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

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Contributed Paper prepared for presentation at the 96th Annual Conference of the Agricultural Economics Society, K U Leuven, Belgium

4 – 6 April 2022

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Acknowledgements: We are grateful to Alex Inman, Ian Bateman and Stephane Hess for advice on choice experiment design and to Georgie Bray and Dave Jones for feedback on the choice experiment survey. We thank Malcolm Ausden for sharing bird data from RSPB reserves. We thank Pete Carey and Aiden Keane for their help on the compliance monitoring analysis.

Abstract

The biodiversity and climate crises demand ambitious policies for lowering the environmental impacts of farming. Most current interventions incentivise so-called land-sharing approaches to addressing the widespread trade-off between farm yields and on-farm environmental outcomes, typically compensating farmers who adopt yield-reducing measures that encourage wildlife or reduce net emissions within farmed land. Here, we present the first quantification of the likely costs 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 UK, we used a choice experiment to explore farmer preferences and estimate the overall costs of contrasting agri-environment schemes that delivered populations of well-studied farmland birds and reduced net carbon emissions in England. We included capital, administration and monitoring costs, and lost food production. Sparing delivered our target outcomes for bullfinches, lapwings, yellowhammers and carbon emissions at 71% 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. 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 UK. Our results suggest that continuing a land-sharing approach in countries such as the UK is not only an inefficient use of government funds but also undermines conservation and food security in food-exporting countries who bear the burden of compensating domestic production forgone in the name of conservation.

Keywords Environmental economics, land-use policy, agriculture, biodiversity, carbon

JEL code Q58: Environmental economics, government policy

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, Cornell, Scharlemann, & Balmford, 2005). However, 15 years of empirical data from 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, with higher yields on already-cleared land freeing-up land elsewhere for the retention or restoration of natural habitats (Balmford, 2021; Dotta, Phalan, Silva, Green, & Balmford, 2016; Finch et al., 2019; Finch, Green, Massimino, Peach, & Balmford, 2020; Kamp et al., 2015; Phalan, Onial, Balmford, & Green, 2011; Williams et al., 2017). However, to date, there has been no attempt to estimate and compare the costs, particularly to taxpayers, of pursuing these alternative approaches to reducing the environmental footprint of farming.

Here we address this important gap using data for the UK. Agriculture constitutes only 0.58% of the UK's GDP (World Bank, 2021), yet covers 70% of its land surface (Defra, 2018a). Brexit offers an opportunity to review current sharing-oriented environmental policies which are widely perceived, in the UK and the European Union, as having delivered relatively little for biodiversity (Batáry, Dicks, Kleijn, & Sutherland, 2015; Dicks, Ashpole, Dänhardt, & James, 2014; D. Kleijn et al., 2006; David Kleijn & Sutherland, 2003; Pe'er et al., 2020), despite public expenditures of €3.2bn/y across Europe (Batáry et al., 2015) and >£500m/y in the UK (RSPB, 2020). Importantly, Europe sources most of its food from overseas; nearly 60% of the land needed to meet demand for agricultural and forestry products comes from elsewhere (Friends of the Earth, 2011), so any conservation efforts that reduce domestic production risk increasing off-shored demand and thus exacerbating, rather than alleviating, the global extinction and climate crises.

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. Compensation payments are expected to cover the opportunity costs of forgone profits which, if biodiversity outcomes for a given level of food production are greater under land sparing, are anticipated to be lower with 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, Barreiro-Hurlé, & Van Bavel, 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. There are other important costs to consider: 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, Kirk, Jones, & Williams, 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 studied outcomes deliverable by both sharing- and sparing-style interventions on arable land. 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.

Methods

Identification of sharing and sparing interventions

We assessed the costs of meeting hypothetical but plausible targets for conserving three bird species and delivering net reductions in carbon emissions. We chose species that all occur on farmland but that differ in their response to changes in farm yield (Finch et al., 2019): Northern Bullfinch (*Pyrrhula pyrrhula*), Northern Lapwing (*Vanellus vanellus*) and Yellowhammer (*Emberiza citrinella*). 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). We studied two different types of sharing intervention: in-field, which affects food-producing practices across the whole field, and marginal, which involves addition of an intervention outside the area used to produce food, typically the field margin. We calculated the associated per-area benefit delivered by the in-field sharing, marginal sharing and sparing options (Table 1; Supplementary Information, Section Cii). 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, Gough, Fuller, & Brierley, 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.

