A comparison of the costs of delivering conservation through land sharing and land sparing

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This thesis is submitted for the degree of Doctor of Philosophy

April 2022

DECLARATION

This thesis is the result of my own work and includes nothing which is the outcome of work done in collaboration except where specifically indicated in the text. It is not substantially the same as any that I have submitted, or, is being concurrently submitted for a degree or diploma or other qualification at the University of Cambridge or any other University or similar institution except as declared in the Preface and specified in the text. It does not exceed the prescribed word limit for the relevant Degree Committee.

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Summary

Globally, drastic biodiversity declines and the worsening climate crisis demand overhaul of existing land use policies which have failed to reconcile food production and environmental conservation. In Europe, most existing policies compensate farmers to voluntarily implement land-sharing measures, commonly referred to as wildlife-friendly farming, which seeks to deliver conservation benefits on the farmed land through agri-environment schemes (AES) offering a fixed price per hectare. Investment into sharing has continued despite the accumulation of evidence showing that, for the same amount of lost food production, substantially more would be delivered for conservation and climate change mitigation with the contrasting approach of land sparing, where high-yield farming allows large areas to be spared elsewhere in the landscape as (semi-)natural habitat. Following Brexit, the UK has the opportunity to rethink this approach; but until now policy decisions have had to be made without estimates of the relative taxpayer costs of using sharing and sparing to deliver target conservation outcomes. Addressing this critical research gap was the primary aim of this thesis, as follows.

In this thesis, I sought to uncover the taxpayer and food production costs of delivering meaningful conservation outcomes with land sharing and sparing. First, I conducted a novel comparison of the costs of monitoring sharing and sparing schemes for compliance and effectiveness. Monitoring is a fundamental, though often overlooked, taxpayer cost. In terms of effectiveness monitoring, I found current monitoring levels to be insufficient to precisely determine the effects on wild species of sharing schemes; in contrast, the same effort could deliver relatively precise estimates of the much larger effects of sparing. Furthermore, turning to compliance monitoring, I found the cost-effectiveness of existing English AES could be vastly improved with more compliance monitoring; however, this may be politically unpopular with farmers. It is therefore notable that I also found relatively less money was wasted when monitoring sparing at a sub-optimal rate compared to sharing. Second, I used a discrete choice experiment involving 118 arable farmers to establish their willingness to accept (WTA) payment to participate in sharing and sparing schemes that delivered the same biodiversity and carbon outcomes. I found that all but the most farmland-tolerant outcomes were delivered at less taxpayer expense with sparing. Third, combining this assessment of farmer WTA with knowledge of how much schemes must be monitored, I compared the taxpayer costs of delivering the same environmental outcomes with fixed-price sharing and sparing schemes which paid all recruits at the WTA of the least-willing farmer required in the scheme to deliver the target outcome. I found that sparing delivered the same outcomes at less than half the taxpayer cost of sharing; and, importantly, sparing saw only 79% of the food production lost under sharing. Fourth, I examined the distribution of farmer stated

WTA, finding that variation in responses was mostly driven by factors other than lost gross margin. Given marked inter-farmer variation in their stated WTA, variable-price schemes, which pay farmers their stated WTA rather than the rate required by the least-willing participant, offered savings to both sharing and sparing schemes. However, even under variable pricing, sharing was not cheaper than sparing in delivering our more farmland-sensitive outcomes. Finally, I examined whether a land-purchase strategy, where the government purchases land and then contracts organisations to manage and create habitat on it, would deliver sparing at less expense than a farm-subsidy approach. I found land purchase was more cost effective than the farm-subsidy approach if long timeframes, low discount rates and large budgets were considered; however the impacts on farming communities of largescale ownership changes warrant further consideration.

To conclude, I found overwhelming evidence for UK arable farming that land sparing can deliver biodiversity and carbon outcomes at substantially lower cost than land sharing both in terms of taxpayer costs and lost food production. The relative costs of sharing would increase even more with consideration of species that do not tolerate farmland, in a country with a shorter history of agriculture where fewer habitat specialists have gone extinct compared the UK, and if the production required elsewhere to compensate greater volume of food production lost under sharing was taken into account. Furthermore, the effects of sharing may be near-impossible to precisely determine with current monitoring efforts and continued sub-optimal compliance monitoring would increase the costs of sharing-like options relatively more than the more sparing-like options of existing AES. Whilst variable pricing and land purchase may further reduce the costs of sparing, the costs of delivering meaningful environmental outcomes are most substantially reduced by pursuing a land-sparing, rather than land-sharing, approach. This work is of considerable significance to the UK government, given that prevailing land-sharing policy approaches can at best deliver less than half the environmental outcomes delivered by the same budget spent on land sparing.

Preface

This thesis is presented by manuscript. Each chapter received invaluable contributions beyond my own. A statement of contributions can be found for each chapter below. Given the contributions of others, throughout Chapters 2-6, I refer to what "we" did.

Chapter 2

The idea to assess and compare the costs of monitoring sharing and sparing schemes originated from Pete Carey. I developed the ideas, led the data collection (which was from existing studies and via Freedom of Information requests to the UK government), analysed the data and wrote the paper. Andrew Balmford and Pete Carey gave feedback throughout.

Chapter 3

Rhys Green initially proposed this study. I developed the idea in collaboration with Rhys Green and Andrew Balmford, with contributions from Tom Finch, Nick Hanley and Alex Inman. I designed the experiment, collected the data, analysed it, and wrote the paper, with feedback from Andrew Balmford throughout. Rhys Green, Nick Hanley and Tom Finch also provided feedback on written drafts. This paper will be submitted to *Ecological Economics.*

Chapter 4

This chapter brought together the ideas of Chapters 2-3. I analysed the data, with assistance on the choice experiment analysis from Romain Crastes dit Sourd. I wrote the paper; Andrew Balmford, Rhys Green, Nick Hanley and Romain Crastes dit Sourd gave feedback and all appear as authors on this paper which is under review at *Global Change Biology.*

Chapter 5

Rhys Green conceived the idea for this chapter. I developed the idea, conducted the analysis and wrote the paper with input from Andrew Balmford throughout. This paper will be submitted to *Land Use Policy.*

Chapter 6

The idea for this chapter came from a discussion I had with Richard Bradbury. I developed the idea, collected the data, conducted the analysis and wrote the paper. Andrew Balmford gave advice throughout and Anthony Waldron gave feedback on written drafts. This paper will be submitted to *Biological Conservation.*

Finally, a note on my appendices. The contents of Appendix A are common to all chapters. To prevent repetition, it appears as a separate appendix to those that come afterwards, Appendices B-F, where one appendix corresponds to each chapter.

Acknowledgements

There are many people that I want to thank for supporting me throughout my PhD. First and foremost my supervisor, Andrew Balmford, for your incredible vision and hard work to realise it. You have been so dedicated to me and this project while offering such incredible attention to detail, knowledge and confidence that it would turn into something. I have learnt a huge amount that will always stay with me in the time that we have worked together, thank you.

I want to thank my advisors, Julia Jones and Becky Kilner, who oversaw the development of this thesis over the years and provided useful guidance on the feasibility of my plans. I also want to thank my two examiners, Ian Bateman and Bill Sutherland, who helped to improve this thesis . Hughes Hall has been a very supportive College to me; in particular, I want to thank Victoria Espley for helping me to realise I have useful and important contributions to give (sometimes, anyway) and my College Tutor, Arthur Hibble, who checked up on my progress with an encouraging but non-intrusive interest and has readily written various funding references for which I am grateful. On that note, I must thank my funders: primarily the Cambridge Trust and Hughes Hall who provided the bulk of my funding, but also the Cambridge Philosophical Society and the Department of Zoology. Thanks also to the Cambridge Society for the Application of Research who gave me an award as well as an opportunity to share my work more widely.

I want to thank Rhys Green who initially conceived the idea for much of this work, and who has also taught me a great deal about experimental design, statistics, the need to know what you are talking about, grammar and punctuation which has certainly improved the clarity of my work. For any remaining overuse of brackets and dashes, I apologise. Pete Carey provided helpful input on how agri-environment schemes are currently administered and monitored, for which I am very grateful. Research by Tom Finch provided the background for this study and I want to thank him for his generous feedback and support during this process. Richard Bradbury shared his encouragement and knowledge with me in the early days, for which I am grateful, and gave me the idea for the land purchase analysis which was developed with useful advice from Anthony Waldron. Alex Inman provided important insight into farmer attitudes which helped to improve the design of my choice experiment. The Morley Agricultural Foundation were very supportive of my work, and I want to thank Dave Jones in particular, as well as Georgie Bray of the RSPB, for casting a farmer's eye over my choice experiment. Thank you also to the Agriculture and Horticulture Development Board, particularly Teresa Meadows, who gave me access to farmers to recruit to my work, as did many other organisations. On that note, I must thank all of the anonymous farmers that shared their thoughts in my survey; my work would not exist without you, thank you.

I would not have been able to run a choice experiment without the wonderful course led by Stephane Hess at Leeds University's Choice Modelling Centre which taught me more than I could have hoped in just a few days. Stephane then continued to help me afterwards, crucially by directing me to Romain Crates dit Sourd who was extremely generous in his assistance with choice experiment analysis. Thank you, Romain, there is no way I would have bootstrapped confidence intervals without you. Nick Hanley also provided extremely useful insight that helped me to interpret the choice experiment results, and really gave me the confidence that this work is sound and important. I am most grateful for Nick's encouragement to attend the Annual Conference of the Agricultural Economics Society which gave me absolutely unparalleled access to key policymakers from Natural England, Defra, HM Treasury and elsewhere. For anyone reading this that does relevant work: please attend, even if you do not consider yourself an economist, it is an extremely important forum for exchanging and challenging ideas.

The David Attenborough Building has been an incredible place to work. Many of those I have already thanked were people I was able to meet by working there. However, the unequivocally best part of the building is the Conservation Science Group which has been an absolutely wonderful thing to be part of for the last 3.5 years. Many people have come and gone in that time, but particularly I want to thank Kirsten Russell, Emma Garnett, Harriet Bartlett, Imogen Cripps, Tom White, Will Morgan, Fernando Goncalves, Ali Johnston, Ali Eyres, Andrew Bladon, Roberto Correa, Ellie Tew, Phil Erm, Nicky Swan, Tania Maxwell, Kate Kincaid, Silviu Petrovan, Kristian Nielsen, Tom Swinfield, Alec Christie and Hannah Wauchope who have all helped me and enriched my time here in different ways. Working through a pandemic was difficult, and I want to thank Gianluca Cerullo and Francisco d'Albertas Gomes de Carvalho for entertaining lunchtime excursions. Thank you also to Tom Worthington for teaching me that it is easy to find rare birds, provided he is there to spot them and position the scope. And to everyone else that has made CSG so vibrant, thank you.

Lastly, I must thank my friends and family. In particular, Adora, Imogen, Jodie, Cassius, Ismail and Pippa who were there through all the ups and (lock)downs at 83 Mawson Road which has been such a lovely home to me for nearly 3 years. The first lockdown sent me back to Guernsey; that time that would have been so much worse without Dannie, Em and Flinty who accompanied me on so many sanity-restoring daily swims. There are a whole host of other people back home, and further afield, for whom I am so grateful, not least my family who have given me so much support and motivation to do whatever it is that I want to do; for you I will write the words thank you, but really they are nowhere near enough.

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Chapter 1: Introduction

Every year, farming produces more than 7.9 quadrillion calories of food (Bahadur et al. 2018). This enormous feat is achieved at the expense of the environment: agriculture has altered over 70% of the Earth's ice-free land surface (IPCC 2020), is the single greatest threat to biodiversity (Tilman et al. 2017) and accounts for a third of global anthropogenic greenhouse gas emissions each year (Crippa et al. 2021). As biodiversity loss and climate change accelerate, so too does the demand for food. Continued present trends may see a further billion hectares of land (an area the size of Canada) cleared to deliver an anticipated potential doubling of demand for crops by 2050 from a population growing in both size and wealth (Tilman et al. 2011). At a time when already more than 28% of all assessed species are threatened with extinction (IUCN 2020), and well-monitored populations of vertebrates have fallen in size by 68% since 1970 (WWF 2020), continued agricultural expansion at this scale is untenable. Changes in diet and food waste could substantially decrease the pressure on land by reducing demand (Tilman & Clark 2014; Williams et al. 2021), but unprecedented change is needed to meet the remaining demand without gravely threatening the integrity of Earth's life support systems.

Strategies for reconciling food production and environmental conservation can be characterised along a spectrum between land sharing, i.e. wildlife-friendly farming, and land sparing, where high yield farming is combined with the restoration or protection of large areas of (semi-)natural habitat elsewhere in the landscape (Green et al. 2005). Given these strategies operate at large scales, and aim to deliver public goods, that would not be delivered by markets without intervention, the decision to pursue either strategy often falls to governments. In Europe, much more has been invested in the land-sharing approach, mainly through fixed-price (i.e. where the same per-ha, annual price is paid to all participants) agri-environment schemes (AES), one element of the UK's Common Agricultural Policy, that compensate farmers for the yield reductions associated with implementing wildlife-friendly farming techniques (Defra 2019a). More than £5 billion is invested across Europe in these policies annually, with over £600m spent every year in the UK (Kettunen et al. 2011). Unfortunately, evidence suggests these schemes have delivered little for farmland birds, a major target of these schemes (Kleijn & Sutherland 2003; Kleijn et al. 2006; Pe'er et al. 2014; Batáry et al. 2015), or carbon sequestration, a more recent focus (Pe'er et al. 2019, 2020). Some have suggested insufficient monitoring has failed to detect effects that are present (MacDonald et al. 2019); though in already co-opting vast citizen science datasets (e.g. Baker et al. 2012; Gillings et al. 2005; MacDonald et al. 2019), it is difficult to see how monitoring efforts could be greatly increased without a much larger budget.

The alternative strategy of land sparing has received much less public investment despite growing evidence that, for a given level of food production, it delivers far more than land sharing in terms of biodiversity, carbon sequestration, recreation, and water quality(Phalan et al. 2011; Gilroy et al. 2014; Lamb et al. 2016a; Dotta et al. 2016; Williams et al. 2017; Finch et al. 2019, 2020, 2021; Feniuk et al. 2019). This pattern has been reported in studies from five continents and nine countries (Balmford 2021), including the UK (Finch et al. 2019, 2020, 2021) where the history of agriculture dates back 6000 years (Collard et al. 2010). Some sparing-like interventions are found in existing English AES, but these tend to be relatively small-scale, and collectively receive only 12% of the budget spent on the sharing interventions that spread production over a larger area (Rural Payments Agency, *pers. comm.*). Land sparing remains the only viable approach for simultaneously meeting food demand without substantially increasing imports while also conserving the many species in the UK (and relatively more elsewhere in the world) that cannot persist on land farmed at any yield (Lamb et al. 2019). Despite the apparent merits of sparing, investment into sharing has continued without explicit consideration of the relative taxpayer costs. This thesis addresses this major shortcoming by conducting the first assessment of the taxpayer costs of delivering the same environmental outcomes with sharing and sparing. I established the payments required by farmers to implement sharing and sparing interventions using a choice experiment and combined this with estimates of monitoring and administration costs. I compared the taxpayer and food production costs of delivering the same outcomes with sharing and sparing. I then explored the drivers of the payments required by farmers, as well as the cost implications of changing other aspects of scheme design.

Paying farmers to participate in AES

The most substantial taxpayer cost of sharing and sparing schemes are the payments made to farmers who voluntarily enrol in schemes that typically offer a fixed-price payment to cover the production costs of an action intended to benefit the environment (Defra 2019b). Amongst the criticisms of sparing is the suggestion that farmers are unwilling to implement large-scale habitat creation (Fischer et al. 2008). Advocates of this opinion have discounted sparing as too unpalatable, and therefore expensive (given the higher levels of required compensation), to implement despite there being no empirical evidence comparing the costs of paying farmers to share and spare. Sparing interventions may indeed require more compensation per unit area, given that production typically ceases, rather than continuing in a wildlife-friendly way, and possibly also since it is less familiar, and requires more drastic change. However, a cost-benefit analysis, of which none exists, may find this to be offset by the greater benefit delivered per unit area for many outcomes (Finch et al. 2019). Furthermore, in the UK, the greater permanency and larger spatial scale of sparing interventions may be attractive given uncertainty in future policy prescriptions, with many areas of the UK expected to become unprofitable to farm (Acs et al. 2010; Arnott et al. 2021). Clearly, for policymakers to make the important decision between taking a sharing or sparing policy approach, farmers' willingness to participate in sharing vs sparing schemes must be empirically compared.

Exploring the distribution and composition of payments required by farmers

To investigate the compensation required by farmers to participate in sharing and sparing schemes, I undertook a choice experiment with 118 arable farmers in England. Choice experiments can be used to uncover the distribution of preferences amongst a population by asking participants to make a series of choices between sets of options that differ in several attributes including the payment rate (Hanley et al. 1998). This allows estimation of participants' willingness to accept (WTA) payment to participate in a scheme with a prescribed set of attributes (Villanueva et al. 2015). Much variation in farmers' willingness to participate in AES has been uncovered using choice experiments (e.g. Kuhfuss et al. 2016). Past studies have found that farmers generally require greater compensation, per year and per unit area, to implement options over longer timescales and larger areas and also to accept higher monitoring rates (Ruto & Garrod 2009; Espinosa-Goded et al. 2010; Christensen et al. 2011; Villanueva et al. 2015; Kuhfuss et al. 2016b; Gómez-Limón et al. 2019). Despite this work on relative preferences, no study of farmers has scaled-up their findings to estimate the relative costs of competing approaches, as I do here. In comparing sharing and sparing, given the great variation between farmers, and given that schemes tend to pay all farmers at the rate required by the least-willing participant, it may be important that sparing generally delivers greater benefit per unit area, such that fewer participants are required to deliver the same benefit compared to a sharing scheme.

The compensation paid to farmers is intended to cover the value of forgone production (Defra 2019b). Given yields and profits vary across space, fixed-price schemes cannot avoid paying some participants above their lost gross margin. Furthermore, lost gross margin is unlikely to be the only driver of WTA, though I know of no study that has dissected its composition beyond preliminary calculations reported in Armsworth et al. (2012) that only \$0.12-\$0.46 per dollar paid to farmers in upland English AES compensated forgone income. In addition to compensation for lost gross margin, we might expect farmers to require compensation to cover the labour and administration costs of participating in the scheme. Farmers might simply not particularly want to participate in government schemes, or dislike the compromised flexibility, or indeed the appearance of the implemented changes (reviewed in Dessart et al. 2019). Whilst these factors may increase WTA beyond the value of lost gross margin, positive effects of interventions and the guarantee of income may decrease the payment which farmers, who are typically risk-averse (Groom et al. 2008), are willing to accept. To provide novel insight into the relative importance of these factors in driving WTA, in my study of farmer preferences I dissected individual farmers' WTA into that to cover their lost gross margin and the remaining difference.

We term the difference, which may be positive or negative, between WTA and lost gross margin the residual requirement and explored whether its size differs between sharing and sparing schemes. This is primarily of interest since it may lead to differences in the taxpayer costs of sharing and sparing schemes, but also for three other related reasons. First, we consider the residual requirement to be a measure of underlying preferences, besides those related to forgone returns, for sharing or sparing when the same gross margin is lost to either approach. This is important to debate of the relative merits, and feasibility, of sharing and sparing. Not least since, second, taxpayers may be less keen to spend on this residual requirement, compared to the value of lost production. Third, lost gross margin, but not residual requirement, would be expected to correlate to lost food production which is indicative of the overseas impacts associated with the imports required to make-up for domestic production forgone. Therefore, taxpayer costs may mischaracterise overseas impacts if the cost of paying farmers is dominated by their residual requirement, rather than lost gross margin. For all these reasons, the relationships between WTA, lost gross margin and the fixed-price payment are of interest.

Other AES costs: Monitoring, administration and lost food production

The taxpayer costs of AES do not end with payments to farmers; successful schemes must be monitored for both compliance and effectiveness. Without monitoring, the rate of non-compliance can be expected to increase, with consequent negative impacts for the benefit delivered (Hart & Latacz-Lohmann 2005). However, monitoring is expensive; so the optimal rate of monitoring is a trade-off between the cost of monitoring more participants and the cost of paying more participants to enrol in the scheme to make-up the benefit lost to non-compliance (Keane et al. 2008). Existing research has modelled this trade-off following utility theory and explored the implications to scheme outcomes of monitoring at varying rates (Ozanne et al. 2001), as I do here in the first comparison of sharing and sparing schemes. Whilst this modelling can identify the most efficient monitoring rate, issues of unfairness and the possibility that monitoring may deter participants from entering schemes may keep monitoring rates low, despite non-compliance (Keane et al. 2008; Broch & Vedel 2012). Indeed, in England, only 6% of AES participants were monitored each year between 2015-2017, and 46% of those monitored were fined for being in breach of scheme requirements (Rural Payments Agency, *pers. comm.*). Therefore, it may be pragmatic to consider the cost of monitoring at sub-optimal rates. Whilst they have never been studied, the costs of compliance monitoring may differ between sharing and sparing, given differences in the payoff from non-compliance and the number of participants required to deliver the same outcome, as well as the relative permanency and spatial scale of the options. Because non-compliance reduces the benefit delivered, spend on compliance monitoring may also impact the costs of effectiveness monitoring, discussed next.

To compare the cost-effectiveness of different approaches, and establish whether targets have been delivered, we must monitor the effects of AES. Governments should be concerned both with the size of effects, and the associated costs of achieving them. Many existing studies consider whether effects are statistically significant (e.g. Baker et al. 2012; Bright et al. 2015; Davey et al. 2010; Gillings et al. 2005; MacDonald et al. 2019), but ignore their size; therefore, such studies cannot be used to assess cost-effectiveness. Measurements of effect sizes are inherently not certain; and whilst the precision of estimates increases with sample size, so do costs of effectiveness monitoring. Given existing suggestions that current AES are inadequately monitored (MacDonald et al. 2019), I considered the amount of data needed to determine effects with adequate certainty and to assess the feasibility, and therefore costs, of delivering this. Despite the vast spend on AES, and on monitoring programs, I am aware of no study that has done this, except in terms of statistical significance (Geijzendorffer et al. 2016). Likewise the relative costs of monitoring the effectiveness of sharing and sparing schemes have not been compared, until now, but are potentially variable, given the larger effects of sparing and potential differences in variability between sites. Given that current monitoring approaches are failing to determine concrete effects of most current AES investments, my assessment of the scale of monitoring necessary to deliver precise estimates should be of great interest to governments in deciding their policy approach.

As well as likely differences between sharing and sparing approaches in the payments required by farmers and the costs of compliance and effectiveness monitoring, administration costs may also differ, given likely differences in the number of participants required to deliver particular outcomes. Last, and alongside these taxpayer costs, it is critical that policymakers consider the food production lost in implementing AES, since any domestic production losses are likely compensated by increased imports, with potentially significant taxpayer, political and environmental consequences (including for conservation in food-exporting countries) (Smith et al. 2019).

Impacts on scheme costs of variable pricing and land purchase

There are other aspects of scheme design that affect taxpayer costs, besides whether the scheme encourages sharing or sparing, In my thesis, I explore two key aspects. First, most existing schemes use fixed pricing to pay all participants at the same per-hectare annual rate. This might be considered fair, until differences in the value of lost food production are considered. Under fixed pricing, farmers may differ widely in the subsidy they receive for a given action, net of its direct and opportunity costs. Variable pricing, where participants are paid at their true supply price, rather than the rate of the least-willing participant, has been suggested as a more efficient payment structure (Armsworth et al. 2012). An even more efficient payment structure would also take into account spatial variation in the benefits delivered per unit compensation (Gibbons et al. 2011), though that is beyond the scope of this thesis. The relative savings offered by variable pricing will depend on the distribution of farmer WTA and the number of participants that must be recruited into the scheme to deliver a given outcome. Again, this may differ between sharing and sparing, but has not been studied. Here, I compare the cost of delivering the same environmental outcomes first with fixed-price, and then with variable-price, schemes. Second, I explored the implications of altering a different aspect of the scheme: whether sparing could in principle be delivered not by paying farmers to undertake specified actions but by purchasing land and then contracting conservation organisations for habitat creation and maintenance. Whilst land purchase clearly involves high up-front costs, over a long enough timeframe it may become cheaper than paying farmers who must be compensated for the value of lost production every year. Given the different timescales over which these costs are incurred, it follows that the timescale and value associated with future costs/benefits, i.e. the discount rate, might drive conclusions, as the few existing studies comparing these approaches elsewhere in the world have found (Curran et al. 2016; Schöttker et al. 2016; Schöttker & Wätzold 2018). Therefore, in my thesis, I explore the timescales, discount rates, as well as the budgets, over which land purchase is more cost effective than the annual farm-subsidy approach. Large-scale public purchase of land may deliver additional benefits that I do not quantify: the greater permanence may facilitate restoration of longer-term ecological processes thereby generating greater benefits, it may facilitate agglomeration of spared habitat, which would likely increase the benefit delivered per unit area of spared land (Lamb et al. 2016b), as well as potentially offering other benefits, such as recreation (Seddon et al. 2020). However, a programme of purchases would take income away from farming communities, amongst other potentially negative effects.

The challenge addressed by this thesis

At this moment, the UK has the opportunity to launch an agricultural policy that delivers much needed action to address biodiversity declines and reduce the net emissions associated with agriculture. A complete assessment of the costs of delivering the same outcomes with sharing and sparing could facilitate a transition away from a policy that has disproportionately supported large landowners (Bateman & Balmford 2018), mostly for action entirely devoid of environmental benefit (Pe'er et al. 2014). Despite the previously discussed evidence from the UK that conservation would be delivered with less lost food production under sparing, and acknowledgements of the merits of sparing in some high-level plans (National Food Strategy 2021), initial policy outlines of the UK's forthcoming Environmental Land Management Scheme suggest much more funding might be available for the sharing-like approach of the 'Sustainable Farming Incentive' than for sparing-style action such as the 'Landscape Recovery Program', while the 'Local Nature Recovery' AES may continue to invest largely in sharing (Defra 2022). Here I provide the first comparison of the taxpayer costs of sharing and sparing schemes to provide the first insight into whether less taxpayer money would be needed to deliver the same outcomes by sharing or sparing.

Thesis aims

In light of this, this thesis provides the first ever assessment of the taxpayer costs of delivering the same outcomes by sharing and sparing (Chapter 4), through explorations of the required level of monitoring (Chapter 2) and farmer preferences (Chapter 3). Having established the relative costs of contrasting approaches that deliver the same levels of a suite of agri-environmental outcomes, I examine the savings offered by running schemes with variable pricing (Chapter 5) and delivering sparing with land purchase (Chapter 6). The aims of this thesis are thereby summarised as follows:

- Establish appropriate levels of monitoring for current schemes

- Identify and then compare the payments required by farmers to implement sharing and sparing via a large-scale choice experiment
- Estimate the total taxpayer (compensation, monitoring and administration) costs of delivering the same environmental outcomes via sharing and sparing, and compare this to lost food production
- Explore how the distribution of WTA varies across farmers, including how much of it is driven by lost gross margin, to consider whether variable pricing offers a fairer and more cost-effective payment structure
- Examine how the costs of sparing change under a land-purchase approach

Thesis structure

The structure of my thesis is described below. Much of this study considers four key environmental outcomes: supporting populations of three species of conservation concern that differ in their response to farming, the red-listed Yellowhammer (*Emberiza citrinella*), the amber-listed Eurasian Bullfinch (*Pyrrhula pyrrhula*) and the red-listed Northern Lapwing (*Vanellus vanellus*) (Finch et al. 2019); and delivering net carbon emission reductions, a growing focus of AES. All outcomes can be delivered on farmland which, given the many species and services that cannot be delivered on land farmed at any yield, biased this study in favour of sharing. My findings are interpreted in this light.

In **Chapter 2**, I examine how cost efficiency varies with the compliance monitoring rate and study how the rate of effectiveness monitoring affects our knowledge of scheme performance. For compliance, I identify sharing and sparing options in current schemes. Then, I use an approach grounded in utility theory to assess the monitoring rate at which the combined monitoring and compensation costs of these options are minimised, as well as the cost of monitoring at sub-optimal rates. For effectiveness, I use the findings of current monitoring schemes to assess the sample size required to deliver estimates of effects to a target level of precision, and the cost of delivering these. I also discuss the limitations of my effectiveness monitoring analysis, which preclude it from contributing to the overall estimates of taxpayer costs developed in Chapter 4.

Chapter 3 examines the payments required by farmers to implement sharing and sparing interventions that deliver target biodiversity (increased populations of bullfinches, lapwings and yellowhammers) and carbon outcomes. I use a choice experiment to establish the distribution of farmer WTA compensation to implement sharing and sparing options across arable farms in England. I use this distribution to simulate the delivery of my focal outcomes for the same spend on sharing and sparing. I also examine some of the reasons behind farmers' willingness to engage in sharing and sparing options.

Having established the costs of monitoring schemes for compliance, and the payments required by farmers, in **Chapter 4** I present my estimates of the combined taxpayer costs of delivering the target biodiversity and carbon outcomes with sharing and sparing. Using data collected during the choice experiment, I also estimate the food production lost by farmers in delivering these outcomes, and compare these, alongside the taxpayer costs, for sharing and sparing schemes.

I explore the distribution and composition of farmer WTA in **Chapter 5**. I examine the relative contribution of lost gross margin to WTA and quantify the remaining difference – the residual requirement. For sharing and sparing schemes, I compare the spend on this residual requirement relative to the target delivered and lost gross margin. From the WTA distribution, I also estimate the savings achievable by using variable, rather than fixed, pricing.

Finally, in **Chapter 6**, I explore how the taxpayer costs of delivering land sparing change when the government purchases land, which is then managed by contracted organisations, rather than paying farmers in AES. To recognise the difference in the temporal distribution of costs, I explore the timescales and discount rates, as well as budgets, under which land purchase is more cost effective than the farm-subsidy approach.

Chapter 2: How much should we monitor agri-environment schemes, and do monitoring costs differ between land-sharing and land-sparing interventions?

Abstract

To address the severe and urgent threats posed by climate change and continued biodiversity loss, governments must re-assess agricultural policies that have so-far failed to reconcile food production and conservation. The UK's withdrawal from the European Common Agricultural Policy offers an opportunity to reconsider the current approach which mainly involves encouraging landscape-wide so-called wildlife-friendly farming (i.e. land sharing). Critical to scheme success is monitoring, both for compliance (whether participants meet scheme requirements) and effectiveness (whether the intended benefit is delivered). We examined the appropriateness of current levels of monitoring and explored the likely cost of applying current monitoring approaches to English schemes illustrative of both land sharing and land sparing, in which high-yield farming is combined with the conservation of non-farmed land for nature. For an array of target environmental outcomes delivered through sharing- and sparing-style measures in the current Countryside Stewardship Scheme (CSS), we found monitoring costs trade-off dramatically with scheme payments. Increasing the present ~6% monitoring rate – under which there is high non-compliance which must be offset with many more scheme participants – up to ~34% would reduce combined monitoring and payment costs by between 33-47% for sharing- and sparing-style interventions. Given CSS payment rates, sparing always delivered these environmental outcomes at less cost than sharing, whether considering combined costs or monitoring costs alone. We found high uncertainty in the known effectiveness of sharing interventions for bird species, despite extensive monitoring; current volunteer efforts must generally triple, likely at high cost, to be confident the true effect lies within 50% of the mean measurement. The same precision can be reached for most types of sparing interventions with no increase in current sampling, though differences in the design of studies available for parameterising our estimates account for an unquantified portion of this difference. Overall, our results show that sparing would be cheaper to monitor than sharing; but also, for both sharing and sparing, that increasing current levels of monitoring would cut total costs and increase confidence in scheme effectiveness.

Introduction

The UK Government spends ~£600m each year subsidising farmers in English agri-environment schemes (AES) to deliver environmental outcomes at the cost of food production (Natural England 2009). The severe threats posed by continued biodiversity loss and climate change demand that England's post-Brexit Environmental Land Management Scheme is both more ambitious and more effectively implemented than existing AES. While scheme costs are dominated by payments to farmers, monitoring is a fundamental – but commonly overlooked – aspect of implementation. Schemes require monitoring for compliance (whether participants meet scheme requirements) and effectiveness (whether the intended benefit is delivered). Monitoring for compliance is costly but deters cheating so participants deliver more of the intended benefit (Keane et al. 2008). With greater compliance, fewer participants are required to deliver the target benefit; so the total costs of delivering a target outcome may be reduced by spending more on compliance monitoring (Ozanne et al. 2001). Effectiveness monitoring determines the impact of the implemented actions and must be sufficiently extensive to deliver estimates with some certainty. Knowledge of effects allows schemes to be revised; and schemes are thought more likely to deliver the intended benefit given this regular review and revision (Geijzendorffer et al. 2016).

Current schemes mainly incentivise wildlife-friendly farming by subsidising farmers for the yield reductions and other costs that arise from measures intended to benefit the environment. This so-called land-sharing approach (Green et al. 2005) has delivered little for biodiversity (Kleijn & Sutherland 2003; Kleijn et al. 2006; Dicks et al. 2014; Batáry et al. 2015; Pe'er et al. 2020), despite great investment (Kettunen et al. 2011). Some have suggested that monitoring is insufficient to detect effects that are present (e.g. MacDonald et al. 2019); but in already co-opting vast citizen science datasets (e.g. Baker et al. 2012; Gillings et al. 2005; MacDonald et al. 2019), it is difficult to see how sample sizes could be vastly increased without great investment into a bespoke monitoring program. Therefore, it is important to compare the feasibility of concretely estimating the effects of sharing schemes to the alternative approach of land sparing where high-yield farming allows large areas of land to be left unfarmed and spared for nature (Green et al. 2005); in England, this requires restoration of farmland to (semi-)natural habitat. Growing evidence from England suggests that, for a given level of overall food production from a landscape, land sparing may deliver greater benefits for biodiversity (Lamb et al. 2019; Finch et al. 2019, 2020), carbon sequestration, nature-based recreation and water quality (Finch et al. 2021), particularly when some spared land is managed for nature as low-yielding farmland. However, the costs of implementing these contrasting strategies have not been compared. Here, we conduct a novel exploration of the costs of monitoring sharing and sparing schemes for compliance and effectiveness.

The costs of monitoring are largely determined by the number of participants that must be monitored. For compliance monitoring, this is driven by the number of participants required to deliver the target given the per-area benefit of the intervention as well as the relative payoffs, and detectability, of non-compliance (Ozanne et al. 2001). For effectiveness monitoring, required sample sizes will vary according to the size of the effect of the intervention, how much this varies across sites, and differences in monitoring design (Nakagawa & Cuthill 2007).

Here, we used information from the Countryside Stewardship Scheme (CSS), the AES currently in operation across England, to explore the efficacy of current monitoring and assess the likely costs of using existing approaches to monitor hypothetical sharing and sparing schemes in England. We studied interventions supported under current schemes that illustrate what may be pursued under sharing and sparing; though, in principle sparing may involve larger-scale actions than considered here. Thus, for these contrasting sharing and sparing interventions, we examined the following:

a. **Compliance monitoring:** Is increased spend on compliance monitoring made worthwhile by reductions in the number of farms needed to deliver environmental targets? At different monitoring rates, what is the cost of delivering target environmental outcomes with sharing- and sparing-style approaches? What is the additional financial cost of delivering the target outcomes with monitoring below the optimal rate? How sensitive are the relative costs of sharing- and sparing-style schemes to variation in assumed parameter values?
b. **Effectiveness monitoring**: How certain are existing estimates of the effectiveness of AES? What sample sizes are required to determine the effects of schemes to target degrees of certainty? What sample sizes are required to determine statistical significance?

Methods

- **Compliance monitoring**

Under current European AES, farmers deliver environmental benefits at a cost which is compensated by the scheme payment. However, farmers may accept the payment without complying with requirements, thereby avoiding the cost of participation (the so-called principal-agent problem in the moral-hazard literature; Ozanne et al. 2001). Utility theory has often been applied to model compliance behaviour in AES (e.g. Gómez-Limón et al. 2019; Hart and Latacz-Lohmann 2005; Ozanne et al. 2001), and argues that farmers adopt the strategy that confers greatest utility. Increased monitoring reduces the utility of non-compliance due to the increased risk of being caught and fined (Ozanne et al. 2001). However, monitoring is expensive: in England, it is typically conducted with in-person visits, though improvements in remote sensing may offer a less expensive option in the near-future (Sadlier et al. 2018). Thus the economically efficient monitoring rate is a trade-off between the costs of monitoring more participants and the costs of paying more farmers to participate to make up the benefit lost to non-compliance. This means the costs of compliance monitoring can only meaningfully be assessed in combination with scheme payments, plus the associated administrative costs and recovery of payments through fines.

Here, we applied insight from utility theory, and adopted the payment rates of the CSS (Defra 2019), to study the efficacy of current monitoring. For a range of monitoring rates, we compared the costs (considering scheme payments, administrative costs and compliance monitoring) of delivering the same environmental outcomes with sharing- and sparing-style options. In doing so, we studied two types of sharing-style intervention: in-field (which sees changes to practices that do produce food, e.g. post-harvesting stubble retention) and field-edge (which sees a habitat added to farmed land that does not produce food, e.g. hedgerows). We considered sparing as habitat creation in large blocks. Based on this, we categorised CSS options as in-field sharing, field-edge sharing, and sparing: then, in order to make meaningful comparisons, we selected environmental outcomes for study which can be delivered by all three intervention types. This led us to focus on three bird species which vary in their habitat requirements and sensitivity to farming (Finch et al. 2019): the most farmland-tolerant species studied were Yellowhammers (*Emberiza citronella)* which require scrub habitat, next most tolerant were Eurasian Bullfinches (*Pyrrhula pyrrhula)*, which require scrub or early-woodland habitat, and the most farmland-sensitive species studied were Northern Lapwings (*Vanellus vanellus*) which require wetlands. We also studied interventions that reduce net carbon emissions (Table 2.1). We calculated the per-area delivery of our four environmental outcomes by these identified interventions based on existing literature (Appendix A). For context, of the options in Table 2.1, sharing receives 89% of CSS payments, while sparing receives only 11% (Rural Payments Agency (RPA)*, pers. comm*.; though other sparing schemes, primarily for woodland creation, do exist).

Setting targets for study in a hypothetical landscape of farmers

Next we established a hypothetical landscape of 1000 farmers and set environmental targets to deliver with sharing and sparing action. To set these targets, we calculated the maximum benefit delivered by each option in the CSS assuming current participation rates and full compliance (Table B1). The benefit delivered by existing sharing and sparing options varied (Table 2.1). We were careful not to set targets above these maximum benefits because this would necessitate greater-than-current payments (and we do not know how participation varies with payment rate). Furthermore, we wanted to be able to meet our targets despite some participants not complying with requirements. So, for each outcome, we identified the poorest-performing option (that which delivers the lowest total benefit at present); in all cases this was field-edge sharing (see Table B1, for full details). Then, to allow the target outcome to be delivered despite non-compliance amongst participants, we set our targets at 50% of the benefit delivered by total compliance amongst participants of the existing field-edge sharing option.

Table 2.1. Interventions that deliver the studied outcomes[1], the maximum benefit deliverable amongst 1000 farmers (assuming 100% compliance and current CSS participation rates) and, from this, the target set for each outcome.

Environmental outcomes	**Intervention type**	**Intervention**	**Max benefit delivered in landscape of 1000 farmers** (birds or tonnes carbon/y)	**Target adopted in study**
Yellowhammer	In-field sharing	Stubble, spring cropping	627	94
	Field-edge sharing	Winter bird cover	1239	94

		Hedgerows	188	94
	Sparing	Scrub	333	94
Bullfinch	Field-edge sharing	Hedgerows	36	18
	Sparing	Scrub	91	18
	Sparing	Woodland	39	18
Lapwing	In-field sharing	Stubble, spring cropping	63	10
	Field-edge sharing	Fallow	19	10
	Sparing	Wet grassland	86	10
Carbon	In-field sharing	Nil fertiliser use	81	41
	Field-edge sharing	Hedgerows	124	41
	Sparing	Woodland	3139	41

[1] No in-field sharing option was identified as beneficial for bullfinches.

Modelling the strategies adopted by farmers

Next, we sought to determine which of our 1000 farmers would participate in each option in turn. We assumed farmers would join a scheme when the net benefits of participation outweighed the costs. To reflect the great variation in the cost to farmers of joining these schemes, we assumed costs varied within the farmer population and generated a cost for each farmer (as a multiple of mean cost) by sampling from a normal distribution with a mean of 1 and standard deviation of 0.2. We next set the mean cost for each intervention such that the participation rate generated matched that seen in the CSS (see Table B1 for current participation rates). Then, following Ozanne et al. (2001), we assumed not participating offers a utility of 0, so farmer *n* would participate in a scheme that cost c_n (£/y) and is compensated at rate *P* (£/y) when:

$$P - c_n > 0 \quad \textbf{[1]}$$

We then considered who would participate but fail to comply. Following Gómez-Limón et al. (2019), we explored different degrees of non-compliance: total (where farmers incur 0% of the cost and deliver 0% of the benefit) and minor (where farmers pay 70% of the cost and deliver 70% of the benefit). Minor non-compliance may be deliberate or accidental (e.g. arising from not fully studying the agreement terms; Finn et al. 2009). The 70% bound is arbitrary but was set according to the extent to which we judged it possible to accidentally not comply (P. Carey, *pers. obs.*). We assumed the utility of non-compliance is the payoff given if it is undetected, less the cost if caught and fined; and farmers will cheat when this exceeds the utility of complying (as in Ozanne et al. 2001), i.e.:

$$(P - xc_n)(1 - l) - Pfl > P - c_n \quad \textbf{[2]}$$

where x is the extent to which cheating reduces the cost of compliance (for total non-compliance, $x = 0$, and for minor non-compliance, $x = 0.7$), l is the likelihood that detected non-compliance is fined, and f is the fine as a proportion of the payment rate. Last, following Hart and Latacz-Lohmann (2005), we assumed some 'honest' farmers simply do not consider cheating an available strategy. Based on reported non-compliance in the CSS and the Environmental Stewardship Scheme of 46% in 2015-2017 (RPA, *pers. comm.*; Table B2), we set the proportion of honest farmers at 54% (to accurately predict the current non-compliance rate given the present 6% monitoring rate). This is well below the 82.5% of Hart and Latacz-Lohmann (2005), but a sensitivity test of this assumption did not change our conclusions (Figure B1).

Parameterising these equations

We parameterised equations [1] and [2] using the best available parameter estimates, as follows.

Payment rates: CSS payment rates were used (Table B1; Defra 2019).

Fines: We set fines at 1x and 2x the annual payment rate for minor and major non-compliance respectively. This approximately follows the fines applied when participants overstate the area enrolled in an option (Defra 2019, p.78).

Areas: For each option, we assumed all participants enrolled the mean area currently enrolled by CSS participants (Table B1).

Administrative costs: We assumed a per-agreement administrative cost of £458/y, based on the reported £6.48m spent administering 19,118 CSS agreements in 2009 (Natural England 2009, p. 26), less IT costs (which were assumed independent of the number of participants) and adjusted for inflation.

Probability non-compliance is fined: We assumed that majorly non-compliant participants would always be fined when monitored (following Gómez-Limón et al. 2019). For in-field sharing and sparing, we assumed minor non-compliance (e.g. removing stubble early, over-grazing of grassland, etc.) would only be detected in 50% of cases. Given differences in the nature of the interventions, minor non-compliance of field-edge sharing options was considered detectable in 100% of cases (since it would most probably involve actions being implemented over less than the required area, which should be detected during in-person visits).

Remote monitoring: Given recent developments in remote sensing, we assumed major non-compliance could be detected remotely (as well as in-person) for field-edge sharing and sparing but assumed detection of minor non-compliance required in-person checks. For in-field sharing options, we assumed remote sensing could not detect non-compliance at all (J. Griffin, *pers. comm.*).

Monitoring costs: We used cost estimates of current CSS monitoring to estimate the costs of monitoring participants (P. Carey; *pers. obs*.; Appendix B).

To acknowledge the uncertainty in many of these parameters, we explored the sensitivity of our conclusions to variation in the input parameters in terms of: (i) the proportion of honest farmers, (ii) the rate of fines, (iii) monitoring costs, (iv) the area enrolled by each participant, (v) the biodiversity benefit delivered per unit area and (vi) the availability of remote monitoring (Appendix B).

Simulating total costs

Based on this parameterisation, we simulated the costs of 10-year schemes which delivered the target outcomes assuming a 3.5% discount rate (following HM Treasury 2018); all costs were adjusted to 2020, using a UK GDP deflator index (Bank of England 2021). We explored costs for a range of monitoring rates and varied the proportion of monitoring that was conducted remotely vs in-person. We calculated total costs as the sum of scheme payments and administrative costs, plus monitoring costs, less fines (Appendix B).

- **Effectiveness monitoring**

Despite much study (e.g. Baker et al. 2012; Bright et al. 2015; Davey et al. 2010; Gillings et al. 2005; MacDonald et al. 2019), there is considerable discussion around whether the land-sharing approach emphasised in much of recent UK agricultural policy has been effective (Chamberlain 2018). Despite indication that some interventions may have had positive impacts on some species (Davey et al. 2010b; Baker et al. 2012; MacDonald et al. 2019), there has been little, if any, impact on overall population trends: populations of farmland bird species, the major focus of AES, continue to decline (Hayhow et al. 2017). If current monitoring efforts are insufficient to deliver conclusive results (as suggested by MacDonald et al. 2019), it would seem prudent to establish the monitoring effort required to determine the effect of interventions with adequate certainty. To date, studies have typically focused on proving statistical significance (e.g. Baker et al. 2012; Bright et al. 2015; Davey et al. 2010; Gillings et al. 2005; MacDonald et al. 2019); i.e. whether the intervention has had an effect, regardless of its size. This approach cannot be used to assess cost-effectiveness, for which the size of effects is important (Amrhein et al. 2017). Since we, and presumably policymakers, are interested in cost-effectiveness, here we take a different approach. We consider the size of effects (i.e. the additional birds delivered per unit area by the sharing or sparing intervention) and the associated uncertainty, quantified using confidence intervals (CI's). If a population were repeatedly resampled and 95% CI's calculated, we would expect 95% of these CI's to contain the true population mean. The width of CI's is dictated by variation in the data (reflected in standard error), the sample size and the target accuracy (here set at 95%), according to:

$$CI\ width = z \times SE \qquad \textbf{[3]}$$

where z is the test statistic dictated by the desired accuracy (for 95% CI's, z = 1.96) and SE is the standard error, calculated as:

$$SE = \frac{\sigma}{\sqrt{n}} \qquad \textbf{[4]}$$

where σ is the standard deviation and n the sample size.

Here, we consider uncertainty as the width of the CI as a proportion of the estimated mean effect; so differences in effect size also drive uncertainty. We rely on existing studies to compare the certainty of sharing and sparing effects and, somewhat problematically, these studies differ in their approach. Sharing is typically monitored by studying population growth rates over time on intervention and non-intervention land (e.g. Baker et al. 2012; Bright et al. 2015; Gillings et al. 2005). Trend-based studies typically deliver smaller estimates of effect, and greater uncertainty, than the single-time studies comparing population density on intervention and non-intervention land (Christie et al. 2019) that are typically applied to monitor sparing. However, given that we could find no studies of population density for sharing interventions that reported standard errors, we had to compare studies of different demographic metrics. To explore whether current monitoring approaches are appropriate, we calculated the uncertainty associated with estimates produced by these studies. Then, we manipulated sample size (i.e. the number of squares surveyed) – the feature of study design that has a readily calculable effect on CI width – to determine the minimum required to deliver estimates to the target degrees of certainty.

Compared to the compliance monitoring analysis, a different set of environmental outcomes are considered here because data were not available for all the species/interventions considered for compliance monitoring. We did not study interventions that reduce net carbon emissions since the empirical values used in the compliance monitoring analysis (from IPCC 2019) do not readily allow quantification of uncertainty.

