payments to farmers roughly tracked lost food production implies that the payments required are driven by the value of lost production, and other attitudes that affect farmer's minimum supply price (WTA) do not substantially differ between sharing and sparing. However, elsewhere, we have shown that to deliver higher targets than those assessed here, schemes must recruit farmers who require more compensation above the value of lost production (i.e. lost gross margin), with this effect substantially more marked for sharing than for sparing (Collas et al., unpublished). This suggests that, provided their lost gross margins are covered, farmers can be considered to prefer sparing (ibid). This is an important evidence-based challenge to previously unquantified suggestions that farmers prefer sharing (Jiren et al., 2018; Kremen, 2015; Quandt, 2016). We found more divergence between sparing and sharing for compliance monitoring costs. Elsewhere we have shown that current schemes are inadequately monitored for compliance and effectiveness which both increases costs and reduces the likelihood that schemes deliver target outcomes (Pe'er et al., 2020); policymakers should thus be encouraged that sparing interventions require less monitoring than sharing.

Given that some species, particularly in countries with long histories of agriculture such as the United Kingdom, depend on farmland for all or part of their life cycle, Finch et al. (2019) found bird densities were highest under a three-compartment strategy where high-yield farming is used to enable large areas to be spared for nature both in the form of (semi)-natural habitat and low-yield farmland. In the first assessment of the relative costs, we found that this three-compartment sparing strategy, which combined sparing- and sharing-style interventions, was two-thirds the taxpayer cost of the purely sparing strategy, although it offered little savings in terms of lost food production (Figure S6). These taxpayer savings largely arise because yellowhammers, the species found at highest densities on farmland of those considered, were readily delivered by sharing interventions which some farmers were willing to implement at little cost (Figure S7a), while other species and carbon were delivered at less cost with predominantly sparing interventions.

Given we studied only three species and one ecosystem service under arable farming, it is important to question whether our findings would hold across a wider array of outcomes and farming systems; on this, we suggest two interesting lines of thought. First, it is important to note that compensation payments to farmers were consistently the largest component of the schemes we considered and that differences in compensation costs for sharing and sparing roughly tracked differences in lost food production. Other studies in the United Kingdom have compared a much broader range of conservation outcomes and farming systems but only compared them with the food production consequences of sharing and sparing. Evidence from two regions of the United Kingdom and over 100 bird species found land sparing increased the populations of more species and resulted in better outcomes in terms of global warming potential, nitrogen and phosphorus pollution and outdoor recreation than did land sharing (Finch et al., 2019, 2021). This broad conclusion for biodiversity has been replicated on five continents for over 1500 species of trees, sedges, greases, forbs and insects, with the result typically more marked than for bird species (see review by Balmford, 2021). Unless the link we found between lost food production and compensation payments does not hold more broadly, which seems unlikely, this evidence suggests land sparing may indeed be less expensive to the taxpayer in delivering given improvements across a much broader range of species and ecosystem services than those we were able to study here. Further study, particularly of other farm systems (e.g. upland livestock production) and other species and ecosystem services, is needed to explore this.

Second, importantly, our analysis underestimates the costs of sharing relative to sparing in at least three ways. First, we do not explicitly consider the taxpayer and environmental consequences of increasing imports to compensate for the 1.3× greater loss, relative to sparing, in domestic food production. This is particularly concerning given that the biodiversity and climate impacts of food produced overseas may be even greater than domestic production. Currently, the United Kingdom imports half its food (Department for Environment Food & Rural Affairs, 2021); but imported food, along with other imported commodities, already causes 5× more species threats overseas than domestically (Lenzen et al., 2012). Furthermore, food imported to meet consumer demand in developed countries is known to increase carbon emissions elsewhere in the world (Smith et al., 2019) and the carbon footprints of imported goods account for 64% of all food emissions, suggesting they have disproportionately greater footprints than the 46% of food produced domestically (De Ruiter et al., 2016). Second, our assessment was deliberately conservative in considering only those environmental outcomes that are deliverable on farmland. However, nearly one in four of the lowland bird species found in England/Wales do not occur on land farmed at any intensity (Lamb et al., 2019) (Supporting Information), many of which are in need of conservation (Finch et al., 2019); and land sharing cannot aid the recovery of these species at all. Therefore, the inclusion of other habitat specialist species, which often show much more marked differences in population densities on spared versus farmed land, would greatly increase the estimated cost-efficiency of sparing relative to sharing. This is an important consideration in the United Kingdom, but likely even more so in countries where habitat conversation for agriculture is more recent and less widespread such that habitat specialists are likely to make up a higher proportion of the biota. Third, the cost-efficiency of sparing may be further improved with the agglomeration of spared areas, possibly achieved through changes in AES to encourage spatial coordination (Liu et al., 2019). The competitive tender of contracts through auction, or the differentiation of payments on the basis of the results delivered, may further improve cost-efficiency (Armsworth et al., 2012; Elliott et al., 2015). It is unclear whether any such improvement in cost-efficiency would differ systematically between sharing and sparing, although the implementation costs of a results-based payments approach may be lower for land sparing on the basis of its larger scale (Bartkowski et al., 2021; Herzon et al., 2018) and potential to deliver the same conservation outcome with fewer participants than land sharing.

In conclusion, based on our study of three species and one ecosystem service across arable farms in England, we found strong economic evidence in favour of a land-sparing approach to reconciling environmental conservation and food production. Consideration of the consequences of increased food imports, the species/services that do not persist on land farmed at any yield, and efficiency-improving measures, would only serve to increase the margin by which sparing would cost taxpayers less than sharing interventions that achieve the same outcomes. Prolonging the current predominance of land-sharing interventions risks delivering environmental outcomes at a greater cost to the taxpayer while potentially increasing environmental damage in food-exporting countries and reducing the space available for wild species that do not tolerate conditions on farmed land.

AUTHOR CONTRIBUTIONS

Lydia Collas, Rhys Green and Andrew Balmford conceived the ideas and designed methodology with input from Tom Finch; Lydia Collas collected the data; Lydia Collas and Romain Crastes dit Sourd analysed the data with input from Rhys Green, Nick Hanley and Andrew Balmford; Lydia Collas led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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CONFLICT OF INTEREST

I confirm none of the authors have any conflicting interests surrounding this publication.

DATA AVAILABILITY STATEMENT

Modelling data available on Dryad Digital Repository https://doi.org/10.5061/dryad.j0zpc86jd.

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SUPPORTING INFORMATION

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

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