Table 1. The sharing and sparing interventions that deliver the conservation outcomes studied and their estimated per-area benefit.

Conservation outcome	**Intervention type**	**Intervention**	**Benefit** *(birds/ha or tC/ha/y)*	**Source**
Yellowhammer	In-field sharing	Stubble, spring cropping on wheat, barley and/or oats	0.26	Hancock and Wilson (2003)
	Marginal sharing	Winter bird cover	0.83	Henderson et al. (2012); Parish and Sotherton (2004); Stoate et al. (2003)
	Marginal sharing	Hedgerow creation	4.67	Macdonald and Johnson (1995); Bradbury et al. (2001)
	Sparing	Scrub	0.59	Morgan (1975); Donovan (2013)
Bullfinch	Marginal 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)
	Marginal 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	0.27 [1]	Kindred et al. (2008)

	beet and/or potatoes		
Marginal sharing	Hedgerows	1.84	IPCC (2019)
Sparing	Woodland	3.77	Falloon et al. (2004)

[1] Benefit shown here was estimated according to mean rates of fertiliser application (Farm Business Survey, 2019); our study estimated the benefit delivered based on participants' reported fertiliser application rates.

Choice experiment setup

We conducted a 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, though 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, marginal 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. Refs. (Barreiro-Hurlé, Espinosa-Goded, & Dupraz, 2010; Christensen et al., 2011; Villanueva, Rodríguez-Entrena, Arriaza, & Gómez-Limón, 2016)). These attributes were set at the following levels (summarised in Table S1):

- Areas were set to be achievable on most arable farms. In-field and sparing areas were set at 10, 20 and 50ha (with 50ha excluded for farms <100ha), and all marginal sharing options set at 5, 10 and 20ha (except hedgerow creation, where we set smaller areas of 2, 4 and 8ha which, for simplicity, were presented to participants as km lengths [assuming 6m hedgerow width]).
- 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.
- 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/y) were set at approximately 0.33x, 0.67x, 1x, 1.33x and 2x the average participant's estimated lost gross margin from participating in the scheme (calculated

using means from the Farm Business Survey (Farm Business Survey, 2019); Supplementary Information, Section C), rounded to the nearest GBP100. Where appropriate, capital costs were stated to be covered separately and in full.

Figure 1. Sample choice card.

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 50ha interventions to be removed for those farming <100ha). 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 variable costs, in order to allow calculation of each farmer's food energy and gross margin lost by implementing each of the studied options.

Choice experiment data collection

We obtained ethics approval from the University of Cambridge Psychology Research Ethics Committee (HVS/2018/2582) and 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 76,072ha, i.e. 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.

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 (see Table S3 for variations, all of which worsened model fit). Under mixed logit, the probability of individual *n* choosing alternative *j* is:

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

[1]

where X_{ni} is the vector of explanatory variables for alternative *j* faced by participant *n,* and β 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, i.e.:

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

[2]

Choice experiment analysis

We calculated participants' 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. 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 non-monetary parameter (WTA_{NM}) was given by the ratio of non-monetary parameters (β_{NM}) to the payment parameter (β_M), i.e.:

$$WTA_{NM} = -\frac{\beta_{NM}}{\beta_M}$$

[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 (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 marginal 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.

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/y. 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 non-compliance; this reduces the benefit delivered by scheme participants, such that the target may not be delivered in full. Increased monitoring deters non-compliance, 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 non-compliance (Ozanne, Hogan, & Colman, 2001). In summary, our approach to estimating non-compliance, and the cost of delivering targets in spite of it (detailed in Supplementary Information, Section C), used utility theory to assess the non-compliance 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 non-compliance, 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 (Supplementary Information, Section C). The per-agreement administration costs were set at £458/y, estimated from the reported £6.48m 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 (Supplementary Information, Section C). In doing so, we took account of the fact that yields vary across farms, and the likelihood that land would be spared from the least productive fields; and that yields vary within fields, with marginal sharing options probably being implemented on the least productive parts of the field. 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 vs livestock, the edible proportion, and the per-weight energy content (as per Finch et al., 2019; Supplementary Information, Section C).