Identifying existing studies

For sharing, we used data from Baker et al. (2012) as this reports on a wide range of interventions and species. The authors use data from the Breeding Bird Survey (BBS) to estimate the additional population growth rate ($\widehat{\alpha_9}$, and associated 95% CI's) attributable over a 9-year period to sharing interventions in arable, mixed and pastoral farming systems. Sample sizes were not explicitly stated but could be inferred from other information reported (Appendix B). For sparing, we used data from Newson et al. (2005), which provides estimated population densities, and bootstrapped 95% CI's, in farmland and (semi-)natural habitat using BBS data. The BBS is not explicitly designed for monitoring AES; however, its volunteers collect huge amounts of data so it is often co-opted for this purpose, saving the government the great expense of a bespoke monitoring program.

Table 2.2. The in-field sharing, field-edge sharing and sparing interventions selected for analysis of effectiveness monitoring, along with the species that they benefit in the given farmland system (arable, mixed or pastoral). WBC= winter bird cover.

Environmental outcome	System	Intervention		
		In-field sharing	**Field-edge sharing**	**Sparing**
Bullfinch	Arable		Hedgerow management	Woodland

	Mixed		Hedgerow management	Woodland
	Pastoral		Hedgerow management	Woodland
Lapwing	Arable	Grass management		Wet grassland
	Mixed	Grass management		Wet grassland
Reed bunting	Arable	Stubble, grass management	WBC, grass margins, hedgerow management, ditch management	Wet grassland
	Mixed	Stubble, Grass management	WBC, grass margins, hedgerow management, ditch management	Wet grassland
	Pastoral	Stubble	WBC, grass margins, hedgerow management, ditch management	Wet grassland
Skylark	Arable	Grass management	WBC	Dry grassland
	Mixed	Stubble	WBC	Dry grassland
	Pastoral	Stubble	WBC	Dry grassland
Song thrush	Arable	Grass management	WBC, hedgerow management	Woodland
	Mixed		WBC, grass margins, hedgerow management	Woodland
	Pastoral		Grass margins, hedgerow management	Woodland

We next used these estimates to identify species that benefitted from at least one sharing and one sparing option (Table 2.2). Despite having to use different effect-size metrics for sharing (the additional population growth rate attributable to the intervention) and sparing (the density difference between intervention and non-intervention land), we can still compare the certainty of these estimates by calculating the width of the corresponding CI's as a proportion of the mean; from this we can calculate the sample sizes required to reduce those CI's to a specified target width. We explored the first of these issues – the certainty of estimates given existing monitoring protocols – by calculating CI's (from Equation [3]), using each study's standard errors and adjusting our CI's to account for the increase in monitoring effort seen through to 2018 (Appendix B). We then estimated the sample sizes required to determine effects to two target degrees of certainty: a precise degree where CI's were no wider than 20% of the mean and a less precise interval of 50%. We assumed standard deviation and effect size do not change with increasing sample sizes, such that:

$$CI\ width \propto \frac{1}{\sqrt{n}} \qquad \textbf{[5]}$$

from which we calculated the change in *n* (sample size; i.e. the number of squares monitoring by volunteers in the Breeding Bird Survey) required to deliver CI's of the desired widths. To illustrate the feasibility of obtaining these sample sizes, we estimated them as a proportion of current sampling effort (see Appendix B). Last, to estimate sample sizes required to show statistical significance, rather than to deliver certain estimates of effect sizes, we calculated the minimum sample sizes required to deliver a CI that does not overlap 0.

Results

- **Compliance monitoring**

1. Is increased spend on compliance monitoring made worthwhile by reductions in the number of farms needed to deliver environmental targets?

We explored the costs of sharing and sparing schemes that deliver the same environmental outcome at a range of monitoring rates. Differences among schemes are explored in detail below, but all schemes followed the schematic pattern in Figure 2.1: costs varied dramatically with monitoring rates but were lowest for all schemes when approximately a third of participants were checked for non-compliance – far more than the current 6% monitoring rate (RPA, *pers. comm.*). Monitoring costs were a relatively small – but extremely important – component of total cost. Overall costs fell dramatically with relatively smaller increases in spend on monitoring because this reduced non-compliance, so fewer participants were required to deliver the target benefit. Indeed, monitoring costs actually fell when monitoring was increased from 25% to 35% since there are fewer participants in the scheme, so monitoring a greater proportion equated to fewer actual checks. Overall costs did increase when monitoring was increased further since little non-compliance remained.

Heavier fines could reduce the monitoring required; but for the current monitoring rate to deliver the lowest scheme costs, our model suggested fines for major non-compliance would need to be >11x the payment rate (Figure B7).

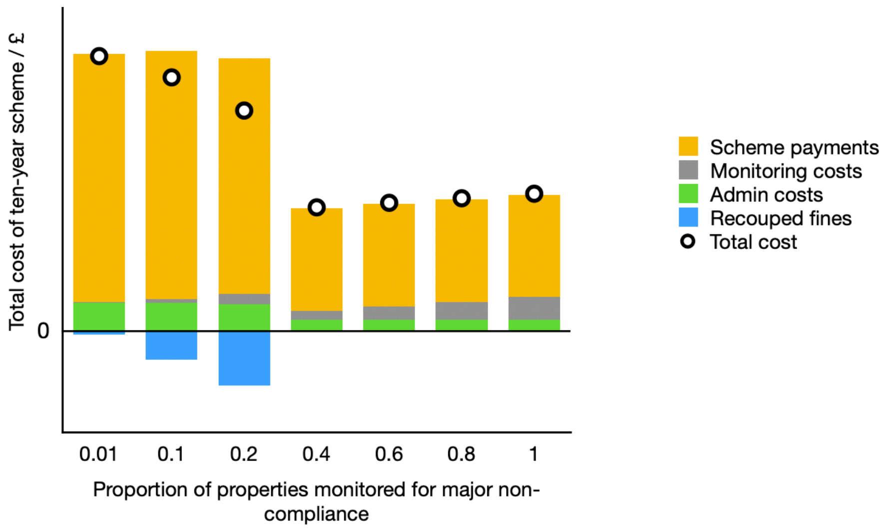

Figure 2.1. Schematic illustrating the total cost of a hypothetical scheme (black circles) broken down into scheme payments (orange), monitoring costs (grey), administrative costs (green), less recouped fines (blue).

2. At different monitoring rates, what is the cost of delivering target environmental outcomes with sharing- and sparing-style approaches?

The costs of schemes that delivered the target outcomes in terms of yellowhammers, bullfinches, lapwings and net carbon emission reductions are shown in Figure 2.2. At the optimum monitoring rate, the target outcomes were always delivered at least cost by sparing, whether considering all costs together (scheme payments, monitoring costs, administrative costs and fines) or monitoring costs alone (dotted lines; Figure 2.2).

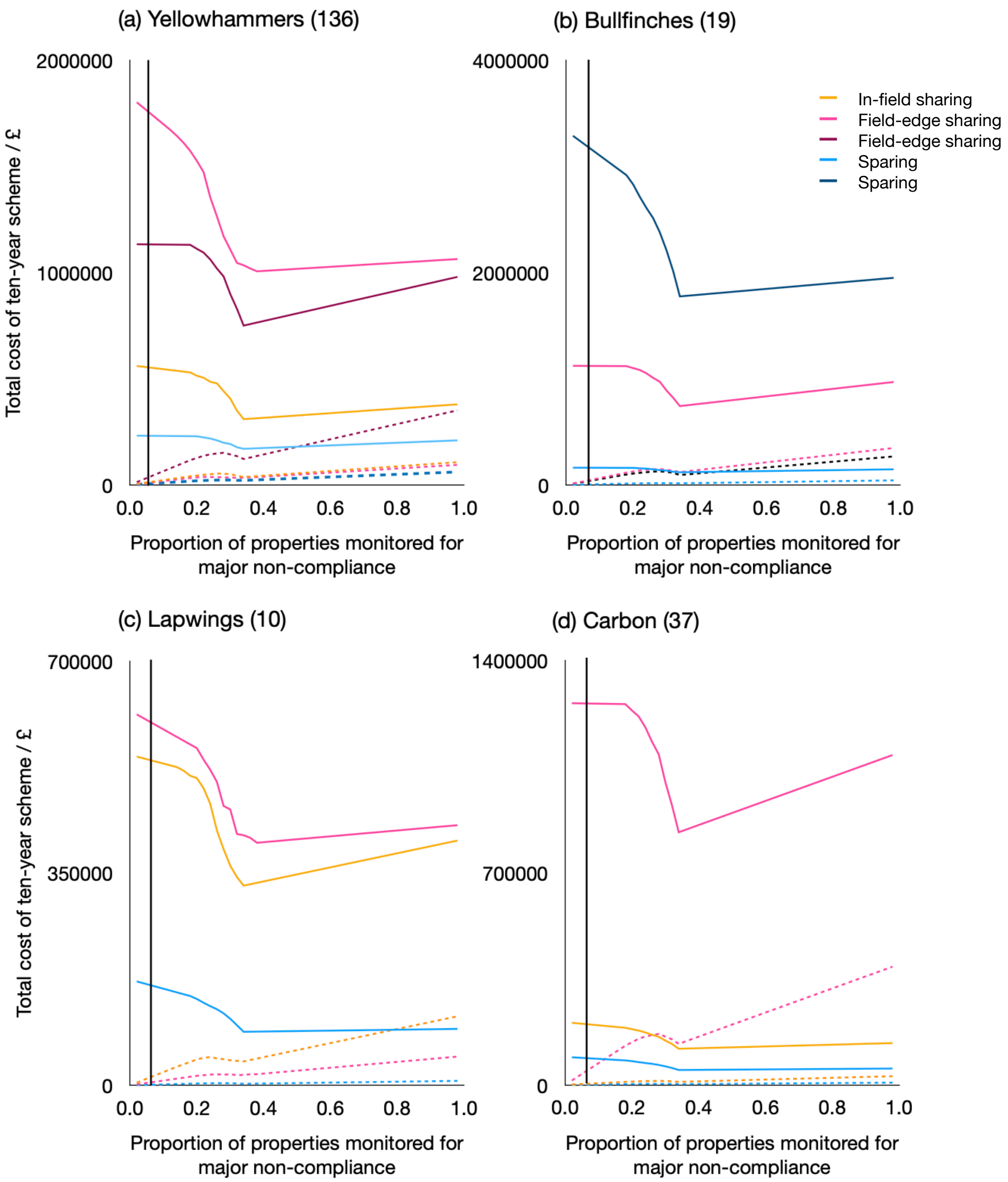

Figure 2.2. For varying monitoring rates, the total cost (solid lines; i.e. scheme payments and monitoring plus administrative costs less fines) and monitoring costs only (dotted lines) for 10-year schemes involving in-field sharing (orange), field-edge sharing (pink) and sparing (blue), where the schemes deliver the same amount of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) reduced net carbon emissions. Here we only plotted costs for the ratio of in-person to remote monitoring that delivered lowest cost: 0.4 for winter bird cover and fallow plots, 0.9 for hedgerows and sparing and 1 for in-field sharing (remote monitoring not possible). Of the two field-edge sharing interventions for yellowhammers, the lighter line is winter bird cover and the darker line is hedgerow creation. Of the two sparing interventions for bullfinches, the lighter line is scrub and the darker is woodland creation. Black vertical line shows present monitoring rate. Target outcomes are shown in brackets in the headings – units are birds or tonnes carbon/y.

3. What is the cost of monitoring at a suboptimal rate?

Monitoring at the present suboptimal rate is extremely costly (Figure 2.2; y-axis variation). We estimated that increasing the current monitoring rate from its current level of 6% to 34% would reduce the cost of delivering the target outcomes by between 33-47%. In absolute terms, it was relatively far more costly to monitor sharing-style interventions at a suboptimal rate compared to sparing. Furthermore, the costs of monitoring at a suboptimal rate would rise further if the prevalence of 'honest' farmers was lower than considered here (Figure B1).

4. How sensitive are the relative costs of sharing- and sparing-style schemes to variation in assumed parameter values?

We parameterised this model using our best estimates of real-life values of a number of parameters. To recognise the uncertainty in these values, we explored the sensitivity of the relative costs of sharing vs sparing to variation in key parameters (presented in full in Appendix B). Varying the proportion of farmers assumed to be honest, the rate of fines, monitoring costs, the area enrolled by participants and the availability of remote monitoring did not substantially alter the patterns presented above. However, and unsurprisingly, the least expensive strategy did change from sparing to sharing if the sparing payment rate was made substantially greater (or the benefit of sparing was substantially reduced; Figure B5) while the sharing values were unchanged – by 2.6x for yellowhammers, 10x for bullfinches, 4.0x for lapwings and 3.1x for carbon (Figure 2.3; intersections between blue and other coloured lines).

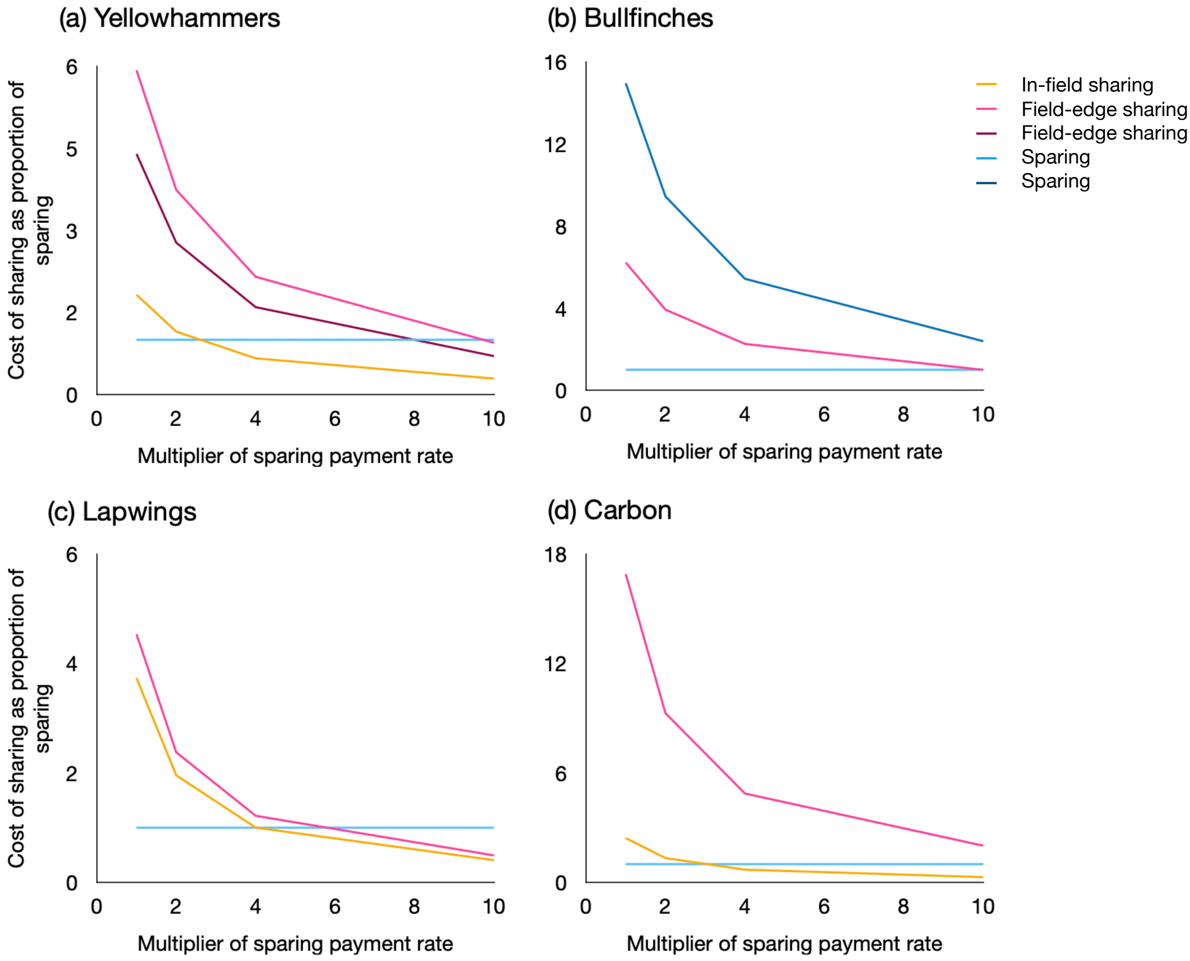

Figure 2.3. The cost of sharing schemes presented as a proportion of sparing schemes when the sparing payment rate was increased by the range of factors shown (whilst sharing payment rates do not change). Within each plot, schemes deliver the same amount of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) carbon.

- **Effectiveness monitoring**

When identifying sharing- and sparing-style interventions to compare for effectiveness, we found that some species benefitted from multiple interventions of a single type (where the types were in-field sharing, field-edge sharing and sparing). In presenting the results for each species, we show only the most certain intervention of each type, i.e. the intervention with the CI that is the smallest proportion of the mean effect. Furthermore, Baker et al. (2012) monitored some sharing interventions at multiple spatial scales; we present only the scale at which we are most certain. The benefit delivered by stubble for skylarks in pastoral systems was excluded owing to the huge associated uncertainty which obscured other patterns on these plots.

1. How certain are existing estimates of the effectiveness of AES?

There is considerable uncertainty in estimates of the effects of all studied interventions, though more so for sharing than sparing for every species and every farm system studied (where uncertainty is defined as CI width as a percentage of the mean effect; Figure 2.4). CI width was <50% of the mean for only one sharing intervention (out of 20), but for most sparing interventions (10 of the 14 studied).

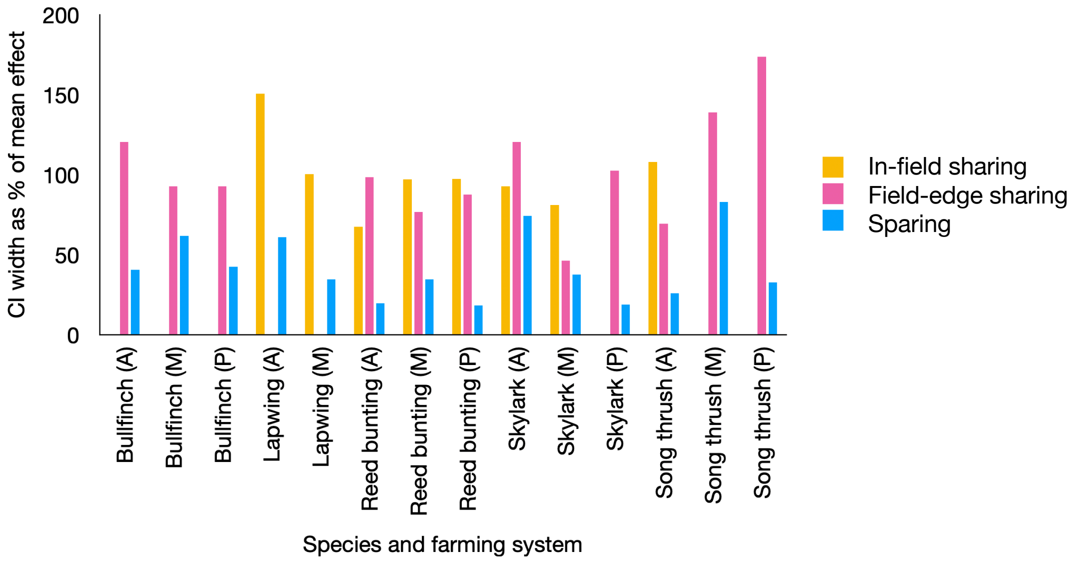

Figure 2.4. The uncertainty associated with reported estimates of the effect of sharing (in-field – orange; field-edge – pink) and sparing (blue) schemes, presented as a % of the mean estimated effect, for a range of species and farming systems (A=arable, M=mixed, P=pastoral). Calculations assume 2018 BBS sampling effort in England.

2. What sample sizes are required to determine the effects of schemes to target degrees of certainty? To estimate the mean effects of sharing with associated CI's no wider than 20% of the mean, current effort must increase >20x for 15 of the 20 studied species-farm system combinations (Figure 2.5a). Sparing required smaller – though still substantial – increases to deliver this level of precision (10 out of 14 require <5x increases). For sharing interventions, achieving a less-demanding CI width of 50% of the mean would still require >3x increases in the 2018 sampling effort for 17 of the 20 studied species (Figure 2.5b). For sparing, this certainty was often obtained without increasing current sampling (10 out of 14; Figure 2.5b).

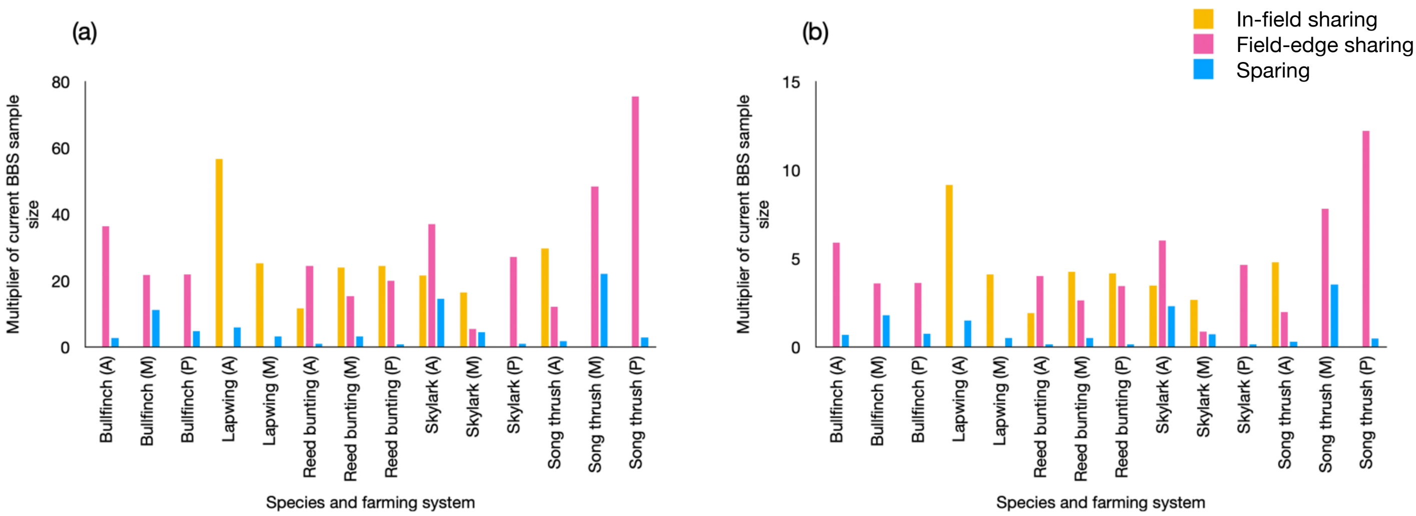

Figure 2.5. The sample sizes required to determine the effect of sharing (in-field – orange; field-edge – pink) and sparing (blue) schemes with CI width (a) <20% of the mean and (b) <50% of the mean presented as a multiplier of 2018 BBS sample size in England for a range of species and farm systems (A=arable, M=mixed, P=pastoral).

3. What sample sizes are required to determine statistical significance?

Far smaller samples sizes were needed to find statistically significant – rather than precise – effects. Typically, sample sizes below the 2018 BBS sampling effort in England delivered statistical significance (14 out of 20 sharing interventions, all 14 sparing interventions; Figure 2.6).

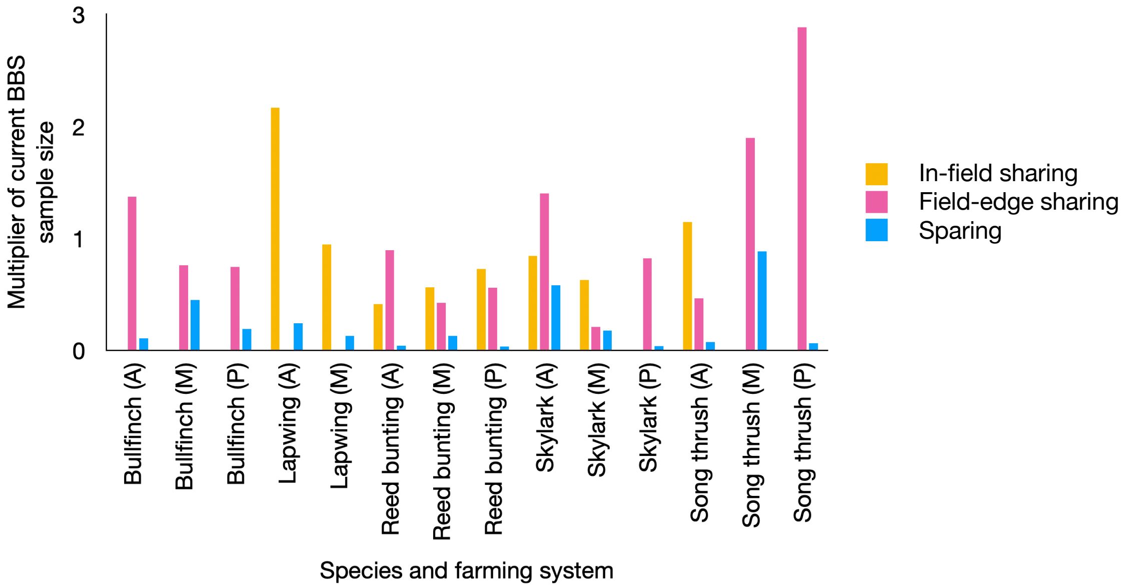

Figure 2.6. The sample sizes (presented as multipliers of 2018 BBS sample sizes in England) required to determine the effects of sharing and sparing schemes as statistically significant.

Discussion

We found that a dramatically more farmland must be monitored to precisely determine the effect of sharing-style agri-environment schemes. The costs of this increase, though not quantified, are probably substantial compared to the costs of monitoring sparing, where effects are already known with far greater certainty, albeit via a study design that may be less robust (see below).Furthermore, for compliance monitoring, our results suggest the current low levels of monitoring in English AES deliver environmental outcomes inefficiently and with great uncertainty. Despite the cost of additional monitoring, increasing the rate of current compliance monitoring from ~6% to ~34% would greatly reduce the costs of delivering biodiversity and net reductions in carbon emissions by 33-47% for in-field or field-margin improvements (characteristic of land sharing) and schemes delivering habitat restoration (as envisaged in land sparing). We found the same outcomes to conservation were always delivered at less cost with sparing-style interventions compared to sharing, whether considering combined costs or monitoring costs alone, given current CSS payment rates.

Our compliance monitoring analysis was limited in the payment rates, and therefore environmental targets, that could be studied. Given current CSS payment rates, the costs of sparing were lower because greater benefit is delivered per unit compensation and, in most cases, per unit area (Appendix A). Therefore, our focal environmental targets could be met with fewer participants, which reduces spend on scheme payments and monitoring, compared to sharing schemes. However, in some cases, CSS payment rates delivered greater participation than required; so presumably a lower payment rate would have sufficed. In testing the sensitivity of our conclusions, we predictably found that sharing does become the less expensive strategy when the sparing payment rate is increased substantially, or the benefit delivered by sparing interventions is decreased, whilst that of sharing is held constant (Figure 2.3 & B5). The relative performance of sharing and sparing did not, however, change with variation in the proportion of honest farmers, size of fines, monitoring costs, area enrolled by participants or the availability of remote monitoring (Figures S3-8). Given current AES have failed to reverse biodiversity declines (Hayhow et al. 2017; Chamberlain 2018), policymakers may well be interested in delivering more ambitious conservation targets than those considered here, which would require wider uptake of interventions. A separate study, that explores how participation varies with payment rate, would be needed to quantify the costs of delivering greater environmental outcomes and how these may vary across contrasting intervention types (Chapters 3 & 4).

That the effects of sharing are difficult to determine precisely is well known (McCracken et al. 2015). Uncertainty is affected by study design: the design of Baker et al. (2012), which compares population growth rates through time relative to the area of intervention, would be expected to deliver less biased results than the single-time control-impact design of sparing studies which cannot control for differences between control and impact sites that may exist prior to the intervention (Christie et al. 2020). From our study, we cannot quantify the difference in required sample sizes that is attributable to these aspects of study design; however, we should factor additional uncertainty into the findings of sparing studies given the likely biases introduced by study design. Independent of study design, fundamental differences in species' responses are probably also important. Sharing interventions are smaller in spatial- and temporal-scale than the larger-scale habitat restoration characteristic of sparing; this means that the scale of sharing may be insufficient to influence national population changes of farmland birds in England (Bright et al. 2015). Their small scale may also mean there is greater variation within BBS squares, since features of the surrounding land may dilute effects (Scheper et al. 2013). For these reasons, further study of sharing interventions with existing protocols will probably continue yielding highly uncertain results; and single-time studies of population density (as we used for sparing) may not be appropriate (Nielsen et al. 2009; Baker et al. 2012). Past monitoring of sparing, on the other hand, has delivered precise estimates; though we should consider whether more robust approaches are required (e.g. before-after control-impact design; Christie et al. 2020).

Policymakers must consider whether monitoring for compliance and effectiveness can be upscaled to the levels required. Current compliance monitoring results in high levels of cheating (46% of participants monitored in English AES between 2015-2017 did not comply with all requirements; RPA, *pers. comm.;* Table B2). However, the government may not recognise that the cost of increased monitoring would be outweighed by the benefit delivered by greater compliance since their policy does not in practice set out to deliver quantified environmental outcomes (Pe'er et al. 2014). Furthermore, increased monitoring may be undesirable: farmers may require more compensation to enrol in schemes with higher monitoring rates (Broch & Vedel 2012) or be discouraged from participating at all (Keane et al. 2008). The need to monitor more may be reduced if more farmers than the 54% considered here are in fact 'honest' (though this is unlikely given the current rate of non-compliance) or if fines are increased; although, again, this may discourage participation (Keane et al. 2008).

The extent to which effectiveness monitoring must be increased depends on the level of certainty sought. It is unlikely that the 2018 BBS volunteer effort could be upscaled to deliver precise estimates of the effects of sharing, though it is generally sufficient for monitoring sparing. The BBS is not explicitly designed to monitor sharing and sparing schemes but the vast amount of data collected by its volunteers mean it is often usefully co-opted for this purpose; collecting data through bespoke monitoring schemes would be far more expensive. Studying for statistical significance (e.g. Baker et al. 2012; Bright et al. 2015; Dadam and Siriwardena 2019; Gillings et al. 2005), rather than precision, does dramatically reduce the required sample size but is prone to Type I errors (false positives; as identified by Baker et al. (2012)), ignores the issue that statistically significant effects may be extremely small (Nakagawa & Cuthill 2007), and negates our ability to set targets or compare cost-effectiveness. Therefore, precise determination of effects is better than focusing on statistical significance (Nakagawa & Cuthill 2007). Finally, non-compliance may contribute to high variability in the data since some survey areas considered to have implemented the interventions may not in practice have done so. Therefore, increased compliance monitoring may reduce the required spend on effectiveness monitoring.

Our results indicate that current monitoring for both compliance and effectiveness is insufficient. Policymakers should consider the feasibility, and estimate the costs, of upscaling monitoring to an appropriate level when considering potential approaches to delivering conservation on farmland. Our study was the first to compare the efficacy of using current approaches to monitor compliance and effectiveness for interventions characteristic of sharing and sparing. Our findings suggest that future policy should not be constrained by the sharing approach that has dominated AES to-date. Indeed, not only does sharing deliver less for conservation (Lamb et al. 2016a; Finch et al. 2019): our results indicate it is also more expensive to monitor for compliance (given current payment rates) and policymakers are largely in the dark about its effectiveness. However, comparison of farmers' attitudes towards the implementation of sharing and sparing, and estimation of how both approaches impact food production, are needed to fully compare the relative costs of delivering the ambitious environmental targets required to reverse biodiversity declines and realise the potential for farmland to store carbon. Therefore, in Chapter 3, we estimate farmers' willingness to accept compensation for implementing sharing and sparing agri-environment options before combining this, in Chapter 4, with estimates of monitoring costs, administration costs and lost food production in the most complete known comparison of the costs of delivering environmental outcomes with sharing and sparing.

Chapter 3: Exploring the relative preferences of farmers for land sharing and land sparing as revealed by a choice experiment

Abstract

Action to address biodiversity declines and climate change in Europe relies largely on farmers who voluntarily accept compensation for yield-reducing management designed to benefit the environment. In England, such schemes have so far mostly encouraged so-called wildlife-friendly farming within land still managed for food production; a land-sharing approach. However, despite annual investment in excess of £600m, conservation gains from this strategy appear limited as wildlife declines continue Land sparing, in contrast, involves concentrating food production into a smaller, higher-yielding farmland area, and safeguarding or restoring land for conservation elsewhere. To understand the relative costs to the taxpayer of land sharing and land sparing, we assessed farmer preferences for contrasting land-sharing and land-sparing interventions. We conducted an online choice experiment with 118 farmers in England. Using latent class modelling, we found preferences for sharing and sparing varied widely. Given equal compensation, more farmers were willing to enrol in sharing-style interventions. However, since sparing often delivers greater environmental benefit per unit area, spending our budget paying farmers to spare delivered more of two farmland bird species targeted in our hypothetical scheme, bullfinches (at low budgets) and lapwings, as well as greater reductions in carbon emissions. In contrast, for a given budget, sharing delivered more yellowhammers, which are found at comparable densities on farmland and semi-natural habitat, and bullfinches when the budget was large. The farmers most willing to spare land farmed, on average, a larger area and currently participated in stewardship schemes. This is the first evidence that farmer preferences do not preclude the adoption of a land-sparing policy approach: indeed, they may require less compensation to spare than share to deliver all but the most farmland-tolerant environmental outcomes.

Introduction

There are calls to reform agricultural policies across Europe in light of continued biodiversity declines and the threats posed by climate change (Pe'er et al. 2020). The UK's withdrawal from the European Union's Common Agricultural Policy presents an opportunity to redefine a long-term policy which has broadly failed to stem biodiversity declines (Kleijn & Sutherland 2003; Kleijn et al. 2006; Dicks et al. 2014; Batáry et al. 2015; Pe'er et al. 2020) or realise the potential for farmland to store carbon (Pe'er et al. 2019). Contrasting supply-side approaches to reconciling food production with environmental conservation can be conceived as lying along a continuum between land sharing, i.e. wildlife-friendly farming across the landscape, and land sparing where high-yield, biodiversity-poor farming is combined elsewhere in the landscape with large blocks of natural or (semi-)natural habitat. Sparing requires retention of natural habitats or, in regions with little remaining natural habitat, restoration of (semi-)natural habitat on previously farmed land (Green et al. 2005). In England, land sparing has been estimated to deliver more biodiversity and carbon sequestration than land sharing, when comparing across landscapes producing the same total amount of food (Finch et al. 2019; Lamb et al. 2019; Finch et al. 2020; Lamb et al. 2016b). Despite this, English agri-environment policy has continued to invest predominantly in land sharing, such as through Pillar 2 of the Common Agricultural Policy.

One objection to land sparing, and a possible driver of the continued land-sharing approach, is the assumption that wildlife-friendly farming is a lower cost option for farmers (Fischer et al. 2008) and therefore cheaper to compensate. However, no previous study has compared the payments required by farmers to implement sharing and sparing interventions. Current AES in England predominantly fund sharing options; sparing-like interventions receive only 12% of what is spent on sharing (Appendix B), with far lower compensation rates, whether considered per unit area or per unit of lost production, reflected in much lower participation rates. To compare the cost-effectiveness of sharing and sparing, we must consider the benefit delivered when the same budget is spent on sharing and sparing; and, for this, we require a novel study of how participation in sharing- and sparing-style options varies with the subsidy offered to farmers.

Farmers are known to vary widely in their willingness to alter their practices and forgo food production for the benefit of the environment (e.g. Broch and Vedel 2012). Relative preferences will vary due to differences in the value of forgone production, because of attitudes towards specific impacts of the intervention (e.g. associated benefits, the time required for management, visual implications and effects on pest and disease control), and behavioural characteristics such as risk aversion or inclinations towards conservation (reviewed in Dessart et al. 2019). Variation amongst farmers is of particular interest when, as in English AES, fixed-price schemes see all farmers paid at a uniform rate per hectare instead of payments being differentiated based on individual willingness to accept compensation (Ferraro 2008). If sparing delivers, on average, greater environmental benefit per unit area, then delivery of a specified outcome would require fewer participants than if sharing approaches were adopted. The larger spatial scale of each sparing intervention may also make it attractive to farmers. However the novelty and unfamiliarity of sparing approaches might result in farmers requiring large payments per hectare than more familiar sharing approaches (Defra 2018a).

Stated preference choice experiments have often been used to explore farmer preferences towards alternative AES prescriptions, including aspects of pesticide management, afforestation, and varying area and duration requirements (Ruto & Garrod 2009; Broch & Vedel 2010; Villanueva et al. 2015; Kuhfuss et al. 2016a). In eliciting stated preferences, choice experiments are a useful tool; however, in relying from statement of intent rather than actual behaviour, error is introduced (Christie & Azevedo 2009). However, we do not know of a choice experiment that has compared preferences towards sharing- and sparing-style interventions, nor combined that with data on environmental benefits in order to assess the relative cost-effectiveness of alternative approaches.

Here, we used a choice experiment to explore the preferences of arable farmers towards land-sharing and land-sparing interventions. We used our findings to compare the delivery of a range of outcomes when paying the farmers in our sample to enrol in fixed-price schemes with a budget of up to £20m over a 20-year period. We investigated how far these results held among different types of environmental outcomes (populations of three bird species; carbon sequestration), which varied in their response to sharing and sparing interventions. Finally, we explored how farmer preferences varied with observable characteristics of farmers and the land they manage.

Methods

First, we identified environmental outcomes that could be delivered by both sharing and sparing on arable farms in the UK. Since not all of these interventions would apply to pastoral systems, we limited our study to farmers who managed at least some arable land. We chose populations of three bird species of conservation concern that differ in their response to farming, the red-listed Yellowhammer (*Emberiza citrinella*), the amber-listed Eurasian Bullfinch (*Pyrrhula pyrrhula*) and the red-listed Northern Lapwing (*Vanellus vanellus*). The population densities of these species are affected in different ways by farming practices (Finch et al. 2019); we speculated that sharing was most likely to favour yellowhammers whilst sparing was most likely to favour lapwings. Given the importance of farmland in government net-zero plans (Committee on Climate Change 2020), we also studied interventions that reduce, and in several instances reverse, on-farm carbon emissions. For sharing, we considered a 50% reduction in inorganic fertiliser use, which avoids carbon emissions, as well as the creation of hedgerows, which sequesters carbon emissions. For sparing, we studied woodland creation. We identified sharing and sparing interventions (changes to land management) applicable to arable farms that deliver these outcomes. In doing so, we divided sharing interventions into two types: "in-field sharing" methods, which affect food-producing practices across the whole field, and "field-edge sharing" methods, which involve an intervention outside the area used to produce food, often the field margin. We identified sharing and sparing interventions that delivered our biodiversity and climate mitigation outcomes and, from existing literature, estimated their per unit area delivery of the outcome (Table 3.1; calculations presented in Appendix A). All interventions had a close analogue in England's Countryside Stewardship Scheme (CSS) and we calculated the annual spend on the equivalent sharing and sparing options in the CSS: as of 2019, 89% of payments compensated sharing options whilst sparing received 11% (Rural Payments Agency; *pers. comm.*).

Table 3.1. The sharing and sparing interventions that deliver the environmental outcomes studied and their per-area benefit.

Environmental outcome	Intervention type	Intervention	Environmental Benefit *(birds/ha or tC/ha/y)*	Source
Yellowhammer	In-field sharing	Stubble, spring cropping	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)
	Sparing	Scrub	0.59	Donovan (2013); Morgan (1975)
Bullfinch	Field-edge sharing	Hedgerows	0.92	Macdonald and Johnson (1995)
	Sparing	Scrub	0.20	Morgan (1975); Knepp Estate
	Sparing	Woodland	0.05	Gregory and Baillie (1998); Lamb et al. (2018); Newson et al. (2005)
Lapwing	In-field sharing	Stubble, spring cropping	0.05	Shrubb et al. (1991); Wilson et al. (2001)
	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	0.27[1]	Kindred et al. (2008)
	Field-edge sharing	Hedgerows	1.84	IPCC (2019)
	Sparing	Woodland	3.77	Falloon et al. (2004)

[1] Value here is based on mean fertiliser application on English cereal farms but values used in analyses were calculated individually for each participant according to their reported fertiliser application rates (Appendix A).

Next, we developed a choice experiment to explore the minimum payments farmers would be willing to accept (WTA) to implement each of these interventions. Each round of the choice experiment asked farmers to choose between AES contracts involving an in-field sharing, field-edge sharing and sparing option which differed in several attributes: the type of intervention, and its area, duration and payment rate. Participants could also choose not to accept any contract offer (Figure 3.1). The attributes and levels used in the experimental design were as follows (Table 3.2):

(i) *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 field-edge 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).

(ii) *Contract 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.

(iii) *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 estimated lost gross margin based on the average farm reported by the Farm Business Survey (calculated as output minus input costs based on means from the Farm Business Survey (2019)). Where appropriate, capital costs were stated to be covered separately and in full.

Table 3.2. The attributes and levels used in the choice experiment.

Attribute	**Levels**							
Type of intervention	In-field sharing		Field-edge sharing			Sparing		
Within type intervention	Stubble	50% N fertiliser reduction	Winter bird cover	Fallow plots	Hedge[1]	Scrub	Woodland	Wet grass
[2]**Area(ha)**	10, 20, 50	10, 20, 50	5, 10, 20	5, 10, 20	2, 4, 8	10, 20, 50	10, 20, 50	10, 20, 50
Duration(years)	5, 10, 20	5, 10, 20	5, 10, 20	5, 10, 20	10, 20, 50	10, 20, 50	10, 20, 50	10, 20, 50
Payment rates £/ha	40, 80, 120, 160, 240	130, 260, 400, 550, 800	175, 350, 525, 700, 1050	170, 340, 500, 700, 1000	459, 918, 1360, 1700, 2720	300, 600, 900, 1100, 1700	300, 600, 900, 1100, 1700	300, 600, 900, 1100, 1700

[1]For hedgerow creation, areas and payment rates were presented per km length hedgerow.

[2]50ha area requirements were not presented to participants farming <100ha.

Figure 3.1. Sample choice card.

We asked each participant to make a sequence of 12 contract choices (Greiner et al. 2014). These attributes and levels were combined in an efficient design generated using Ngene with priors obtained from a pilot study (as per Rose and Bliemer 2013). This design comprised 12 blocks, each containing 12 choices, with each participant randomly assigned to a specific block.

The choice experiment was set within an online Qualtrics survey. Before presenting the 12 choice tasks, we asked participants for information on their farm size so that larger area requirements could be removed for participants with small farms. After the choice tasks, questions explored why those choices were made and gathered information to allow estimation of lost food production and gross margins for each crop. Finally, we asked about several characteristics of the participant and their farm: age, education, perceived likelihood of future profitability, proportion of land owned vs rented, and present participation in AES.

We gained ethics approval from the University of Cambridge Psychology Research Ethics Committee (HVS/2018/2582) and informally piloted our survey in May 2019 to gather preliminary feedback. Formal piloting took place in June/July 2019 with 11 respondents. The final survey was launched in September 2019 and generated 118 useable responses from farmers in England and bordering arable areas in Wales, who between them farmed 1.7% of all English lowland arable land (Defra 2019c) by June 2020. Since we studied interventions applicable to arable land, some of which could not be undertaken in pastoral systems, participant farmers had to manage some arable land to be eligible. Participants were recruited through a variety of channels including farming newsletters and magazines, Twitter, and online fora. Our therefore non-random sample is over-representative of younger farmers and larger farms (Figure C1). Participants were offered a summary of the findings, personalised estimates of the cost of implementing the studied options on their land, and the chance to win a subscription to Farmers Weekly.

Analysis

The choice data were analysed in R using Apollo (Hess & Palma 2019). We were interested in understanding the heterogeneity of preferences across the studied population and exploring the characteristics associated with that distribution. A latent class model is well suited for this purpose (Hess 2014). Latent class modelling assumes preferences can be grouped into a discrete number of classes (Hess 2014). The appropriate number of classes for a given choice dataset can be determined using a number of criteria. Each participant was assumed to belong to these classes according to a set of probabilities. Individuals were assumed to choose between alternative options based on their relative utility (Luce 1959). The utility of alternative *j* to individual *n* who belonged to class *c,* i.e. V_{jnc}, was assumed to be the sum of the individual's preference towards each attribute *S* (as per Lancaster 1966):

$$V_{jnc} = \beta_0 + \beta_{c1} S_{1j} + \beta_{c2} S_{2j} + \ldots + \beta_{ck} S_{kj} \qquad \textbf{[1]}$$

Where the coefficients β reflect sensitivity to each attribute and β_0 is a constant which accounts for preference heterogeneity unexplained by the other attributes. The modelling process estimated values for the β coefficients that best predict the observed choices where the probability of respondent *n* choosing alternative *j* in choice situation *t,* assuming they belong to class *c*, i.e. $P_{njt}(\beta)$ was:

$$P_{njt}(\beta_c) = \frac{e^{V_{njt}}}{\sum_{j=1}^{J} e^{V_{njt}}}$$

[2]

where *J* is the total number of alternatives: i.e., the probability of a choice is the utility (given membership in class *c*) associated with that alternative divided by the summed utility of all alternatives. Exponential transformations are simply used to avoid division by zero or negative numbers.

However, no individual belonged perfectly to a single class; thus the likelihood with which farmers belonged to each class must be incorporated into the model. We assumed individual *n* belonged to class *c* with probability π_{nc} where $0 \leq \pi_{nc} \leq 1$ and $\sum_{c=1}^{C} \pi_{nc} = 1$. These class allocation probabilities were assigned during modelling to maximise the likelihood with which the model reproduced respondents' stated choices. So, the likelihood of an individual (L_n) making their set of choices for a given set of β coefficients ($L_n(\beta)$) was:

$$L_n(\beta) = \sum_{c=1}^{C} \pi_{nc} \left(\prod_{t=1}^{T_n} P_{njt}\,(\beta_c) \right)$$

[3]

Where *njt* refers to the alternative *j* chosen by individual *n* in choice task *t* and T=12*.* So, the likelihood of an individual's choices was the product of the probabilities of making their choices in each of the 12 tasks given their membership in class *c*. Then, to recognise that participants belonged partially to all classes, that product was calculated for all *C* classes and multiplied by the probability that the individual belonged to that class. Finally, these results were summed across classes.

We assessed the performance of the model in terms of its log-likelihood (LL)*,* which across all *N* sampled individuals was given by:

$$LL = \sum_{n=1}^{N} \ln L_n(\beta)$$

[4]

The modelling process iteratively sought to find the β coefficients and the class membership probabilities π that reproduced the respondents' choices with the greatest LL.

In specifying our model, we linear-coded the area, duration and payment attributes, so that the output parameters indicated the utility of a one-unit change. We dummy-coded the "type of intervention" parameters, so that the output parameters indicated their utility relative to not participating in any contract.

We excluded the six participants that opted out of every contract choice from our analysis as this improved model fit. These participants were less likely to be participating in current AES (17% vs 62% across the remainder of the sample) and were more confident of that they will be profitable in the future (3.2 vs 2.4 on a five-point scale where higher numbers indicate greater confidence), compared to the sample mean.

To determine the appropriate number of classes, we considered LL, McFadden's pseudo-R^2, Akaike Information Criterion (AIC) and minimum Bayesian Information Criterion (BIC) statistics (Table 3.3). All statistics improved by increasing the number of classes from two to three, but there was no clear improvement across model statistics for the four-class model compared with the three-class model (LL, R^2 and AIC improved but BIC worsened). Therefore, we selected the three-class model (following Villanueva et al. 2015), since this allows a more parsimonious description of preference heterogeneity. The three-class model has a pseudo R^2 of 0.22; in such models, values between 0.20-0.40 are thought to indicate excellent fit (McFadden 1973).

Table 3.3. Criteria used for setting the optimal class number. Higher values of LL and R^2 and lower values of AIC and BIC imply better fit.