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 1). To eliminate the effects of protest votes (Adamowicz, Boxall, Williams, & Louviere, 1998) we excluded six participants who opted out of every choice as this improved model fit (Table S3). 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 five-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 schemes, with the number of participants required for each option to achieve a given outcome driven by the area required to deliver that outcome (Supplementary Information, Section C).

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 S3 for other distributional assumptions). Standard errors for mean WTA calculated via bootstrapping. * = significant at 0.05

	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					

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 1557tC/y is 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 S1).

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 less compensation 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).

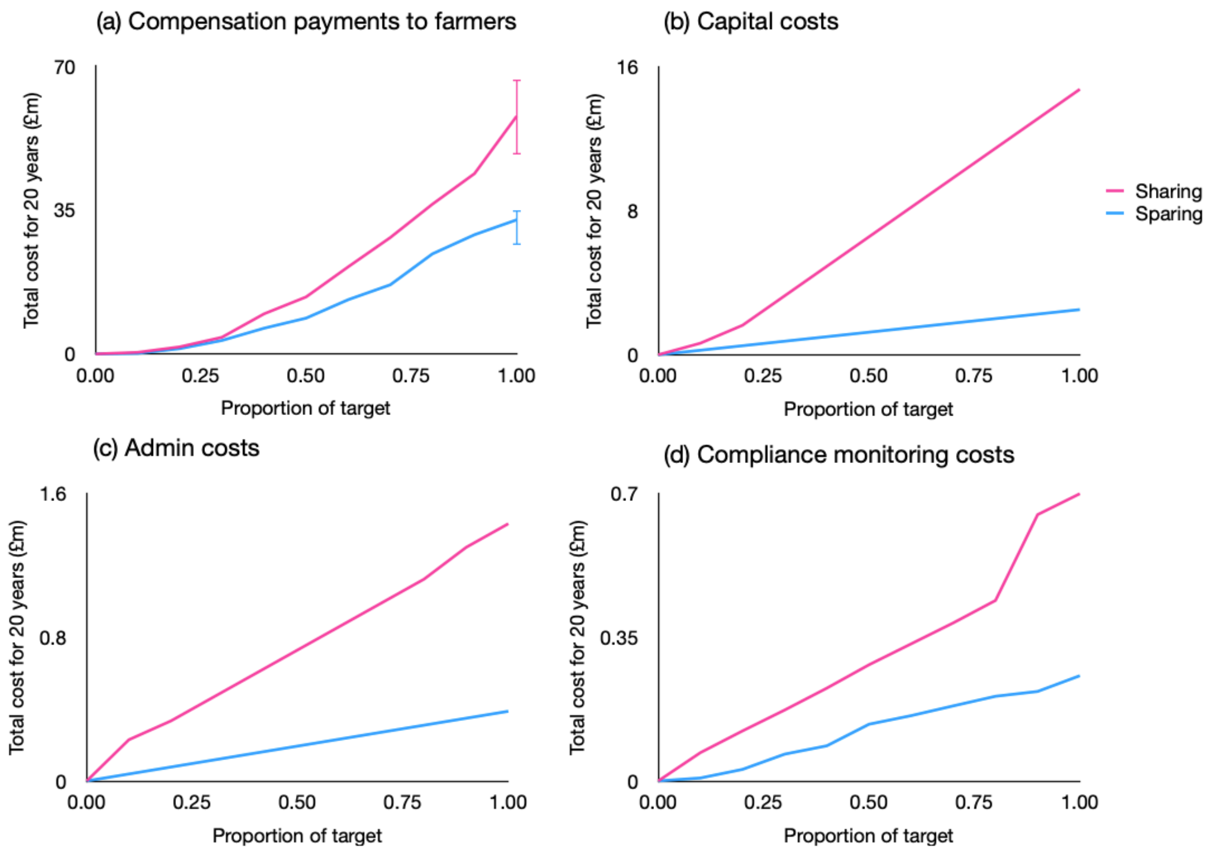

Figure 2. The component taxpayer costs of sharing (pink; stubble/spring cropping, 50% reduction in N fertiliser, winter bird seed plots, fallow plots and hedgerow creation) and sparing (blue; creation of scrub, wet grassland and woodland) schemes that delivered varying proportions of the combined target outcomes of yellowhammers, lapwings, bullfinches and net carbon emissions. 95% 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 (HM Treasury, 2018).

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 (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, though 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 non-compliance.

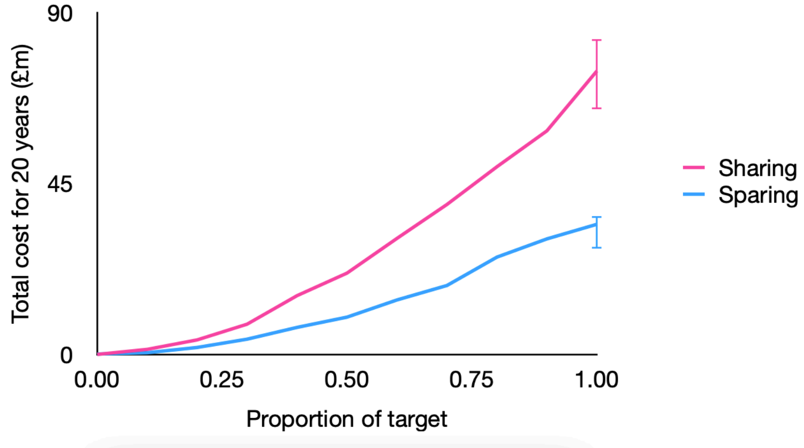

Figure 3. The overall costs to the taxpayer (compensation payments, capital, administration and compliance monitoring) of 20-year sharing (pink) and sparing (blue) schemes that delivered a range of proportions of the combined target outcomes of biodiversity and net carbon emissions. 95% bootstrapped confidence intervals reflect uncertainty in compensation payments to farmers; other sources of error exist but were not quantified (see Discussion).

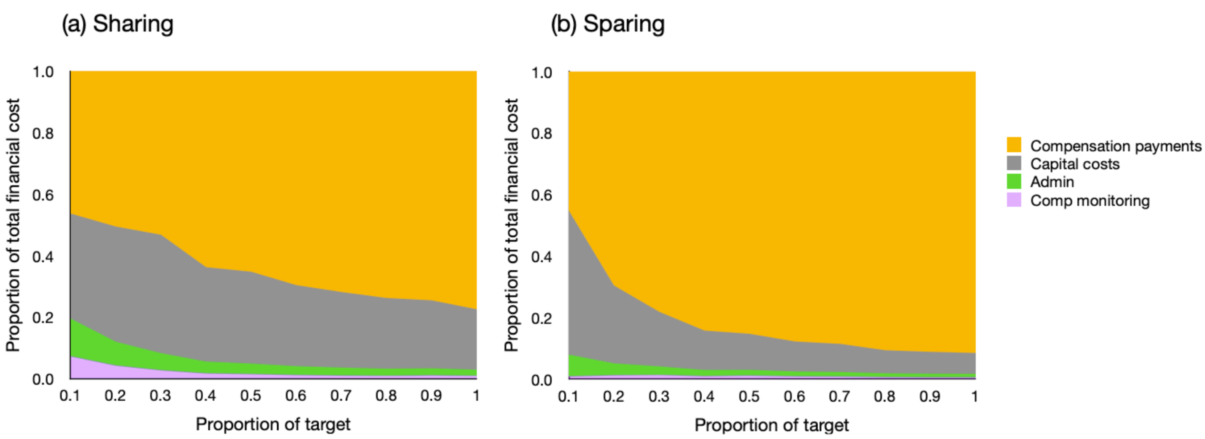

Figure 4. The proportion of taxpayer costs of (a) sharing and (b) sparing schemes that delivered varying proportions of the combined target outcomes that were compensation payments to farmers (orange), capital costs (grey) administration costs (green) and compliance monitoring (pink).

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 71% 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).