No. classes	Parameters (P)	LL	McFadden's pseudo-R^2	AIC	BIC
1	11	-1363	0.12	2748	2803
2	22	-1298	0.21	2640	2700
3	33	-1215	0.22	2495	2661
4	44	-1178	0.24	2462	2698

Parameter estimates for each non-price attribute were converted into WTA estimates, which reflect the amount by which contract offer prices must change for farmers to be willing to participate in a specific contract, by taking the ratio of non-monetary contract parameters (β_{NM}) to the payment parameter (β_M), and since this is a study of willingness-to-accept, rather than willingness-to-pay, the negative of this ratio must be found such that farmers prefer higher payment rates i.e.:

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

[5]

This gave the compensation required to offset a unit change in that parameter if linear-coded (area, duration), or a shift to the baseline state if dummy-coded (type of intervention).

Finally, we estimated the environmental benefit that would be delivered for a range of budgets over 20 years. We discounted future spend at a rate of 3.5% (following HM Treasury 2018). Given continued biodiversity declines and the scale of the climate crisis, we wanted to explore ambitious action, so we considered a budget of up to £1m/y, so up to £20m spent across the 20-year period, for each of our four outcomes (yellowhammers, bullfinches, lapwings, carbon). Current spend is relatively lower: interventions that benefit yellowhammers receive the greatest combined compensation but even then only £4.03m would be spent across an equivalent area in a 20-year time period (RPA, *pers. comm.)*.

We calculated the confidence intervals associated with our estimates of the environmental benefit delivered to reflect uncertainty in our assessment of farmer preferences. To do so, we estimated the 95% confidence intervals associated with our estimates of class WTA using the delta method in Apollo (Bliemer & Rose 2013). We then re-ran our simulation of the environmental benefit delivered for a given spend assuming first the lower, and then the upper, CI bound of WTA for each class. We did not consider other costs associated with these schemes such as capital, administration and monitoring costs, or lost food production.

Results

The three classes characterising farmer preferences differed markedly in the payments required for different types of interventions (see Table 3.4 for parameter values, converted to marginal WTA estimates in Figure 3.2). We found that farmers most likely to belong to Class 2 were the most willing to participate in all interventions, with sharing options generally requiring lower compensation payments than sparing. Class 3 were quite unwilling to participate in any intervention, particularly in land-sparing options. Class 1 were relatively willing to participate in some sharing options, though they required more compensation than Class 3 for interventions designed to reduce fertiliser and create hedgerows, suggesting some specific aspects of these interventions divided preferences within the sample. Longer durations and larger areas required more compensation across all classes; Class 2 required the least additional compensation to enrol an additional hectare or year.

Table 3.4. Parameters derived from latent class modelling. Standard errors are given in brackets.

	Intervention								**Area**	**Duration**	**Payment**	**Mean probability of class membership**
	Stubble	50% fertiliser reduction	Winter bird cover	Fallow plots	Hedgerow	Scrub	Wood	Wetland				
Class 1	0.3376	-1.5897*	-1.1663*	-0.9940*	-3.8626*	-2.0011*	-2.4607*	-4.0123*	-0.0123	-0.0223*	0.0022*	0.37
	(0.3369)	*(0.3519)*	*(0.3889)*	*(0.3184)*	*(0.4924)*	*(0.4944)*	*(0.4726)*	*(0.72)*	*(0.0071)*	*(0.0085)*	*(0.0001)*	
Class 2	1.4689*	0.5650	1.1621*	1.1576*	-0.6892*	0.1747	-1.4903*	0.3815	-0.0045	-0.0059*	0.0022*	0.39
	(0.4261)	*(0.3684)*	*(0.4199)*	*(0.4399)*	*(0.4004)*	*(0.4892)*	*(0.4809)*	*(0.5391)*	*(0.0051)*	*(0.005)*	*(0.0001)*	
Class 3	-2.5655*	-1.3462*	-1.8161*	-2.5824*	-2.4011*	-3.4637*	-4.7106*	-6.0601*	-0.0082	-0.0232*	0.0022*	0.24

(0.6406)	*(0.4299)*	*(0.3791)*	*(0.4951)*	*(0.4436)*	*(0.6988)*	*(0.952)*	*(1.9477)*	*(0.0143)*	*(0.0124)*	*(0.0001)*

* Significant at 5% level

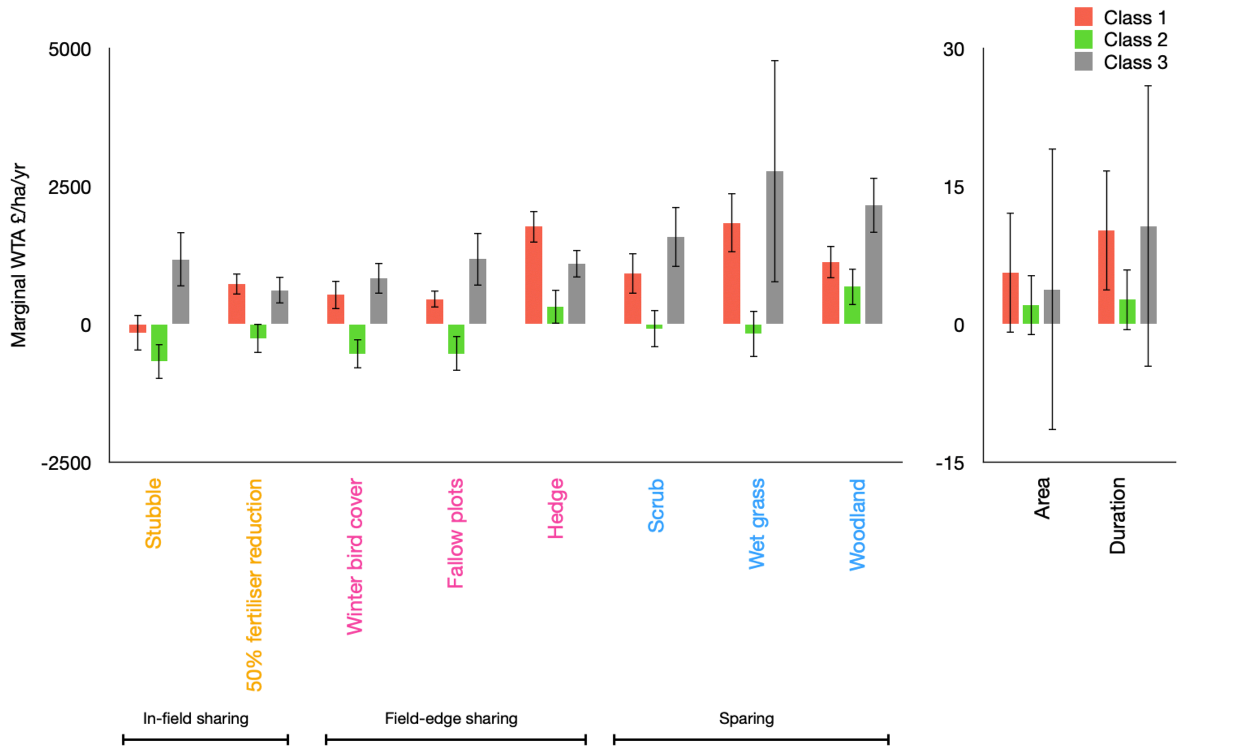

Figure 3.2. The marginal WTA compensation, for each class of farmers, to implement a range of in-field sharing (orange labels), field-edge sharing (pink labels) and sparing (blue labels) interventions, relative to not participating, and the additional compensation for each hectare (area) and year (duration) with 95% confidence intervals calculated by the delta method in Apollo (Bliemer & Rose 2013).

We next used the distribution of preferences amongst the sample to estimate the environmental outcomes delivered by fixed-price 20-year schemes, at successive budget increments up to £20m. Any negative WTA values were treated as zeros, i.e. we assumed such farmers would participate at a zero subsidy. This is visible on Figure 3.3 where lines do not begin at the origin. For all outcomes, we found a rapid increase in the area enrolled, and therefore the environmental benefit delivered, at low budgets. As the total budget increased, participants required higher payments to be recruited; i.e. the marginal supply price increased with the area of land enrolled. Since fixed-price schemes pay all participants the same subsidy, large increases in spending delivered little additional environmental benefit. We found that more yellowhammers, the species found at highest densities on farmland, could be delivered with a given budget than any other species, and that field-edge sharing delivered the greatest number; for a £20m spend, hedgerow creation delivered 2.6x more yellowhammers than sparing (Figure 3.3a). In contrast, more lapwings and greater reductions in carbon emissions were delivered for a given spend with sparing (Figure 3.3c-d): £20m delivered 3.8x and 4.6x more lapwings and carbon respectively than the best sharing strategy. At budgets below £10m, more bullfinches were delivered by sparing, but field-edge sharing (in this case creating hedgerows) became the more cost-effective strategy at higher budgets (far in excess of what is currently spent; Figure C2). Hedgerow creation does, however, involve considerable capital costs which are not considered here.

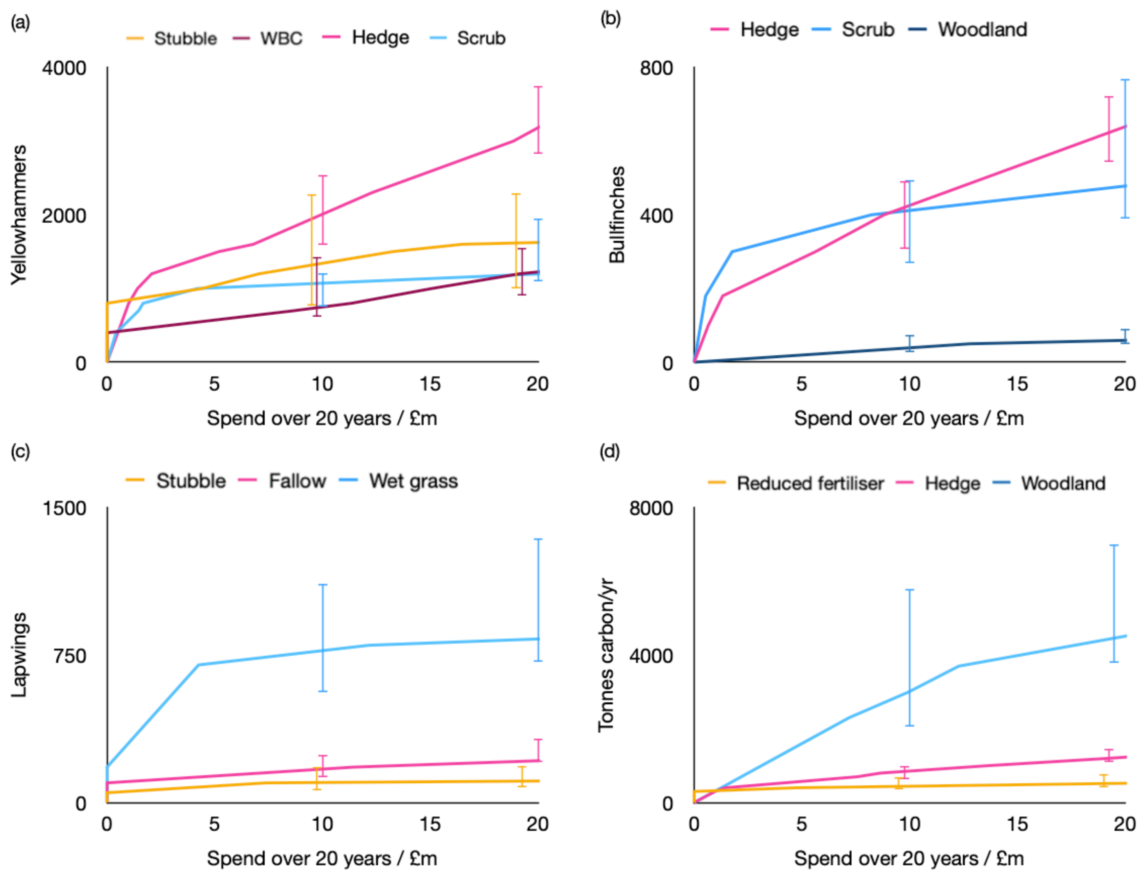

Figure 3.3. The number of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) tonnes carbon/y delivered amongst the studied 118 farmers by spending up to £20m across 20 years on in-field sharing (orange), field-edge sharing (pink) and sparing (blue) schemes. 95% confidence intervals were calculated by the delta method in Apollo (Bliemer & Rose 2013) and are jittered along x-axis where necessary to make all visible.

Next we explored the farm and farmer characteristics associated with latent classes of preference. To do so, we assigned each participant to the class they belonged to with the highest probability and then explored differences in mean characteristics between classes (Figure 3.4; Table 3.5). Membership of Class 2, the most willing to share and spare, was associated with farming a larger area of land than Class 3 (Figure 3.4a). The proportion of land owned differed little across classes (Figure 3.4b). Class 2 was associated with significantly higher current participation in AES than membership in either Class 1 or 3 (Figure 3.4c). Despite the unveiled differences across farmers in willingness to engage in our AES, all classes were highly uncertain of future profitability (Figure 3.4d). Age and education differed little across classes (Figure 3.4e&f).

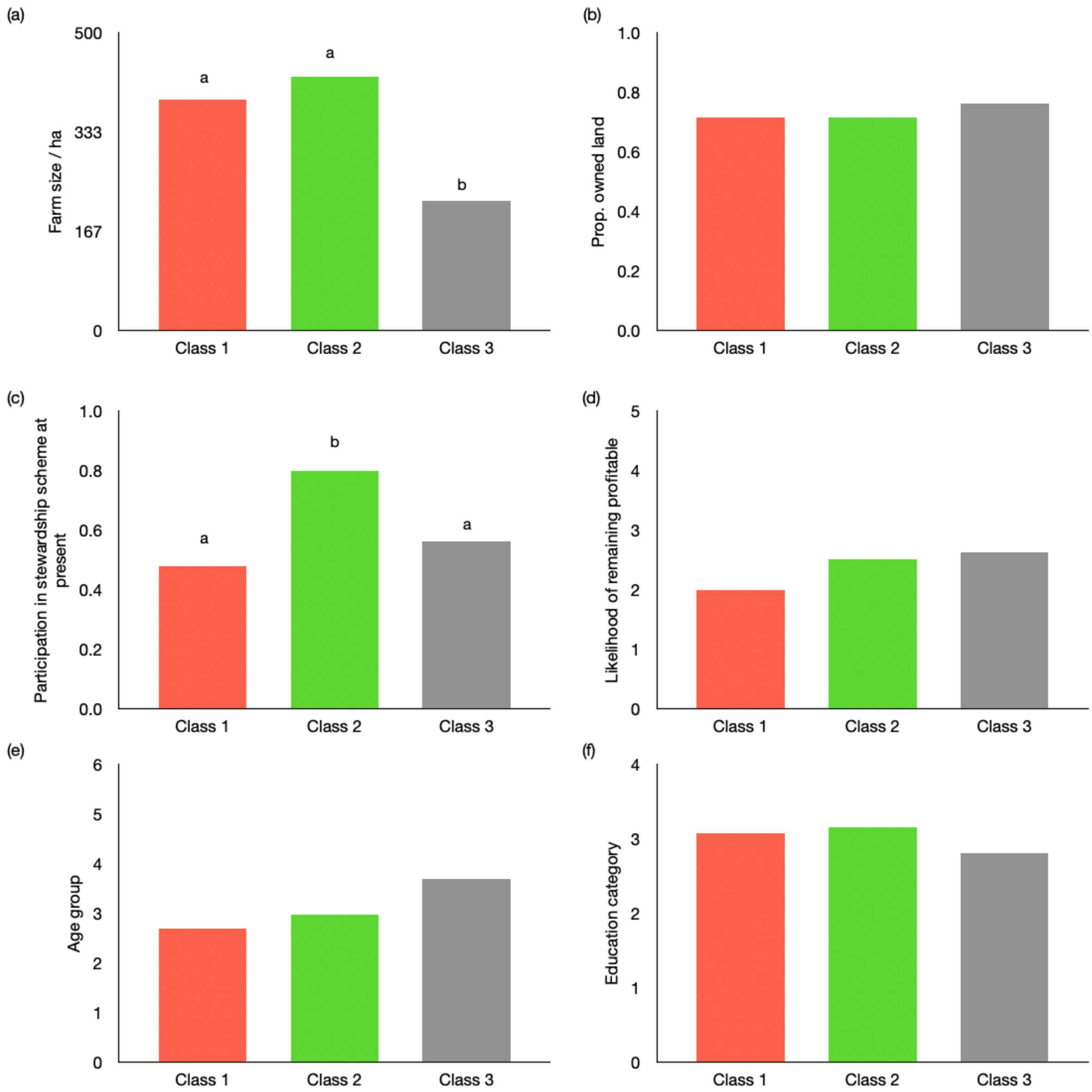

Figure 3.4. Mean characteristics associated with the classes identified by latent class analysis. For likelihood of remaining profitable despite forthcoming policy change (d), 1= disagree, 2= somewhat disagree, 3= neither agree nor disagree, 4= somewhat agree, 5= agree; for age group (e), 1= 18-24, 2= 25-34, 3= 35-44, 4= 45-54, 5= 55-64, 6=65+ and for education (f), 1= secondary school, 2= Higher National Certificate/Diploma, 3= college/undergraduate degree, 4= postgraduate qualification;. Letters indicate significant differences between classes according to Kruskal-Wallis (linear variables) and chi-squared tests (categorical variables; see Table 3.5).

Table 3.5. Posterior analysis of the distribution of characteristics across latent classes using chi-squared tests for categorical variables and Kruskal-Wallis testing for continuous variables.

Characteristic	**Chi-squared, χ^2**	**Kruskal-Wallis, *H***	**P-value**	**Degrees freedom**
Farm size		5.2	0.07	2
Prop owned		0.4	0.82	2

Stewardship	5.8	0.05	2
Profit	13.1	0.11	8
Age	12.7	0.24	10
Education	6.4	0.78	10

Discussion

Land sharing and land sparing are contrasting approaches to delivering improved biodiversity outcomes from farmland. Our study uncovered great variation in farmer preferences for participating in sharing- and sparing-style contracts. A portion of farmers were very willing to participate, particularly in sharing, but also in sparing schemes; these participants offered very cost-effective delivery of the target outcomes. Due to the greater environmental benefit delivered per unit area, sparing saw the most bullfinches, lapwings and carbon delivered at low budgets. Larger budgets could recruit less-willing farmers, but we found cost-efficiency declined dramatically because in a fixed-price scheme all must be paid at the rate demanded by the least-willing participant. At budgets well above current spend in the case study country (England), hedgerow creation delivered up to 1.3x more bullfinches than sparing – although note that the substantial associated capital costs were not considered here. At the max £20m budget, across the 20-year period, sparing delivered 3.8x and 4.6x more lapwings and carbon respectively than the best sharing strategy. Across all budgets, more yellowhammers, which are found at similar densities on farmland and in (semi-)natural habitat, were delivered by sharing, and a given budget delivered more yellowhammers than any other species.

Our ability to make policy recommendations based on these results is limited in that they are based on hypothetical choice situations rather than observed behaviour. Estimates of minimum supply prices derived from choice experiments are typically associated with large errors (Hensher 2010). Indeed, our 95% confidence intervals for the first and second-best strategy for delivering our target outcomes overlap for bullfinches and at low budgets for yellowhammers. Moreover, these intervals do not incorporate the uncertainty associated with the estimates of the environmental benefit delivered by the studied changes in farm management, because errors were not reported in all studies from which these estimates were derived. Our modelling predicted some farmers would participate in some options without any compensation. Whilst this would be expected for stubble/spring cropping, which is commonly practised not for conservation but for weed/pest control, it is unclear whether our projections of participation in other options without compensation is due to the existence of other such benefits for farmers, such as utility derived from on-farm improvements in biodiversity (Kuhfuss et al. 2016b). We did compare the outcomes predicted by our model to those of current schemes (Figure C2). Based on the area enrolled in options of existing AES that are most analogous to those studied here, we predicted the outcome generated by the government's current spend given the compensation rates offered. We compared this to our model predictions for that spend and found most matched relatively well; though we did overestimate the environmental benefit delivered for a given spend on stubble/spring cropping and winter bird seed plots (Figure C2), suggesting these sharing interventions may, in practice, be more costly to implement than our results suggest.

Our results are limited in two further ways. First, our conclusions may not hold if farmers are willing to enrol far larger areas than we considered. This could delay the plateau in environmental benefit delivered for a given scheme spend (Figure 3.3). And second, we cannot make projections for all environmental outcomes since we studied only three bird species and one ecosystem service (reduced net carbon emissions). Species commonly found on farmland are, like yellowhammers, likely to do better under sharing than sparing. Some of these species have declined as farming practices have intensified, and so are of conservation concern. However, these species which live at higher densities on farmland than natural habitats ('winner' species; Finch et al. 2019) should perhaps receive less priority. Our study did not consider specialists of (semi-)natural habitat, which cannot persist on land farmed at any yield, and for which land sparing constitutes the only feasible approach (Finch et al. 2019).

That farmers are heterogenous in their attitudes towards participating in AES is well known (e.g. Broch and Vedel 2012). We found those most willing to participate typically farmed a larger area of land and were more likely to be currently participating in stewardship schemes. These associations make intuitive sense but other important drivers probably exist. Personal traits, values and attitudes towards specific aspects of interventions are likely important (Dessart et al. 2019; Cortés-Capano et al. 2021; Kuhfuss et al. 2022). We briefly explored the latter in follow-up questions which asked participants' reasoning for not implementing any of the studied interventions as well as reasons for being encouraged/discouraged from implementing specific options. In general, participants reported more reasons not to implement sparing than sharing, with particular concerns for cultivating land after contracts end; and, as expected, wet grassland was not considered possible to create on some farms (Figure C3). Time was the most common reason for not implementing field-edge sharing whilst results for in-field sharing were mixed (Figure C3). Participants found more reasons to be discouraged (Figure C4b) than encouraged (Figure C4a) from implementing both sharing and sparing interventions. Participants were far more encouraged to implement stubble/spring cropping than any other intervention due to the associated benefits for weed/pest control and soil health; this follows that farmers were most willing to implement stubble/spring cropping of all the studied interventions. Further exploration of the reasons underlying preferences could aid understanding of variation in preferences across farmers whilst also identifying ways to increase uptake; for example, longer-term contracts for sparing interventions may alleviate concerns about cultivating land after contracts end.

Knowledge of the distribution of farmers' WTA payment to participate in land-sparing and land-sharing schemes, which we have uncovered here for arable farmers in England for the first time, can help policymakers develop more effective and more efficient AES policies in future. Here, as in much current policy practice in the European Union, we modelled fixed-price schemes. However, particularly if larger environmental targets are sought, governments should consider the substantial savings potentially achieved by offering variable, rather than fixed, pricing (Ferraro 2008; Armsworth et al. 2012); although transaction costs and perceptions of fairness may limit how far this can be pursued (Latacz-Lohmann & Schilizzi 2005). A further alternative could see the government deliver land sparing by purchasing land which is then contracted to conservation organisations for management (an idea explored in detail in Chapter 6) However, lack of capital may constrain use of this approach. Such land purchase could also facilitate the otherwise costly spatial agglomeration of spared areas with the potential to deliver greater environmental benefit (Lamb et al. 2016b).

In conclusion, we have demonstrated, for the first time, that farmers are willing to participate in sparing-style schemes such that they deliver more of several environmental outcomes than when the same budget is spent paying farmers to land-share. This contradicts widely held but empirically unquantified beliefs that farmers are not willing enough to spare to make land sparing a viable approach. Current policy debates are arguably the most important in recent history, given the increasingly urgent need for action to slow climate change and mitigate the unfolding biodiversity crisis. This work should prompt policymakers to fully consider the potential for land sparing to deliver the outcomes that sharing-style interventions, which dominate the current policy landscape, have to date generally failed to deliver (Pe'er et al. 2014, 2019; Harris et al. 2019). In doing so, policymakers should seek to add consideration of other scheme costs, such as monitoring, administration, capital payments and lost food production, to this assessment of the payments required by farmers, as we do next in Chapter 4.

Chapter 4: The costs of delivering environmental outcomes with land sharing and land sparing

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 interventions 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 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. 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: land use policy; land sharing; land sparing; environmental economics; choice experiment; agri-environment schemes; biodiversity conservation; carbon emissions

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 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 (Phalan et al. 2011; Kamp et al. 2015; Dotta et al. 2016; Williams et al. 2017; Finch et al. 2019, 2020; Balmford 2021), 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). 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 over 70% of its land surface (Defra 2018b). 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 (Kleijn & Sutherland 2003; Kleijn et al. 2006; Dicks et al. 2014; Batáry et al. 2015; Pe'er et al. 2020), despite public expenditures of €3.2bn/y across Europe (Batáry et al. 2015) and >£600m/y in the UK (RSPB 2020a). 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 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 2018a), but sharing may be more familiar, which may reduce the payments farmers require to participate. Indeed, past criticisms of land sparing have included the unquantified suggestion that farmers prefer wildlife-friendly farming (Fischer et al. 2008). 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 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 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 4.1). We studied 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. We calculated the associated per-area benefit delivered by the in-field sharing, field-edge sharing and sparing options (Table 4.1; Appendix A). In line with evidence of the rapid recovery of birds on previously farmed land restored to natural habitat (Vanhinsbergh et al. 2002; Eglington et al. 2007; Marren 2016), 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 4.1. The sharing and sparing interventions that deliver the environmental outcomes studied and their estimated per-area benefit.

Environmental 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)
	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 [1]	Kindred et al. (2008)
	Field-edge 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 2020); 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, field-edge sharing and sparing option, plus the option not to select any of the contracts (see Figure 4.1 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 D1):

a. 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 field-edge 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]).

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.

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/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 2020); Appendix D). Where appropriate, capital costs were stated to be covered separately and in full.

Figure 4.1. 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 input 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 2019c). 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 D2 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 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.

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 et al.

2001). In summary, our approach to estimating non-compliance, and the cost of delivering targets in spite of it (detailed in Appendix D), 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 (Appendix D). 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 (Appendix D). 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 detract land from less profitable aspects of the rotation. 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; Appendix D).

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 4.2). 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 D2). 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, with the number of participants required for each option to achieve a given outcome driven by the area required to deliver that outcome (Appendix D).

Table 4.2. 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 D2 for other distributional assumptions). Standard errors for mean WTA calculated via bootstrapping. * = significant at 5% level

	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 4.2 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 D1).

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 4.2). Although the average farmer was willing to accept less compensation per hectare for sharing interventions (Table 4.2), 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 4.2b). 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 4.2c&d).

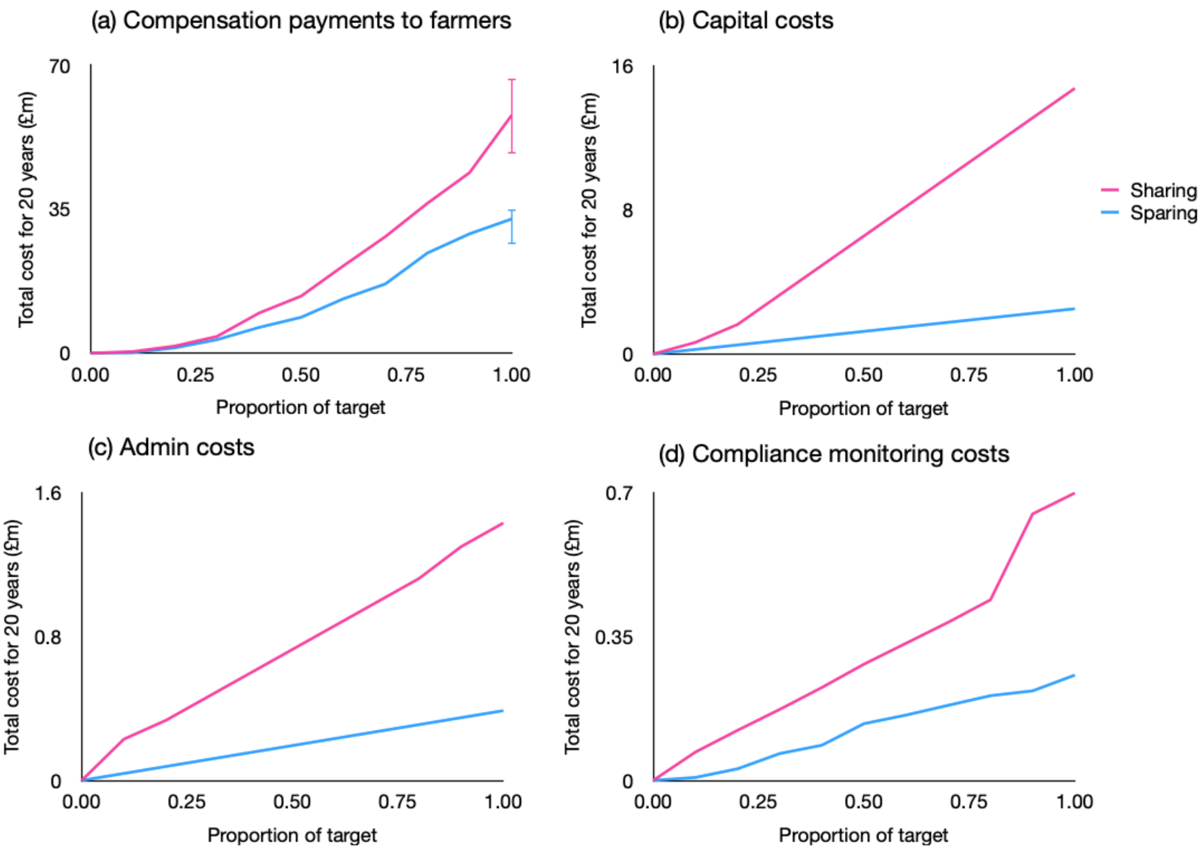

Figure 4.2. The component taxpayer costs of sharing (pink; stubble/spring cropping, 50% reduction in N fertiliser, winter bird seed plots, fallow plots and hedgerow creation) and sparing (blue; creation of scrub, wet grassland and woodland) schemes that delivered varying proportions of the combined target 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 4.2, we found that sparing delivered the target outcomes at 48% of the cost of sharing (Figure 4.3). These taxpayer costs were dominated by compensation payments to farmers (Figure 4.4; 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.

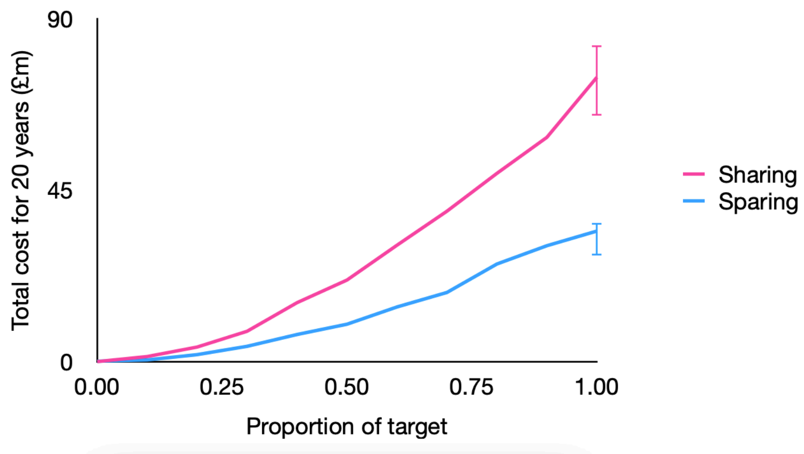

Figure 4.3. The overall costs to the taxpayer (compensation payments, capital, administration and compliance monitoring) of 20-year sharing (pink) and sparing (blue) schemes that delivered a range of proportions of the combined target outcomes of biodiversity and net carbon emissions. 95% bootstrapped confidence intervals reflect uncertainty in compensation payments to farmers; other sources of error exist but were not quantified (see Discussion).

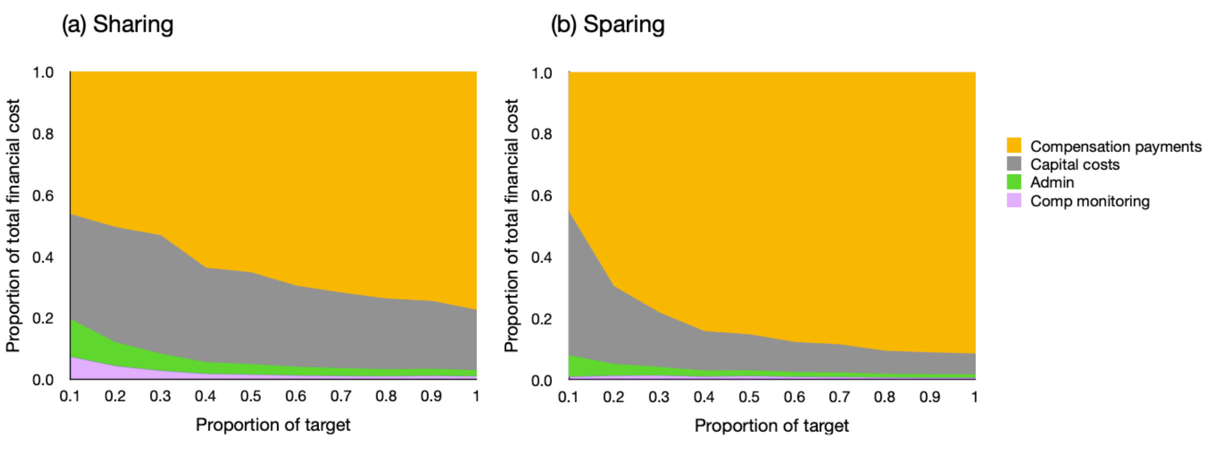

Figure 4.4. The proportion of taxpayer costs of (a) sharing and (b) sparing schemes that delivered varying proportions of the combined target outcomes that were compensation payments to farmers (orange), capital costs (grey) administration costs (green) and compliance monitoring (pink).

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 79% of the food lost in delivering the same outcomes with sharing (Figure 4.5a). This difference is approximately in line with the relative difference in compensation payments to farmers (Figure 4.5b, orange vs black line). The relative difference, between sharing and sparing schemes, was greater for other costs (capital, administration and compliance monitoring; Figure 4.5b, 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 4.5b, red vs black lines).

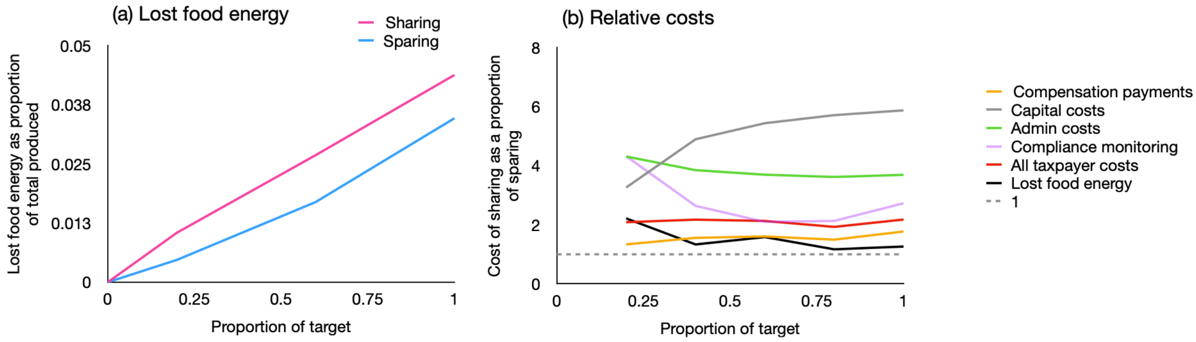

Figure 4.5. (a) The food energy lost, as a proportion of the total produced by the sampled farmers, in delivering the target environmental outcomes with sharing (pink) and sparing (blue). (b) The costs of sharing as a proportion of sparing.

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 79%). To our knowledge this is the first evidence that sparing schemes cost the taxpayer less than sharing schemes to deliver the same environmental 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 schemes 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 (Appendix D). 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 D6). 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. 2016b); however, such an analysis would also have to consider the financial incentives needed to encourage spatial coordination (Liu et al. 2019; Banerjee et al. 2021). 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 D4). This underscores the huge efficiency gains generated by using sparing rather than sharing interventions to reduce net carbon emissions, particularly at higher targets (Figure D5).

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 (Chapter 5). 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 the long-held belief that farmers prefer sharing (Fischer et al. 2008) . 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 (Chapter 2; see also 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 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 D2). 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 D3a), 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 1.3x 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 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) (Appendix 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.

Next, this thesis will further examine the distribution of the payments required by farmers to share and spare with exploration of how much of their WTA is to cover lost gross margin and, based on the distribution of WTA across farmers, explore the savings that could be achieved by using fixed, rather than variable, pricing.

Chapter 5: Understanding what farmers seek in agri-environment payments and how to make payments more cost-efficient

Abstract

Action to reconcile food production and environmental conservation in Europe relies on farmers to voluntarily participate in agri-environment schemes (AES) that typically offer a fixed-price payment in return for management actions that reduce production. Fixed pricing is inefficient because farmers vary in the payments they are willing to accept (WTA), both due to differences in lost gross margin and in other factors that may increase or decrease WTA. For the first time, we studied the relative contribution of a farmer's lost gross margin to their WTA and quantified the WTA net of lost gross margin, which we termed the farmer's residual requirement. We compared spend on lost gross margin vs this residual requirement, and estimated the costs added by fixed compared with variable pricing, where each farmer is paid their WTA. We focused on schemes that delivered the same environmental outcomes (bullfinches, lapwings, yellowhammers, and reduced net carbon emissions) through either the currently predominant land-sharing approach of wildlife-friendly farming, or the alternative land-sparing approach where farmers are paid to create or maintain large areas of habitat for nature. To do so, we studied a distribution of farmers' WTA payments for sharing and sparing interventions obtained from a choice experiment conducted with 118 English arable farmers in 2019/20. We found, firstly, that the differences between farmers in their WTA compensation for sharing and sparing interventions was shaped more by the payment they required above lost gross margin, i.e. their residual requirement, than by their lost gross margin. Second, for all of our target outcomes, we found that less was spent compensating the residual requirement in sparing compared to sharing schemes. Relatedly, third, for the same spend on lost gross margin, we found less was almost always spent on the residual requirement under sparing than under sharing. Fourth, because our farmers varied widely in their WTA, we found variable pricing did offer substantial savings for the Exchequer; however, for most of our outcomes, scheme costs differed more according to whether a sharing or sparing approach was taken than whether the scheme used fixed or variable pricing. Our results suggest that, irrespective of pricing structure, both farmers and taxpayers may prefer a land-sparing approach in AES designed to deliver ambitious biodiversity and climate mitigation targets in the UK.

Introduction

Publicly funded agri-environment schemes (AES) constitute the main approach to reconciling biodiversity loss and food production in Europe (Batáry et al. 2015). Climate mitigation is also a growing focus of AES given the importance of farmland-based sequestration to national net-zero targets (Committee on Climate Change

2020; Pe’er et al. 2020). AES offer a payment for a given action which farmers can voluntarily choose to accept. In Europe, most compensation is offered for so-called wildlife-friendly farming (i.e. land-sharing interventions) which typically reduce farm yields (Green et al. 2005). Much research suggests that the alternative land-sparing approach, where high-yield farming is combined with the sparing of large areas for nature elsewhere within the landscape, would deliver more biodiversity and carbon sequestration for a given level of lost food production (Phalan et al. 2011; Gilroy et al. 2014; Lamb et al. 2016a; Dotta et al. 2016; Williams et al. 2017; Finch et al. 2019, 2020, 2021; Feniuk et al. 2019). However, investment into sharing continues, partly due to the entirely untested assumption that farmers are less willing to adopt land sparing (Fischer et al. 2008).

Farmer attitudes towards AES are reflected in the payment they are willing to accept (WTA) to voluntarily participate. Meeting farmers’ WTA is the major taxpayer cost of AES. We know that WTA varies dramatically across farmers (e.g. Kuhfuss et al. 2016). This presumably reflects, at least in part, differences in the value of the production forgone in implementing the action (i.e. the farmer's lost gross margin), which we would expect to vary across space, as yields and profits do. Variation in WTA may also arise from differences in the additional compensation farmers require to cover other aspects of implementation, such as the associated time and labour, the changed look of the land, and transaction costs (reviewed in Lastra-Bravo et al. 2015; Pavlis et al. 2016; Wilson and Hart 2016). It is also possible that WTA may be reduced (potentially even below lost gross margin) if there is some benefit associated with the action, or because AES payments offer an appealing guaranteed income source to typically risk-averse farmers (Iyer et al. 2020). Despite much interest in the factors influencing AES participation, no study has sought to explore the relative contribution of lost gross margin to WTA i.e. to quantify WTA net of lost gross margin (which we term farmers’ residual requirement). We suggest that this residual requirement is an appropriate measure of farmers' underlying preferences for sharing and sparing, given that it nets out forgone returns from farming. More positive residual requirements may be seen where farmers feel there is more risk associated with implementing the agri-environment option or where it is deemed to be inconvenient or visually unappealing. A negative residual requirement may indicate a farmer who is willing to take on the net costs of enhancing the environment.

Beyond simply quantifying the residual requirement, we are interested in the relative contribution of lost gross margin and the residual requirement to WTA. Governments justify spending taxpayer money on AES with the reasoning that compensation covers the value of forgone production (Defra 2022). It follows that taxpayers may be less keen on interventions for which farmers require compensation well above their lost gross margin. Moreover, the relative contribution of lost gross margin and residual requirement to WTA may differ between sharing and sparing schemes. Lost gross margin is typically greater per unit area under sparing, since all production is forgone whilst under sharing the land is not entirely removed from production, instead it continues to be farmed in a wildlife-friendly way. Attitudes affecting residual requirement may also differ: sharing is perhaps more familiar, but the more concentrated spatial scale and greater permanency of sparing may be attractive given declining farm profitability and uncertainty in forthcoming policy changes. Here, for the first time, we explore whether the relative contribution of lost gross margin and residual requirement differs between sharing and sparing schemes that deliver the same environmental outcomes.

Whether schemes used fixed or variable pricing adds a third element, in addition to lost gross margin and residual requirement, to the costs of paying farmers. Under fixed pricing, all participants are paid at the rate required by the least-willing participant (i.e. the highest fixed price sufficient to get sufficient engagement to satisfy targets). The costs of fixed, relative to variable, pricing will thus increase where WTA is more different amongst farmers, and where more participants, and therefore participants with a greater range of WTAs, must be recruited into a scheme. Given that land-sparing schemes deliver more benefit per unit area (Appendix A), they typically require fewer participants (Chapter 4), so whilst variable pricing will still offer savings under sparing, there is likely less difference in costs compared to sharing. the relative savings available from variable pricing may be lower compared to sharing. Moreover, the relative composition of WTA is important here too: variable pricing may be considered fairer if WTA largely covers lost gross margin, for which it could be argued that all farmers deserve full compensation (Armsworth et al. 2012). However, this argument may be less valid if WTA is dominated by farmers' residual requirements, for which differential compensation may be harder to justify. In any case, governments may be unlikely to pursue variable-price schemes given the high implementation costs (Ferraro 2008) and the resultant disproportionate payment cuts to farms in the least profitable areas (Acs et al. 2010).

To address these issues, we conducted a novel investigation into the distribution and makeup of farmers' WTA compensation for implementing a suite of sharing and sparing interventions. We sought to (i) characterise WTA as the compensation required to cover lost gross margin and farmers' residual requirements; (ii) compare the cost of compensating the residual requirement for sharing vs sparing schemes that deliver the same environmental outcome; (iii) compare the residual requirement when the same value of gross margin is lost to sharing vs sparing; and (iv) explore the savings offered by delivering the same environmental outcomes with sharing and sparing schemes using variable vs fixed pricing. We based our assessment on estimates of farmers' WTA compensation to implement sharing and sparing interventions derived from a choice experiment conducted in 2019/20 with 118 farmers in England and bordering areas in Wales. The choice experiment assessed farmers' WTA payment to undertake contrasting actions that delivered a common set of environmental outcomes: increased populations of yellowhammers, bullfinches, lapwings, and reduced net carbon emissions.

Methods

We wanted to compare the distribution and composition of the payments required by farmers to undertake sharing vs sparing approaches to delivering the same environmental outcomes. To do this, we first established environmental outcomes that can be delivered by both approaches using the literature and in consultation with relevant experts. We decided to study three bird species which differ in their responses to farming: Yellowhammer (*Emberiza citrinella*), Eurasian Bullfinch (*Pyrrhula pyrrhula*) and Northern Lapwing (*Vanellus vanellus*), which decrease in that order in their abundance on farmed land relative to that on (semi-)natural habitats (Newson et al. 2005). Given the importance of farmland as a carbon sink in the UK's plan to meet net-zero targets (Committee on Climate Change 2020), we also studied interventions that reduce net carbon emissions. Next, we used the literature to identify sharing and sparing interventions that delivered these target outcomes on arable farms in the UK. Given potential differences in farmer attitudes, we divided sharing interventions into two types: field-edge, which sees the addition of habitat to farmed land that does not produce food (e.g., hedgerows), and in-field, which sees changes to practices that do produce food (e.g., post-harvesting stubble retention). Sparing required farmers to create and maintain areas of (semi-)natural habitat up to 50ha in size. For each intervention we assessed the per-area benefit it delivered in terms of the additional number of birds or tonnes carbon per year. We did this by systematically reviewing the white literature, and thoroughly searching the grey literature, to identify all relevant estimates, from which we calculated the mean effect of each intervention on the relevant outcome(s) (Table 5.1; full details in Appendix A).

Table 5.1. The sharing and sparing interventions that deliver the environmental outcomes studied and their estimated per-area benefit (expanded in Appendix A).

Environmental outcome	Intervention type	Intervention	Benefit *(birds/ha or tC/ha/y)*	Source
Yellowhammers	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)
Bullfinches	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)

Lapwings	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 [1]	Kindred et al. (2008)
	Field-edge sharing	Hedgerows	1.84	IPCC (2019)
	Sparing	Woodland	3.77	Falloon et al. (2004)

Estimation of farmer WTA

To estimate the payments farmers required to implement these interventions we used the results of a choice experiment (Chapter 4). This estimated farmers' WTA payment to implement these sharing and sparing interventions (Table 5.1) across a range of areas set to be achievable on the average arable farm in the UK (up to 50ha for in-field sharing and sparing; up to 8ha for hedgerow creation and up to 20ha for other field-edge sharing options) and contract durations. Delivered online, using Qualtrics, the survey ran between September 2019 and June 2020 and was completed by 118 farmers in arable England, and bordering areas in Wales, who were responsible for farming 1.7% of England's lowland arable farmland. Participants were informed that capital costs would be covered separately, and in full (the methods for their estimation can be found in Chapter 4). Since we wanted to explore preferences at an individual, rather than group, level, we analysed this choice data with a mixed logit model (see Chapter 4). Mixed logit modelling uses a maximum likelihood approach to estimate the parameters that describe each participant's sensitivity to each of the attributes studied in the choice experiment (type of intervention, area, duration, payment rate; Train 2009). The ratio of each parameter to the payment rate parameter illustrates the additional compensation required for the individual to accept a marginal change in that parameter. We used this relationship, known as the marginal rate of substitution, to estimate each participant's WTA payment to accept a contract that requires the implementation of a given intervention over a stated area and duration. Throughout this thesis, we have examined the delivery of quantified target outcomes; here we again consider the baseline target outcomes of delivering an additional 300 yellowhammers, 300 bullfinches, 300 lapwings and 1557 tC/yr across our 118 study farms.