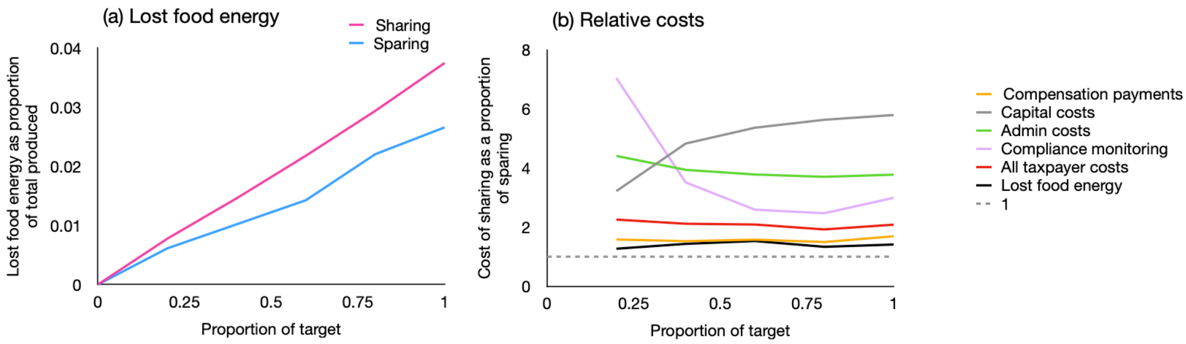

Figure 5. (a) The food energy lost, as a proportion of the total produced by the sampled farmers, in delivering the target environmental outcomes with sharing (pink) and sparing (blue). (b) The costs of sharing as a proportion of sparing.

Discussion

We found 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, though 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 71%). To our knowledge this is the first evidence that sparing schemes cost the taxpayer less than sharing schemes which deliver the same conservation 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, whilst 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 of which did not report standard errors of effect sizes (Supplementary Information, Section C). 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 S6). 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" UK 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 actions, whose conservation benefits are likely to increase disproportionately in larger, and better connected, patches (Lamb, Balmford, Green, & Phalan, 2016); however, such an analysis would also have to consider the financial incentives needed to encourage spatial coordination (Banerjee, Cason, Vries, & Hanley, 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, though for biodiversity it was 77% the cost of sparing, compared to 11% when only carbon was considered (Figure S4). This underscores the huge efficiency gains generated by using sparing rather than sharing interventions to reduce net carbon emissions, particularly at higher targets (Figure S5).

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 actions relative to sharing. Past criticisms to land sparing have included the unquantified suggestion that farmers prefer wildlife-friendly farming (Fischer et al., 2008). 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. In contrast, 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 (Collas, Carey, & Balmford, 2021, in prep.; see also Pe'er et al., 2020); policymakers should thus be encouraged that sparing interventions require less monitoring than sharing actions.

Given that some species, particularly in countries with long histories of agriculture such as the UK, depend on farmland for all or part of their lifecycle, Finch et al. (2019) found bird densities were highest under a 3-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 3-compartment sparing strategy, which combined sparing- and sharing-style interventions, was two-thirds the taxpayer cost of the purely sparing strategy, though it offered little savings in terms of lost food production (Figure S2). These taxpayer 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 S3a), whilst other species and carbon were delivered at less cost with predominantly sparing interventions.

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 1.4x greater loss, relative to sparing, in domestic food production. Food imported to meet consumer demand in developed countries is known to threaten biodiversity (Lenzen et al., 2012) and increase carbon emissions (Smith et al., 2019) elsewhere in the world. Second, our assessment was deliberately conservative in considering only those conservation 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., 2018) (Supplementary Information, Section D), 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 market differences in population densities on spared vs farmed land, would greatly increase the estimated cost-efficiency of sparing relative to sharing. This is an important consideration in the UK, 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). Differentiated pricing, or the competitive tender of contracts through auction, may further improve cost efficiency (Armsworth et al., 2012; Elliott et al., 2015); though it is unclear whether any such improvement in cost efficiency would differ systematically between sharing and sparing.

In conclusion, 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.

Contributions

L.C., A.B., R.E.G., and T.F. designed the study, L.C. performed the research, L.C. and R.C.dS. analysed the data and all authors contributed substantially to the interpretation of results and writing of the manuscript.

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