Estimation of lost gross margin

Next, we estimated how much of the payment farmers were WTA was to compensate the gross margin lost in implementing each intervention using information gathered in the choice experiment survey. We asked farmers about the crop/livestock types they produced as well as the associated areas, yields, selling prices and variable costs for the 2018 harvest year. Participants provided an amalgamated value of variable costs which included fertiliser, crop protection, seed, water and haulage (where relevant), as well as reporting fertiliser use and fertiliser costs separately. Any missing information was completed using averages from the Farm Business Survey (FBS 2020). In summary, we found lost gross margins by multiplying lost production by its selling price and subtracting the associated variable costs (following FBS 2020). In estimating lost production we assumed that yields vary within and between fields and, where possible, that farmers would implement interventions on the least profitable parts of the farm (see below). We also recognised that crops/livestock are rotated such that interventions cannot only take land away from the least profitable crop/livestock type, unless the interventions can also be moved each year. Where relevant, we also added any implementation costs to our lost gross margin estimates; here we included only the cost of materials, and not the associated labour (which is instead captured in farmers' residual requirements). The full methods for each intervention are described in detail below.

i. In-field sharing: Stubble retention followed by spring cropping

We specified to farmers that, in line with expert guidance, this intervention can only be applied to wheat, barley and/or oats (RSPB 2022). We assumed the area would be met by taking area, in order, from the barley, oats and then wheat grown by the farmer (based on the order in which least gross margin is lost, on average, in data reported in the FBS (2020)). We estimated the gross margins of winter-sowing these crops according to the standard method of multiplying the participant's reported production (yield x area) by selling price and subtracting their variable costs (FBS 2020). Winter-sown crops typically yield more, and have higher gross margins, than spring-sown crops (FBS 2020). We assumed that participants' winter-sown gross margins would be reduced by switching to spring-sowing in line with average national trends. So, using the FBS (2020), we found the average spring-sown gross margin for each crop type as a proportion of the average winter-sown gross margin for the same crop type. Then, for each participant, we multiplied their winter-sown gross margin by this proportion to estimate their spring-sown gross margin. In this way, we estimated lost gross margin as the difference between gross margins for winter- and spring-sown crops.

ii. In-field sharing: 50% reduction in use of N fertiliser

In the choice experiment this intervention required farmers to reduce their current use of inorganic nitrogen fertiliser by 50% on wheat, barley, oil seed rape, sugar beet and/or potatoes. Participants were asked to state their current fertiliser use, either per tonne of product or per unit area, in the survey. Since this option can be rotated each year, we assumed farmers would choose to enrol their crop(s) with the lowest gross margin. We established the relationship between fertiliser use and yield using Kindred et al. (2008). This paper studied only winter wheat, and average fertiliser rates differ across crops, but no comparable studies were available for other crops. To allow us to make inferences for the other relevant crops, we first took the relationship between yield and fertiliser application rate for winter wheat from Kindred et al. (2008; Figure 5.1a). We calculated the fertiliser application rates as a proportion of the FBS (2020) national mean application rate for winter wheat. According to this relationship, we then estimated the yield that would be lost by reducing fertiliser use by 50%. We plotted this new yield that is realised with 50% reduced fertiliser use, as a proportion of the initial yield (i.e. with 100% fertiliser application), against the initial fertiliser application rate, before it was reduced by 50% (Figure 5.1b). Second, for each participant and crop type, we found their stated fertiliser application rate as a proportion of the mean reported by the FBS (2020) for that crop. Third, given this fertiliser use, we used Figure 5.1b to predict their yield when fertiliser use is reduced 50%, as a proportion of their initial yield. We multiplied their initial yield by this proportion to find their new yield at the 50% lower application rate. Then, we calculated lost gross margin by multiplying the difference in yield by the area enrolled in our scheme and their selling price, assuming it was unchanged, before subtracting the variable costs assuming 50% lower spend on fertiliser.

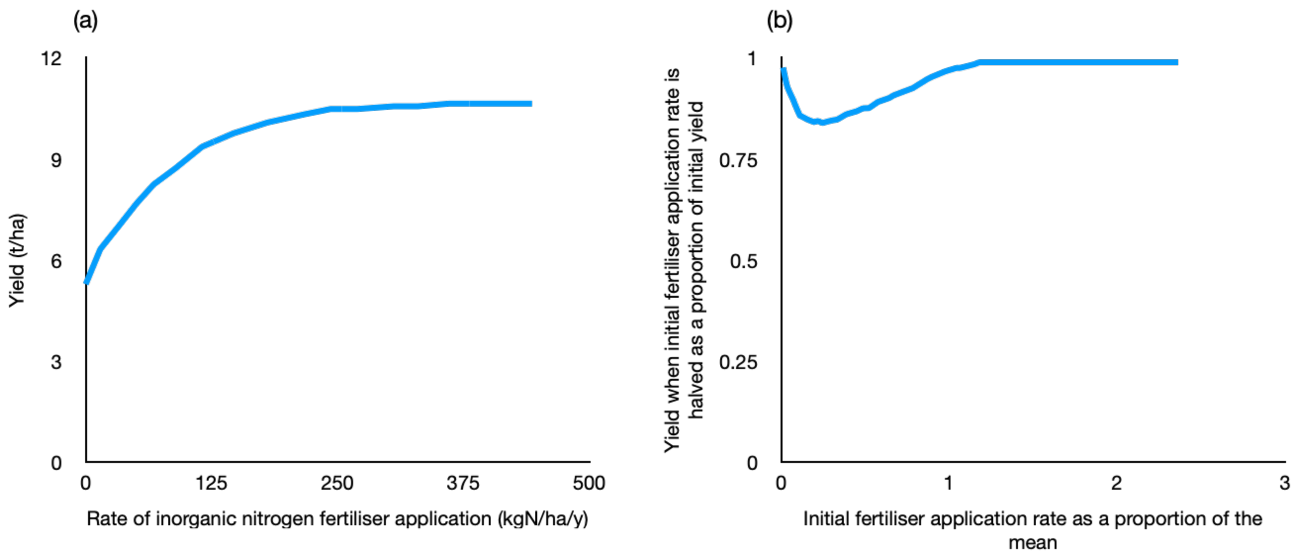

Figure 5.1. (a) From Kindred et al. (2008), the yield associated with varying rates of inorganic N-fertiliser application to winter wheat . (b) Derived from (a), winter wheat yield (as a proportion of the initial yield) when the fertiliser application rate is halved, plotted against the initial fertiliser application rate (as a proportion of the 2018 mean fertiliser application rate reported by FBS (2020) for winter wheat).

iii. Field-edge sharing: Winter bird seed and fallow plots

The choice experiment stated that fallow and winter bird seed plots must be created in arable areas. Since these plots can be moved each year, we assumed they would only be implemented in the crops with the lowest gross margins, though to ensure that plots are well spaced out, we assumed no more than a quarter of each crop type could be plots (as recommended by RSPB (2021)). Because we therefore assumed plots would be located in the least profitable crops as they get rotated around the farm each year, we did not consider between-field variation to be important (as we do below for hedgerows and sparing below). It made sense to allocate plots to the least profitable crops, rather than to the least profitable fields, because our data suggested that crops varied more in their gross margins than fields. We did, however, assume the plots would be allocated to the least profitable parts of fields, as described below.

We assumed farmers would allocate plots to the lowest-yielding, and therefore least profitable, parts of fields first, and that the amount of this more marginal land would scale with the size of the farm. We explored the cumulative production, as a percentage of the total field production, that would be lost by taking incrementally larger areas of the field out of production in Figure 5.2, given likely variation within fields. We plotted the production lost against the area taken out of production according to the yields measured in 5x5m patches across a field reported by Muhammed et al. (2016), plotting the lowest-yielding 5x5m patches first, and the highest-yielding patches last (blue line). This showed that cumulative production is not directly proportional to field area: some parts of a field yield far less than others. However, we assumed that not all the lowest-yielding 5x5m patches would be adjacent to each other such that in implementing a fallow or winter bird seed plot, some higher-yielding land would unavoidably be taken out of production. We assumed that 2/3rds of the area for the plot would be the lowest-yielding patches, and 1/3 would yield the average field-wide yield: we again plotted this relationship in Figure 5.2 (green line). Then, to estimate the production lost by participants, we calculated the area of plots they implemented as a percentage of the area of the least profitable crop(s) in which we assumed they were established. We used the green-line relationship in Figure 5.2 to establish the production lost on the crop area taken out by the plots, as a percentage of the total field's production. We multiplied this by farmer's reported yield, and the area of plots, to find the lost production from the area enrolled in our schemes. We assumed that all yield was lost where plots were established, with no yield lost in surrounding areas. We estimated the value of this lost production using farmers' reported selling prices and variable costs, assuming these did not vary across the field. Last, we allowed allocation to move to the next least profitable crop if less gross margin was lost by adding to the worst parts of those fields, rather than continuing to use more land in the fields growing the least profitable crop. We estimated lost gross margin in this way for fallow plots and for winter bird seed plots where we also added the cost of seed, which Nix (2018) reported at £48/ha/yr.

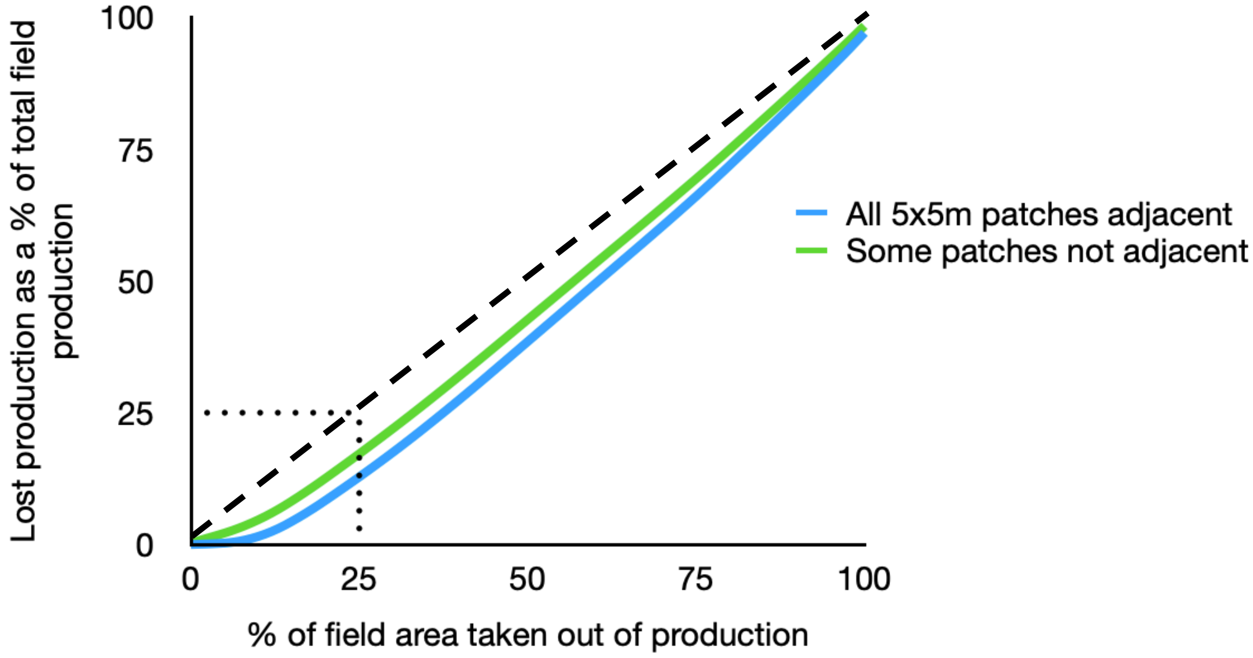

Figure 5.2. The cumulative production lost by taking incrementally larger proportions of a field out of production when assuming (a) that yields vary according to Muhammed et al. (2016) and blocks of only the lowest-yielding patches can be taken out of production (blue line) and (b) that only two-thirds of blocks can be established in the lowest-yielding patches, and one-third established in areas supporting the field-average yield (green line). We assumed interventions could only take-up one quarter of field area, so the dotted box shows the part of the graph relevant to our calculations. The relationship shown by the dashed black line would be seen if yields did not vary within fields.

iv. Field-edge sharing: Hedgerow creation

In the choice experiment farmers were told that hedgerows could only be created in arable areas (since this affected the carbon sequestered; IPCC 2019). In creating hedgerows, we assumed all production was lost in the area occupied by the hedge and some production was lost in the area surrounding the hedge due to shading (Raatz et al. 2019); we deal with each loss in turn below.

We estimated the production lost to the hedge based on it being 6m wide. To allow for crop rotation, and given the hedge cannot be moved each year, we assumed this area would come from all crop types, weighted by their relative areas. To minimise lost gross margin, we assumed farmers would create hedges in the lowest-yielding fields. We established how yields vary between fields using data from Muhammed et al. (2016) who reported wheat yields for a number of fields across one farm. We plotted the production across the whole farm against farm area, assuming that fields did vary in their production potential according to Muhammed et al. (2016; Figure 5.3). In applying Figure 5.3 to our data, first we assumed all crops/livestock were rotated around all the fields on the farm, where the relative areas of each crop/livestock remained the same as in the year we collected data. Second, we calculated the area of fields that would be bordered by hedgerows, assuming fields were 9ha in area (Marshall et al. 2006) and square. Third, we calculated this area of fields as a percentage of the total farm area and used Figure 5.3 to estimate the percentage of the production that would be lost assuming the hedgerows were allocated to the lowest-yielding fields. In addition, we assumed the edges of fields, where hedges were located, would yield less than field centres. As for fallow and winter bird seed plots, we used the dimensions of the hedge and Figure 5.2 (green line) to estimate the difference in yield between where the hedge would be implemented and the field-average yield. We thus assumed that the production lost was from the area occupied by the hedge from all crops, weighted by their relative areas, and adjusted for both between-field and within-field yield variation. We used farmers' reported crop-specific selling prices and variable costs to estimate the gross margin associated with this lost production, assuming crop-specific selling prices and variable costs did not vary within or between fields.

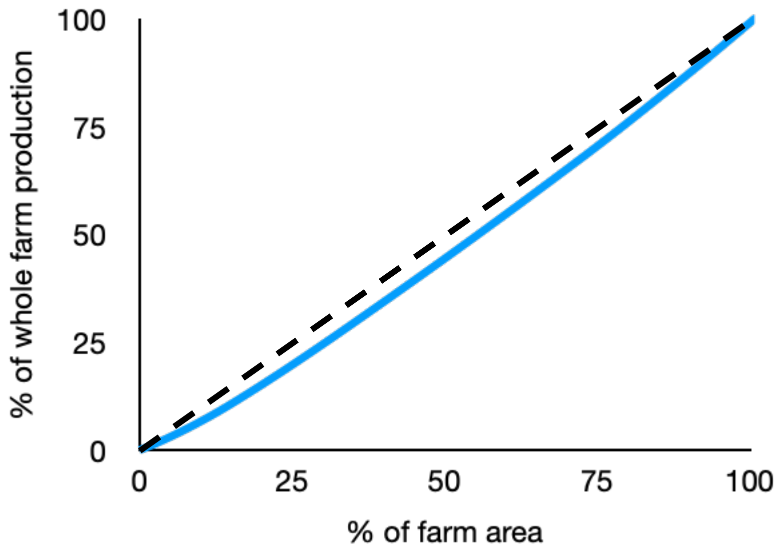

Figure 5.3. Cumulative production, as a percentage of total production, across the whole farm plotted for incrementally larger proportions of the total farmed area assuming that yields vary between fields (blue line). In plotting this, we averaged across the rotation assuming all crops/livestock were rotated around all fields (derived from Muhammed et al. (2016)). Only between-field, and not within-field, variation is included here. The relationship shown by the dashed would be seen if yields did not vary across the farm.

Hedgerows not only cause production to be lost from the area they cover: yields also decline in surrounding areas due to shading. We calculated the production lost from hedgerow shading based on Raatz et al. (2019) who found yields adjacent to hedgerows were 14.5% lower, on average, than the in-field yield up to 17.85m from the field edge. However, that the field-edge yielded less than the middle was not wholly attributable to the hedgerow: due to edge effects, in a field without a bordering hedgerow, 7.8% of yield was estimated to be lost up to 6.93m into the field. So we assumed the production lost to shading from the hedgerow was the difference between these two estimates. However, this was based on full-size hedgerows. Therefore, we assumed that the full shading effect would not be realised until year 7 (based on a growth rate of 0.6m/y and that the full effect occurs when the hedges are 4m high (Raatz et al. 2019)) with yield lost to shading increasing linearly in years 1-6. Furthermore, we assumed half of hedges would border one field, and half of the hedges created would border two fields and thereby shade twice as much crop. Given these estimates of yield loss to shading and the area of each crop affected by shading for each year of the 20-year scheme, we used farmers' reported selling prices and variable costs to estimate the value of this lost production. Finally, we estimated lost gross margin by adding the value of production lost to shading to that lost to creation, and added in average annual maintenance costs of £40/ha/y (Nix 2018).

v. Sparing

Sparing results in total loss of gross margin on the land taken out of production. Given rotation, we assumed land for sparing would come from all crops/livestock, weighted by their relative areas. We also assumed that fields vary in their profitability, and that the least productive fields would be spared first; within-field variation was not relevant here since the whole field is taken out of production. We used the extent of yield variation across fields again based on Figure 5.3 (blue line; derived from Muhammed et al. (2016)). Using this relationship, we adjusted the production of the relevant crop/livestock types lost according to the proportion of the farm that was spared. We used this to estimate the crop/livestock-specific tonnes of product lost and calculated the output forgone given participants' reported selling prices. We assumed variable costs did not differ across fields, and therefore used the reported mean variable costs for the relevant crop/livestock types. We also added the annual maintenance costs associated with materials for the spared habitats in our estimates of lost gross margin. These were estimated, in 2018 GBP, at £0/ha/y for scrub (given restoration typically involves natural regeneration; RSPB 2020) , £50/ha/y for woodland (based on Nix 2018) and £50/ha/y for wet grassland (based on Ausden and Hirons 2002).

Estimation of the residual requirement

The above estimations of lost gross margin allowed us to calculate the difference between WTA and lost gross margin, which we termed farmers' residual requirement. To explore the relative spend on this residual requirement between sharing and sparing schemes, we simulated fixed-price schemes that delivered varying amounts of our target outcomes (estimated using the per-area benefit estimates reported in Appendix A). In these schemes, participants with the lowest WTA were enrolled first; larger targets were met either by existing participants enrolling more land or with more scheme participants. Knowing the overall enrolment needed to meet our target, we then quantified how much of the compensation required by the participants (i.e., their combined WTA) covered lost gross margins vs residual requirements.

Estimation of the additional costs of fixed pricing

To explore the relative merits of fixed vs variable pricing we first estimated the total costs of our sharing and sparing schemes when participants were all paid the same rate. On top of WTA payments, we included capital, compliance monitoring, and administration costs in these estimates (for which the methods can be found in Chapter 4). Next, we estimated the costs of the same schemes run instead with variable pricing, assuming each enrolled participant was paid at their WTA. The difference in cost between schemes that delivered the same outcomes with fixed vs variable pricing is what we term the additional cost of fixed pricing.

Results

1. Characterising WTA as lost gross margin and residual requirement

In Figure 5.4, we ranked our 118 participants in order of their WTA payment (grey line) to implement sharing (top two rows, except woodland creation for bullfinches) and sparing (bottom row) for 20 years and over the maximum areas studied in the choice experiment (50ha for in-field sharing and sparing; 8ha for hedgerow creation and 20ha for other field-edge sharing options). We plotted individuals participants' WTA against the cumulative outcome delivered by the area enrolled by each participant.

We found WTA varied dramatically (Figure 5.4). Breaking down participants’ WTA values into lost gross margin (blue) and residual requirement (green) showed that for all but the most-willing farmers (with the lowest WTA), farmers' residual requirement exceeded their lost gross margin. Therefore, when ranking participants in order of their WTA, the distribution of WTA was shaped more by differences in residual requirement than lost gross margin. In Figure 5.4 it is also evident that sparing could typically deliver our baseline targets (black lines) with far fewer participants compared to sharing; hence the black lines in Figure 5.4 are substantially further to the left of the x-axis in the bottom, vs other, rows.

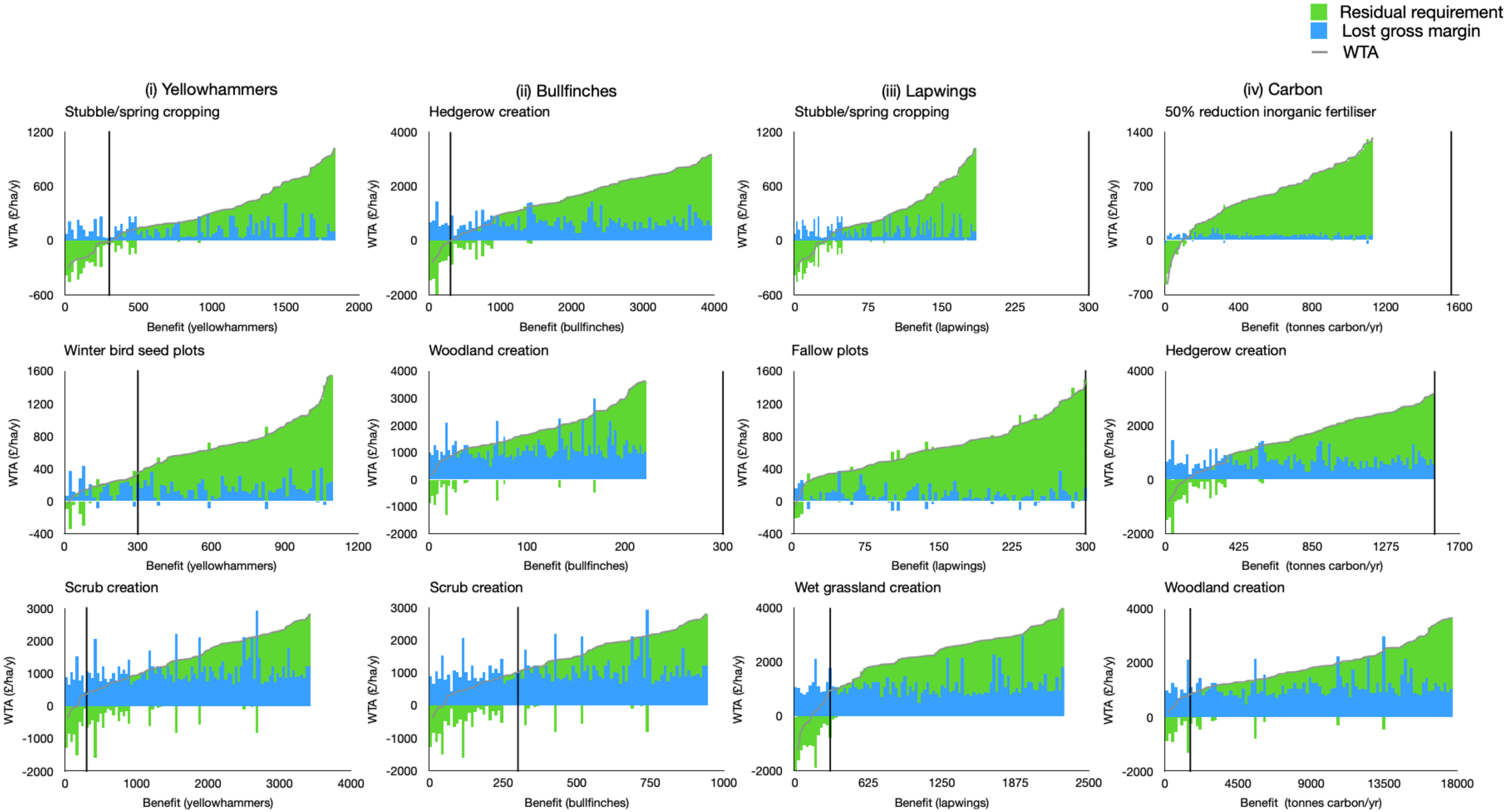

Figure 5.4. Surveyed participants' willingness-to-accept (WTA; grey line) compensation, split into that to cover lost gross margin (blue bar) and the residual requirement above lost gross margin (green bar) to participate in 20-year schemes that deliver increased populations of (i) yellowhammers, (ii) bullfinches, (iii) lapwings and (iv) reduced net carbon emissions. Note x- and y-axis scales differ between plots. Vertical lines show baseline target outcomes: 300 yellowhammers, 300 bullfinches, 300 lapwings, and 1557 tC/yr.

2. Costs of farmers' residual requirements

We simulated schemes that offered a fixed payment rate and, based on the area enrolled by each of our 118 participants at that payment rate (Chapter 4), estimated the amount of our environmental outcomes delivered. As we increased the offered payment rate, more participants enrolled more land, and so the outcome delivered also increased. We ran simulations of fixed price schemes offering successively higher payment rates until all farmers were enrolled; for some interventions, this delivered outcomes far in excess of our baseline targets. Then, across our simulations, we estimated the absolute spend to meet the residual requirement of all participants and plotted this for incremental increases in the outcomes delivered (Figure 5.5a-d; for comparability, we calculated the delivered outcomes as a proportion of our baseline target outcomes of 300 bullfinches, 300 lapwings, 300 yellowhammers and 1557tC/yr). Since sparing schemes generally delivered greater benefit per unit area, the total outcome that could be delivered by enrolling all our farmers in sparing schemes was greater; hence in each case the sparing lines extend further along the x-axis.

For all environmental outcomes, we found achieving higher targets required more to be spent compensating farmers' residual requirements (the lines in Figure 5.5a-d generally curve upwards). In delivering the same outcome, we found more was always spent compensating residual requirements under sharing, compared to sparing, across all of the outcomes considered (within a plot in Figure 5.5a-d the lines describing sharing interventions lie above those for sparing interventions).

Next, for the same set of simulations, we plotted spend on residual requirement against the enrolled participants' lost gross margins (Figure 5.5e). We generally found that for the same spend on lost gross margin, more was spent on meeting farmers' residual requirements under sharing, compared to sparing, interventions (sharing curves again lie above sparing curves). This did not always hold at low lost gross margin values for stubble/spring cropping and hedgerow creation, where less was sometimes spent on residual requirements compared to woodland and scrub schemes. Again, the sparing lines extend much further along the x-axis, this time because more gross margin is lost when implementing a sparing action, since production is entirely halted (rather than continuing in a wildlife-friendly way).

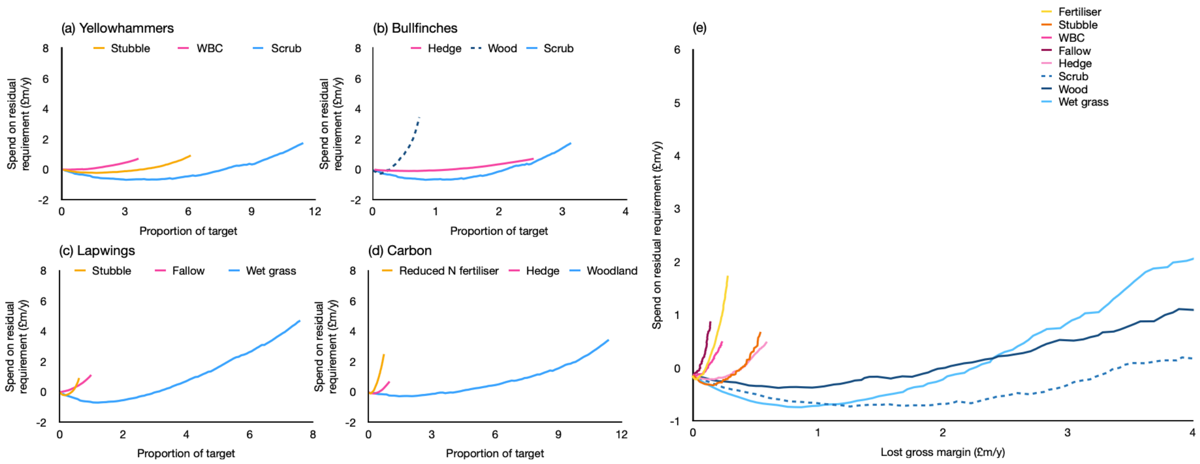

Figure 5.5. Annual spend on the residual requirements (WTA net of lost gross margin) of farmers participating in in-field sharing (orange), field-edge sharing (pink) and sparing (blue) schemes that deliver incrementally more ambitious outcomes plotted as a proportion of our baseline target outcomes of (a) 300 yellowhammers, (b) 300 bullfinches, (c) 300 lapwings and (d) 1557 tC/yr. Plot (e) shows the spend on residual requirement plotted against the gross margin lost by enrolling increasingly less-willing farmers to in-field sharing (orange/yellow), field-edge sharing (pink/purple) and sparing (blue) schemes.

3. Costs of fixed vs variable pricing

In the final analysis we estimated the total costs to the taxpayer (including capital costs, administration costs, compliance monitoring costs and payments to farmers) for delivering a range of our target outcomes via schemes run with fixed- and variable-price structures. As we would expect, scheme costs were always greater with fixed pricing, where all participants are paid at the rate required by the least-willing participant required in the scheme to deliver the conservation target (Figure 5.6, upper bounds for each intervention type) compared with variable pricing, where participants are paid at their WTA (Figure 5.6, lower bounds). The additional costs of fixed pricing (the shaded areas between bounds) were often substantial and increased at higher targets (shown by the widening of the shaded area moving left to right). However, even if sharing schemes for lapwings (Figure 5.6c) and carbon (Figure 5.6d) were run with variable pricing, they would not be cheaper than sparing schemes run with fixed pricing. Similarly, yellowhammers were always delivered at least cost by in-field sharing, regardless of the pricing structure. The only case where the pricing structure changed whether sharing or sparing was cheaper was for bullfinches: we found a sparing scheme ran with variable pricing delivered bullfinches at less cost than fixed-price but not variable-price sharing.

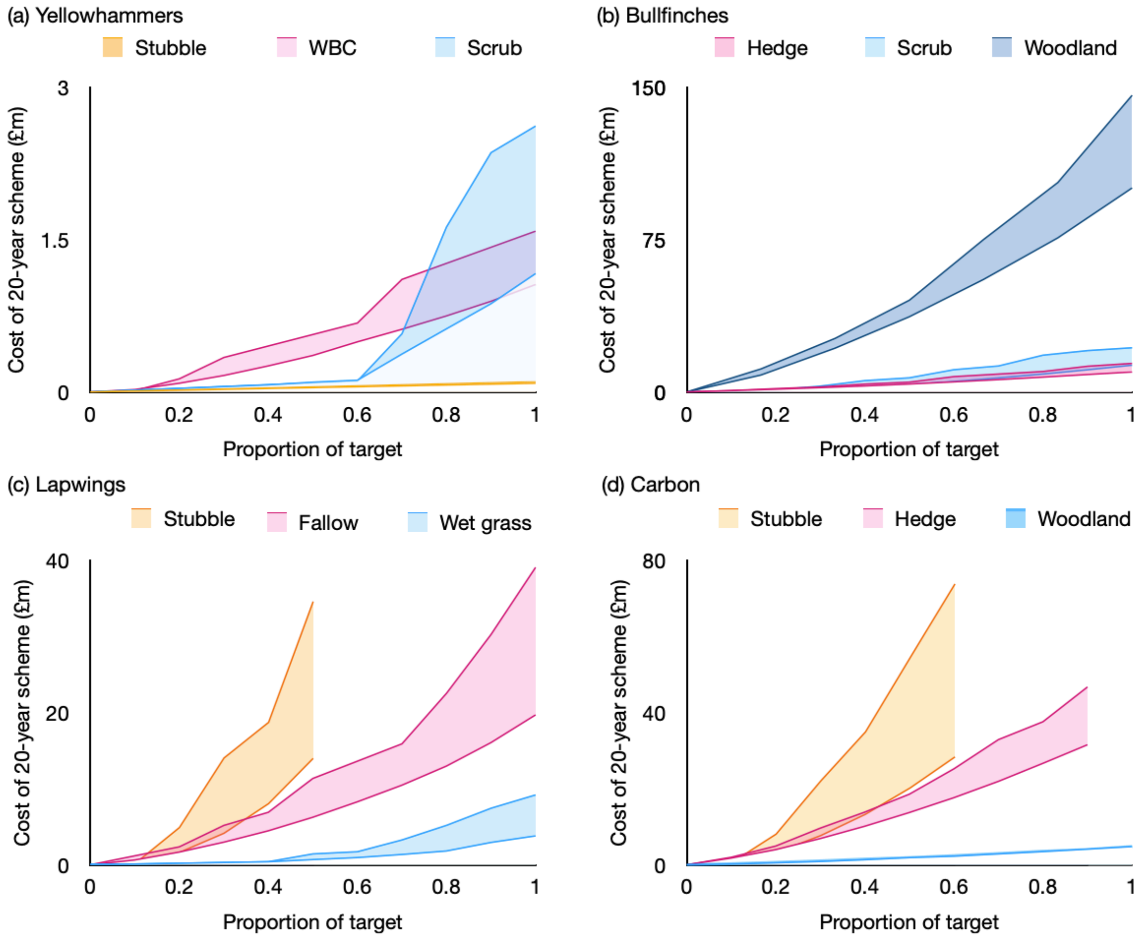

Figure 5.6. The difference in costs of fixed-price (upper bound) and variable-price (lower bound) schemes that deliver varying proportions of (a) 300 yellowhammers, (b) 300 bullfinches, (c) 300 lapwings, and (d) 1557tC/yr with in-field sharing (orange), field-edge sharing (pink) and sparing (blue) interventions. Shading shows the difference in cost between a fixed-price and variable-price scheme (the additional cost of fixed pricing). Lines do not extend across the x-axis if required payments exceeded £2000/ha/yr (which is well above spend on comparable options in current AES).

Discussion

Here we have shown for the first time that farmers vary substantially in their WTA payment to implement sharing and sparing interventions for reasons other than simply the costs of lost production. For schemes that delivered the same environmental outcome, we found that more compensation was always required to meet farmers' residual requirement (the difference between WTA and lost gross margin) under sharing, compared to sparing. Furthermore, when we explored farmers' residual requirements and lost gross margins together, we found that the residual requirement was generally greater under sharing compared to when the same gross margin was lost to sparing (Figure 5.5e). Given the great variation in farmer WTA, we found that fixed pricing does, as anticipated, add substantially to scheme costs. However, the overarching decision of whether to pursue a sharing or sparing approach had more marked consequences for overall AES costs than the pricing structure. Even with variable pricing, sharing could not deliver lapwings or carbon at less cost than a fixed-price sparing scheme, whilst sharing was always cheaper for yellowhammers (the species studied that is most abundant on farmland), regardless of pricing structure. Indeed, the only outcome for which the price structure changed conclusions was bullfinches, where a variable-price sparing scheme was cheaper than a fixed-price, but not variable-price, sharing scheme.

Inevitably, our study is limited. Three key limitations stand out. First, our estimates of lost gross margin were based on information from a single year, but we compared them to the annual payments required by participants to take part in 20-year schemes. This was unavoidable since we considered it unreasonable to ask farmers to enter this large amount of information for more than one year. Inter-annual profit variation may explain why some farmers appeared willing to participate at payment rates below what we estimated to be their lost gross margin. Farmers are generally risk-averse (Iyer et al. 2020), so in deciding whether to participate they may assume their future gross margins to be those of poorer years (whereas in 2018 harvest, whilst yields were down, high prices saw above-average gross margins for major arable crops; Lang 2020). Second, our estimates of the benefit delivered by interventions involve unquantified uncertainty; we could not estimate this since standard errors were missing from many of the studies on which our estimates were based (Appendix A). Third, there is uncertainty in our estimates of the payments that farmers were willing to accept (Chapter 4). Choice experiments offer an approximation of the payments farmers would be observed to accept in real-life situations (Christie & Azevedo 2009). Nonetheless, the participation we predicted at the payment rates of current schemes did match observed participation rates quite well (Figure E1).

Our findings are important to governments seeking to justify the spending of taxpayer money. Taxpayers may find sparing more palatable than sharing since a greater proportion of the compensation required by farmers is for lost gross margin; and we found this difference only increased for more ambitious action. Furthermore, under sparing, since more of farmer WTA is lost gross margin, action to limit spend would have the additional benefit of limiting forgone production (which is predicted by lost gross margin but not by residual requirement) and hence the need to increase food imports. In time, our novel dissection of WTA could facilitate understanding of the drivers that change WTA over time. Whilst lost gross margin would be expected to fluctuate with selling prices, input costs and yield, residual requirement may vary for other reasons, such as changing attitudes to government and to the roles of land ownership. Neither lost gross margin nor residual requirement are necessarily more predictable, but both are important to WTA. s

Perhaps our most important finding is that when we controlled for the costs of implementing sharing and sparing, farmers generally required less compensation to spare than to share. For equal spend on lost gross margin, our schemes always spent more compensating the residual requirement for fertiliser, winter bird cover and fallow schemes, compared to any sparing scheme. This also held for stubble/spring cropping and hedgerow creation for all but the least ambitious schemes (which saw relatively little lost gross margin). This first dissection of farmer preferences thus suggests that, when controlling for forgone returns, farmers can largely be considered to prefer sparing to sharing. This contradicts the untested assumption, prevalent among some critics of sparing, that sparing is unviable because farmers are unwilling to do it (Fischer et al. 2008). It would be inappropriate to justify a continued sharing approach to agricultural policy based on unsupported – indeed apparently incorrect – notions of farmer preferences.

Given the variation in farmer WTA, it is unsurprising that variable pricing offered cost savings. For some time research has discussed variable pricing as a more efficient pricing structure (Armsworth et al. 2012). Variable pricing could also be considered fairer on the basis that farmers deserve full compensation for the costs of forgone production (Ferraro 2008), which differ widely across farms. However, this argument is weakened by our finding that much of the difference in farmer WTA does not appear to be driven by lost gross margin. Variable-price schemes would likely be implemented with auctions, where farmers bid the price they are WTA to participate in the scheme. In such circumstances, their residual requirement may be fickle and increase, for example, on learning that another farmer has been paid more for the same action. This may lead farmers to increase their bids over successive rounds towards the maximum price they know is likely to be accepted, as has been observed in practice (Khanna & Ando 2009).

There is much interest in the pros and cons of variable and fixed pricing. However, governments should perhaps be even more interested in the merits of sharing vs sparing approaches, given that we found this choice had more marked consequences for AES costs than the pricing structure of either scheme. The most efficient policy designs may be those that consider costs and benefits together, e.g., through outcome-based payments (Hanley et al. 2012). Such an approach would likely favour the creation of larger areas of habitat, since these would deliver disproportionately greater conservation benefit per unit area (Lamb et al. 2016b). Given participants required, on average, greater compensation to spare larger areas (Chapter 4), governments could consider paying premiums for existing participants to enrol additional land, rather than recruiting new participants. This style of variable pricing may be justifiable, though it may disproportionately favour large landowners.

In conclusion, for the first time we have examined the contribution of lost gross margin to the payments farmers require to implement agri-environment options. Contrary to apparent government intentions, we have shown that large amounts of the compensation required by farmers is to compensate above lost gross margin; and this is more pronounced for more ambitious targets, and in land-sharing (compared with land-sparing) schemes. In addition, farmers tend to require more compensation to cover their residual requirement for sharing schemes compared to sparing interventions that result in the same lost gross margin. Altogether, we suggest this is evidence that both taxpayers and farmers may prefer policymakers to take a sparing approach to delivering conservation in the UK.

Finally, this thesis will explore the implications of changing yet another aspect of scheme design in comparing the relative cost-effectiveness of delivering sparing with a land-purchase policy approach, rather than paying farmers through agri-environment schemes.

Chapter 6: Comparing the cost-effectiveness of delivering environmental benefits through subsidies to farmers vs land purchase

Abstract

Action to address biodiversity declines and climate change must be taken on farmland, which covers half of all habitable land on Earth. In Europe, governments have predominantly invested in fixed-price, voluntary agri-environment schemes (AES) which pay farmers to change their management to the benefit of the environment. We conducted the first UK-based comparison of the cost-effectiveness of this approach, in terms of the environmental benefit delivered for a fixed spend, with an alternative land-purchase strategy in which we assumed organisations were contracted to manage land for nature that had been purchased by government. To estimate the relative costs of these approaches, we took the novel approach of using the results of a discrete choice experiment conducted among 118 farmers in the UK arable sector to establish the payments they required to create and manage habitat. To establish the costs of land purchase and its subsequent management we used estimates based on the literature. Given an equal annual spend on both strategies, we estimated the benefit delivered in terms of a suite of environmental outcomes (bullfinches, lapwings, yellowhammers and carbon sequestration) and explored how relative cost-effectiveness varied in relation to the total available budget, discount rates and timescales. We found that at budgets in line with current spend, AES that paid farmers to manage land were more cost effective in delivering biodiversity outcomes, provided the evaluated timescale was <50 years. Low discount rates and larger budgets, however, favoured land purchase; and because we assumed climate mitigation would receive more funding than any single biodiversity outcome, a land-purchase strategy was most cost effective in delivering carbon at all timescales and discount rates considered. Sensitivity testing revealed that the effectiveness of the land-purchase strategy was affected by fluctuations in future land prices which, given their inherent uncertainty, presents a challenge to long-term policymaking, which must also consider the impacts on farming communities of large-scale changes in land ownership.

Introduction

Addressing the twin biodiversity and climate crises will require extensive and costly government intervention. Given that agriculture covers half of Earth's habitable land (Ritchie 2019), action to increase the biodiversity and carbon sequestered on currently farmed land is essential. Indeed, governments, particularly in Europe, have invested substantially since the 1980s in a subset of agri-environment schemes (AES) that pay farmers to alter their management practices for the benefit of the environment (Batáry et al. 2015). However, attracting farmers into such schemes requires compensation for, amongst other things, the forgone profit of lost production. As such, it might instead be more cost effective to buy currently farmed land outright and contract organisations to manage it for environmental outcomes. AES have been favoured by European governments over larger-scale habitat creation approaches: between 2007-2013, the Natura 2000 scheme, which creates protected areas, received just 1% of what was invested by the UK, and 10% of what was invested across Europe, in smaller-scale AES approaches (Kettunen et al. 2011). Clearly, a land-purchase strategy involves high up-front costs. These costs may, however, be overcome by relatively cheaper long-term maintenance costs which may arise both because there is no need to pay for forgone production and because organisations contracted to manage the land may achieve economies of scale that farmers cannot.

One way of assessing the cost-effectiveness of these two approaches is to compare the long-run environmental outcomes they achieve when they both receive the same level of annual investment. Because the schedule of environmental benefits delivered by these approaches differs through time, we must consider how to value future benefit. In economics, discount rates are commonly applied to reflect our preference for more immediate gains (Gowdy et al. 2010). The same approach can be deployed when assessing long-run environmental outcomes: the concept of the social cost of carbon, for example, is based on the damage over time of carbon emissions, expressed (using discounting) in present-day terms (Stern 2008). Of the very few studies to have explored the relative merits of ongoing farm subsidies vs land purchase, there is some evidence that the discount rate affects the relative cost-effectiveness of the two strategies we explore here: Curran et al. (2016) found low discount rates favoured a land-purchase strategy given its initial high costs. Since the discount rate may therefore be central to conclusions, it is important to assign a discount rate that is appropriate and relevant to the UK government, though there is much discussion over what this should be (Armsworth 2018).

Conclusions may also change based on available budget and timescale. Larger budgets allow purchase of more expensive patches or, in the alternative AES approach, recruitment of less-willing farmers. Therefore, larger budgets may favour land purchase, where each patch is bought at its true price, compared to fixed-price AES where all participating farmers are paid at the rate required by the least-willing farmer (Schöttker et al. 2016). A land-purchase strategy may also be favoured when outcomes are compared over longer timescales, since this allows for more years of relatively cheaper maintenance costs to outweigh the initial high cost of land purchase (Curran et al. 2016; Schöttker & Wätzold 2018).

In what we believe to be the first UK-based comparison of its kind, we explored the relative cost-effectiveness of a one-off land-purchase strategy and an agri-environment style annual farm-subsidy strategy in delivering biodiversity and carbon sequestration outcomes through habitat creation and maintenance. We estimated the costs of the land-purchase strategy based on the literature and the subsidies required by farmers from a discrete choice experiment (Chapter 4). Combining this with literature-based estimates of the biodiversity and carbon sequestration delivered by our habitat types, we explored how the relative performance of our contrasting approaches varied with total available budget, discount rate, and the timescale over which outcomes were evaluated. Importantly, in using the results of a choice experiment to establish the subsidy payments they require, our study provides the most accurate assessment to date of the outcomes delivered by a range of budgets. This is an important advance on previous studies which have assumed either the subsidy payment rates of current AES (thus limiting potential to make inferences at different budgets) or estimates of the value of lost production (which represent only a portion of the compensation required by farmers; Chapter 5).

Methods

Scenario set-up

To begin, we established our environmental outcomes of interest. We selected three bird species for study which differed in their habitat requirements and response to farming (Newson et al. 2005; Finch et al. 2019):, Yellowhammer (*Emberiza citrinella*), Eurasian Bullfinch (*Pyrrhula pyrrhula*) and Northern Lapwing (*Vanellus vanellus*). Given plans for interventions on farmland to deliver climate mitigation in the UK (Committee on Climate Change 2020), we also studied carbon sequestration. We used existing studies to identify the (semi-)natural habitats that deliver each of these four outcomes in greater quantity than conventionally managed farmland: scrub for bullfinches and yellowhammers, wet grassland for lapwings and woodland for carbon sequestration (Appendix A). From all relevant studies, we found the difference in the delivery of our four outcomes (in terms of birds, or tonnes carbon, per unit area) on the relevant (semi-)natural habitat vs on farmland. We then calculated the benefit delivered in converting a unit of farmland to (semi-)natural habitat by taking the mean of these difference-in-density estimates (Appendix A).

A government could seek to deliver these environmental outcomes either with AES that pay farmers to create and maintain habitat (the "farm-subsidy strategy"), or by purchasing land on which organisations are contracted to create and maintain habitat (the "land-purchase strategy"). We explored the cost-effectiveness of these two strategies, in terms of our four key environmental outcomes, when the same budget was spent on both strategies each year. The outcomes delivered by each strategy depended on how much habitat our budget could afford to create, and then maintain, each year. As described in detail below, for the farm-subsidy strategy, we estimated habitat maintenance costs from a choice experiment (Chapter 4) and habitat creation costs from the literature. For the land-purchase strategy, we estimated the costs of purchasing land according to market prices, and based creation and maintenance costs on the literature. For the same budget spent on both strategies every year, we found the benefit delivered by summing the birds/carbon delivered across all years of the fixed-length scheme (with future benefit discounted according to our choice of discount rate). In our analysis, we expected the relative cost-effectiveness of the two strategies to differ, given the presumed higher initial costs of land purchase potentially offset by lower on-going maintenance costs. Since we also expected that the area under conservation management, and therefore benefits, would accumulate at different rates under the two strategies, we also expected the assumed discount rate and timescale to alter conclusions. Our analysis considered each habitat type separately; complexities not considered here would arise if all outcomes were pursued simultaneously. We did not include the administration costs which we would expect to differ across strategies but to be a small proportion of the overall costs of either strategy. I

Costs: Farm-subsidy strategy

First, we considered the costs of habitat maintenance under the farm-subsidy strategy. This strategy required farmers to voluntarily choose to participate in an agri-environment style scheme that paid a fixed price to all participants, as in most existing AES across Europe (Armsworth et al. 2012). Farmers vary in their willingness-to-accept a payment to enter such a scheme (Ruto & Garrod 2009). Therefore, we used the findings of a choice experiment (detailed in Chapter 3) to establish the relationship between the maintenance payment offered and the area likely to then be enrolled in the scheme across a given population. For arable farmers managing 1.7% of the total area of lowland arable farmland in England, this experiment estimated their willingness-to-accept payment to create scrub, wet grassland and woodland habitats over varying areas and scheme durations (Appendix F; Chapter

4). Using the preferences of the participant farmers revealed by this experiment, we simulated the area enrolled across all studied farmers for a given maintenance payment rate for a range of scheme durations; the area-payment distribution (Figure 6.1 shows the area enrolled for 20-year schemes). The area enrolled increased with the maintenance payment rate both because new farmers entered the scheme and because existing participants enrolled more land (Appendix F). For a given scheme duration and available budget, we set the rate paid to farmers by identifying from the area-payment distribution the maximum area that could be afforded given its cost. For example (black crosses on Figure 6.1), a £100,000 annual budget enrolled 204 ha of woodland in a 20-year scheme at a payment rate of £490/ha/y, whilst doubling the budget to £200,000 enrolled 303ha for a payment rate of £660/ha/y. Since farmers typically required more compensation to enrol in longer schemes (Appendix F; Chapter 4), and we were interested in exploring conclusions across a range of timescales, we repeated these estimations given the payments required by farmers across all studied timescales (established below).

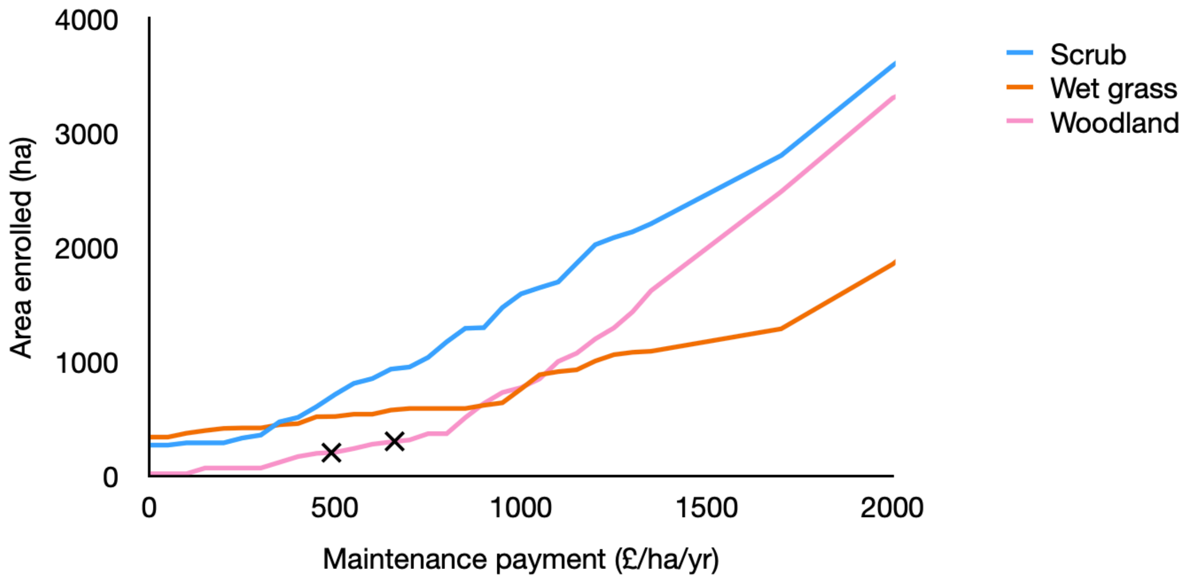

Figure 6.1. Across all studied 118 farmers, the cumulative area enrolled in scrub, wet grass and woodland creation schemes lasting 20 years for a range of maintenance payment rates. Curves were derived from the results of a choice experiment (Appendix F; Chapter 4) which estimated the payments required by farmers to enrol varying areas in schemes of varying duration. Black crosses are for illustration only (see text).

In addition to these habitat maintenance costs, the farm-subsidy strategy also involved habitat creation costs. These creation costs were not reflected in the maintenance payment since participants of the choice experiment were told that one-off capital costs of habitat creation would be covered separately, and in full (Appendix F; Chapter 4). Instead we used the literature to estimate the habitat creation costs associated with planning, infrastructure and labour (Table 6.1).

Costs: Land-purchase strategy

Turning to the land-purchase strategy, we estimated the costs of purchasing land by assuming that our scenarios would purchase the cheapest land first. We also assumed that the benefit delivered per area of habitat was uniform across land regardless of its sale price, and that all land was suitable for creating each of the habitat types studied. We established a distribution of land prices in 2019 GBP according to data from Strutt & Parker (2021; Figure 6.2). To make our two strategies comparable, we assumed the land-purchase strategy to also apply to 1.7% of arable land in England – the area studied in the choice experiment. Therefore, we assumed the area available for purchase each year was 1.7% of the mean arable area marketed annually in England across the last 10 years (Savills UK 2020). Given the trend in land prices through time (Savills UK 2020), we assumed future land prices would increase only with inflation, but explored the sensitivity of our conclusions to steeper increases.

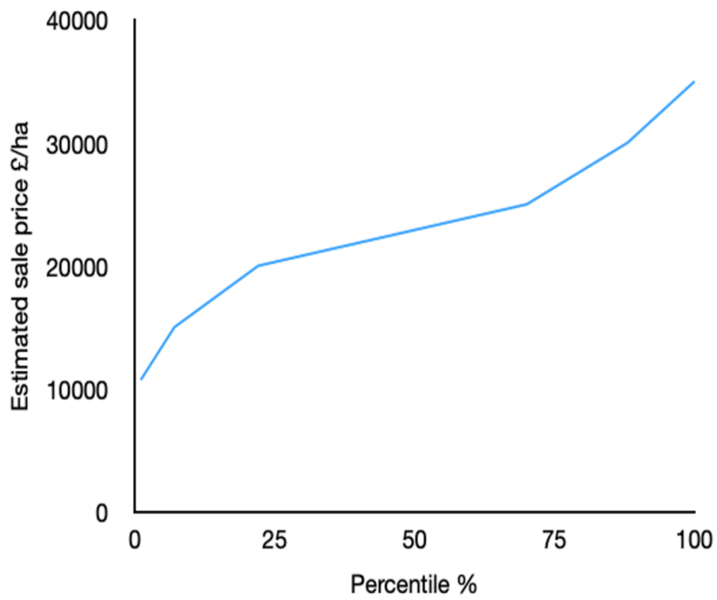

Figure 6.2. The assumed distribution of land prices in the study region based on Strutt & Parker (2021).

Next we estimated the costs of paying contracted organisations to create and maintain habitat on the purchased land. We assumed that the costs of paying contracted organisations to create habitat were the same as for farmers, though we tested the sensitivity of our conclusions to this assumption. However, the costs of maintaining habitat were assumed to differ, both because the payments required by farmers, and not contracted organisations, included compensation for forgone production, and because contracted organisations may have access to economies of scale that reduce maintenance costs. We therefore used estimates of annual maintenance costs reported by organisations specialised in habitat management, where these included the costs of infrastructure (including materials) and labour (Table 6.1).

Table 6.1. The costs under the farm-subsidy and land-purchase strategies of purchasing land, and then creating and maintaining scrub, wet grassland and woodland habitats. All estimates are in 2019 GBP; we used an inflation calculator (Bank of England 2021) to adjust estimates made in different years.

Habitat type	Farm-subsidy strategy		Land-purchase strategy		
	Creation costs	**Maintenance costs**	**Land purchase costs**	**Creation costs**	**Maintenance costs**
Scrub	Planning: Scrub is typically created by allowing natural regeneration (RSPB 2020). The costs associated with planning scrub creation were not well documented in the literature, so we employed generic values taken from Ausden and Hirons (2002) who estimated the planning costs for three wet grassland sites as £278/ha. Infrastructure/labour: We assumed there to be no infrastructure or labour costs because scrub was assumed to regenerate naturally. Combining these estimates, the creation costs for scrub creation were estimated as £278/ha.	Based on choice experiment; vary depending on area enrolled (Fig. 5.1).	Based on distribution of land prices; vary depending on area bought (Fig. 5.2).	As for farm-subsidy	We did not find estimates of the costs of maintaining scrub in the UK. However, a study from Denmark examined the annual maintenance costs of heathland from which we took the mean cost of £69/ha/y.
Wet grassland	Planning/infrastructure/labour: The creation costs of planning, infrastructure were estimated at £1454/ha based on Ausden & Hirons (2002) who explored the costs associated with creating three wet grassland sites.				As for creation, the annual maintenance costs for wet grassland were taken from Ausden & Hirons (2002). Combining the management, staff and 'other' costs associated with habitat maintenance gave an estimate of £419/ha/y.
Woodland	We assumed woodland creation required planting since natural regeneration is slow and can only be achieved with exclusion of grazers. Planning: Planning costs were not well documented so we again used the estimate of £278/ha from Ausden & Hirons (2002). Infrastructure: We estimated infrastructure costs according to the Countryside Stewardship Woodland Creation Grants Scheme (Natural England 2018). The costs of trees, planting, weeding and guards, plus replacements following mortality, gave an estimated infrastructure cost of £1732/ha. Labour: CJC Consulting (2014) estimated the labour costs of woodland in Wales at £457/ha and at £1652/ha for a broadleaved woodland managed for game/biodiversity in south-west England. We took the mid-point of these estimates. Combining these costs, the creation costs of woodland creation were estimated at £2976/ha.				Nix (2018) estimated the costs of maintaining woodland at £60-90/ha/y but it was unclear whether this included labour costs. Therefore, to ensure labour costs were included, we estimated maintenance costs at the £200/ha/y rate paid under the Countryside Stewardship Woodland Creation Grant Scheme for woodland maintenance (Natural England 2018).

Estimating the benefit delivered by a given budget

Given these costs, our two strategies were conceptualised as follows:

- Under the annual farm-subsidy strategy, in the first year, all budget was spent paying farmers to create habitat; in subsequent years budget was first allocated to paying farmers to maintain existing habitat before any surplus was allocated to generate further habitat creation.
- In the land purchase strategy, to avoid introducing long-term repayments and the associated complexities into only one strategy, we assumed land was purchased outright (following Schöttker et al. 2016). So, in the first year, we assumed all budget was spent purchasing land and then paying for habitat creation; in the following years, budget was first spent maintaining habitat that had been bought and created in previous years, before any surplus budget was allocated to further land purchase and creation of habitat.

Based on all budget being spent in all years, we estimated the area under each strategy each year. We assumed costs would increase with inflation, as would the budget, so our relative purchasing power remained constant through time.

For each of our four outcomes, we defined the benefit as the sum of the additional birds/tonnes carbon delivered on maintained habitat across each and all years of the scheme; this assumed the change in birds/tonnes carbon was realised in full in the first year following habitat creation. We compared the long-run benefits achieved by each approach, capturing the present value of future benefits using a discount rate. We explored how these benefits varied with differences in discount rate, budget and timescale relative to a baseline scenario with fixed discount rate and budget, as follows:

1. Environmental discount rates were varied between 0 and 5% following Curran et al. (2016), and to incorporate the 3.5% economic discount rate advocated by HM Treasury (2018). In line with this, we fixed the discount rate at 3.5% in the baseline scenario.
2. Given uncertainty in what the government would be willing to spend on delivering these outcomes, we explored difference arising when between £0-£10m/y (an upper bound of what might be spent, particularly on carbon sequestration; Committee on Climate Change (2018)) was spent amongst our population of farmers, who farmed 1.7% of lowland arable England. For illustration, we considered a baseline scenario where we set these budgets in line with spend in current AES. We wanted to consider ambitious conservation action, so we based our baseline budget on current spend on interventions that deliver yellowhammers since our other outcomes receive less funding (RPA, *pers. comm.).* Therefore, in the baseline case, we set the budget for each biodiversity outcome as 1.7% (i.e. the percentage of arable farmland

studied here) of the current spend on interventions in the Countryside Stewardship Scheme that deliver yellowhammers (RPA, *pers. comm*.); this gave a budget of £0.3m/y. We assumed the combined budget for our biodiversity outcomes would be spent on carbon sequestration, i.e. £0.9m/y.

3. We varied the timescales over which outcomes were evaluated between 10 and 100 years; this upper limit corresponds to that applied in other environmental planning exercises (Defra & Environment Agency 2006; Rayment et al. 2011).

In addition, to recognise the uncertainties in other parameters involved in our analysis, we explored the effect on relative cost-effectiveness of changing our assumptions in four key ways. Relative to the baseline scenario, we explored the effects of when: (1) the maintenance costs and (2) the creation costs on purchased land were halved, in order to reflect the economies of scale that may be achieved by contracted organisations; (3) 10% more benefit was delivered per unit area on purchased land, given the potential for more experienced contractors to deliver greater outcomes than farmers, and; (4) land prices increased at 1%/y above inflation. We evaluated the impacts of each of these changes across a range of timescales and discount rates.

Results

First, we found that the relative cost-effectiveness of our two strategies differed markedly depending on the budget and timescale considered. For a 3.5% discount rate, the farm-subsidy strategy was more cost effective at very small budgets and short timeframes (Figure 6.3). This is because small budgets could only enrol the most-willing farmers who sought less compensation for habitat maintenance than contracted organisations, particularly for wet grassland (Figure 6.3c). Given our farm-subsidy strategy involved fixed-price payments, increasing budgets saw all farmers paid at the rate of the least-willing participant, so cost-effectiveness declined. Thus, the land-purchase strategy, where the per-ha maintenance payments made to contracted organisations were assumed to remain unchanged regardless of how much land was purchased, was more cost effective at all but the lowest budgets. Land purchase became favourable at longer timescales because the high up-front costs of land purchase delayed the rate at which benefit accumulated relative to the farm-subsidy strategy.

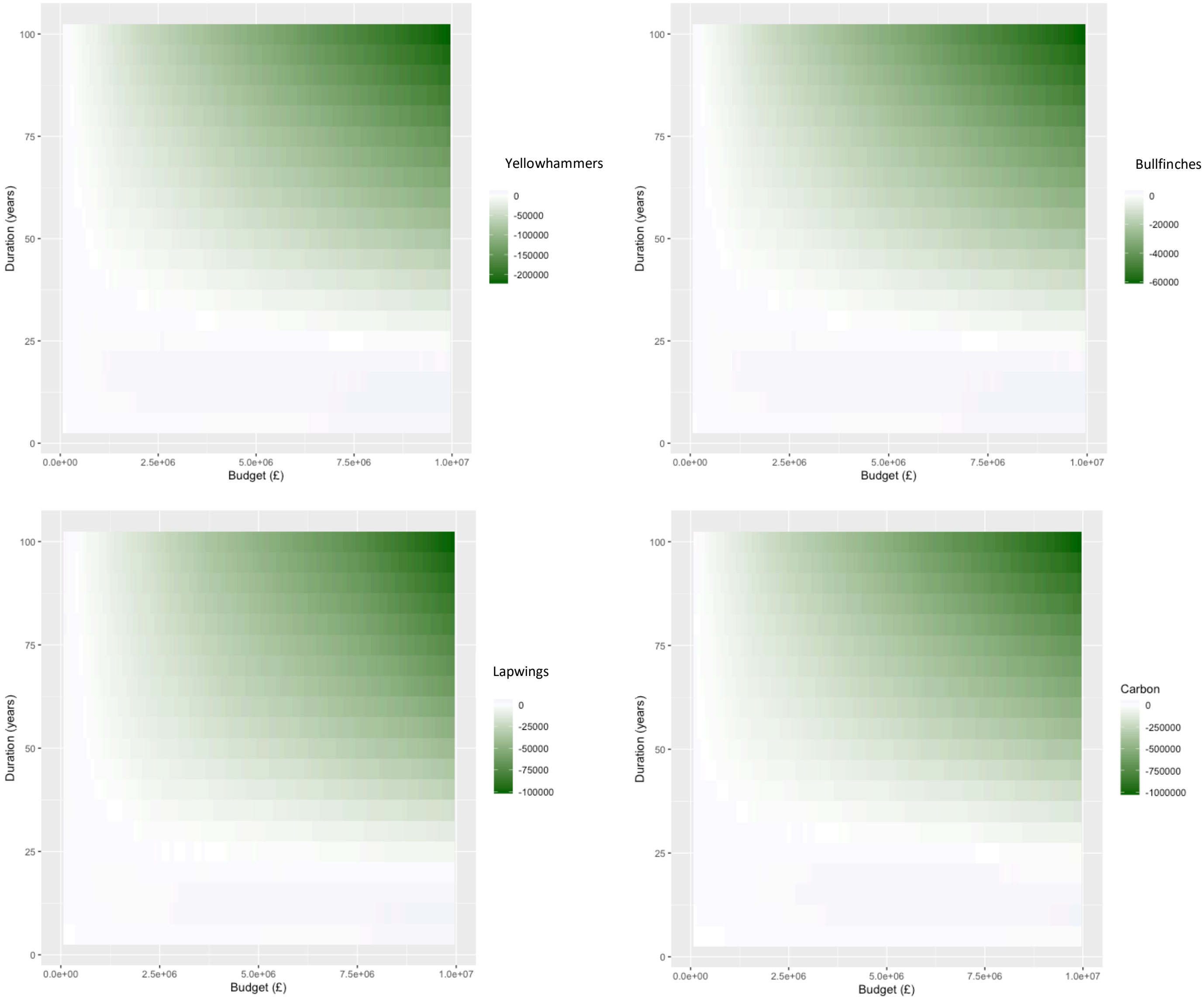

Figure 6.3. The difference in benefit delivered under the farm-subsidy strategy and the land-purchase strategy, for budgets up to £10m/y (2019 GBP), a range of scheme durations, and a fixed discount rate of 3.5%, in terms of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) carbon. Positive (blue) values indicate greater benefit was delivered by the farm-subsidy strategy; negative (green) values indicate greater benefit was delivered by the land-purchase strategy. Note: colour scales vary across plots.

Turning now to explore the impacts of varying discount rates, when we fixed the budget in line with current spend on existing AES at £0.3m/y for each of the three biodiversity outcomes, we found it was more cost effective to deliver biodiversity outcomes with the farm-subsidy strategy at all but the longest timescales and lowest discount rates (Figure 6.4). Higher discount rates gave more weight to the environmental benefits accumulated early on which were lower under the land-purchase strategy (due to the high up-front costs of land purchase). However, in line with our findings on the effects of available budgets (Figure 6.3), because in this analysis we assumed a larger budget was allocated to carbon sequestration (£0.9m/y, to match the combined spend on biodiversity outcomes), we found the land-purchase strategy to be more cost effective in delivering carbon sequestration across a wider range of discount rates and timescales than for our biodiversity outcomes.

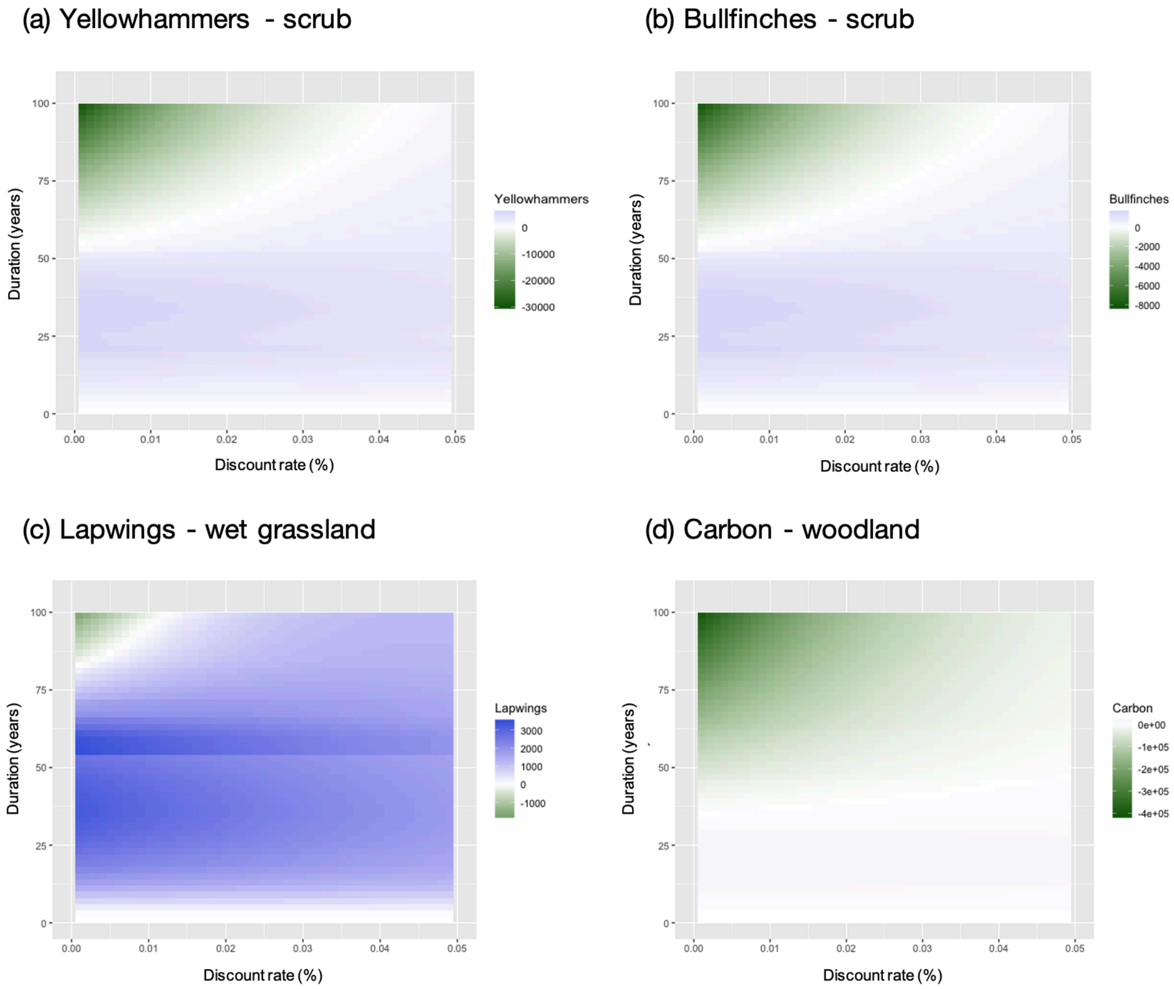

Figure 6.4. The difference in benefit delivered by the farm-subsidy strategy and the land-purchase strategy, for a range of scheme durations and discount rates, and a fixed budget (£0.3m/y for each biodiversity outcome and £0.9m/y for carbon in 2019 GBP), in terms of (a) yellowhammers, b) bullfinches, (c) lapwings and (d) carbon. Positive (blue) values therefore indicate greater benefit was delivered by the farm-subsidy strategy; negative (green) values indicate greater benefit was delivered by the land-purchase strategy. Note: colour scales vary across plots.

Last, we explored the sensitivity of our findings to changes in other parameters relative to the baseline scenario (the black line in Figure 6.5). In the baseline scenario, land purchase was favoured by higher durations and lower discount rates (i.e. the top-left corner of Figure 6.5). We found the land-purchase strategy became more cost effective at shorter timescales and higher discount rates when we assumed purchased land would deliver greater benefit per unit area relative to farmers' land (Figure

6.5; black vs green line). Reducing the costs of creating habitat on purchased land by 50% (Figure 6.5; black vs blue line) only slightly lowered the discount rates and timescales at which the land-purchase strategy became more cost effective; the effect was slightly larger when maintenance costs were assumed 50% lower (Figure 6.5; black vs orange line), but the impact was only marked for lapwing conservation (Figure 6.5c). Assuming the price of land marketed for sale increased at 1% each year above inflation (red line) substantially reduced the relative cost-effectiveness of the land-purchase strategy for delivering yellowhammers, bullfinches and lapwings, though even over quite short timescales it remained favourable for carbon sequestration across a wide range of discount rates (Fig. 5d).

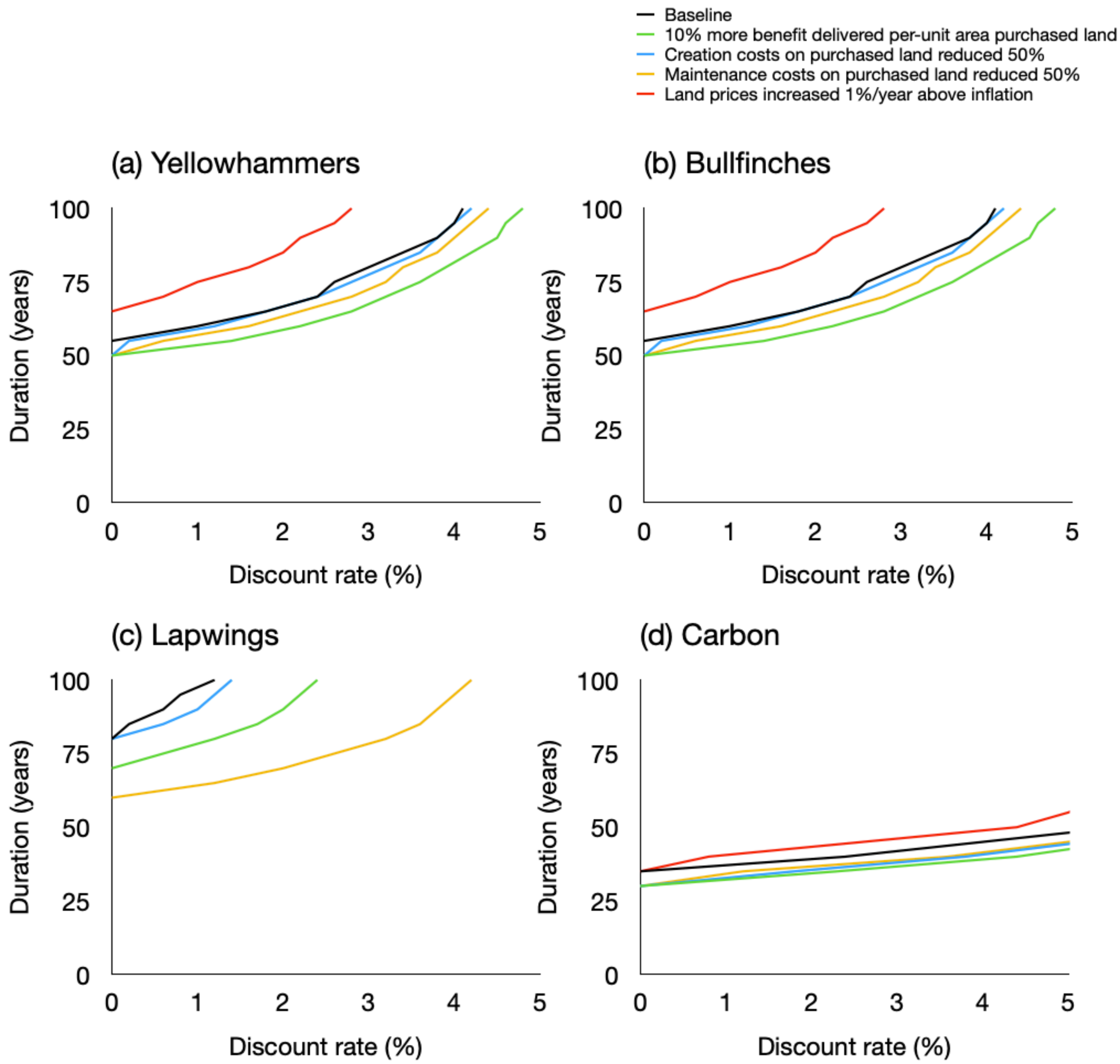

Figure 6.5. The duration and discount rate at which the more cost-effective strategy switched from land purchase (top left corner on each plot) to farm subsidy (bottom right corner) for: the baseline condition (black); when 10% more benefit was assumed to be delivered per unit area of habitat under the land-purchase strategy

compared with the farm-subsidy strategy (green); when the costs of creation (blue) and maintenance (orange) in the land-purchase strategy were halved, and; when the price of land for sale was assumed to increase by 1%/y in addition to inflation (red). (For lapwings (c), the farm-subsidy strategy was always more cost effective than land purchase when land prices increased at 1% above inflation, hence there is no red line).

Discussion

Using a combination of new choice experiment data, market prices for land purchase and literature-based habitat creation and maintenance costs, we discovered that the current AES approach of paying farmers to create and maintain habitat was less cost effective than purchasing land at long timescales, low discount rates and high budgets. This, the first UK-based study, mirrored the direction in which these parameters have been found to favour land purchase in other studies based elsewhere (Curran et al. 2016; Schöttker et al. 2016; Schöttker & Wätzold 2018). Our work advances previous studies by basing the payments required by farmers on the results of a choice experiment. Recognising that farmers vary in their willingness to accept (WTA) payment to participate gives rise to a more accurate picture of the benefit delivered by a range of budgets compared to other studies that have assumed the payment required by all potential participants is that offered by current schemes or the participant's lost gross margin, which is known to not be the only determinant of WTA (Chapter 5).

Our conclusions are limited in three key ways. Firstly, land prices and land availability vary unpredictably through time (Savills UK 2020; Strutt & Parker 2021) and fluctuations in either (which could arise if government was purchasing large areas of land) would change the relative cost-effectiveness of the two strategies, potentially quite substantially (Figure 6.5). Second, we used the best available cost estimates for creating and maintaining habitat on purchased land but information was scarce (White et al. 2022). Moreover, contracted organisations may require compensation above their costs, though competitive tendering of contracts may limit the extent of this and our sensitivity analysis did suggest that conclusions were fairly robust to even large deviations in these payment rates (Figure 6.5). Third, we did not explore the administration costs associated with delivering either strategy, which may differ under the two approaches (Ferraro 2008), but are a small component of scheme costs (administration costs account for <4% of the annual spend on AES; Natural England 2009).

Given we found that conclusions changed depending on the timescale, discount rate and budget assumed, it is important to consider how the parameter values we considered reflect UK government practice. An evaluation timescale of 100 years, which would favour land purchase, does appear to be considered when planning land purchase for flood mitigation or offsetting the biodiversity impacts of development (e.g. Defra and Environment Agency 2006; Rayment et al. 2011). A 3.5% economic discount rate is commonly advocated (HM Treasury 2018); however we also explored lower discount rates which may be more appropriate for environmental outcomes (Gollier 2010; Armsworth 2018) and which favour land purchase, though there is little consensus around an appropriate environmental discount rate (Groom et al. 2022). At the budgets currently spent in AES, farm-subsidy schemes would likely be more cost effective at all but the longest timescales and lowest discount rates for delivering bullfinches, lapwings and yellowhammers. However, climate mitigation action may see spend on carbon sequestration even in excess of what was explored in Figure 6.4 (e.g. as outlined in Committee on Climate Change (2018)), which would favour land purchase regardless of the timescale and discount rate considered. We did not consider together the costs of simultaneously delivering our biodiversity and carbon outcomes. Habitat creation for carbon sequestration can deliver some biodiversity benefits (Griscom et al. 2017; Chausson et al. 2020; Di Sacco et al. 2021), but may also increase the costs of delivering other biodiversity benefits by increasing competition for land; and we should be wary of land management practices that may deliver net zero at the expense of biodiversity (Bradfer-Lawrence et al. 2021).

A land-purchase strategy presents several possible advantages. First, per-area costs fall when a large area is managed (Armsworth et al. 2011). Indeed, pursuing this strategy may unlock economies of scale; our analysis may therefore be conservative in assuming that the per-ha costs of paying conservation organisations to manage land do not change as more land is purchased. Second, land purchase may facilitate the agglomeration of spared land which might increase the benefit delivered per unit area (Lamb et al. 2016b). Third, the land purchase strategy offers the advantage of future environmental programs being potentially able to address other environmental concerns using land already under government ownership (Schöttker & Wätzold 2018).

Whilst land purchase presents some advantages, there are also potential risks. We did not explore the implications of the lost livelihoods by the farming community in selling land which is then not primarily managed for food production (Fairhead et al. 2012). Lost cultural value, which is not considered in our analysis, may not be compensated by other possible co-benefits such as flood alleviation and recreation (Kirchhoff 2012). The alternative farm-subsidy strategy of paying farmers to manage habitat has potentially less risky implications for farming communities: farmers retain land ownership and so benefit from the payments issued to manage it whilst retaining the capacity to make future land-management decisions.

In conclusion, we found that if the UK government assumes short timescales, high discount rates and prevailing budget allocations, they would deliver more yellowhammers, bullfinches, lapwings and carbon sequestration with their predominant strategy of offering payments to farmers to create and maintain habitat through voluntary AES. However, if environmental goals become more ambitious into the future and are supported by correspondingly larger budgets, land purchase becomes favourable, particularly if AES continue to use fixed, rather than discriminatory, pricing (Ferraro 2008), and if land prices do not rise above inflation. Especially for carbon sequestration, growing interest and investment may deliver the conditions which make land purchase more cost effective, though impacts on farming communities of large-scale land ownership changes must be explored further.

Chapter 7: Discussion

Summary of results

In this thesis, the first-ever exploration of the relative costs of implementing land-sharing and land-sparing interventions to achieve environmental outcomes on farmed land, I have presented evidence that land sparing is more cost effective to implement than land sharing in my UK study system. First, in Chapter 2 I showed that the sample size required to precisely estimate the effects of sharing may be near- impossible to deliver, whilst sparing effects can be determined with little increase to current efforts. Furthermore, I found it to be relatively less costly to monitor compliance in sparing schemes at current low and financially sub-optimal rates, accounting for the increased level of farmer participation needed to make up non-compliance. In Chapter 3, I found that despite more farmers being willing to share than spare for the same compensation payment, conservation outcomes that are less tolerant of farming (lapwings and reduced net carbon emissions) were delivered at much less expense by sparing, because of the far greater benefit sparing schemes delivered per unit area. Combining farmer preferences with monitoring requirements, in Chapter 4 I found that sparing delivered my combined target outcomes in terms of increased populations of bullfinches, lapwings, yellowhammers and reduced net carbon emissions at <50% of the taxpayer cost, and 79% of the food production cost, of sharing. In Chapter 5, I discovered that more compensation was needed to cover farmers' residual requirement (WTA net of lost gross margin) under sharing vs sparing; this suggests farmers could actually be considered to prefer sparing when the same returns are forgone to either approach. I also found the cost of these fixed-price schemes would be reduced by instead paying farmers their true supply price under variable pricing; however, whether a sharing or sparing approach was taken had a greater impact on taxpayer costs than the pricing structure. Finally, in Chapter 6 I showed that ambitious sparing action may be delivered at less cost with a land-purchase strategy, rather than making indefinite compensation payments to farmers, provided policymakers assumed long timescales and low discount rates. Altogether, this first exploration of their relative taxpayer costs showed land sparing may deliver a range of environmental outcomes at substantially less taxpayer cost than land sharing.

The implications of our monitoring studies

I conducted this work at a time when European and UK AES are dominated by land-sharing measures (evident from the relative spend on sharing vs sparing interventions; Rural Payments Agency, *pers. comm.*) which have uncertain effects (Kleijn & Sutherland 2003; Kleijn et al. 2006; Pe'er et al. 2014; Batáry et al. 2015). My effectiveness monitoring analysis sheds some light on why this uncertainty persists, despite extensive monitoring efforts (e.g. Baker et al. 2012; Bright et al. 2015; Davey et al. 2010; Gillings et al. 2005; MacDonald et al. 2019): very large sample sizes are required to deliver precise estimates of sharing actions because effects are relatively small and vary between sites. Many existing studies measure significance (e.g. Baker et al. 2012; Bright et al. 2015; Davey et al. 2010; Gillings et al. 2005; MacDonald et al. 2019), for which I found current monitoring efforts are near-sufficient. However, focusing only on whether an effect is present, and ignoring its size, totally precludes comparison of the cost-effectiveness of different measures (Nakagawa & Cuthill 2007; Amrhein et al. 2017) – a key focus of this thesis, and presumably also of policymakers.

That said, whilst I found that sparing required smaller sample sizes to deliver precise estimates, because I based my analysis on existing studies, I unavoidably had to compare studies of population density for sparing (Newson et al. 2005) with studies of population growth rate for sharing (Baker et al. 2012). Precise estimates of population growth rates would be expected to require larger sample sizes relative to studies of population density; therefore, we cannot know how much of the difference in my results was due to this methodological difference, rather than fundamental differences between sharing and sparing. I did use existing studies, which probably have good reasons for taking different approaches to monitoring sharing and sparing, but this shortcoming prevented the inclusion of effectiveness monitoring in my combined assessment of taxpayer costs (Chapter 4), a key limitation of my work. We might reasonably expect some of the difference in cost between sharing and sparing to be retained even if a common metric was measured, given that sparing generally delivers larger effects. In this case, the inclusion of effectiveness monitoring in the combined cost estimate would likely only increase the extent to which sparing was cheaper than sharing. In addition to reducing the benefits delivered, the low compliance monitoring rate at present may also be adding to the variability between sites, which increases the sample size of the effectiveness monitoring required to deliver precise results.

We found that the cost efficiency of current schemes could be dramatically increased if participants were monitored more for compliance, but that the relative costs of sub-optimal monitoring were lower for sparing. As I have discussed, increasing the compliance monitoring rate to what I found to be financially optimal in Chapter 2 may be undesirable since it may discourage farmers from joining schemes altogether (Keane et al. 2008), or increase the payments they require (Broch & Vedel 2012). My estimate of the financially optimal rate is limited in that it does not include these potential feedbacks, and so I may be underestimating the financial costs of increased monitoring. There may also be political costs to increasing monitoring rates; farmers may perceive increased monitoring as unfair, intrusive, or a sign of mistrust (Vedel et al. 2015), and therefore oppose it. Given how unpopular increasing monitoring may be, it is important that I found sparing was relatively less costly to monitor below the optimal rate, compared to sharing. Again, the extent to which sparing is less costly than sharing would only increase with the assumption that the current low compliance monitoring rate would continue.

Farmer preferences and the related costs of delivering each environmental outcome

Perhaps the most important aspect of this work is the finding that farmers are willing enough to spare that larger-scale habitat creation delivered less farmland-tolerant outcomes at much less cost than sharing. Opponents of land sparing have cited, without empirical evidence, farmers' unwillingness to spare as a reason for it not to be pursued (Fischer et al. 2008). Per unit area, farmers generally did require more compensation to spare, presumably because more gross margin was typically lost by sparing interventions. However, controlling for this lost gross margin by estimating farmers’ residual WTA, net of lost gross margin, showed that farmers required more compensation to implement sharing. I suggest this is preliminary evidence that, when the same gross margin is lost to either approach, farmers may actually prefer to spare – which begs the question of why sparing is not being pursued in government policy.

Sparing was not cheaper for delivering every outcome studied. In Chapter 3, I showed more yellowhammers, and narrowly bullfinches, would be delivered for equivalent spend on sharing, compared to sparing. Yellowhammers were particularly cheap to deliver given they benefit from retention from stubble prior to spring cropping (Hancock & Wilson 2003; Baker et al. 2012) which is a practice many farmers implement annually for weed/pest control (AHDB 2018). That farmers could be paid to deliver bullfinches by hedgerow creation, a sharing option, at slightly less cost than scrub creation follows from literature-based estimates that bullfinches are quite abundant in hedgerows (Macdonald & Johnson 1995). All my estimates of the biodiversity benefit of options are limited in that I did not estimate associated uncertainties, as these were not reported in many of the studies from which I extracted data. The studies available on hedgerow creation for bullfinches were particularly limited, so I am cautious of this particular result.

Combining costs for all outcomes

In Chapter 4, for the benefit of policymakers who must consider the best approach for all environmental outcomes, we combined the costs of paying farmers to deliver each outcome, as presented separately in Chapter 3, and added in the associated capital, monitoring and administration costs. Despite farmers requiring lower payments for yellowhammers and bullfinches in Chapter 3, when we combined costs for all outcomes in Chapter 4, we found sparing was overwhelmingly less expensive for the following reasons. Both yellowhammers and bullfinches, which are cheaper to deliver by sharing, are relatively abundant on farmland (Newson et al. 2005), and so the absolute difference between the cost of sharing and sparing was less than for lapwings and carbon, which are far more costly to deliver with sharing. For carbon and lapwings, sharing interventions delivered much less benefit per unit area, so in order to achieve my target outcomes, a fixed-price scheme had to appeal to relatively less-willing farmers with a higher WTA. Given farmers varied greatly in their WTA, such schemes therefore had to pay all participants the price of a farmer relatively far along the supply curve. Therefore, when I combined the costs of paying farmers to deliver lapwings and carbon with yellowhammers and bullfinches, I found sparing was the cheaper strategy by a considerable margin. This difference only increased when I included the capital costs, the costs of monitoring and the administration costs, which scaled with the number of required participants, and were therefore less under sparing.

The present analysis is the only estimate of which I am aware of the relative taxpayer costs of sharing and sparing. It builds on evidence that more biodiversity and carbon sequestration is delivered when the same volume of food production is lost by sparing compared to sharing (Phalan et al. 2011; Gilroy et al. 2014; Lamb et al. 2016a; Dotta et al. 2016; Williams et al. 2017; Finch et al. 2019, 2020, 2021; Feniuk et al. 2019). I found that, at the target levels studied, the payments required by farmers roughly tracked lost food production. Residual payments, that cover the surplus required by farmers above lost gross margin, were substantial, particularly if more ambitious conservation targets were considered. To deliver higher targets, less-willing farmers must be recruited who require much more compensation not, as I have shown, because they lose more gross margin, but for entirely different (and unknown, in my work) reasons. Importantly, the amount spent compensating the residual requirement, i.e. factors other than gross margin, was increasingly greater under sharing compared to sparing. The importance of this is two-fold: one, the relative costs of sharing will only increase relative to sparing at higher targets, and two, taxpayers may be less sympathetic towards compensating factors aside from the value of lost production. So, if the government were to pursue more ambitious action, I suggest that not only farmers, but also taxpayers, could be expected to prefer a land-sparing approach.

Other drivers of scheme costs: Variable pricing and land purchase

Variable pricing, where farmers are paid at their true supply price, is more cost effective than fixed pricing provided participants vary in their WTA. Indeed, previous research has shown fixed pricing to be a hugely important cause of cost inefficiency in current schemes (Armsworth et al. 2012). However, even with variable pricing, the costs of delivering lapwings and carbon via sharing interventions were not reduced to those of fixed-price sparing schemes. Furthermore, proponents of variable pricing justify the need for differential payments on the grounds of farmers losing different amounts of gross margin, which deserves equal compensation (Armsworth et al. 2012). This argument is perhaps weakened, therefore, by my finding that a large portion of WTA is driven by factors besides lost gross margin. The nature of these factors – not identified here, but potentially including labour costs, risk aversion and attitudes towards the government – probably influences whether taxpayers consider them sufficient grounds to differentiate payments. Residual requirements may also be fickle, potentially fluctuating, for example, with the knowledge that another farmer is being paid more for the same action. This may explain why bids made in reverse auctions, designed to reveal farmers' true supply prices, increase through time toward the maximum bid accepted (Hailu and Schilizzi 2004; reviewed in Ferraro 2008). Given perceptions around fairness, and probably also the operating costs (Ferraro 2008), variable-price schemes may be unlikely to be pursued.

Factors aside from taxpayer costs may also drive whether land sparing might be implemented via land-purchase or ongoing farm subsidy. I found land purchase offered some cost savings over a farm-subsidy approach; however, policymakers may be unlikely to consider the ambitious targets, long timeframes and low discount rates under which land purchase is cheaper. Indeed, land purchase is not a new possibility, yet it has received little investment (Kettunen et al. 2011). Perhaps this is due to the marked and lasting effects that land purchase could have on farming communities who are left with no control over future land management; even under long-term land sparing some degree of control would be retained, given that contracts would be time limited.

There may be benefits associated with land purchase that were not captured in our analysis. Land purchase may offer a way to deliver the agglomeration of spared areas that would likely increase the benefit per unit area of sparing (Laurance et al. 2011; Didham & Ewers 2012; Lamb et al. 2016b), while potentially decreasing taxpayer costs, of sparing. Indeed, in my thesis I considered spared areas up to 50ha in size since this was achievable on most English arable farms; however, environmental gains may be much greater with the creation of much larger areas of semi-natural habitat (Lamb et al. 2016b). This could perhaps be delivered with bonus payments that reward the creation of larger spared areas, including through coordination across farms (e.g. Banerjee et al. 2014; Hanley et al. 2012; Liu et al. 2019); but may also arise to a totally unstudied extent without explicit compensation due to the spatial coarseness of farm profitability across the UK (Defra 2018b).

Changes in ownership may arise in any case due to the forthcoming downscaling of the Basic Payment Scheme (Defra 2022) that has maintained the profitability of farming in less productive areas of the UK. Alternatively, a sparing scheme may offer a continued way of life for these farms; sharing may also do that but, as I have shown, it would be far less cost effective. Growing interest in private biodiversity and carbon offsets may also be a lifeline (Green Alliance 2022); though this market should be regulated, for many reasons (Victor & Cullenward 2007; Wara 2007; Haya et al. 2020), including so that the most cost-effective measures are implemented to avoid increasing the pressure on land by using inefficient sharing approaches.

Implications of our findings for other species, ecosystem services and locations

Given that the relative performance of sharing and sparing varies across different conservation outcomes it is important to consider whether more species are like yellowhammers or like lapwings, and whether our findings for reduced net carbon emissions would be mirrored across other ecosystem services. Whilst the taxpayer costs of using sharing and sparing to deliver outcomes beyond the four considered here have not been studied, we do know from studies that have assessed >2,500 species across 5 continents (Balmford 2021) that less food production would be lost when using a sparing, rather than sharing, approach to deliver biodiversity, as well as carbon sequestration, nature-based recreation, nitrogen pollution abatement and phosphorus pollution abatement (Finch 2021). In Chapter 4 we found sparing to be both the strategy that resulted in the least lost food production and the least costly strategy to the taxpayer; this alignment might be expected to hold across other outcomes, since taxpayer costs are, in part, compensation for the value of forgone production (Chapter 5). This provides a basis to suggest that we might expect sparing to be the least costly strategy to the taxpayer for delivering this much broader array of outcomes for which we know sparing sees less forgone production compared to sharing.

Furthermore, for two reasons, I argue that my assessment of taxpayer costs was biased in favour of sharing. First, one in four lowland species in the UK cannot be delivered on farmland at all (Lamb et al. 2019). These species are therefore even more costly than lapwings to deliver with sharing. Including any of these species, which I deliberately did not do, would drive conclusions further in favour of sparing. Evidence that sparing delivers nature-based recreation as well as nitrogen and phosphorus pollution mitigation more efficiently than sharing suggests that this is likely also true for many ecosystem services (e.g. Finch et al. 2021). Second, conservation targets that do not persist well on farmland are arguably more in need of conservation, given the strikingly limited area under semi-natural habitat in the UK vs the domination of 70% of the land surface by farming (National Food Strategy 2021). The prevalence of habitat-specialists that do not tolerate conditions on farmed land is likely to be greater elsewhere in the world where countries typically have a shorter history of agriculture than the UK, and thus have not already driven many such species to extinction (Balmford 1996). Consequently, the relative costs of sharing would likely only increase were this study conducted in a different country.

We should also be aware that the domestic conservation efforts studied here may themselves have impacts on species/services in other countries via the consequences of such actions for domestic food production, and hence for food imports (Lambin & Meyfroidt 2011; WWF & RSPB 2020). Unless sharing-style conservation efforts and linked to marked changes in diets and/or food waste (Van Zanten et al. 2018; Springmann et al. 2018) and sparing efforts are not, sharing will inevitably see a greater rise in food imports, to keep pace with demand. My work suggests sparing is less expensive both in terms of its domestic cost and its impacts overseas. This framing is important in the context of net-zero targets: countries such as the UK will likely meet their target by offshoring the production of carbon-intensive products and therefore the associated emissions overseas (WWF & RSPB 2020). Indeed, research suggests the emissions associated with the imports needed to compensate yield losses to organic farming, a land-sharing approach, would entirely outweigh on-farm emissions reductions (Smith et al. 2019). Further work should look to conduct the same assessment for biodiversity: global biodiversity will not benefit if policymakers ignore the offshored impacts that arise from their decisions.

Conclusion

In summary, I have provided new evidence showing that a sparing approach would deliver meaningful biodiversity and climate mitigation in the UK at around half the taxpayer cost and 79% the food production cost of sharing. My taxpayer cost assessment is wide-ranging, encompassing farmer attitudes as well as capital, monitoring and administration costs. Consideration of the costs of compensating lost food production would only increase the extent to which sparing is cheaper, as would consideration of the many species/services that cannot be delivered on farmland, those that are most in need of conservation, and those living in regions likely to see increased agricultural activity to meet shortfalls in UK production. Variable pricing and land purchase may reduce taxpayer costs, but I have shown that the most important decision is whether or not to pursue a land-sparing approach. I have refuted suggestions that farmers are not willing enough to spare to make this approach viable; moreover, sparing would in practice see a greater proportion of payments spent compensating the value of lost food production than addressing other aspects of farmer preferences – something which is likely to be more palatable to the taxpayer. Additional work is needed: in deciding the direction of future policy, governments would be helped by clearer estimates of the costs of comparable effectiveness monitoring, by explicit comparison of the taxpayer and environmental costs of compensating domestic production losses, and by analyses of the likely costs of delivering larger-scale spared areas. However, even without this information, I have provided robust evidence that sparing is a much cheaper approach to delivering biodiversity conservation and reducing carbon emissions in the UK.

References

Acs S, Hanley N, Dallimer M, Gaston KJ, Robertson P, Wilson P, Armsworth PR. 2010. The effect of decoupling on marginal agricultural systems: Implications for farm incomes, land use and upland ecology. Land Use Policy **27**:550–563.

Adamowicz W, Boxall P, Williams M, Louviere J. 1998. Stated preference approaches for measuring passive use values: choice experiments and contingent valuation. American Journal of Agricultural Economics **80**:64–75.

Aebischer NJ, Bailey CM, Gibbons DW, Morris AJ, Peach WJ, Stoate C. 2016. Twenty years of local farmland bird conservation: The effects of management on avian abundance at two UK demonstration sites. Bird Study **63**:10–30.

AHDB. 2018. Managing weeds in arable rotations-a guide Incorporating WRAG guidelines.

Alder DC, Fuller RJ, Marsden SJ. 2018. Implications of transformation to irregular silviculture for woodland birds: A stand wise comparison in an English broadleaf woodland. Forest Ecology and Management **422**:69–78.

Amrhein V, Korner-Nievergelt F, Roth T. 2017. The earth is flat (p > 0:05): Significance thresholds and the crisis of unreplicable research. PeerJ **2017**:3544.

Armsworth PR. 2018. Time discounting and the decision to protect areas that are near and threatened or remote and cheap to acquire. Conservation Biology **32**:1063–1073.

Armsworth PR, Acs S, Dallimer M, Gaston KJ, Hanley N, Wilson P. 2012. The cost of policy simplification in conservation incentive programs. Ecology Letters **15**:406–414.

Armsworth PR, Cantú-Salazar L, Parnell M, Davies ZG, Stoneman R. 2011. Management costs for small protected areas and economies of scale in habitat conservation. Biological Conservation **144**:423–429.

Arnott D, Chadwick DR, Wynne-Jones S, Jones DL. 2021. Vulnerability of British farms to post-Brexit subsidy removal, and implications for intensification, extensification and land sparing. Land Use Policy **107**:104154.

Atkinson PW, Fuller RJ, Vickery JA, Conway GJ, Tallowin JRB, Smith REN, Haysom KA, Ings TC, Asteraki EJ, Brown VK. 2005. Influence of agricultural management, sward structure and food resources on grassland field use by birds in lowland England. Journal of Applied Ecology **42**:932–942.

Ausden M, Hirons GJM. 2002. Grassland nature reserves for breeding wading birds in England and the implications for the ESA agri-environment scheme. Biological Conservation **106**:279–291.

Bahadur K et al. 2018. When too much isn't enough: Does current food production meet global nutritional needs? PLoS ONE **13**:e0205683.

Baker DJ, Freeman SN, Grice P V., Siriwardena GM. 2012. Landscape-scale responses of birds to agri-environment management: a test of the English Environmental Stewardship scheme. Journal of Applied Ecology **49**:871–882.

Balmford A. 1996. Extinction filters and current resilience: the significance of past selection pressures for conservation biology. Trends in Ecology & Evolution **11**:193–196.

Balmford A. 2021. Concentrating vs spreading our footprint: how to meet humanity's needs at least cost to nature. Available from https://www.repository.cam.ac.uk/handle/1810/325360 (accessed September 7, 2021).

Banerjee S, Cason TN, Vries FP de, Hanley N. 2021. Spatial Coordination and Joint Bidding in Conservation Auctions. https://doi.org/10.1086/714601 **8**:1013–1049.

Banerjee S, de Vries FP, Hanley N, van Soest DP. 2014. The Impact of Information Provision on Agglomeration Bonus Performance: An Experimental Study on Local Networks. American Journal of Agricultural Economics **96**:1009–1029.

Bank of England. 2021. Inflation Calculator. Available from https://www.bankofengland.co.uk/monetary-policy/inflation/inflation-calculator (accessed July 5, 2021).

Barreiro-Hurlé J, Espinosa-Goded M, Dupraz P. 2010. Does intensity of change matter? Factors affecting adoption of agri-environmental schemes in Spain. Journal of Environmental Planning and Management **53**:891–905.

Batáry P, Dicks L V., Kleijn D, Sutherland WJ. 2015. The role of agri-environment schemes in conservation and environmental management. Conservation Biology **29**:1006–1016.

Bateman IJ, Balmford B. 2018. Public funding for public goods: A post-Brexit perspective on principles for agricultural policy. Land Use Policy **79**:293–300.

Bellamy PE, Hinsley SA, Newton I. 1996. Factors Influencing Bird Species Numbers in Small Woods in South-East England. Journal of Applied Ecology **33**:249–262.

Benford K. 2016. The British Survey of Fertiliser Practice: Fertiliser use on farm crops for crop year 2015.

Bennett AF, Hinsley SA, Bellamy PE, Swetnam RD, Mac Nally R. 2004. Do regional gradients in land-use influence richness, composition and turnover of bird assemblages in small woods? Biological Conservation **119**:191–206.

Bevington A. 1991. Habitat selection in the Dunnock Prunella modularis in northern England. Bird Study **38**:87–91.

Bird JP, Martin R, Akçakaya HR, Gilroy J, Burfield IJ, Garnett ST, Symes A, Taylor J, Şekercioğlu ÇH, Butchart SHM. 2020. Generation lengths of the world's birds and their implications for extinction risk. Conservation Biology **34**:1252–1261.

Blackstock TH, Stevens DP, Howe EA. 1996. Biological components of Sites of Special Scientific Interest in Wales. Biodiversity and Conservation **5**:897–920.

Bliemer MCJ, Rose JM. 2013. Confidence intervals of willingness-to-pay for random coefficient logit models. Transportation Research Part B: Methodological **58**:199–214.

Bolton M, Tyler G, Smith K, Bamford R. 2007. The impact of predator control on lapwing Vanellus vanellus breeding success on wet grassland nature reserves. Journal of Applied Ecology **44**:534–544.

Bradbury RB, Browne SJ, Stevens DK, Aebischer NJ. 2004. Five-year evaluation of the impact of the Arable Stewardship Pilot Scheme on birds. Ibis **146**:171–180.

Bradbury RB, Kyrkos A, Morris AJ, Clark SCSC, Perkins AJ, Wilson JD. 2000. Habitat associations and breeding success of yellowhammers on lowland farmland. Journal of Applied Ecology **37**:789–805.

Bradbury RB, Payne RJH, Wilson JD, Krebs JR. 2001. Predicting population responses to resource management. Trends in Ecology and Evolution **16**:440–445.

Bradfer-Lawrence T, Finch T, Bradbury RB, Buchanan GM, Midgley A, Field RH. 2021. The potential contribution of terrestrial nature-based solutions to a national 'net zero' climate target. Journal of Applied Ecology **58**:2349–2360.

Bright JA, Field RH, Morris AJ, Cooke AI, Fern J, Grice P V., Peach W. 2014. Influence of crop type and Agri-Environment management on the use of over-winter stubbles by farmland birds in England. Bird Study **61**:332–339.

Bright JA, Morris AJ, Field RH, Cooke AI, Grice P V., Walker LK, Fern J, Peach WJ. 2015. Higher-tier agri-environment scheme enhances breeding densities of some priority farmland birds in England. Agriculture, Ecosystems and Environment **203**:69–79.

Broch SW, Vedel SE. 2010. Heterogeneity in landowners' agri-environmental scheme preferences. Pages 62–86 Proceedings of the Economic Council Conference on Environmental Economics.

Broch SW, Vedel SE. 2012. Using Choice Experiments to Investigate the Policy Relevance of Heterogeneity in Farmer Agri-Environmental Contract Preferences. Environmental and Resource Economics **51**:561–581.

Broome A, Bellamy C, Rattey A, Ray D, Quine CP, Park KJ. 2019. Niches for Species, a multi-species model to guide woodland management: An example based on Scotland's native woodlands. Ecological Indicators **103**:410–424.

Buckingham DL, Evans AD, Morris AJ, Orsman CJ, Yaxley R. 1999. Use of set-aside land in winter by declining farmland bird species in the UK. Bird Study **46**:157–169.

Butler SJ, Bradbury RB, Whittingham MJ. 2005. Stubble height affects the use of stubble fields by farmland birds. Journal of Applied Ecology **42**:469–476.

Butler SJ, Mattison EHA, Glithero NJ, Robinson LJ, Atkinson PW, Gillings S, Vickery JA, Norris K. 2010. Resource availability and the persistence of seed-eating bird populations in agricultural landscapes: A mechanistic modelling approach. Journal of Applied Ecology **47**:67–75.

Carrasco L, Norton L, Henrys P, Siriwardena GM, Rhodes CJ, Rowland C, Morton D. 2018. Habitat diversity and structure regulate British bird richness: Implications of non-linear relationships for conservation. Biological Conservation **226**:256–263.

Chamberlain D. 2018. Agri-environment schemes and farmland bird populations: Is the glass half-full or half-empty? Animal Conservation **21**:193–194.

Chamberlain D, Gough S, Anderson G, Macdonald M, Grice P, Vickery J. 2009. Bird Study Bird use of cultivated fallow “Lapwing plots” within English agri-environment schemes. Bird Study **56**:289–297.

Chamberlain DE, Freeman SN, Vickery JA. 2007. The effects of GMHT crops on bird abundance in arable fields in the UK. Agriculture, Ecosystems and Environment **118**:350–356.

Chamberlain DE, Fuller RJ, Bunce RGH, Duckworth JC, Shrubb M. 2000. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. Journal of Applied Ecology **37**:771–788.

Chamberlain DE, Fuller RJ, Garthwaite DG, Impey AJ. 1999. A comparison of farmland bird density and species richness in lowland England between two periods of contrasting agricultural practice. Bird Study **88**:307–320.

Chausson A et al. 2020. Mapping the effectiveness of nature-based solutions for climate change adaptation. Global Change Biology **26**:6134–6155.

Christensen T, Pedersen AB, Nielsen HO, Mørkbak MR, Hasler B, Denver S. 2011. Determinants of farmers’ willingness to participate in subsidy schemes for pesticide-free buffer zones-A choice experiment study. Ecological Economics **70**:1558–1564.

Christie AP et al. 2020. Quantifying and addressing the prevalence and bias of study designs in the environmental and social sciences. Nature Communications **11**:1–11.

Christie AP, Amano T, Martin PA, Shackelford GE, Simmons BI, Sutherland WJ. 2019. Simple study designs in ecology produce inaccurate estimates of biodiversity responses. Journal of Applied Ecology **56**:2742–2754.

Christie M, Azevedo CD. 2009. Testing the Consistency Between Standard Contingent Valuation, Repeated Contingent Valuation and Choice Experiments. Journal of Agricultural Economics **60**:154–170.

CJC Consulting. 2014. Assessing the Cost-effectiveness of Woodlands in the Abatement of Carbon

Dioxide Emissions.

Collard M, Edinborough K, Shennan S, Thomas MG. 2010. Radiocarbon evidence indicates that migrants introduced farming to Britain. Journal of Archaeological Science **37**:866–870.

Committee on Climate Change. 2018. Land use: Reducing emissions and preparing for climate change:100.

Committee on Climate Change. 2020. Land use: Policies for a Net Zero UK.

Cornulier T, Robinson RA, Elston D, Lambin X, Sutherland WJ, Benton TG. 2011. Bayesian reconstitution of environmental change from disparate historical records: hedgerow loss and farmland bird declines. Methods in Ecology and Evolution **2**:86–94.

Cortés-Capano G, Hanley N, Sheremet O, Hausmann A, Toivonen T, Garibotto-Carton G, Soutullo A, Di Minin E. 2021. Assessing landowners' preferences to inform voluntary private land conservation: The role of non-monetary incentives. Land Use Policy **109**:105626.

Crastes dit Sourd R. 2021. A New Shifted Log-Normal Distribution for Mitigating "exploding" Implicit Prices in Mixed Multinomial Logit Models. SSRN Electronic Journal.

Crippa M, Solazzo E, Guizzardi D, Monforti-Ferrario F, Tubiello FN, Leip A. 2021. Food systems are responsible for a third of global anthropogenic GHG emissions. Nature Food **2**:198–209.

Critchley CNR, Allen DS, Fowbert JA, Mole AC, Gundrey AL. 2004. Habitat establishment on arable land: Assessment of an agri-environment scheme in England, UK. Biological Conservation **119**:429–442.

Curran M, Kiteme B, Wünscher T, Koellner T, Hellweg S. 2016. Pay the farmer, or buy the land?—Cost-effectiveness of payments for ecosystem services versus land purchases or easements in Central Kenya. Ecological Economics **127**:59–67.

Dadam D, Siriwardena GM. 2019. Agri-environment effects on birds in Wales: Tir Gofal benefited woodland and hedgerow species. Agriculture, Ecosystems and Environment **284**:106587.

Davey CM, Vickery JA, Boatman ND, Chamberlain DE, Parry HR, Siriwardena GM. 2010a. Assessing the impact of Entry Level Stewardship on lowland farmland birds in England. Ibis **152**:459–474.

Davey CM, Vickery JA, Boatman ND, Chamberlain DE, Siriwardena GM. 2010b. Entry Level Stewardship may enhance bird numbers in boundary habitats. Bird Study **57**:415–420.

Defra. 2018a. Health and Harmony: the future for food, farming and the environment in a Green Brexit.

Defra. 2018b. Agriculture in the United Kingdom 2017:1–111.

Defra. 2019a. Countryside Stewardship Mid Tier and Wildlife Offers Manual.

Defra. 2019b. Countryside Stewardship: Higher Tier Manual The European Agricultural Fund for Rural Development Europe investing in rural areas.

Defra. 2019c. Structure of the agricultural industry in England and the UK at June: Results by farm type.

Defra. 2019d. Farming Statistics: Final Crop Areas, Yields, Livestock Populations and Agricultural Workforce at 1 June 2014 - United Kingdom. Page National Statistics.

Defra. 2022. Environmental Land Management schemes: overview. Available from https://www.gov.uk/government/publications/environmental-land-management-schemes-overview/environmental-land-management-scheme-overview (accessed March 11, 2022).

Defra, Environment Agency. 2006. National evaluation of the costs of meeting coastal environmental requirements.

Dessart FJ, Barreiro-Hurlé J, Van Bavel R. 2019. Behavioural factors affecting the adoption of sustainable farming practices: A policy-oriented review. European Review of Agricultural Economics **46**:417–471.

Di Sacco A et al. 2021. Ten golden rules for reforestation to optimize carbon sequestration, biodiversity recovery and livelihood benefits. Global Change Biology **27**:1328–1348.

Dicks L, Ashpole J, Dänhardt J, James K. 2014. Farmland Conservation: Evidence for the effects of interventions in northern and western EuropeVolume 3.

Didham RK, Ewers RM. 2012. Predicting the impacts of edge effects in fragmented habitats: Laurance and Yensen’s core area model revisited. Biological Conservation **155**:104–110.

Dolman PM, Hinsley SA, Bellamy PE, Watts K. 2007. Woodland birds in patchy landscapes: the evidence base for strategic networks. Ibis **149**:146–160.

Donald PF, Fuller RJ, Evans AD, Gough SJ. 1998. Effects of forest management and grazing on breeding bird communities in plantations of broadleaved and coniferous trees in western England. Biological Conservation **85**:183–197.

Donald PF, Haycock D, Fuller RJ. 1997. Winter bird communities in forest plantations in western England and their response to vegetation, growth stage and grazing. Bird Study **44**:206–219.

Donovan I. 2013. Wild Thing: the effect of re-wilding on the densities of a group of Bird of Conservation Concern species.

Dotta G, Phalan B, Silva TW, Green R, Balmford A. 2016. Assessing strategies to reconcile agriculture and bird conservation in the temperate grasslands of South America. Conservation Biology **30**:618–627.

Douglas DJT, Benton TG, Vickery JA. 2010. Contrasting patch selection of breeding Yellowhammers Emberiza citrinella in set-aside and cereal crops. Bird Study **57**:69–74.

Draycott RAH, Hoodless AN, Cooke M, Sage RB. 2012. The influence of pheasant releasing and associated management on farmland hedgerows and birds in England. European Journal of

Wildlife Research **58**:227–234.

Eglington SM, Gill JA, Bolton M, Smart MA, Sutherland WJ, Watkinson AR. 2007. Restoration of wet features for breeding waders on lowland grassland. Journal of Applied Ecology **45**:305–314.

Eglington SM, Gill JA, Smart MA, Sutherland WJ, Watkinson AR, Bolton M. 2009. Habitat management and patterns of predation of Northern Lapwings on wet grasslands: The influence of linear habitat structures at different spatial scales. Biological Conservation **142**:314–324.

Elliott J, Day B, Jones G, Binner AR, Smith G, Skirvin D, Boatman ND, Tweedie F. 2015. Scoping the strengths and weaknesses of different auction and PES mechanisms for Countryside Stewardship.

Ellis M V., Taylor JE. 2018. Effects of weather, time of day, and survey effort on estimates of species richness in temperate woodlands. Emu - Austral Ornithology **118**:183–192.

Elton C. 1935. A Reconnaissance of Woodland Bird Communities in England and Wales. Journal of Animal Ecology **4**:127–136.

Espinosa-Goded M, Barreiro-Hurlé J, Ruto E. 2010. What do farmers want from agri-environmental scheme design? A choice experiment approach. Journal of Agricultural Economics **61**:259–273.

Fairhead J, Leach M, Scoones I. 2012. Green Grabbing: a new appropriation of nature? Journal of Peasant Studies **39**:237–261.

Falloon P, Powlson D, Smith P. 2004. Managing field margins for biodiversity and carbon sequestration: a Great Britain case study. Soil Use and Management **20**:240–247.

Farm Business Survey. 2020. FBS Definitions. Available from http://www.farmbusinesssurvey.co.uk/benchmarking/Default.aspx?module=DefOfTerms (accessed July 11, 2020).

Feniuk C, Balmford A, Green RE. 2019. Land sparing to make space for species dependent on natural habitats and high nature value farmland. Proceedings of the Royal Society B **286**.

Ferraro PJ. 2008. Asymmetric information and contract design for payments for environmental services. Ecological Economics **65**:810–821.

Field RH, MORRIS AJ, Grice P V., Cooke A. 2011. The provision of winter bird food by the English Environmental Stewardship scheme. Ibis **153**:14–26.

Finch T, Day BH, Massimino D, Redhead JW, Field RH, Balmford A, Green RE, Peach WJ. 2021. Evaluating spatially explicit sharing-sparing scenarios for multiple environmental outcomes. Journal of Applied Ecology **58**:655–666.

Finch T, Gillings S, Green RE, Massimino D, Peach WJ, Balmford A. 2019. Bird conservation and the land sharing-sparing continuum in farmland-dominated landscapes of lowland England. Conservation Biology:cobi.13316.

Finch T, Green RE, Massimino D, Peach WJ, Balmford A. 2020. Optimising nature conservation outcomes for a given region-wide level of food production. Journal of Applied Ecology **57**:985–994.

Finn JA, Bartolini F, Bourke D, Kurz I, Viaggi D. 2009. *Ex post* environmental evaluation of agri-environment schemes using experts' judgements and multicriteria analysis. Journal of Environmental Planning and Management **52**:717–737.

Fischer J et al. 2008. Should agricultural policies encourage land sparing or wildlife-friendly farming? Frontiers in Ecology and the Environment **6**:380–385.

Friends of the Earth. 2011. Europe's land import dependency.

Fuller RJ, Atkinson PW, Garnett MC, Conway GJ, Bibby CJ, Johnstone IG. 2006. Breeding bird communities in the upland margins (ffridd) of Wales in the mid-1980s. Bird Study **53**:177–186.

Fuller RJ, Chamberlain DE, Burton NHK, Gough SJ. 2001. Distributions of birds in lowland agricultural landscapes of England and Wales: How distinctive are bird communities of hedgerows and woodland? Agriculture, Ecosystems and Environment **84**:79–92.

Fuller RJ, Crick HQP. Broad-scale patterns in geographical and habitat distribution of migrant and resident passerines in Britain and Ireland. Ibis **134**:14–20.

Fuller RJ, Hinsley SA, Swetnam RD. 2004. The relevance of non-farmland habitats, uncropped areas and habitat diversity to the conservation of farmland birds. Ibis **146**:22–31.

Fuller RJ, Trevelyan RJ, Hudson RW. 1997. Landscape composition models for breeding bird populations in lowland English farmland over a 20 year period. Ecography **20**:295–307.

Fuller RM, Devereux BJ, Gillings S, Amable GS. 2005. Indices of bird-habitat preference from field surveys of birds and remote sensing of land cover: a study of south-eastern England with wider implications for conservation and biodiversity assessment. Global Ecology and Biogeography **14**:223–239.

Fuller RM, Devereux BJ, Gillings S, Hill RA, Amable GS. 2007. Bird distributions relative to remotely sensed habitats in Great Britain: Towards a framework for national modelling. Journal of Environmental Management **84**:586–605.

Gardner E, Hesselberg T, Grabowska-Zhang A, Hughes J. 2019. The effect of woodland area on avian community composition in a fragmented southern UK landscape and associated management recommendations. Bird Study **66**:293–305.

Geiger F et al. 2010. Landscape composition influences farm management effects on farmland birds in winter: A pan-European approach. Agriculture, Ecosystems and Environment **139**:571–577.

Geijzendorffer IR et al. 2016. How much would it cost to monitor farmland biodiversity in Europe? Journal of Applied Ecology **53**:140–149.

Gibbons JM, Nicholson E, Milner-Gulland EJ, Jones JPG. 2011. Should payments for biodiversity conservation be based on action or results? Journal of Applied Ecology **48**:1218–1226.

Gillings S, Austin GE, Fuller RJ, Sutherland WJ. 2006. Distribution shifts in wintering Golden Plover Pluvialis apricaria and Lapwing Vanellus vanellus in Britain. Bird Study **53**:274–284.

Gillings S, Fuller RJ, Henderson ACB. 1998. Avian community composition and patterns of bird distribution within birch-heath mosaics in north-east Scotland. Ornis Fennica **75**:27–37.

Gillings S, Newson SE, Noble DG, Vickery JA. 2005. Winter availability of cereal stubbles attracts declining farmland birds and positively influences breeding population trends. Proceedings of the Royal Society B: Biological Sciences **272**:733–739.

Gillings S, Wilson AM, Conway GJ, Vickery JA, Fuller RJ. 2008. Distribution and abundance of birds and their habitats within the lowland farmland of Britain in winter. Bird Study **55**:8–22.

Gilroy J, Woodcock P, Edwards F, Wheeler C, Medina Uribe C, Haugaasen T, Edwards D, 2014 U. 2014. Optimizing carbon storage and biodiversity protection in tropical agricultural landscapes. Global Change Biology **20**:2162–2172.

Godfray HCJ, Garnett T, Charles H, Godfray HCJ, Garnett T. 2014. Food security and sustainable intensification. Philosophical Transactions of the Royal Society B: Biological Sciences **369**.

Gollier C. 2010. Ecological discounting. Journal of Economic Theory **145**:812–829.

Gómez-Limón JA, Gutiérrez-Martín C, Villanueva AJ. 2019. Optimal Design of Agri-environmental Schemes under Asymmetric Information for Improving Farmland Biodiversity. Journal of Agricultural Economics **70**:153–177.

Gowdy J, Howarth RB, Tisdell C, Arnoldus P, Hansjürgens B, Hepburn C, McNeely JA, Mäler K-G, Mäler K-G, Hansjürgens B. 2010. Chapter 6: Discounting, ethics, and options for maintaining biodiversity and ecosystem integrity. Pages 257–284 The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations.

Green Alliance. 2022. Natural Capital: The Battle for Control.

Green RE, Cornell SJ, Scharlemann JPW, Balmford A. 2005. Farming and the fate of wild nature. Science **307**:550–555.

Green RE, Osborne PE, Sears EJ. 1994. The Distribution of Passerine Birds in Hedgerows During the Breeding Season in Relation to Characteristics of the Hedgerow and Adjacent Farmland. Journal of Applied Ecology **31**:677–692.

Gregory RD, Baillie SR. 1998. Large-scale habitat use of some declining British birds. Journal of Applied Ecology **35**:785–799.

Gregory RD, van Strien A, Vorisek P, Gmelig Meyling AW, Noble DG, Foppen RP., Gibbons DW. 2005. Developing indicators for European birds. Philosophical Transactions of the Royal Society B:

Biological Sciences **360**:269–288.

Greiner R, Bliemer M, Ballweg J. 2014. Design considerations of a choice experiment to estimate likely participation by north Australian pastoralists in contractual biodiversity conservation. Journal of Choice Modelling **10**:34–45.

Griscom BW et al. 2017. Natural climate solutions. Proceedings of the National Academy of Sciences of the United States of America **114**:11645–11650.

Groom B, Drupp MA, Freeman MC, Nesje F. 2022. The Future, Now: A Review of Social Discounting. Annual Review of Resource Economics.

Groom B, Koundouri P, Nauges C, Thomas A. 2008. The story of the moment: risk averse cypriot farmers respond to drought management. Applied Economics **40**:315–326.

Hailu A, Schilizzi S. 2004. Are Auctions More Efficient Than Fixed Price Schemes When Bidders Learn? Australian Journal of Management **29**:147–168.

Hancock MH, Wilson JD. 2003. Winter habitat associations of seed-eating passerines on Scottish farmland. Bird Study **50**:116–130.

Hanley N, Banerjee S, Lennox GD, Armsworth PR. 2012. How should we incentivize private landowners to 'produce' more biodiversity? Oxford Review of Economic Policy **28**:93–113.

Hanley N, Bell D, Alvarez-Farizo B. 2003. Valuing the Benefits of Coastal Water Quality Improvements Using Contingent and Real Behaviour. Environmental and Resource Economics **24**:273–285.

Hanley N, Wright RE, Adamowicz V. 1998. Using choice experiments to value the environment. Environmental and Resource Economics **11**:413–428.

Harris SJ, Massimino D, Eaton MA, Gillings S, Noble DG, Balmer DE, Pearce-Higgins JW, Woodcock P. 2019. The Breeding Bird Survey 2018. Page BTO Research Report 717.

Harrison PJ, Buckland ST, Yuan Y, Elston DA, Brewer MJ, Johnston A, Pearce-Higgins JW. 2014. Assessing trends in biodiversity over space and time using the example of British breeding birds. Journal of Applied Ecology **51**:1650–1660.

Hart R, Latacz-Lohmann U. 2005. Combating moral hazard in agri-environmental schemes: a multiple-agent approach. European Review of Agriculture Economics **32**:75–91.

Haukoos JS, Lewis RJ. 2005. Advanced statistics: Bootstrapping confidence intervals for statistics with "difficult" distributions. Academic Emergency Medicine **12**:360–365.

Haya B, Cullenward D, Strong AL, Grubert E, Heilmayr R, Sivas DA, Wara M. 2020. Managing uncertainty in carbon offsets: insights from California's standardized approach. Climate Policy **20**:1112–1126.

Hayhow DB et al. 2017. The State of the UK's Birds 2017.

Henderson IG, Cooper J, Fuller RJ, Vickery J. 2000. The relative abundance of birds on set-aside and neighbouring fields in summer. Journal of Applied Ecology **37**:335–347.

Henderson IG, Holland JM, Storkey J, Lutman P, Orson J, Simper J. 2012. Effects of the proportion and spatial arrangement of un-cropped land on breeding bird abundance in arable rotations. Journal of Applied Ecology **49**:883–891.

Henderson IG, Vickery JA, Carter N. 2004. The use of winter bird crops by farmland birds in lowland England. Biological Conservation **118**:21–32.

Hensher DA. 2010. Hypothetical bias, choice experiments and willingness to pay. Transportation Research Part B: Methodological **44**:735–752.

Hess S. 2014. Latent class structures: Taste heterogeneity and beyond. Pages 311–329 Handbook of Choice Modelling.

Hess S, Palma D. 2019. Apollo Manual.

Hewson CM, Amar A, Lindsell JA, Thewlis RM, Buttler S, Smith K, Fuller RJ. 2007. Recent changes in bird populations in British broadleaved woodland. Ibis **149**:14–28.

Hewson CM, Fuller RJ. 2006. Little evidence of temporal changes in edge-use by woodland birds in southern England. Bird Study **53**:323–327.

Hewson CM, Noble DG. 2009. Population trends of breeding birds in British woodlands over a 32-year period: relationships with food, habitat use and migratory behaviour. Ibis **151**:464–486.

Hiley JR, Bradbury RB, Thomas CD. 2014. Introduced and natural colonists show contrasting patterns of protected area association in UK wetlands. Diversity and Distributions **20**:943–951.

Hill DA, Lambton S, Proctor I, Bullock I. 1991. Winter bird communities in woodland in the Forest of Dean, England, and some implications of livestock grazing. Bird Study **38**:57–70.

Hinsley SA, Bellamy PE, Enoksson B, Fry G, Gabrielsen L, Mccollin D, Schotman A. 1998. Geographical and Land-Use Influences on Bird Species Richness in Small Woods in Agricultural Landscapes. Global Ecology and Biogeography Letters **7**:125–135.

Hinsley SA, Bellamy PE, Hill R, Hinsley SA, Bellamy PE, Hill RA, Dawson A. 2002. Birds and habitat quality: effects of wood size and spatial arrangement in the landscape.

Hinsley SA, Bellamy PE, Newton I. 1995a. Bird Species Turnover and Stochastic Extinction in Woodland Fragments. Ecography **18**:41–50.

Hinsley SA, Bellamy PE, Newton I, Sparks TH. 1995b. Habitat and Landscape Factors Influencing the Presence of Individual Breeding Bird Species in Woodland Fragments. Journal of Avian Biology **26**:94–104.

Hinsley SHA, Redhead JW, Bellamy PE, Broughton RK, Hill RA, Heard MS, Pywell RF. 2010. Testing agri-environment delivery for farmland birds at the farm scale: The Hillesden experiment. Ibis **152**:500–514.

HM Treasury. 2018. The Green Book: Central Government Guidance on Appraisal and Evaluation. Page UK Government.

IPCC. 2019. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Page Volume 4: Agriculture, Forestry and Other Land Use.

IPCC. 2020. Summary for Policymakers — Special Report on Climate Change and Land.

IUCN. 2020. The IUCN Red List of threatened species.

Iyer P, Bozzola M, Hirsch S, Meraner M, Finger R. 2020. Measuring Farmer Risk Preferences in Europe: A Systematic Review. Journal of Agricultural Economics **71**:3–26.

Jackson SF, Austin GE, Armitage MJS. 2006. Surveying waterbirds away from major waterbodies: Implications for waterbird population estimates in Great Britain. Bird Study **53**:105–111.

Kamp J, Urazaliev R, Balmford A, Donald PF, Green RE, Lamb AJ, Phalan B. 2015. Agricultural development and the conservation of avian biodiversity on the Eurasian steppes: a comparison of land-sparing and land-sharing approaches. Journal of Applied Ecology **52**:1578–1587.

Keane A, Jones JPG, Edwards-Jones G, Milner-Gulland EJ. 2008. The sleeping policeman: understanding issues of enforcement and compliance in conservation. Animal Conservation **11**:75–82.

Kershaw M, Cranswick PA. 2003. Numbers of wintering waterbirds in Great Britain, 1994/1995-1998/1999: I. Wildfowl and selected waterbirds. Biological Conservation **111**:91–104.

Kettunen M, Carter O, Gantioler S, Baldock Torkler DP, Arroyo Schnell A, Baumueller A, Gerritsen Rayment EM, Daly E, Pieterse M. 2011. Assessment of the Natura 2000 co-financing arrangements of the EU financing instrument.

Khanna M, Ando AW. 2009. Science, economics and the design of agricultural conservation programmes in the US. Journal of Environmental Planning and Management **52**:575–592.

Kindred DR, Sylvester-Bradley R, Berry PM. 2008. Effects of nitrogen fertiliser use on green house gas emissions and land use change. Aspects of Applied Biology **88**:53–56.

Kirby JS. 1995. Winter population estimates for selected waterfowl species in Britain. Biological Conservation **73**:189–198.

Kirby KJ, Thomas RC, Key RS, McLean IFG, Hodgetts N. 1995. Pasture-woodland and its conservation in Britain. Biological Journal of the Linnean Society **56**:135–153.

Kirchhoff T. 2012. Pivotal cultural values of nature cannot be integrated into the ecosystem services framework. Proceedings of the National Academy of Sciences **109**:E3146.

Kleijn D et al. 2006. Mixed biodiversity benefits of agri-environment schemes in five European countries. Ecology Letters **9**:243–254.

Kleijn D, Sutherland WJ. 2003. How effective are European schemes in and promoting conserving biodiversity? Journal of Applied Ecology **40**:947–969.

Kuhfuss L, Préget R, Thoyer S, de Vries FP, Hanley N. 2022. Enhancing spatial coordination in payment for ecosystem services schemes with non-pecuniary preferences. Ecological Economics **192**:107271.

Kuhfuss L, Préget R, Thoyer S, Hanley N. 2016a. Nudging farmers to enrol land into agri-environmental schemes: the role of a collective bonus. European Review of Agricultural Economics **43**:609–636.

Kuhfuss L, Préget R, Thoyer S, Hanley N, Le Coent P, Désolé M. 2016b. Nudges, Social Norms, and Permanence in Agri-environmental Schemes. Land Economics **92**:641–655.

Kyrkos A, Wilson JD, Fuller RJ. 1998. Farmland habitat change and abundance of Yellowhammers Emberiza citrinella: an analysis of Common Birds Census data. Bird Study **45**:232–246.

Laidlaw RA, Smart J, Smart MA, Gill JA. 2015. The influence of landscape features on nest predation rates of grassland-breeding waders. Ibis **157**:700–712.

Laidlaw RA, Smart J, Smart MA, Gill JA. 2017. Scenarios of habitat management options to reduce predator impacts on nesting waders. Journal of Applied Ecology **54**:1219–1229.

Lamb A et al. 2016a. The potential for land sparing to offset greenhouse gas emissions from agriculture. Nature Climate Change **6**:488–492.

Lamb A, Balmford A, Green RE, Phalan B. 2016b. To what extent could edge effects and habitat fragmentation diminish the potential benefits of land sparing ? Biological Conservation **195**:264–271.

Lamb A, Finch T, Pearce-Higgins JW, Ausden M, Balmford A, Feniuk C, Hirons G, Massimino D, Green RE. 2019. The consequences of land sparing for birds in the United Kingdom. Journal of Applied Ecology **56**:1870–1881.

Lambin EF, Meyfroidt P. 2011. Global land use change, economic globalization, and the looming land scarcity. Proceedings of the National Academy of Sciences of the United States of America **108**:3465–3472.

Lancaster KJ. 1966. A new approach to consumer theory. Journal of Political Economy **74**:132–157.

Lang B. 2020. Crop Production in England. Page Farm Business Survey 2018/19: Crop Production in England.

Lastra-Bravo XB, Hubbard C, Garrod G, Tolón-Becerra A. 2015. What drives farmers' participation in EU agri-environmental schemes?: Results from a qualitative meta-analysis. Environmental Science & Policy **54**:1–9.

Latacz-Lohmann U, Schilizzi S. 2005. Auctions for Conservation Contracts: a Review of the Theoretical and Empirical Literature. Report on Project No: UKL/001/05 to the Scottish Executive

Environment and Rural Affairs Department.

Laurance WF et al. 2011. The fate of Amazonian forest fragments: A 32-year investigation. Biological Conservation **144**:56–67.

Leigh SG, Smart J, Gill JA. 2017. Impacts of grassland management on wader nest predation rates in adjacent nature reserves. Animal Conservation **20**:61–71.

Lenzen M, Moran D, Kanemoto K, Foran B, Lobefaro L, Geschke A. 2012. International trade drives biodiversity threats in developing nations. Nature 2012 486:7401 **486**:109–112.

Liu Z, Xu J, Yang X, Tu Q, Hanley N, Kontoleon A. 2019. Performance of Agglomeration Bonuses in Conservation Auctions: Lessons from a Framed Field Experiment. Environmental and Resource Economics **73**:843–869.

Luce DR. 1959. Individual Choice Behavior: A Theoretical Analysis.

Macdonald DW, Johnson PJ. 1995. The Relationship Between Bird Distribution and the Botanical and Structural Characteristics of Hedges. Journal of Applied Ecology **32**:492–505.

MacDonald MA et al. 2019. Have Welsh agri-environment schemes delivered for focal species? Results from a comprehensive monitoring programme. Journal of Applied Ecology **56**:812–823.

MacDonald MA, Bolton M. 2008. Predation of Lapwing Vanellus vanellus nests on lowland wet grassland in England and Wales: Effects of nest density, habitat and predator abundance. Journal of Ornithology **149**:555–563.

Malpas LR, Kennerley RJ, Hirons GJM, Sheldon RD, Ausden M, Gilbert JC, Smart J. 2013. The use of predator-exclusion fencing as a management tool improves the breeding success of waders on lowland wet grassland. Journal for Nature Conservation **21**:37–47.

Marren P. 2016. British Wildlife | The great rewilding experiment at Knepp Castle. Available from https://www.britishwildlife.com/article/volume-27-number-5-page-333-339 (accessed January 28, 2022).

Marshall EJP, West TM, Kleijn D. 2006. Impacts of an agri-environment field margin prescription on the flora and fauna of arable farmland in different landscapes. Agriculture, Ecosystems & Environment **113**:36–44.

Mason CF. 2001. Woodland area, species turnover and the conservation of bird assemblages in lowland England. Biodiversity and Conservation **10**:495–510.

Mason CF, Macdonald SM. 1999. Winter bird numbers and land-use preferences in an arable landscape in eastern England. Bird Conservation **9**:119–127.

Mason CF, Macdonald SM. 2000. Influence of landscape and land-use on the distribution of breeding birds in farmland in eastern England. Journal of Zoology **251**:339–348.

Mccollin D. 1993. Avian Distribution Patterns in a Fragmented Wooded Landscape (North

Humberside, U.K.): The Role of Between-Patch and Within-Patch Structure. Global Ecology and Biogeography Letters **3**:48–62.

McCracken ME et al. 2015. Social and ecological drivers of success in agri-environment schemes: the roles of farmers and environmental context. Journal of Applied Ecology **52**:696–705.

Méndez V, Austin GE, Musgrove AJ, Ross-Smith VH, Hearn RD, Stroud DA, Wotton SR, Holt CA. 2015. Use of environmental stratification to derive non-breeding population estimates of dispersed waterbirds in Great Britain. Journal for Nature Conservation **28**:56–66.

Mendez V, Gill JA, Burton NHK, Austin GE, Petchey OL, Davies RG. 2012. Functional diversity across space and time: trends in wader communities on British estuaries. Diversity and Distributions **18**:356–365.

Metrics C. 2018. Ngene.

Milsom TP, Langton SD, Parkin WK, Peel S, Bishop JD, Hart JD, Moore NP. 2000. Habitat models of bird species' distribution: An aid to the management of coastal grazing marshes. Journal of Applied Ecology **37**:706–727.

Moorcroft D, Whittingham MJ, Bradbury RB, Wilson JD. 2002. The selection of stubble fields by wintering granivorous birds reflects vegetation cover and food abundance. Journal of Applied Ecology **39**:535–547.

Moreby SJ. 2002. Permanent and temporary linear habitats as food sources for the young of farmland birds. Pages 327–332 Avian Lanscape Ecology: Pure and Applied Issues in the Large-Scale Ecology of Birds.

Morgan R. 1975. Breeding Bird Communities on Chalk Downland in Wiltshire. Bird Study **22**:71–83.

Morris AJ, Whittingham MJ, Bradbury RB, Wilson JD, Kyrkos A, Buckingham DL, Evans AD. 2001. Foraging habitat selection by yellowhammers (Emberiza citrinella) nesting in agriculturally contrasting regions in lowland England. Biological Conservation **101**:197–210.

Muhammed S, Milne A, Marchant B, Griffin S, Whitmore A. 2016. Exploiting yield maps and soil management zones.

Nakagawa S, Cuthill IC. 2007. Effect size, confidence interval and statistical significance: a practical guide for biologists. Biological Reviews **82**:591–605.

National Food Strategy. 2021. The National Food Strategy: The Plan - July 2021.

Natural England. 2009. Agri-environment schemes in England 2009: A review of results and effectiveness.

Natural England. 2013. Higher Level Stewardship: Environmental Stewardship Handbook, Fourth Edition – January 2013 - NE350.

Natural England. 2018. Countryside Stewardship Manual: Woodland Creation Grant.

Neumann JL, Griffiths GH, Foster CW, Holloway GJ. 2016. The heterogeneity of wooded-agricultural landscape mosaics influences woodland bird community assemblages. Landscape Ecology **31**:1833–1848.

Newson SE, Woodburn RJWW, Noble DG, Baillie SR, Gregory RD. 2005. Evaluating the Breeding Bird Survey for producing national population size and density estimates. Bird Study **52**:42–54.

Nielsen SE, Haughland DL, Bayne E, Schieck J. 2009. Capacity of large-scale, long-term biodiversity monitoring programmes to detect trends in species prevalence. Biodiversity and Conservation **18**:2961–2978.

Nix J. 2018. John Nix Farm Management Pocketbook for 2019.

Norton LR et al. 2012. Measuring stock and change in the GB countryside for policy - Key findings and developments from the Countryside Survey 2007 field survey. Journal of Environmental Management **113**:117–127.

O’brien M, Smith KW. 1992. Changes in the status of waders breeding on wet lowland grasslands in England and Wales between 1982 and 1989. Bird Study **39**:165–176.

O’brien M, Wilson JD. 2011. Population changes of breeding waders on farmland in relation to agri-environment management. Bird Study **58**:399–408.

Ozanne A, Hogan T, Colman D. 2001. Moral hazard, risk aversion and compliance monitoring in agri-environmental policy. European Review of Agriculture Economics **28**:329–348.

Parish DMB, Sotherton NW. 2004. Game crops and threatened farmland songbirds in Scotland: A step towards halting population declines? Bird Study **51**:107–112.

Parish T, Lakhani KH, Sparks TH. 1994. Modelling the Relationship Between Bird Population Variables and Hedgerow and Other Field Margin Attributes. I. Species Richness of Winter, Summer and Breeding Birds. Journal of Applied Ecology **31**:764–775.

Parish T, Lakhani KH, Sparks TH. 1995. Modelling the Relationship Between Bird Population Variables and Hedgerow, and Other Field Margin Attributes. II. Abundance of Individual Species and of Groups of Similar Species. Journal of Applied Ecology **32**:362–371.

Pavlis ES, Terkenli TS, Kristensen SBP, Busck AG, Cosor GL. 2016. Patterns of agri-environmental scheme participation in Europe: Indicative trends from selected case studies. Land Use Policy **57**:800–812.

Pe’er G et al. 2014. EU agricultural reform fails on biodiversity. Science **344**:1090–1092.

Pe’er G et al. 2019. A greener path for the EU Common Agricultural Policy. Science **365**:449–451.

Pe’er G et al. 2020. Action needed for the EU Common Agricultural Policy to address sustainability challenges. People and Nature **2**:305–316.

Peach WJ, Baillie SR, Balmer DE. 1998. Long-term changes in the abundance of passerines in Britain and Ireland as measured by constant effort mist-netting **45**:257–275.

Peach WJ, Buckland ST, Baillie SR. 1996. The use of constant effort mist-netting to measure between-year changes in the abundance and productivity of common passerines. Bird Study **43**:142–156.

Perkins AJ, Maggs HE, Wilson JD. 2008. Winter bird use of seed-rich habitats in agri-environment schemes. Agriculture, Ecosystems and Environment **126**:189–194.

Perkins AJ, Whittingham MJ, Morris AJ, Bradbury RB. 2002. Use of field margins by foraging yellowhammers Emberiza citrinella. Agriculture, Ecosystems and Environment **93**:413–420.

Perrins CM, Overall R. 2001. Effect of increasing numbers of deer on bird populations in Wytham Woods, central England. Forestry **74**:299–309.

Phalan B, Onial M, Balmford A, Green RE. 2011. Reconciling Food Production and Biodiversity Conservation: Land Sharing and Land Sparing Compared. Science **333**:1289–1291.

Proffitt FM, Newton I, Wilson JD, Siriwardena GM. 2004. Bullfinch Pyrrhula pyrrhula breeding ecology in lowland farmland and woodland: comparisons across time and habitat. Ibis **146**:78–86.

Raatz L, Bacchi N, Walzl KP, Glemnitz M, Muller MEH, Joshi J, Scherber C. 2019. How much do we really lose? - Yield losses in the proximity of natural landscape elements in agricultural landscapes. Ecology and Evolution **9**:7838–7848.

Rayment M, White A, Dickie I, Pieterse M. 2011. Costing potential actions to offset the impact of development on biodiversity.

Rehfisch MM, Austin GE, Armitage MJS, Atkinson PW, Holloway SJ, Musgrove AJ, Pollitt MS. 2003. Numbers of wintering waterbirds in Great Britain and the Isle of Man (1994/1995-1998/1999): II. Coastal waders (Charadrii). Biological Conservation **112**:329–341.

Rhymer CM, Robinson RA, Smart J, Whittingham MJ. 2010. Can ecosystem services be integrated with conservation? A case study of breeding waders on grassland. Ibis **152**:698–712.

Ritchie H. 2019. Half of the world’s habitable land is used for agriculture. Available from https://ourworldindata.org/global-land-for-agriculture (accessed September 6, 2021).

Robinson J, Pollitt MS. 2002. Sources and extent of human disturbance to waterbirds in the UK: an analysis of Wetland Bird Survey data, 1995/96 to 1998/99. Bird Study **49**:205–211.

Robinson RA, Hart JD, Holland JM, Parrott D. 2004. Habitat use by seed-eating birds: A scale-dependent approach. Ibis **146**:87–98.

Rose JM, Bliemer MCJ. 2013. Sample size requirements for stated choice experiments. Transportation **40**:1021–1041.

RPA. 2012. Spying on Europe’s farms with satellites and drones - BBC News. Available from https://www.bbc.co.uk/news/world-europe-16545333 (accessed March 3, 2021).

RSPB. 2020a. Written evidence submitted by the RSPB.

RSPB. 2020b. Farming for wildlife: Scrub management.

RSPB. 2021. Farming for wildlife: Wild bird seed mixture.

RSPB. 2022. Over-wintered stubble.

Ruto E, Garrod G. 2009. Investigating farmers' preferences for the design of agri-environment schemes: a choice experiment approach. Journal of Environmental Planning and Management **52**:631–647.

Sadlier G, Flytjaer R, Sabri F, Robin N. 2018. Value of satellite-derived Earth Observation capabilities to the UK Government today and by 2020.

Sage R, Cunningham M, Haughton AJ, Mallott MD, Bohan DA, Riche A, Karp A. 2010. The environmental impacts of biomass crops: use by birds of miscanthus in summer and winter in southwestern England. Ibis **152**:487–499.

Sage RB, Hoodless AN, Hewson CM, Wilson S, Le Clare C, Marchant JH, Draycott RAH, Fuller RJ. 2011. Assessing breeding success in common woodland birds using a novel method. Bird Study **58**:409–420.

Sage RB, Parish DMB, Woodburn MIA, Thompson PGL. 2005. Songbirds using crops planted on farmland as cover for game birds. European Journal of Wildlife Research **51**:248–253.

Sage RB, Wilson S, Powell T. 2015. Using fledged brood counts of hedgerow birds to assess the effect of summer agri-environment scheme options. Ecological Indicators **57**:376–383.

Savills UK. 2020. Rural Land Values. Available from https://www.savills.co.uk/property-values/rural-land-values.aspx (accessed March 27, 2020).

Scheper J, Holzschuh A, Kuussaari M, Potts SG, Rundlöf M, Smith HG, Kleijn D. 2013. Environmental factors driving the effectiveness of European agri-environmental measures in mitigating pollinator loss - a meta-analysis. Ecology Letters **16**:912–920.

Schmidt JU, Eilers A, Schimkat M, Krause-Heiber J, Timm A, Siegel S, Nachtigall W, Kleber A. 2017. Factors influencing the success of within-field AES fallow plots as key sites for the Northern Lapwing Vanellus vanellus in an industrialised agricultural landscape of Central Europe. Journal for Nature Conservation **35**:66–76.

Schöttker O, Johst K, Drechsler M, Wätzold F. 2016. Land for biodiversity conservation — To buy or borrow? Ecological Economics **129**:94–103.

Schöttker O, Wätzold F. 2018. Buy or lease land? Cost-effective conservation of an oligotrophic lake in a Natura 2000 area. Biodiversity and Conservation **27**:1327–1345.

Seddon N, Chausson A, Berry P, Girardin CAJ, Smith A, Turner B. 2020. Understanding the value and limits of nature-based solutions to climate change and other global challenges. Philosophical Transactions of the Royal Society B **375**:20190120.

Sheldon R. 2002. Factors affecting the distribution, abundance and chick survival of the Lapwing (Vanellus vanellus).

Sheldon R, Bolton M, Gillings S, Wilson A. 2004. Conservation management of Lapwing Vanellus vanellus on lowland arable farmland in the UK. Ibis **146**:41–49.

Shrubb M, Lack PC, Greenwood JJD. 1991. The numbers and distribution of Lapwings V. vanellus nesting in England and Wales in 1987. Bird Study **38**:20–37.

Siriwardena GM. 2010. The importance of spatial and temporal scale for agri-environment scheme delivery. Ibis **152**:515–529.

Siriwardena GM, Calbrade NA, Vickery JA. 2008. Farmland birds and late winter food: Does seed supply fail to meet demand? Ibis **150**:585–595.

Siriwardena GM, Cooke IR, Sutherland WJ. 2012. Landscape, cropping and field boundary influences on bird abundance. Ecography **35**:162–173.

Siriwardena GM, Crick HQPP, Baillie SR, Wilson JD, Baillic SR, Wilson JD. 2000. Agricultural land-use and the spatial distribution of granivorous lowland farmland birds. Ecography **23**:702–719.

Siriwardena GM, Stevens DK, Anderson GQA, Vickery JA, Calbrade NA, Dodd S. 2007. The effect of supplementary winter seed food on breeding populations of farmland birds: Evidence from two large-scale experiments. Journal of Applied Ecology **44**:920–932.

Smart J, Bolton M, Hunter F, Quayle H, Thomas G, Gregory RD. 2013. Managing uplands for biodiversity: Do agri-environment schemes deliver benefits for breeding lapwing *Vanellus vanellus* ? Journal of Applied Ecology **50**:794–804.

Smart J, Wotton SR, Dillon IA, Cooke AI, Diack I, Drewitt AL, Grice P V., Gregory RD. 2014. Synergies between site protection and agri-environment schemes for the conservation of waders on lowland wet grasslands. Ibis **156**:576–590.

Smith LG, Kirk GJDD, Jones PJ, Williams AG. 2019. The greenhouse gas impacts of converting food production in England and Wales to organic methods. Nature Communications 2019 10:1 **10**:1–10.

Smith P, Milne R, Powlson DS, Smith JU, Falloon P, Coleman K. 2000. Revised estimates of the carbon mitigation potential of UK agricultural land. Soil Use and Management **16**:293–295.

Springmann M et al. 2018. Options for keeping the food system within environmental limits. Nature 2018 562:7728 **562**:519–525.

Stern N. 2008. The economics of climate change. American Economic Review **98**:1–37.

Stoate C, Henderson IG, Parish DMB. 2004. Development of an agri-environment scheme option:

Seed-bearing crops for farmland birds. Ibis **146**:203–209.

Stoate C, Szczur J, Aebischer NJ. 2003. Winter use of wild bird cover crops by passerines on farmland in northeast England. Bird Study **50**:15–21.

Strutt & Parker. 2021. English Estates & Farmland Market Review.

Sullivan MJP, Newson SE, Pearce-Higgins JW. 2015. Using habitat-specific population trends to evaluate the consistency of the effect of species traits on bird population change. Biological Conservation **192**:343–352.

Sullivan MJP, Pearce-Higgins JW, Newson SE, Scholefield P, Brereton T, Oliver TH. 2017. A national-scale model of linear features improves predictions of farmland biodiversity. Journal of Applied Ecology **54**:1776–1784.

Taylor IR, Grant MC. 2004. Long-term trends in the abundance of breeding Lapwing Vanellus vanellus in relation to land-use change on upland farmland in southern Scotland. Bird Study **51**:133–142.

Thaxter CB, Joys AC, Gregory RD, Baillie SR, Noble DG. 2010. Hypotheses to explain patterns of population change among breeding bird species in England. Biological Conservation **143**:2006–2019.

Thompson DBA, MacDonald AJ, Marsden JH, Galbraith CA. 1995, January 1. Upland heather moorland in Great Britain: A review of international importance, vegetation change and some objectives for nature conservation.

Tichit M, Doyen L, Lemel JY, Renault O, Durant D. 2007. A co-viability model of grazing and bird community management in farmland. Ecological Modelling **206**:277–293.

Tilman D, Balzer C, Hill J, Befort BL. 2011. Global food demand and the sustainable intensification of agriculture. Proceedings of the National Academy of Sciences of the United States of America **108**:20260–20264.

Tilman D, Clark M. 2014. Global diets link environmental sustainability and human health. Nature **515**:518–522.

Tilman D, Clark M, Williams DR, Kimmel K, Polasky S, Packer C. 2017. Future threats to biodiversity and pathways to their prevention. Nature **546**:73–81.

Train KE. 2009. Discrete Choise Methods with Simulation - Introduction. Discrete choice methods with simulation, 2nd edition:1–8.

Tucker GM. 1992. Effects of Agricultural Practices on Field Use by Invertebrate-Feeding Birds in Winter. Journal of Applied Ecology **29**:779–790.

Usher MB, Thompson DBA. 1993. Variation in the upland heathlands of Great Britain: Conservation importance. Biological Conservation **66**:69–81.

Van Zanten HHE, Herrero M, Van Hal O, Röös E, Muller A, Garnett T, Gerber PJ, Schader C, De Boer IJM. 2018. Defining a land boundary for sustainable livestock consumption. Global Change Biology **24**:4185–4194.

Vanhinsbergh D, Gough S, Fuller RJ, Brierley EDR. 2002. Summer and winter bird communities in recently established farm woodlands in lowland England. Agriculture, Ecosystems and Environment **92**:123–136.

Vedel SE, Jacobsen JB, Thorsen BJ. 2015. Contracts for afforestation and the role of monitoring for landowners' willingness to accept. Forest Policy and Economics **51**:29–37.

Vickery JA, Sutherland WJ, O'Brien M, Watkinson AR, Yallop A. 1997. Managing coastal grazing marshes for breeding waders and overwintering geese: Is there a conflict? Biological Conservation **79**:23–34.

Vickery JA, Tallowin JR, Feber RE, Asteraki EJ, Atkinson PW, Fuller RJ, Brown VK. 2001. The management of lowland neutral grasslands in britain: Effects of agricultural practices on birds and their food resources. Journal of Applied Ecology **38**:647–664.

Victor D, Cullenward D. 2007. Making carbon markets work. Scientific American **297**:70–77.

Villanueva AJ, Gómez-Limón JA, Arriaza M, Rodríguez-Entrena M. 2015. The design of agri-environmental schemes: Farmers' preferences in southern Spain. Land Use Policy **46**:142–154.

Villanueva AJ, Rodríguez-Entrena M, Arriaza M, Gómez-Limón JA. 2016. Heterogeneity of farmers' preferences towards agri-environmental schemes across different agricultural subsystems. Journal of Environmental Planning and Management **60**:684–707.

Wakeham-Dawson A, Smith KW. 1999. Birds and lowland grassland management practices in the UK: an overview. Pages 77–88 Ecology and conservation of lowland farmland birds.

Walker MP, Dover JW, Hinsley SA, Sparks TH. 2005. Birds and green lanes: Breeding season bird abundance, territories and species richness. Biological Conservation **126**:540–547.

Wara M. 2007. Is the global carbon market working? Nature **445**:595–596.

Westbury DB et al. 2011. Plant and invertebrate resources for farmland birds in pastoral landscapes. Agriculture, Ecosystems and Environment **142**:266–274.

White T, Petrovan S, Christie A, Martin PA, Sutherland W. 2022. What is the Price of Conservation? A Review of the Status Quo and Recommendations for Improving Cost Reporting. BioScience.

Whittingham MJ, Devereux CL, Evans AD, Bradbury RB. 2006. Altering perceived predation risk and food availability: Management prescriptions to benefit farmland birds on stubble fields. Journal of Applied Ecology **43**:640–650.

Whittingham MJ, Swetnam RD, Wilson JD, Chamberlain DE, Freckleton RP. 2005. Habitat selection by yellowhammers Emberiza citrinella on lowland farmland at two spatial scales: implications for

conservation management. Journal of Applied Ecology **42**:270–280.
Williams DR, Alvarado F, Green RE, Manica A, Phalan B, Balmford A. 2017. Land-use strategies to balance livestock production, biodiversity conservation and carbon storage in Yucatán, Mexico. Global Change Biology **23**:5260–5272.
Williams DR, Clark M, Buchanan GM, Francesco Ficetola G, Rondinini C, Tilman D. 2021. Proactive conservation to prevent habitat losses to agricultural expansion. Nature Sustainability **4**:314–322.
Williamson K. 1974. Breeding birds in the deciduous woodlands of mid-argyll, scotland. Bird Study **21**:29–44.
Wilson A, Vickery J, Pendlebury C. 2007. Agri-environment schemes as a tool for reversing declining populations of grassland waders: Mixed benefits from Environmentally Sensitive Areas in England. Biological Conservation **136**:128–135.
Wilson AM, Vickery JA, Brown A, Langston RHW, Smallshire D, Wotton S, Vanhinsbergh D. 2005. Changes in the numbers of breeding waders on lowland wet grasslands in England and Wales between 1982 and 2002. Bird Study **52**:55–69.
Wilson AM, Vickery JA, Browne SJ. 2001. Numbers and distribution of Northern Lapwing Vanellus vanellus breeding in England and Wales in 1998, Bird Study. Bird Study **48**:2–17.
Wilson GA, Hart K. 2016. Financial Imperative or Conservation Concern? EU Farmers' Motivations for Participation in Voluntary Agri-Environmental Schemes: Environment and Planning A: Economy and Space **32**:2161–2185.
World Bank. 2021. Agriculture, forestry, and fishing, value added (% of GDP) - United Kingdom | Data. Available from https://data.worldbank.org/indicator/NV.AGR.TOTL.ZS?locations=GB (accessed August 26, 2021).
WWF. 2020. Living planet report 2020: bending the curve of biodiversity loss.
WWF, RSPB. 2020. Riskier business: The UK's overseas land footprint.
Zmihorski M, Part T, Gustafson T, Berg A. 2015. Effects of water level and grassland management on alpha and beta diversity of birds in restored wetlands. Journal of Applied Ecology **53**:587–595.

Appendices

Appendix A: Conservation benefit of interventions

We estimated the benefit delivered by our studied sharing and sparing interventions using existing literature, as detailed below.

Method

We identified sharing and sparing interventions for delivering the target outcomes (increased populations of yellowhammers, bullfinches and lapwings as well as reduced net carbon emissions) through consultation with experts and the literature (Table 2.1) and sought to quantify the outcome delivered per unit area of each intervention. These estimates are based on a different method for the biodiversity outcomes (where effects are studied by sampling the target species on intervention and non-intervention land) and for reducing net carbon emissions (where effects are quantified from empirical calculations). For biodiversity, we systematically searched the literature to identify all relevant, comparable studies and collated these into a single estimate for each intervention. For interventions that lower net carbon emissions, we took estimates from a single paper for each intervention, typically those used in IPCC reports.

a) Quantifying the effect of interventions that deliver biodiversity

We first systematically searched the white literature, and thoroughly searched the grey literature, for studies investigating the effects of the interventions in Table 2.1. All these interventions increase the populations of the target species by addressing a limiting life-history parameter; in this way, interventions need not meet all of a species' needs year-round. In searching the white literature, we identified search terms for each intervention that would return relevant studies (Table A1) and entered these into the Web of Science in April 2020. We screened the returned papers plus any relevant grey literature for relevance by scanning abstracts (Table A2). To be relevant, papers must have studied our target species in the UK. We read all the papers with promising abstracts and noted those that reported, and then those that quantified, the effects of interventions (Table A3). Finally, we judged whether a paper was 'usable'. Reasons for not being usable were varied, but included multiple interventions being implemented at once or study of a metric other than population density. The effects of sharing interventions were often measured with study of population growth rates, but these studies consistently returned smaller estimates of effects compared to studies estimating the effects of sharing from differences in population density. To ensure our estimates of sharing and sparing effects were comparable, in estimating the effects of sharing and sparing interventions we excluded studies of population growth rate, and only used studies of population density. From those that we deemed usable, we extracted estimates in terms of the additional birds delivered per unit area of the intervention. Few additional usable studies were found in the grey literature, though three estimates from the grey literature did contribute to the final estimates (Table A3).

To collate the identified estimates into a single estimate, we first addressed the issue that most of the identified studies were conducted during the breeding season, but several were conducted during the winter. All bird populations are subject to annual mortality, so population densities for species resident in an area would be expected to be highest at the end of the breeding season and then to decline through the autumn and winter to reach a minimum during the following breeding season. To compare results from studies of the same species conducted at different times of year, we adjusted density estimates according to species-specific estimates of annual adult survival probability, S. We obtained these from the compilation in Bird et al. (2020): 0.43 for bullfinch, 0.79 for lapwing and 0.53 for yellowhammer. To do so, we established the midpoint of survey dates in each winter study t_W and breeding season study t_B. The latest midpoint date at which sampling occurred during the breeding season in any study was the 31 May and there would be very few full-grown young of the year present in populations of these species by that date. We therefore adjusted the estimated densities to those expected on 31 May. To do this, we assumed that the daily probability of death was constant through the year. For each density survey, we determined the number of days elapsed t_D between the survey midpoint date and 31 May and calculated the adjusted density D_{adj} from the observed density D as:

$$D_{adj} = D.S^{(tD/365.25)}.$$

These results are presented in Table A4. We note that populations of birds surveyed in winter will consist of a mixture of adults and first-year birds hatched in the previous breeding season. For most species, we would expect the annual mortality rate of first-year birds to be somewhat higher than that for adults. Hence, our method will probably overestimate adjusted densities derived from winter surveys to a certain extent because we used only the adult value for S. However, in the absence of detailed data on the age composition of the population in winter, we suggest that our method is a reasonable approximation.

From Table A4, we calculated the additional birds delivered per-ha of each intervention by subtracting the 'without intervention' density from the 'with intervention' density. Where papers did not report the without-intervention density, we used densities from Newson et al. (2005) assuming 87.5% of land on arable farms is tilled and 12.5% is improved grassland (based on national averages for arable farms in England (Defra 2019d)); this gives densities of 0.013, 0.017 and 0.23 birds/ha for bullfinches, lapwings and yellowhammers respectively.

We collated these estimates into a single estimate for each intervention-species combination by finding the mean (Figure A1). In doing so, we assumed present-day effects were no different to those detected in the years in which these studies were conducted. Furthermore, we assumed the benefit delivered per unit area did not change according to the patch size of the intervention. For the sparing interventions, we excluded some less relevant studies before calculating the mean. For scrub and wet grassland, we excluded studies that used BBS sampling which allocated land to broad habitat classes including habitat types that would not benefit the target species. For example, the BBS habitat class for scrub includes “young, regenerated woodland, downland scrub, heath scrub, young coppice, young plantation and clear-felled woodland” whilst the class of wet semi-natural grassland includes “machair, water meadow, grazing marsh, reed swap, saltmarsh and other open marsh” (Newson et al. 2005). These definitions are far broader than the habitat specified to participants of the choice experiment, which involved management to prevent succession. We did use BBS-based estimates in determining the effect of woodland creation on bullfinches since there were no alternatives available; therefore, this effect may be underestimated. We assumed these densities would arise within the 20-year timescales studied, following evidence of the rapid recovery of birds on previously farmed land restored to natural habitat (Vanhinsbergh et al. 2002; Eglington et al. 2007; Marren 2016)

Table A1. The search terms entered into Web of Science for each intervention to identify relevant papers.

Intervention	Search terms
Stubble, spring cropping	(Spring cropping OR spring sowing OR stubble OR winter cover) AND (Britain OR UK OR United Kingdom OR England OR Wales OR Scotland) AND (bird OR bullfinch OR yellowhammer OR lapwing)
Winter bird cover	(Britain OR UK OR United Kingdom OR England OR Wales OR Scotland) AND (bird OR bullfinch OR yellowhammer) AND (winter bird cover OR winter bird food OR wild bird cover OR wild bird food)
Fallow plots	(Britain OR UK OR United Kingdom OR England OR Wales OR Scotland) AND (bird OR lapwing) AND (fallow OR lapwing plots)
Hedgerow creation	(Britain OR UK OR United Kingdom OR England OR Wales OR Scotland) AND (bird OR bullfinch OR yellowhammer) AND (hedge OR hedges OR hedgerow OR hedgerows)
Scrub creation	(Britain OR UK OR United Kingdom OR England OR Wales OR Scotland) AND (bird OR bullfinch OR yellowhammer) AND (scrub OR scrubland)
Wet grassland creation	(Britain OR UK OR United Kingdom OR England OR Wales OR Scotland) AND (bird OR lapwing) AND (wet grassland OR wetland)
Woodland creation	(Britain OR UK OR United Kingdom OR England OR Wales OR Scotland) AND (bird OR bullfinch) AND (wood OR woodland)

Table A2. A summary of the number of papers that met the criteria set to identify estimates of the effect of sharing and sparing interventions on yellowhammers (YH), bullfinches (BF) and lapwings (L).

	Stubble, spring cropping			Winter bird cover	Hedge creation		Fallow	Scrub creation		Wood creation	Wet grass creation	Total
	YH	**BF**	**L**	**YH**	**BF**	**YH**	**L**	**YH**	**BF**	**BF**	**L**	**All**
Met search criteria	206	206	206	390	117	117	42	63	63	521	212	**1551**
Relevant abstract	36	36	36	28	26	26	11	20	20	49	37	**206**
Reported direction of effect	7	2	5	7	6	8	5	3	4	7	5	**51**
Quantified effect per unit area	2	0	2	4	3	2	1	3	4	4	5	**28**
Estimate is usable	2	0	2	3	1	2	1	3	4	3	5	**24**

Table A3. Exploration of the papers identified through Web of Science which were thought to potentially contain estimates of the effect of the studied interventions based on their abstracts. In the following, Y=yes, N=no, SE=standard error, YH=yellowhammer, L=lapwing and BF=bullfinch. In the 'Effect reported?' column, '-' refers to papers that could not be located. The table ceases to be completed if any paper does not report an effect, does not quantify an effect or is not usable.

a. Papers studying stubble for bullfinches, lapwings and yellowhammers.

		Reference	Effect reported?	Is effect:			Effect quantified?	How?	Per-area effect quantified?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
				+	--	=								
1	Stubble	Aebischer et al. 2016	N											
2		Bright et al. 2014	N											
3		Baker et al. 2012	Y	YH			Y	Pop^{n} growth rates	N	N	Pop^{n} growth rate			
4		Westbury et al. 2011	N											
5		Field et al. 2011	N											
6		Geiger et al. 2010	N											
7		Hinsley et al. 2010	N											
8		Siriwardena 2010	Y	YH			Y		N	N				
9		Butler et al. 2010	N											
10		Douglas et al. 2010	N											
11		Perkins et al. 2008	Y	YH			N		N	N				
12		Siriwardena et al. 2008	N											
13		Gillings et al. 2008	Y	YH		BF, L	N		N	N				
14		Siriwardena et al. 2007	N											

		Reference	Effect reported ?	Is effect:			Effect quantified ?	How?	Per-area effect quantified ?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
				+	--	=								
15		Chamberlain et al. 2007	N											
16		Whittingham et al. 2006	N											
17		Sage et al. 2005	N											
18		Butler et al. 2005	N											
19		Gillings et al. 2005	Y	YH			Y	Pop^n growth rates	Y	N	Pop^n growth rate			
				L					N					
				BF					N					
20		Whittingham et al. 2005	N											
21		Fuller et al. 2004	N											
22		Bradbury et al. 2004	N											
23		Critchley et al. 2004	N											
24		Parish & Sotherton 2004	N											
25		Hancock & Wilson 2004	Y	YH			Y	Pop^n density	Y	Y		N	Y	
26		Moorcroft et al. 2002	N											
27		Siriwardena et al. 2000	N											
28		Bradbury et al. 2001	N											

		Reference	Effect reported ?	Is effect:			Effect quantified ?	How?	Per-area effect quantified ?	Usable?	Notes on being usable	Uncertaint y reported?	Contributes to final estimate?	Notes on if contributing
				+	--	=								
29		Mason & Macdonald 2000	N											
30		Bradbury & Stoate 2000	-											
31		Buckingham et al. 2010	Y	YH			N		N	N				
32		Mason & Macdonald 1999	N											
33		Wilson et al. 2001	Y	L			Y	Pop^n density	Y	Y		N	Y	
34		Evans 1997	-											
35		Shrubb et al. 1991	Y	L			Y	Pop^n density	Y	Y		N	Y	
36		Tucker 1992	Y			L	N		N	N				

b. Papers studying winter bird cover (WBC) for yellowhammers and bullfinches.

		Reference	Effect reported?	Is effect:			Effect quantified?	How?	Per-area effect quantified?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
				+	--	=								
1	Winter Bird Cover	Aebischer et al. 2016	N											
2		Baker et al. 2012	Y	YH			Y	Pop^n growth rates	N	N				
3		Henderson et al. 2012	Y	YH			Y	Pop^n density	Y	Y		N	Y	
4		Field et al. 2011	N											
5		Geiger et al. 2010	N											
6		Hinsley et al. 2010	Y	YH			N		N	N				
7		Siriwardena 2010	Y	YH			Y	Habitat selection relative to availability	N	N				
8		Butler et al. 2010	N											
9		Perkins et al. 2008	N											
10		Siriwardena et al. 2007	N											
11		Siriwardena et al. 2006	N											
12		Sage et al. 2005	N											
13		Atkinson et al. 2005	N											
14		Robinson et al. 2004	N											
15		Bradbury et al. 2004	N											
16		Stoate et al. 2004	N											

		Reference	Effect reported ?	Is effect: +	--	=	Effect quantified ?	How?	Per-area effect quantified ?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
17		Parish & Sotherton 2004	Y	YH			Y	Pop^n density	Y	Y		N (not readable)	Y	
				BF			N							
18		Henderson et al. 2004	Y	YH			Y	Pop^n density	Y	N	No control farms.			
				BF			Y	Pop^n density	Y	N				
19		Hancock & Wilson 2004	N											
20		Stoate et al. 2003	Y	YH			Y	Pop^n density	Y	Y		Y	Y	
21		Bartram & Perkins 2002	-											
22		Moorcroft et al. 2002	N											
23		Moreby 2002	N											
24		Morris et al. 2001	N											
25		Vickery et al. 2001	N											
26		Henderson et al. 2000	N											
27		Boatman et al. 2000	-											
28		Bradbury & Stoate 2000	-											

c. Papers studying fallow plots for lapwing.

		Reference	Effect reported ?	Is effect: +	--	=	Effect quantified ?	How?	Per-area effect quantified ?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
1	Lapwing plots	Schmidt et al. 2017	N								Not UK			
2		Bright et al. 2015	Y	Y			N		N	N				
3		Henderson et al. 2012	Y	Y			N		N	N				
4		Chamberlain et al. 2009	Y	Y			Y	Pop^n density	Y	Y	Authors report proportion of plots used by lapwing but size of plots is not given. Plots reported to be 2-5ha. Assumed midpoint plot size of 3.5ha.	N	Y	
5		Sheldon et al. 2010	Y	Y			Y		N	N				
6		Critchley et al. 2004	N											
7		Taylor & Grant 2004	N											
8		Siriwardena et al. 2000	N											
9		Chamberlain et al. 2000	N											
10		Buckingham et al. 1999	N											
11		Sheldon et al. 2004	Y	Y			Y		N	N				

d. Papers studying hedgerow creation for yellowhammers and bullfinches.

		Reference	Effect reported ?	Is effect: +	--	=	Effect quantified ?	How?	Per-area effect quantified ?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
1	Hedgerows	Dadam & Siriwardena 2019	Y	YH			Y	Popⁿ growth rates	Y	N	Studied hedge mgmt., not creation			
				BF										
2		Carrasco et al. 2018	N											
3		Sullivan et al. 2017	N											
4		Sage et al. 2015	N											
5		Norton et al. 2012	N											
6		Baker et al. 2012	Y			YH	Y	Popⁿ growth rates	Y	N	Popⁿ growth rate			
				BF										
7		Draycott et al. 2012	N											
8		Siriwardena et al. 2012	N											
9		Cornulier et al. 2011	N											
10		Davey et al. 2010	N											
11		Walker et al. 2005	N											
12		Perkins et al. 2002	N											
13		Moreby 2002	-											
14		Chamberlain et al. 2010	N											
15		Fuller et al. 2001	Y	YH		BF	N		N	N				
16		Bradbury et al. 2000	Y	YH			Y	Popⁿ density	Y	Y		Y	Y	

		Reference	Effect reported ?	Is effect:			Effect quantified ?	How?	Per-area effect quantified ?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
				+	--	=								
17		Mason & Macdonald 2000	Y	YH			Y	Habitat preference index	N	N				
18		Chamberlain & Wilson 1999	-											
19		Chamberlain et al. 1999	N	BF		YH	N		N	N				
20		Kyrkos et al. 2010	Y	YH			Y	Regression to explain habitat associations	N	N				
21		Fuller et al. 1997	Y	YH			Y	Regression to explain habitat associations	N	N				
22		Hinsley et al. 1995	Y	BF		YH	Y	Regression to explain habitat associations	N	N				
23		Parish et al. 1995	N											
24		Green et al. 1994	Y	YH			Y	Regression to explain habitat associations	N	N				
25		Parish et al. 1994	N											
26		Macdonald & Johnson 1995	Y	BF			Y	Pop^n density	Y	Y		N	Y	
				YH			Y	Pop^n density	Y	Y		N	Y	

e. Papers studying scrub creation for yellowhammers and bullfinches.

		Reference	Effect reported ?	Is effect: +	--	=	Effect quantified ?	How?	Per-area effect quantified ?	Usable ?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
1	Scrub	Dadam & Siriwardena 2019	N								Growth rates studied for scrub management, not creation			
2		Sage et al. 2010	N											
3		Hewson & Noble 2009	N											
4		Dolman et al. 2007	N											
5		Fuller et al. 2006	N											
6		Fuller et al. 2004	N											
7		Hancock & Wilson 2003	N											
8		Mason & Macdonald 2000	N											
9		Peach et al. 2010	N											
10		Newson et al. 2005	Y	YH			Y	Pop^n density	Y			Y	N	Crude habitat designation – includes habitat other than that studied.
				BF									N	
11		Gillings et al. 1998	N											
12		Blackstock et al. 1996	N											
13		Peach et al. 1996	N											
14		Thompson et al. 1995	N											

		Reference	Effect reported ?	Is effect:			Effect quantified ?	How?	Per-area effect quantified ?	Usable ?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
				+	--	=								
15		Usher & Thompson 1993	N											
16		Fuller & Crick 1992	N											
17		Morgan (1975)	Y	YH			Y	Pop^{n} density	Y	Y		N	Y	
				BF									Y	
18		Donovan (2013) (grey literature)	Y	YH			Y	Pop^{n} density	Y	Y		N	Y	
19		Gregory & Baillie 1998	Y	BF			Y	Pop^{n} density	Y	Y		Y	N	Crude habitat designation – includes habitat other than that studied.
20		Knepp Estate (grey literature)	Y	BF			Y	Pop^{n} density	Y	Y		N	Y	

f. Papers studying wet grassland to deliver lapwings.

		Reference	Effect reported?	Is effect +	--	=	Effect quantified?	How?	Per-area effect quantified?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
1	Wet grassland	Lamb et al. 2019	Y	Y			Y	Pop^n density	Y	Y		N	N N	Crude habitat designation – includes habitat other than that studied. Crude habitat designation – includes habitat other than that studied.
2		Laidlaw et al. 2017	N											
3		Leigh et al. 2017	N											
4		Zmihorski et al. 2016	N											
5		Mendez et al. 2015	N											
6		Laidlaw et al. 2015	N											
7		Hiley et al. 2014	N											
8		Smart et al. 2014	N											
9		Smart et al. 2013	N											
10		Malpas et al. 2013	N											
11		Mendez et al. 2012	N											
12		O'Brien & Wilson 2011	N											
13		Rhymer et al. 2010	N											

		Reference	Effect reported ?	Is effect			Effect quantified ?	How?	Per-area effect quantified ?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
				+	--	=								
14		Eglington et al. 2009	N											
15		MacDonald & Bolton 2008	N											
16		Eglington et al. 2007	Y	Y			Y	Pop^{n} density	Y	Y		Y	Y	
17		Fuller et al. 2007	N											
18		Tichit et al. 2007	N											
19		Bolton et al. 2007	N											
20		Wilson et al. 2007	Y											
21		Gillings et al. 2006	N											
22		Jackson et al. 2006	N											
23		Wilson et al. 2005	Y											
24		Wilson et al. 2007	N											
25		Newson et al. 2005	Y	Y			Y	Pop^{n} density	Y	Y		Y	N	Crude habitat designation – includes habitat other than that studied.
26		Rehfisch et al. 2003	N											
27		Kershaw & Cranswick 2003	N											
28		Robinson & Pollitt 2002	N											

		Reference	Effect reported?	Is effect +	--	=	Effect quantified?	How?	Per-area effect quantified?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
													Y	
29		Ausden & Hirons 2002	Y	Y			Y	Pop^n density	Y	Y		Y	Y	
30		Milsom et al. 2000	N											
31		Wakeham-Dawson & Smith 1999	N											
32		Peach et al. 1998	N											
33		Vickery et al. 1997	N											
34		Peach et al. 1996	N											
35		Kirby 1995	N											
36		O'Brien & Wilson 1992	N											
37		RSPB Reserves data (grey literature)	Y	Y			Y		Y	Y		Y	Y	

g. Papers studying woodland creation for bullfinches

		Reference	Effect reported?	Is effect: +	--	=	Effect quantified?	How?	Per-area effect quantified?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
1	Woodland	Dadam & Siriwardena 2019	Y	Y			Y	Pop^n growth rate	N	N				
2		Gardner et al. 2019	N											
3		Lamb et al. 2019	Y	Y			Y	Pop^n density	Y	Y		N	Y	
4		Broome et al. 2019	N											
5		Calladine et al. 2019	N											
6		Alder et al. 2018	N											
7		Ellis & Taylor 2018	N											
8		Neumann et al. 2016	N											
9		Sullivan et al. 2015	N											
10		Harrison et al. 2014	N											
11		Norton et al. 2012	N											
12		Siriwardena et al. 2012	N											
13		Sage et al. 2011	N											
14		Thaxter et al. 2010	N											
15		Hewson & Noble 2009	N											
16		Hewson et al. 2007	N											
17		Fuller et al. 2007	N											
18		Dolman et al. 2007	N											

		Reference	Effect reported ?	Is effect:			Effect quantified ?	How?	Per-area effect quantified ?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
				+	--	=								
19		Hewson & Fuller 2006	N											
20		Fuller et al. 2005	N											
21		Newson et al. 2005	Y	Y			Y	Pop^n density	Y	Y		Y	Y	
22		Gregory et al. 2005	N											
23		Proffitt et al. 2004	N											
24		Bennett et al. 2004	N											
25		Bellamy & Hinsley 2004	-											
26		Vanhinsbergh et al. 2002	Y	Y			Y	Pop^n density	Y	N	Most woodlands <2ha			
27		Hinsley et al. 2002	N											
28		Mason 2001	N											
29		Fuller et al. 2001	N											
30		Perrins & Overall 2001	N											
31		Mason & Macdonald 2000	N											
32		Hinsley & Bellamy 2000	-											
33		Gregory & Baillie 1998	Y	Y			Y	Pop^n density	Y	Y		Y	Y	
34		Donald et al. 1998	N											
35		Hinsley et al. 1998	N											
36		Gillings et al. 1998	N											
37		Donald et al. 1997	Y	Y			Y	Timed counts	N	N				

		Reference	Effect reported?	Is effect:			Effect quantified?	How?	Per-area effect quantified?	Usable?	Notes on being usable	Uncertainty reported?	Contributes to final estimate?	Notes on if contributing
				+	--	=								
38		Fuller et al. 1997	N											
39		Bellamy et al. 1996	N											
40		Kirby et al. 1995	N											
41		Hinsley et al. 1995	N											
42		Hinsley et al. 1995	N											
43		McCollin 1993	N											
44		Fuller & Crick 1992	N											
45		Bevington 1991	N											
46		Hill et al. 1991	Y	Y			Y	Timed counts	N	N				
47		McCollin et al. 1987	-											
48		Williamson 1974	N											
49		Elton 1935	N											

Table A4. The intervention and non-intervention bird densities reported in the identified 'usable' studies with adjustment for annual mortality.

Species	**Intervention type**	**Intervention**	**Source publication**	**With or without intervention (Y/N)**	**Density, *D* (birds/ha)**	**Survey date midpoint**	**Days elapsed from survey to 31st May**	**Adjusted density, D_{adj} (birds/ha)**
Bullfinch	Field-edge sharing	Hedgerow creation	Macdonald and Johnson (1995)	Y	0.91[c]	31/05	0	0.91 [c]
	Sparing	Scrub creation	Morgan (1975)	Y	0.20	31/05	0	0.20
			T. Finch (*pers. comm.*)	Y	0.33	15/05	16	0.31
		Woodland creation	Lamb et al. (2019) – BBS	N	0.0069	15/05	16	0.0066
				Y	0.051	15/05	16	0.049
			Lamb et al. (2019) – RSPB	N	0.0069	15/05	16	0.0066
				Y	0.086	15/05	16	0.083
			Newson et al. (2005)	N	0.013	15/05	16	0.013
				Y	0.055	15/05	16	0.053
			Gregory and Baillie (1998)	N	0.010	15/05	16	0.0096
				Y	0.041	15/05	16	0.040
Lapwing	In-field sharing	Stubble, spring cropping	Wilson et al. (2001)	N	0.0017	15/04	46	0.0017
				Y	0.031	15/04	46	0.030
			Shrubb et al. (1991)	N	0.0056	15/04	46	0.0054
				Y	0.056	15/04	46	0.055
	Field-edge sharing	Fallow plots	Chamberlain et al. (2009)	N	0.017	15/05	16	0.017
				Y	0.18	15/05	16	0.18
	Sparing	Wet grass creation	Eglington et al. (2007)	Y	0.68	31/05	0	0.68
			Ausden and Hirons (2002) –ESAs	Y	0.056	15/05	16	0.055

			Ausden and Hirons (2002) – RSPB reserves	Y	0.46	15/05	16	0.46
			M. Ausden *(pers. comm.)*	Y	0.81	15/05	16	0.80
Yellowhammer	In-field sharing	Stubble, spring cropping	Hancock and Wilson (2003)	N	0	09/01	142	0
				Y	0.5	09/01	142	0.39
	Field-edge sharing	Winter bird seed plots	Henderson et al. (2012)	N	0.07	15/05	16	0.068
				Y	0.14[a]	15/05	16	0.14[a]
			Parish and Sotherton (2004)	N	0	30/12	152	0
				Y	0.8	30/12	152	0.61
			Stoate et al. (2003)	N	0.04	30/12	152	0.031
				Y	0.2[b]	30/12	152	0.15[b]
		Hedgerow creation	Macdonald and Johnson (1995)	Y	6.1[c]	31/05	0	6.1[c]
			Bradbury et al. (2001)	N	10[c]	31/05	0	10[c]
	Sparing	Scrub creation	Morgan (1975)	Y	0.74	31/05	0	0.74
			Donovan (2013)	Y	1.2	15/05	16	1.2

[a] Density is birds/ha across entire 100ha farm with 10ha winter bird cover

[b] Density is birds/ha across entire 100ha farm with 32.2ha winter bird cover

[c] Hedge per-ha densities assume hedges are 6m wide.

b) Quantifying the effect of interventions that deliver net carbon emissions reductions

We searched the literature to find papers that quantified the carbon emissions avoided, or the carbon stored, by the interventions identified in Table 2.1. Where possible, we used papers that were referenced in IPCC reports, as detailed below.

i. Reducing fertiliser use

The method for estimating the reductions in net carbon emissions delivered by cutting use of inorganic nitrogen fertiliser differs between Chapter 2, where we study total cessation of fertiliser use by the average farmer enrolled in current schemes, and Chapters 3-5 where we study a 50% reduction by the farmers recruited by our choice experiment. Therefore, here we first describe the method for when we assume the average English arable farmer enrolled in the scheme stops using fertiliser, and then repeat the method for studying a 50% reduction in fertiliser use by participants of the choice experiment.

Method for Chapter 2

To quantify the carbon emissions avoided by eliminating use of inorganic nitrogen (N) fertiliser, we followed Kindred et al. (2008) who studied the emissions associated with the manufacture and application of fertiliser to winter wheat in the UK. The authors reported the greenhouse gas emissions (in kg CO2e) per-tonne of crop yield for a range of fertiliser application rates (Figure A1a). At low levels of fertiliser use, the authors reported low crop yields; thus they quantified the emissions associated with the land-use change necessary to sustain food production. We omitted these emissions since we do not include these rebound effects in calculating the sequestration associated with any other interventions. We converted this plot of emissions per-tonne to emissions per-ha given reported yields. This revealed a linear relationship: i.e., the carbon emitted per unit fertiliser was constant regardless of the application rate. From this, we calculated the tonnes of carbon saved by reducing N-fertiliser use (Figure A1b). The gradient of this line (0.0032) gives the per-ha tonnes carbon saved per-KgN of fertiliser which is not applied.

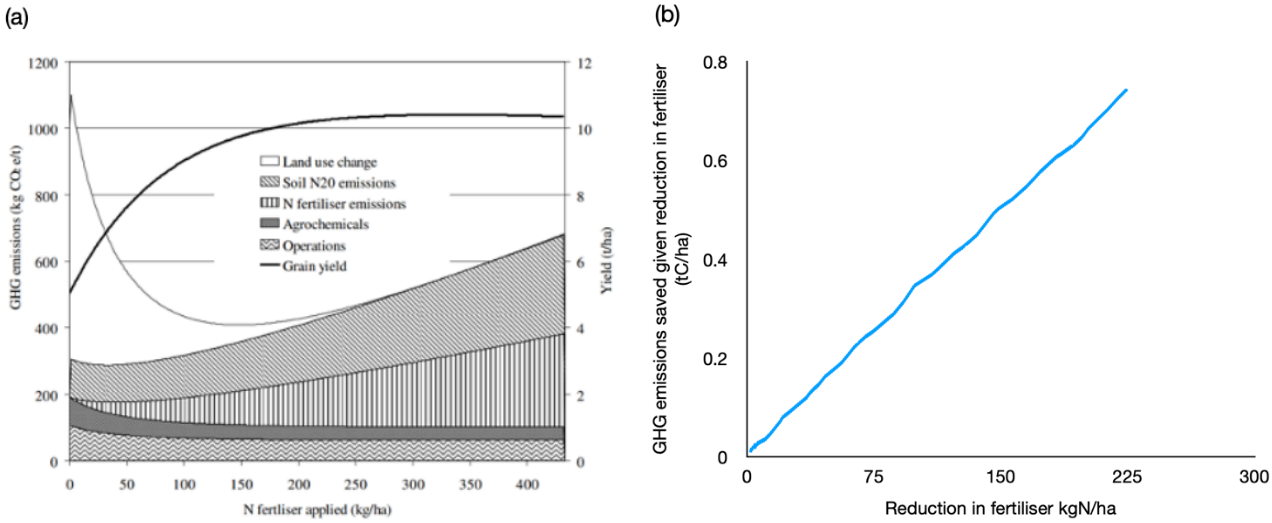

Figure A1. (a) From Kindred et al. (2008), the GHG emissions per-tonne of crop yield associated with varying rates of N-fertiliser application to winter wheat (left axis), and the associated yield (right axis). Derived from Kindred et al. (2008), (b) the tonnes carbon saved by reducing per-ha fertiliser application by varying amounts.

To calculate the emissions avoided by eliminating fertiliser use, we had to establish the rate at which fertiliser is applied on typical arable farms. In doing so, we assumed all crops would behave as winter wheat does here (since comparable data did not exist for other crops). We assumed this option would be applied to areas of wheat, barley and oil seed rape (the most commonly grown crops on UK arable farms) in proportion to the relative area occupied by each crop on an average arable farm (Defra 2019; Table A5). To do so, we took crop-specific fertiliser application rates from Benford (2016; Table B2.1, p. 29) and calculated the average fertiliser application rate (weighted by relative areas; Table A5). We then calculated the carbon emissions saved per-ha by assuming this average application rate (171kgN/ha/y) would be avoided (i.e., 0.0032×171).

Table A5. Fertiliser application rates and average crop areas on an average cereal farm.

	Crop		
	Winter wheat	**Spring barley**	**Oil seed rape**
Crop-specific average fertiliser application rate (kgN/ha/y)	190	105	193
Proportional crop area on average cereal farm	0.60	0.23	0.16
Average fertiliser application rate (kgN/ha/y)		171	

Method for Chapters 3-5:

To quantify the carbon emissions avoided by the participants of our choice experiment reducing their use of inorganic nitrogen fertiliser by 50%, we followed Kindred et al. (2008) which studied the emissions associated with the manufacture and application of fertiliser to winter wheat in the UK. The authors reported the greenhouse gas emissions (in kg CO2e) per-tonne of crop yield for a range of fertiliser application rates (Figure A2a). We converted this plot of emissions per-tonne to emissions per-ha given reported yields. This revealed a linear relationship: i.e., the carbon emitted per unit fertiliser was constant regardless of the application rate. From this, we calculated the tonnes carbon saved by reducing N fertiliser (Figure A2b). The gradient of this line (0.0032) gives the per-ha tonnes carbon saved per-KgN of fertiliser not applied. Participants were told this intervention was only applicable to wheat, barley, oil seed rape, sugar beet and potatoes (crops that typically have high fertiliser requirements) and were asked to report their use of inorganic N fertiliser. We used these reported values to calculate the emissions saved by a 50% reduction, assuming that the crop with the lowest gross margin would be enrolled first. Where respondents did not report fertiliser use, we used average, crop-specific values from the 2019/2020 Farm Business Survey.

The data in Kindred et al. (2008) show that at low levels of fertiliser use yields are reduced; just as with other interventions, meeting the resulting reduction in food production arising from the intervention will thus involve increased GHG emissions and impacts on biodiversity elsewhere, which we did not quantify, but which we note (Main text, Discussion) would be greater the larger the food production forgone.

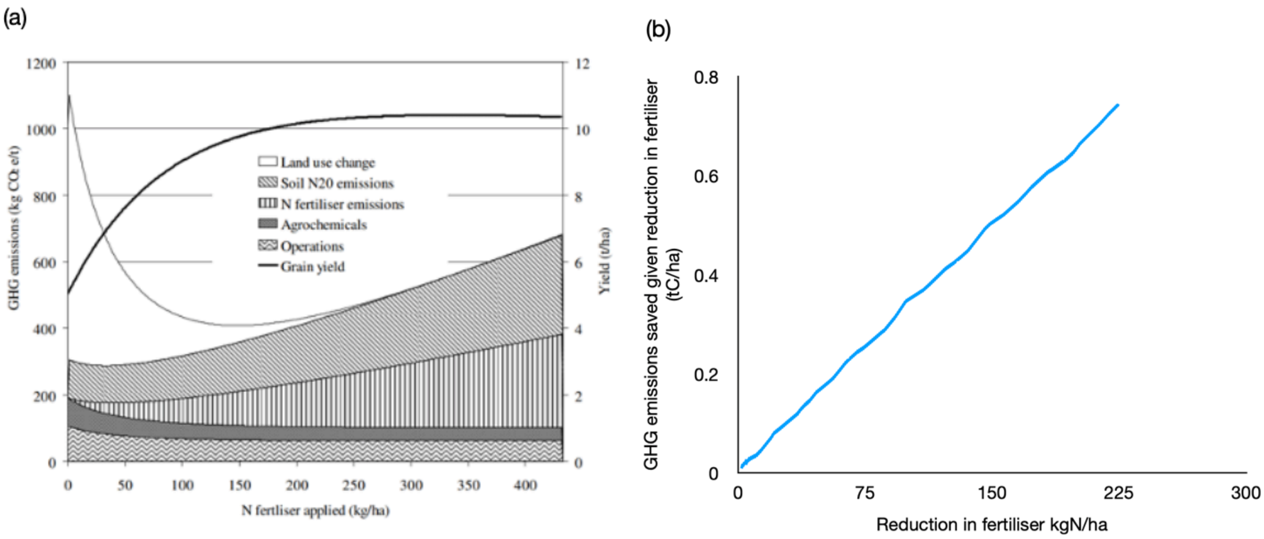

Figure A2. (a) From Kindred et al. (2008), the GHG emissions per-tonne of crop yield associated with varying rates of N-fertiliser application to winter wheat (left axis), and the associated yield (right axis). Derived from Kindred et al. (2008), (c) the tonnes carbon saved by reducing per-ha fertiliser application by varying amounts.

ii. Hedgerow creation

Hedgerows sequester carbon above and below ground. In temperate regions, the maximum above-ground biomass carbon stock at maturity is 26.1 tC/km hedgerow (IPCC 2019). The maturity cycle is estimated at 30 years, which gives an average annual sequestration rate of 0.87tC/km/y for the first 30 years following creation (IPCC 2019). Arable land, in comparison, stores no above-ground carbon. In addition, planting hedgerows on arable land sequesters 0.23tC/km/y below ground (IPCC 2019). Therefore, for the first 30 years after establishment, we estimate that hedgerows sequester, on average, 1.10tC/km/y.

iii. Woodland creation

Creating woodland on arable land also stores carbon above and below ground. To quantify above-ground sequestration of woodland, we used the estimate of Falloon et al. (2004; based on data from the IPCC), at an average sequestration rate 2.8tC/ha/y. The same authors estimated soil carbon to accumulate annually at an average rate of 1.17% when arable land is converted to woodland (Falloon et al. 2004). Assuming soil in arable areas of the UK has a carbon content of 84tC/ha (Smith et al. 2000), the amount of soil carbon sequestered annually over a 20-year period is:

$$\frac{1.17^{20}}{100} \times 84$$
$$= 19.04\text{tC/ha}$$

Therefore, for the 20 years following woodland creation, on average 0.97tC/ha/y is sequestered in soil carbon each year. The average carbon sequestered annually is the sum of that sequestered in the soil and above ground, i.e.:

$$2.8 + 0.97$$
$$= 3.77\text{tC/ha/y}$$

Results

Estimates of the effect of the studied sharing and sparing interventions on bird densities and net carbon emissions reductions are reported in Figure A3.

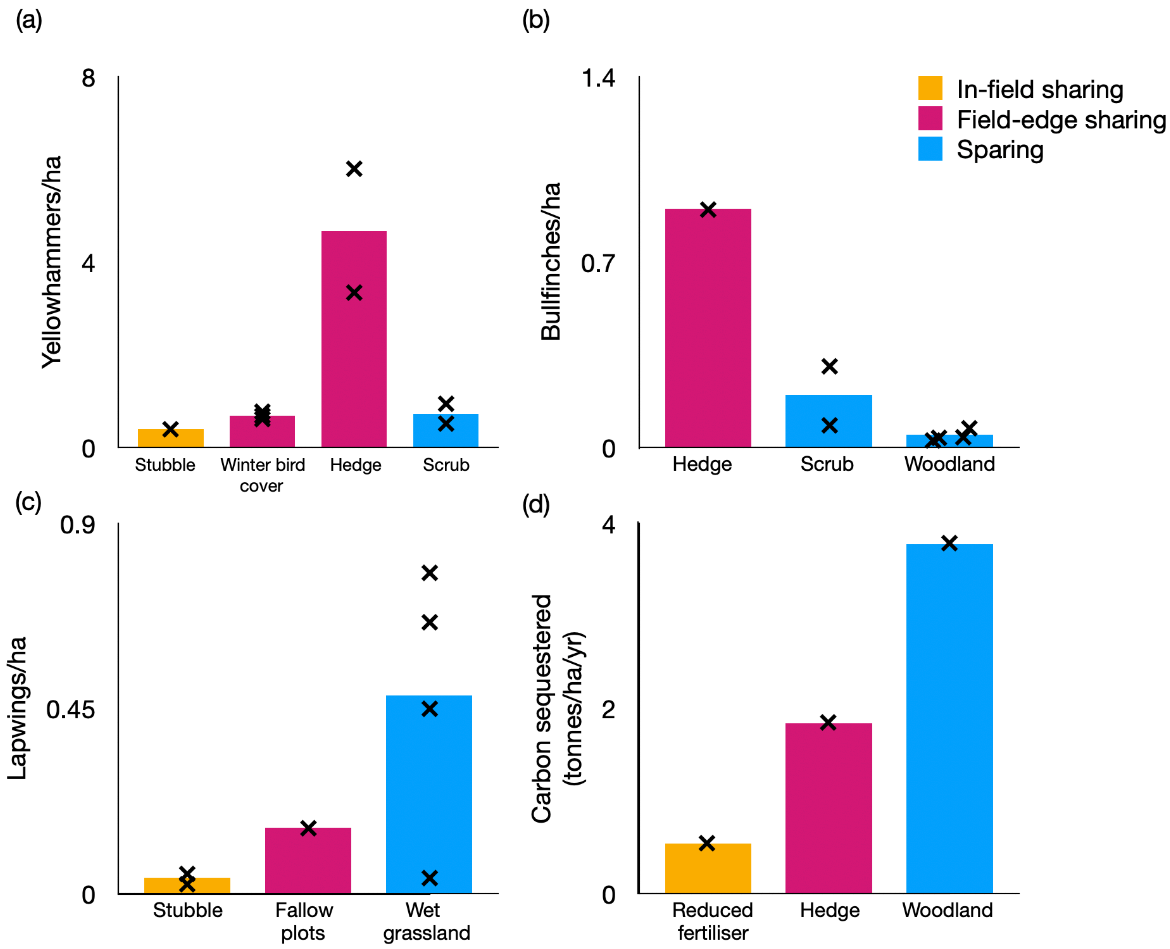

Figure A3. The mean effect of in-field sharing (orange bars), field-edge sharing (pink bars) and sparing (blue bars) interventions that deliver increased populations of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) net carbon emissions reductions with results from individual studies (from which the mean was derived) plotted as crosses (where necessary jittered on the x-axis to make all visible). Hedgerow results are plotted per-ha hedgerow, i.e. per 1.7km length hedge with width of 6m. Reduced fertiliser is plotted for the average English arable farm ceasing use of inorganic nitrogen fertiliser.

Appendix B: Chapter 2

Additional methods for the compliance analysis

(i) Current scheme participation and outcomes delivered

Table B1. The options from the CSS studied in the compliance monitoring analysis, their payment rates and current uptake amongst CSS participants (data from RPA, *pers. comm.),* plus the benefit delivered by the current scheme assuming 100% compliance. (Two codes are given where options have different creation and maintenance payment rates. Woodland creation is funded by the Woodland Creation Grant Scheme which we assumed to cost, on average, £2000/ha).

Type of outcome	**Option**	**CSS Code**	**Payment £/ha/y (or km hedge)**	**Total area enrolled / ha (or km hedge)**	**Average area enrolled per agreement /ha (or km hedge)**	**Proportion of CSS agreements with option**	**Benefit delivered / birds (or tonnes carbon per year)**
Yellowhammer	Stubble	AB2	84	22447	13.39	0.120	8754
	Winter bird cover	AB9	640	18336	4.39	0.409	12651
	Hedge	BN11[a] / BE3	11600 / 80	692	0.68	0.099	1937
	Scrub	WD8[b] / WD7	87 / 74	786	11.39	0.04	574
Bullfinch	Hedge	BN11[a] / BE3	11600 / 80	692	0.68	0.099	374
	Scrub	WD8[b] / WD7	87 / 74	786	11.39	0.04	161
	Woodland	WCG[a] / WD1	2000 / 200	1434	7.71	0.108	67
Lapwing	Stubble	AB2	84	22447	13.39	0.120	875
	Fallow	AB5	524	1225	4.00	0.030	196
	Wetland	GS11[c] / GS9	406 / 264	307	16.06	0.011	149
Carbon	Nil fertiliser	SW14	131	1481	14.96	0.010	859
	Hedge	BN11[a] / BE3	11600 / 80	692	0.68	0.099	763
	Woodland	WCG[a] / WD1	2000 / 200	1434	7.71	0.108	5405

[a] Creation rate paid for 1 year. [b] Creation rate paid for 5 years. [c] Creation rate paid for 10 years.

(ii) Non-compliance in current schemes

Table B2. Rates of monitoring and non-compliance, plus the value of fines awarded to participants of the Countryside Stewardship Scheme (CSS) and Environmental Stewardship Scheme (ESS) between 2015-2017 (data from RPA, *pers. comm.*).

Year	Scheme	Section of scheme	% Properties checked for non-compliance	% Failing checks	Total fines / £	Mean fine £/property	Mean rate of monitoring across all schemes %	Mean rate of non-compliance across all schemes %
2015	**ESS**	ELS	5.9	26.1	636,000	1483	6.0	45.5
		HLS	4.4	70.8	658,000	1485		
2016	**CSS**	Mid Tier	3.0	67.9	67,000	1861		
		High Tier	4.9	65.0	38,000	2923		
	ESS	ELS	3.8	13.6	24,000	320		
		HLS	13.2	58.1	1,043,000	1009		
2017	**CSS**	Mid Tier	4.8	48.0	220,500	1853		
		High Tier	6.6	44.1	61,500	2050		
	ESS	ELS	4.3	12.9	7,200	171		
		HLS	9.3	48.9	498,000	857		

(iii) Monitoring costs

We estimated the costs of monitoring each intervention following estimates of the cost of monitoring participants in current schemes (P. Carey, *pers. obs.*). The time and mileage of professional surveyors travelling to the site and conducting the inspection was valued at £240. Habitat monitoring was estimated at £700 per km^2; so this was adjusted according to the area of the option in question. Finally, aerial analysis was assumed to be used to quantify the area under in-field sharing and sparing options, at the cost of £60. Remote monitoring was assumed to cost £180/site, regardless of the option in question (approximately in line with RPA 2012).

(iv) Calculating total costs

We calculated total costs as the sum of scheme payments and administrative costs, plus monitoring costs, less fines, as follows:

$$(P + A)n + m_r r_r n + m_i r_i n - (r_r d_{r1} + r_i d_{i1}) b_1 n P f_1 - (r_i d_{i2}) b_2 n P f_2$$

Where *P* is the payment rate, *A* is the administrative cost per property, *n* the number of scheme participants, m_r and m_i the costs of monitoring a participant remotely and in-person, r_r and r_i are the proportions monitored remotely and in-person, f_1 and f_2 are the fines for major and minor non-compliance, d_{r1} and d_{i1} are the proportion of majorly non-compliant properties fined for non-compliance if monitored remotely and in-person respectively, and d_{i2} is the proportion of minorly non-compliant properties fined for minor non-compliance if monitored in-person, and b_1 and b_2 are the proportion of scheme participants majorly and minorly not complying.

Sensitivity testing of compliance monitoring analysis

We tested the sensitivity of the results from the compliance monitoring analysis to variation in the following key parameters:

i. Proportion of honest farmers (Figure B1)
ii. Fines (Figure B2)
iii. Monitoring costs (Figure B3)
iv. Area entered by participants (Figure B4)
v. Biodiversity benefit (Figure B5)
vi. Availability of remote monitoring (Figure B6)

We also explored the rate of fines required to make present monitoring rates effective in deterring non-compliance (Figure B7).

(i) Proportion of farmers assumed honest

Assuming more farmers were honest did reduce scheme costs at low rates of non-compliance but had little impact at higher monitoring rates when little non-compliance occurred (Figure B1). Sparing (plots with blue shading) remained the least costly scheme at each rate of honest farmers.

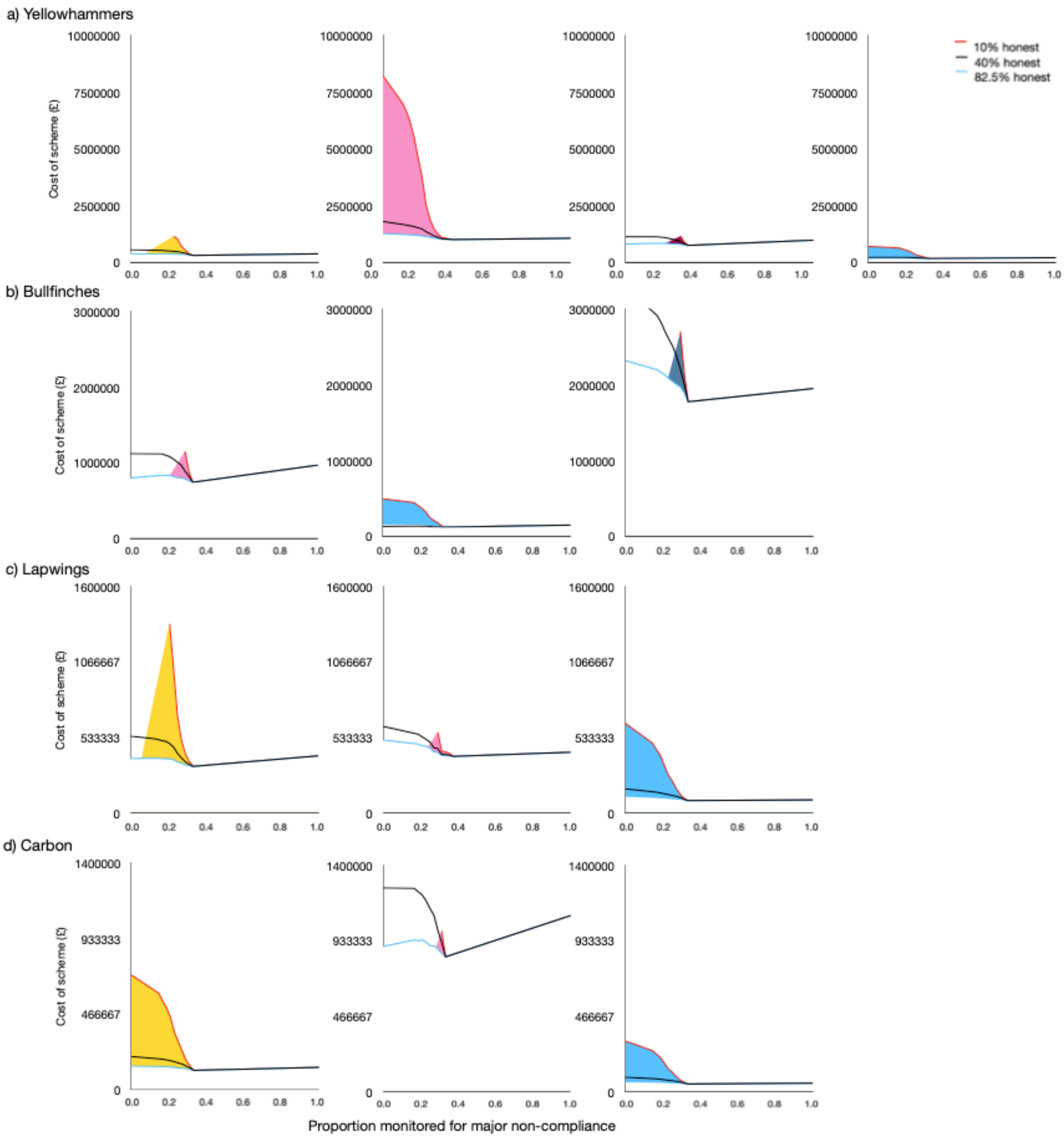

Figure B1. The cost of ten-year schemes at varying rates of monitoring that delivered fixed amounts of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) carbon with in-field sharing (orange), field-edge sharing (pink) and sparing (blue). The black line shows the baseline condition where 54% of farmers were assumed honest; the red line assumed 10% were honest and the blue line assumed 82.5% were honest. Shaded area shows difference in costs between 10% and 82.5% farmers being honest. Of the two field-edge sharing interventions for yellowhammers, the lighter line is winter bird cover and the darker line is hedgerow creation. Of the two sparing interventions for bullfinches, the lighter line is scrub and the darker is woodland creation. (Costs are not plotted where monitoring rate is so low that current AES payment rates would be incapable of meeting the specified target, and hence lines and shading are variable in length.)

(ii) Fines

Higher fines reduced scheme costs at low levels of monitoring and reduced the level of monitoring required (Figure B2). Sparing (plots with blue shading) remained the least expensive scheme in all cases, regardless of the rate of fines.

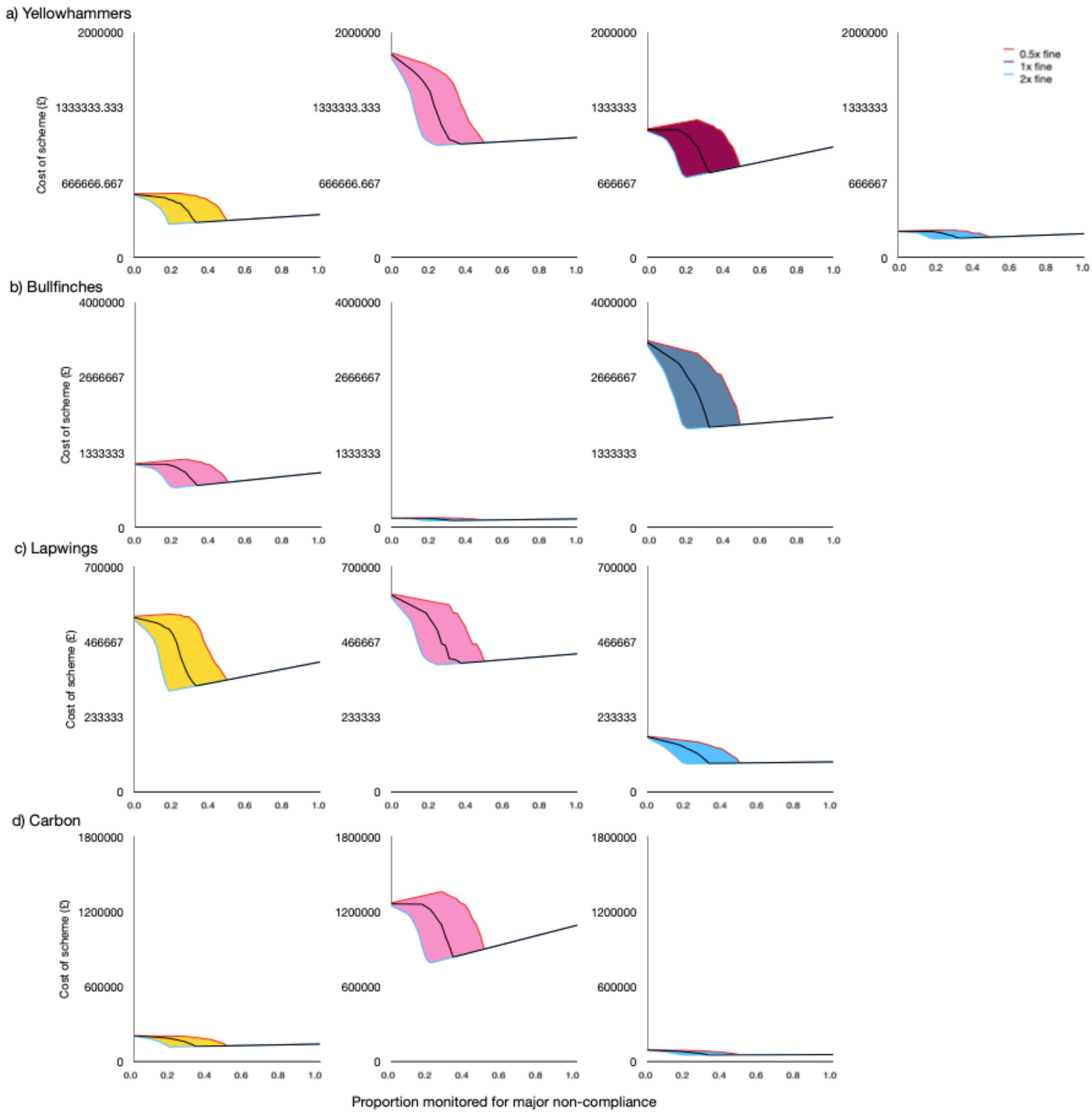

Figure B2. The cost of ten-year schemes at varying rates of monitoring that delivered fixed amounts of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) carbon with in-field sharing (orange), field-edge sharing (pink) and sparing (blue). The black line shows the baseline condition; the blue line shows costs when fines were doubled and the red line when fines were halved. Shading shows difference in costs between fines of 0.5x and 2x the payment rate. Of the two field-edge sharing interventions for yellowhammers, the lighter line is winter bird cover and the darker line is hedgerow creation. Of the two sparing interventions for bullfinches, the lighter line is scrub and the darker is woodland creation.

(iii) Monitoring costs

Scheme costs increased when monitoring was more expensive (Figure B3). However, sparing remained the least expensive scheme in all cases.

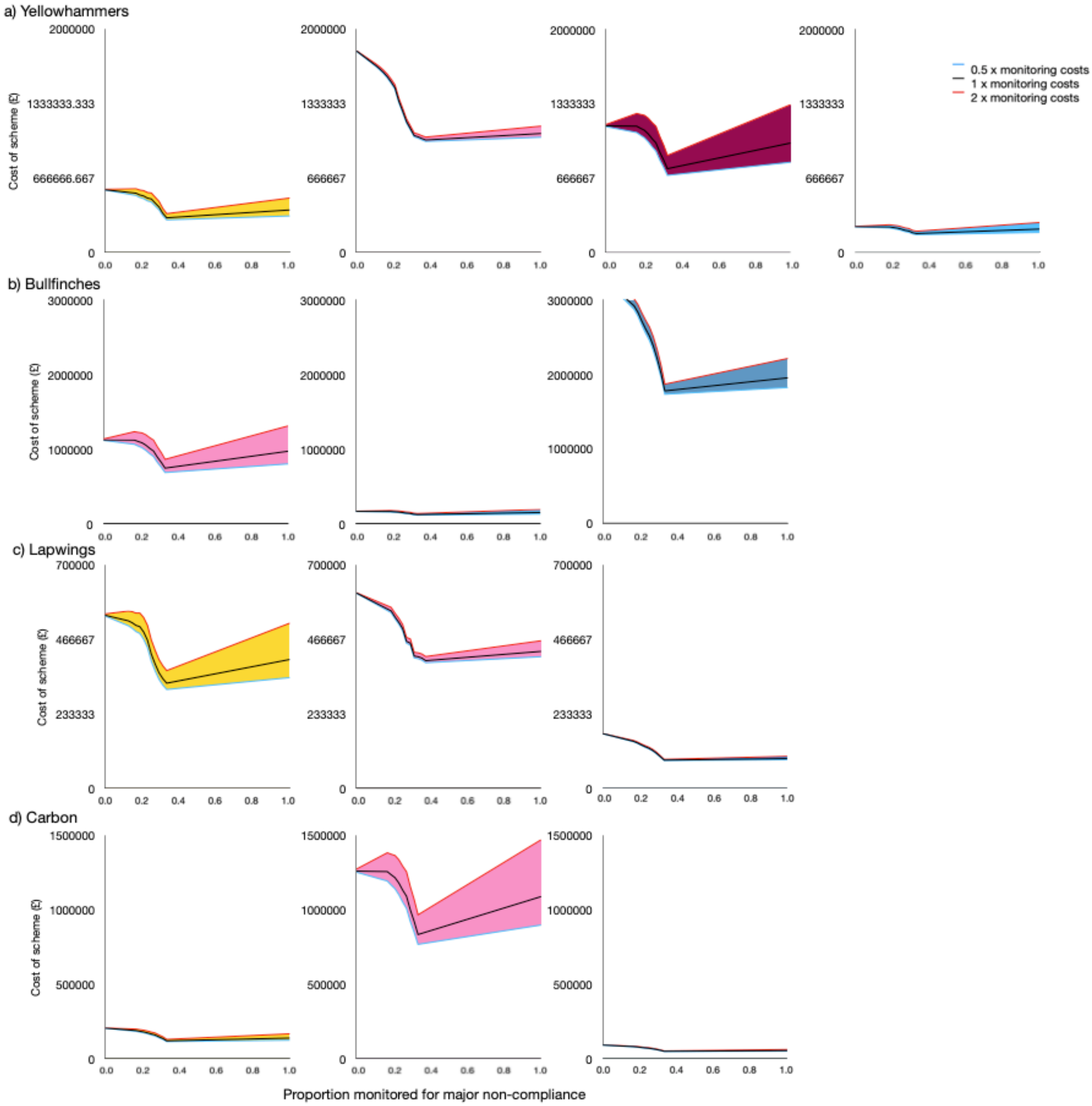

Figure B3. The cost of ten-year schemes at varying rates of monitoring that delivered fixed amounts of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) carbon with in-field sharing (orange), field-edge sharing (pink) and sparing (blue). The black line shows the baseline condition; the blue line shows costs when monitoring costs were halved and the red line when monitoring costs were doubled. Shading shows difference in overall cost when monitoring costs were halved and doubled. Of the two field-edge sharing interventions for yellowhammers, the lighter line is winter bird cover and the darker line is hedgerow creation. Of the two sparing interventions for bullfinches, the lighter line is scrub and the darker is woodland creation.

(iv) Area entered by participants

In the baseline analysis, we assumed all participants entered the mean area currently enrolled by participants in the CSS (Table B1). Allowing participants to enter larger areas reduced scheme costs since fewer participants were required in the scheme and so monitoring costs – which largely scaled with the number of participants, rather than area – were lower (Figure B4).

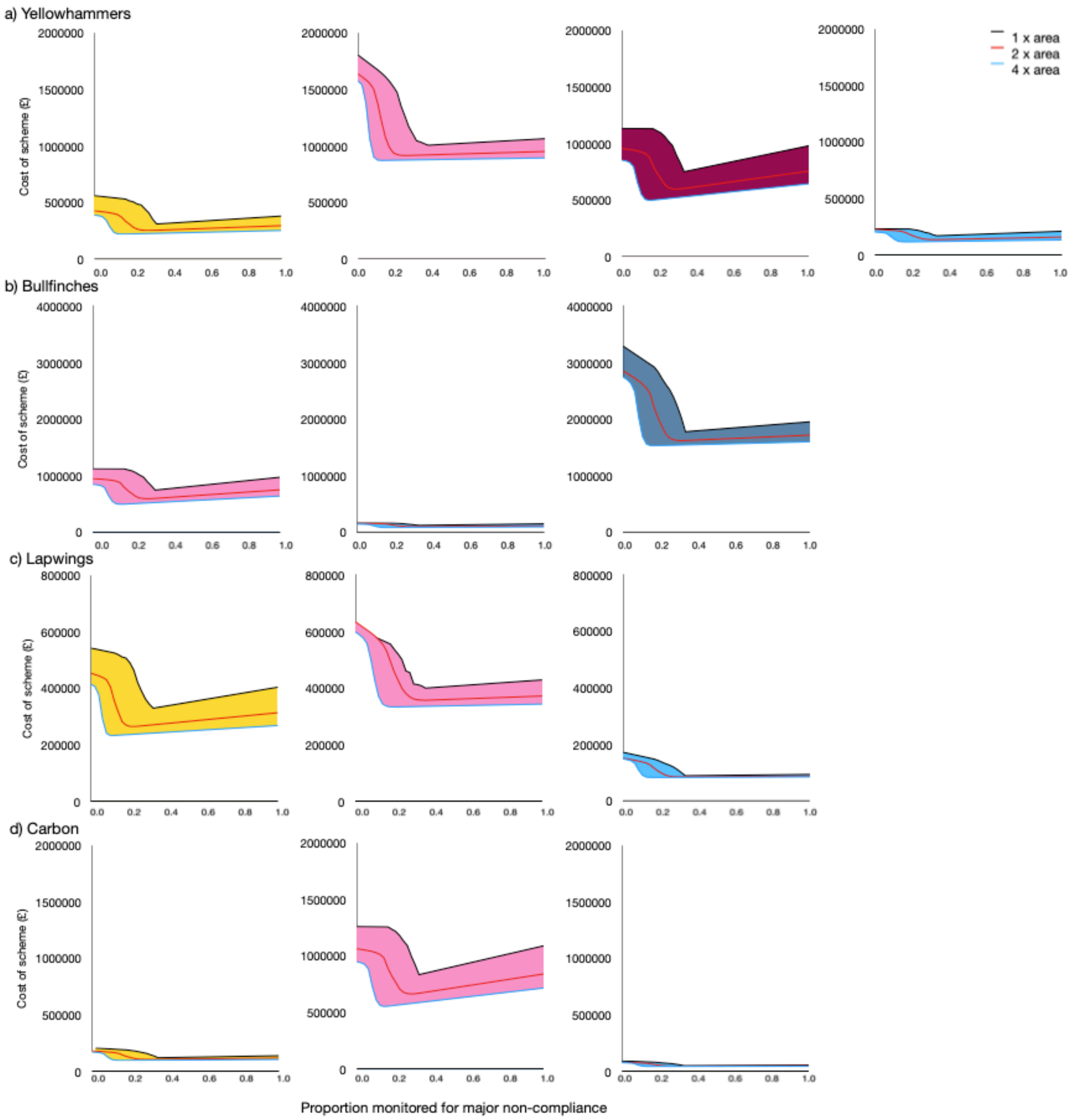

Figure B4. The cost of ten-year schemes at varying rates of monitoring that delivered fixed amounts of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) carbon with in-field sharing (orange), field-edge sharing (pink) and sparing (blue). The black line shows the baseline condition whilst the red line shows costs when the area entered by each participant was doubled and the blue line when it was quadrupled. Shading shows difference in costs when area enrolled was increased 4x vs the baseline. Of the two field-edge sharing interventions for yellowhammers, the lighter line is winter bird cover and the darker line is hedgerow creation. Of the two sparing interventions for bullfinches, the lighter line is scrub and the darker is woodland creation.

(v) Biodiversity benefit

Scheme costs were reduced when the anticipated benefit delivered by each intervention was increased (Figure B5). The cost of sparing exceeded sharing when the benefit associated with sharing was increased by the following factors while that of sparing is held constant: Yellowhammer-stubble, 1.9x; Bullfinch-hedge, 6.5x; Lapwing-stubble, 3.8x; Carbon-nil fertiliser, 2.3x.

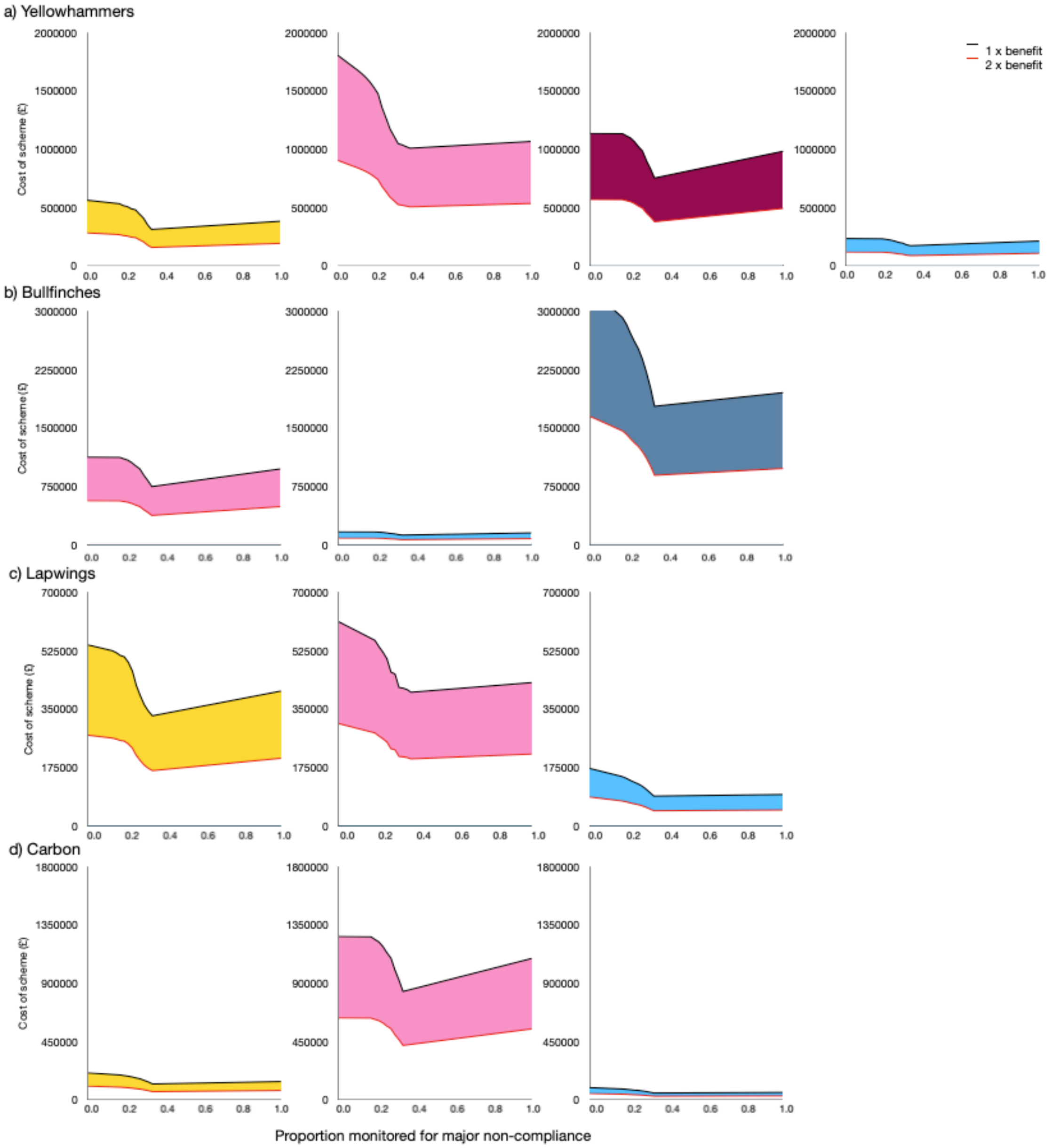

Figure B5. The cost of ten-year schemes at varying rates of monitoring that delivered fixed amounts of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) carbon with in-field sharing (orange), field-edge sharing (pink) and sparing (blue). The black line shows the baseline condition whilst the red line show when the per-area conservation benefit associated with each intervention was doubled; and shading shows the difference in costs. Of the two field-edge sharing interventions for yellowhammers, the lighter line is winter bird cover and the darker line is hedgerow creation. Of the two sparing interventions for bullfinches, the lighter line is scrub and the darker is woodland creation.

(vi) Availability of remote monitoring

We explored the cost implications of using remote monitoring in addition to in-person checks. However, remote monitoring offered only limited cost savings: Figure B6 shows the difference between costs at this level of remote monitoring vs all in-person (difference indicated by shaded area).

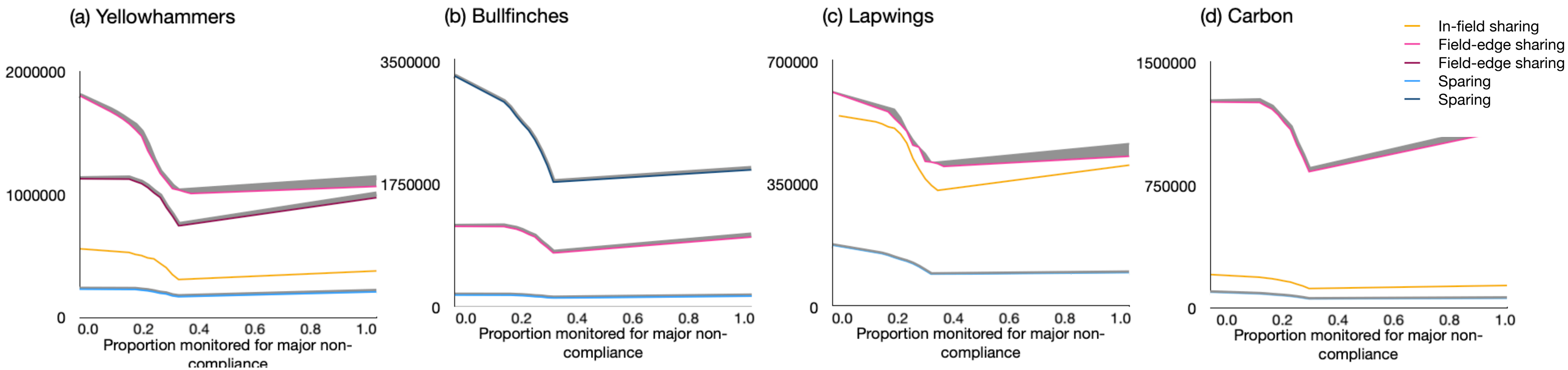

Figure B6. Total costs of in-field sharing (orange), field-edge sharing (pink) and sparing (blue) schemes for a range of monitoring rates that delivered the same amount of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) carbon with the optimal rate of remote monitoring (bottom of shaded band) and without remote monitoring (top of shaded band). (All pink and blue lines have some shading but in some cases it is barely visible). Of the two field-edge sharing interventions for yellowhammers, the lighter line is winter bird cover and the darker line is hedgerow creation. Of the two sparing interventions for bullfinches, the lighter line is scrub and the darker is woodland creation.

(vii) Fine rates required for current monitoring rate to be effective

We explored the extent to which fines must increase for the current monitoring rate to be associated with the lowest scheme cost (Figure B7). Scheme costs were minimised at the current monitoring rate (6%; dotted line) only when fines were increased between 11-18 times the present payment rates.

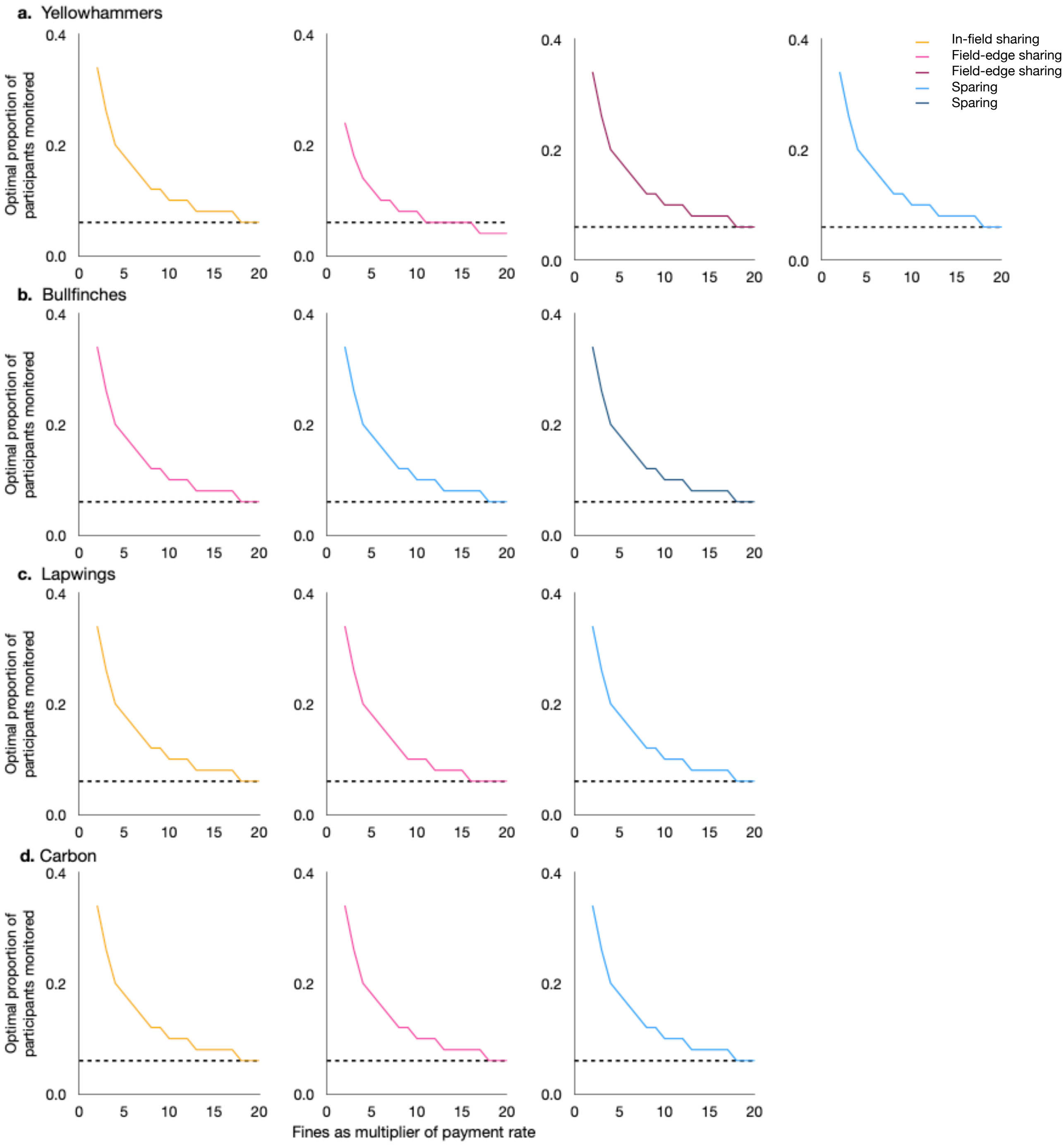

Figure B7. The optimal rate of monitoring for major non-compliance for a range of fine rates (presented as multipliers of the payment rate). All monitoring is in-person. Dotted line shows the current rate of monitoring in the Countryside Stewardship Scheme. Of the two field-edge sharing interventions for yellowhammers, the lighter line is winter bird cover and the darker line is hedgerow creation. Of the two sparing interventions for bullfinches, the lighter line is scrub and the darker is woodland creation.

Additional methods for the effectiveness monitoring analysis

i. Estimating sample sizes of Baker et al. (2012)

Baker et al. (2012) did not explicitly report sample sizes. Instead, they reported the number of unique 1km squares in which each species was recorded across all years separately for the arable, mixed and pastoral farmland classifications. These figures may underestimate sample sizes since any studied squares in which the species was not recorded are excluded. However, they may also overestimate sample sizes due to different squares being studied in different years. Therefore, we calculated sample sizes using the authors' statement that approximately 2000 lowland farmland squares were monitored each year. We assumed these 2000 squares are split between the arable, pastoral and mixed categories according to the ratios of the highest number of squares of each farmland type that any species was recorded in. At the $1km^2$ scale, the species reported in the most mixed and pastoral squares was the chaffinch (771 squares for mixed, 610 squares for pastoral farmland) whilst the skylark was reported in the most arable squares (936 squares). Using these raw numbers would give a total of 2317 squares (a suspected overestimate), and so scaling-down such that the total is 2000 (whilst maintaining relative proportions) gives sample size estimates of 808, 666 and 527 squares for arable, mixed and pastoral respectively. We therefore assumed these were the number of squares of each farmland type surveyed, on average, in each year of the study. Without this scaling-down, the same confidence interval (CI) width would be assumed delivered with a larger sample size, so the data requirement for precise CI's would increase.

ii. Estimating the standard errors of Newson et al. (2005)

Newson et al. (2005) reported bird population densities in a range of habitat types with bootstrapped CI's. Bootstrapped CI's are not calculated from the CI formula, but by repeatedly sampling the data, with replacement, and calculating the associated sample mean. The bootstrapped CI is then dictated by the range of these mean values having removed the most extreme from the distribution, e.g. the lower and upper 2.5% for a 95% CI. We can, however, assume the width of the confidence interval to reflect its standard error (multiplied by the *z* statistic at the level of accuracy studied), provided the upper and lower CI's are roughly equal in width (Haukoos & Lewis 2005). Here, we assumed all CI's to be adequately normally distributed (Figure B8). So, we calculated standard errors by assuming the width of the CI was equal to 1.96 (the z score at 95% confidence) multiplied by the standard error.

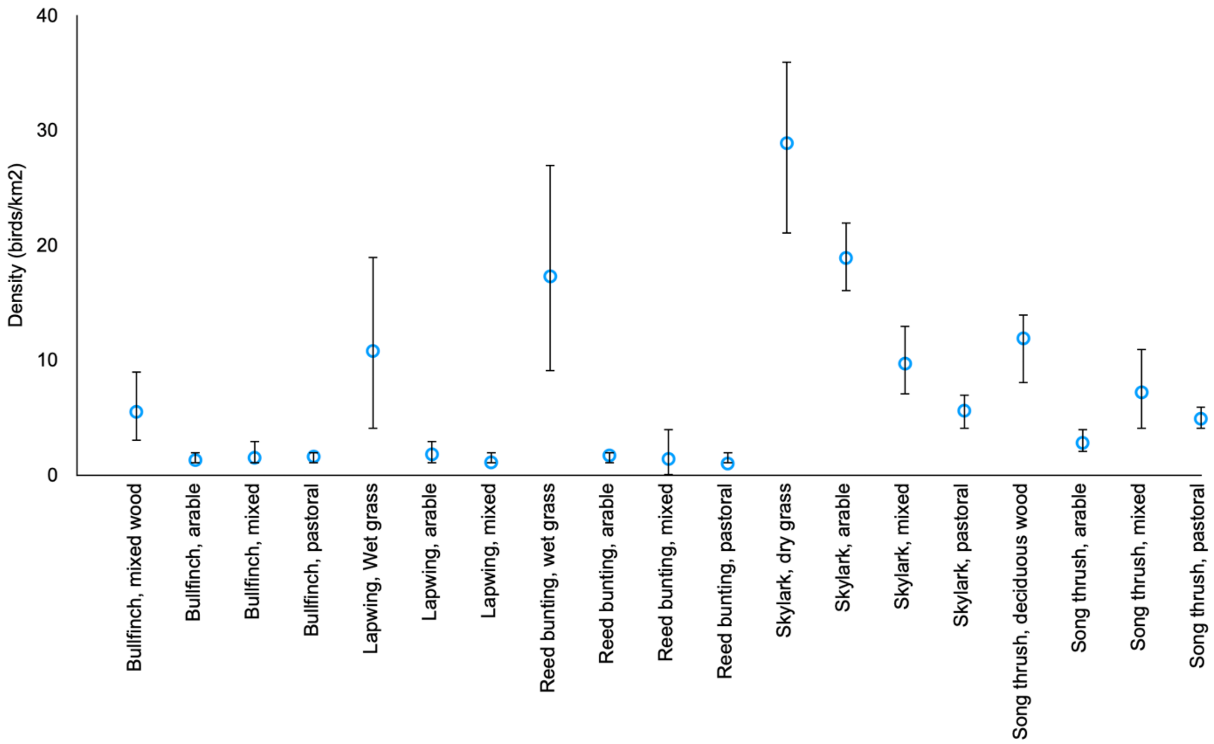

Figure B8. Species-specific densities reported in a range of habitats by Newson et al. (2005). The authors' bootstrapped 95% confidence intervals are shown.

iii. Estimating the sample sizes of Newson et al. (2005)

In Newson et al. (2005), the sample sizes reported are the number of 200m transects of each habitat type surveyed in all of the UK and not the number of 1-km squares surveyed. To be comparable to Baker et al. (2012), we found the number of 1-km squares surveyed by assuming ten 200m transects were surveyed for each square (as per BBS protocol).

iv. Estimating present-day sampling effort in BBS

In the main paper, we explored the precision of estimates delivered by the 2018 BBS volunteer effort. The BBS report the overall increase in area sampled (Harris et al. 2019), but do not provide a breakdown by habitat type. Therefore, for each farmland type, and each category of (semi-)natural habitat, we assumed the sample size had increased by the same factor of increase as the total area sampled in England.

Appendix C: Chapter 3

1. Comparisons to national averages

First, we assessed how representative our sample was of arable farmers in England before considering how well our models predicted participation in the closest analogues to the options studied in existing AES.

i. Assessing how representative our sample is of English arable farmers

We found that our sample over-represented younger farmers and under-represented older farmers compared to the ages of UK farmers (Figure C1a; Defra 2018; data was not available for only arable farmers in England). Our sample also over-represented larger farms compared to data for arable farmers in England (Defra 2019a; Figure C1b).

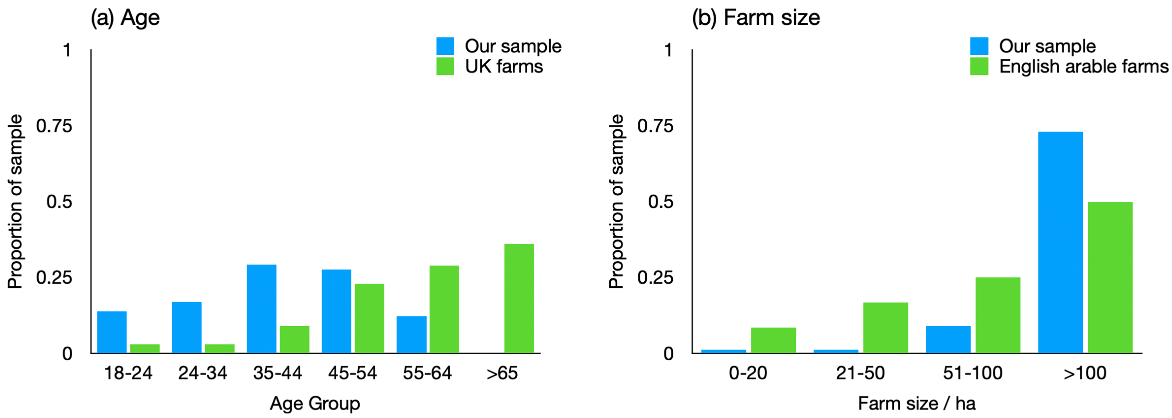

Figure C1. The (a) age groups and (b) farm sizes of our sampled farmers compared to national averages.

ii. Assessing the accuracy of our model in predicting current participation in AES

To assess the accuracy of preferences revealed by stated preference experiments (e.g. choice experiments), it is useful to compare results to preferences revealed by real behaviour (Hanley et al. 2003). To provide some insight into the accuracy of our predictions, we compared our predictions of the benefit, and therefore the participation, delivered for a given spend to the observed participation in current English AES. Here we are only considering spend on maintenance payments to farmers. We obtained data on the participation rates in current AES in England via Freedom of Information requests to the Rural Payments Agency. We received information on the area enrolled in the closest analogues to our studied interventions in the Countryside Stewardship Scheme (CSS) and Environmental Stewardship Scheme (ESS) in 2019. Based on our estimates of the per unit area benefit delivered by each of our interventions (Appendix A), we estimated the total benefit delivered by the area enrolled in each intervention in the CSS and ESS combined. We calculated the cost of delivering that benefit based on the payment rates offered by the CSS and ESS (Natural England 2013; Defra 2019b) over 20 years using a 3.5% discount rate (following HM Treasury 2018). We then adjusted these cost and benefit figures by 0.017, because our sample of 118 farmers managed ~1.7% of the arable land in England (Defra 2019c). The resulting cost/benefit for each intervention was plotted as a black cross in Figure C2, with our model output shown as a coloured line.

Our model explored far greater environmental outcomes than those delivered by current schemes which in many cases are very small (many of the black crosses appear near-zero on the y-axis). We did overestimate the benefit that would be delivered by stubble/spring cropping and winter bird cover for yellowhammers, suggesting we may be overestimating the participation delivered by a given payment rate; so sharing may in practice be more costly than implied by our results.

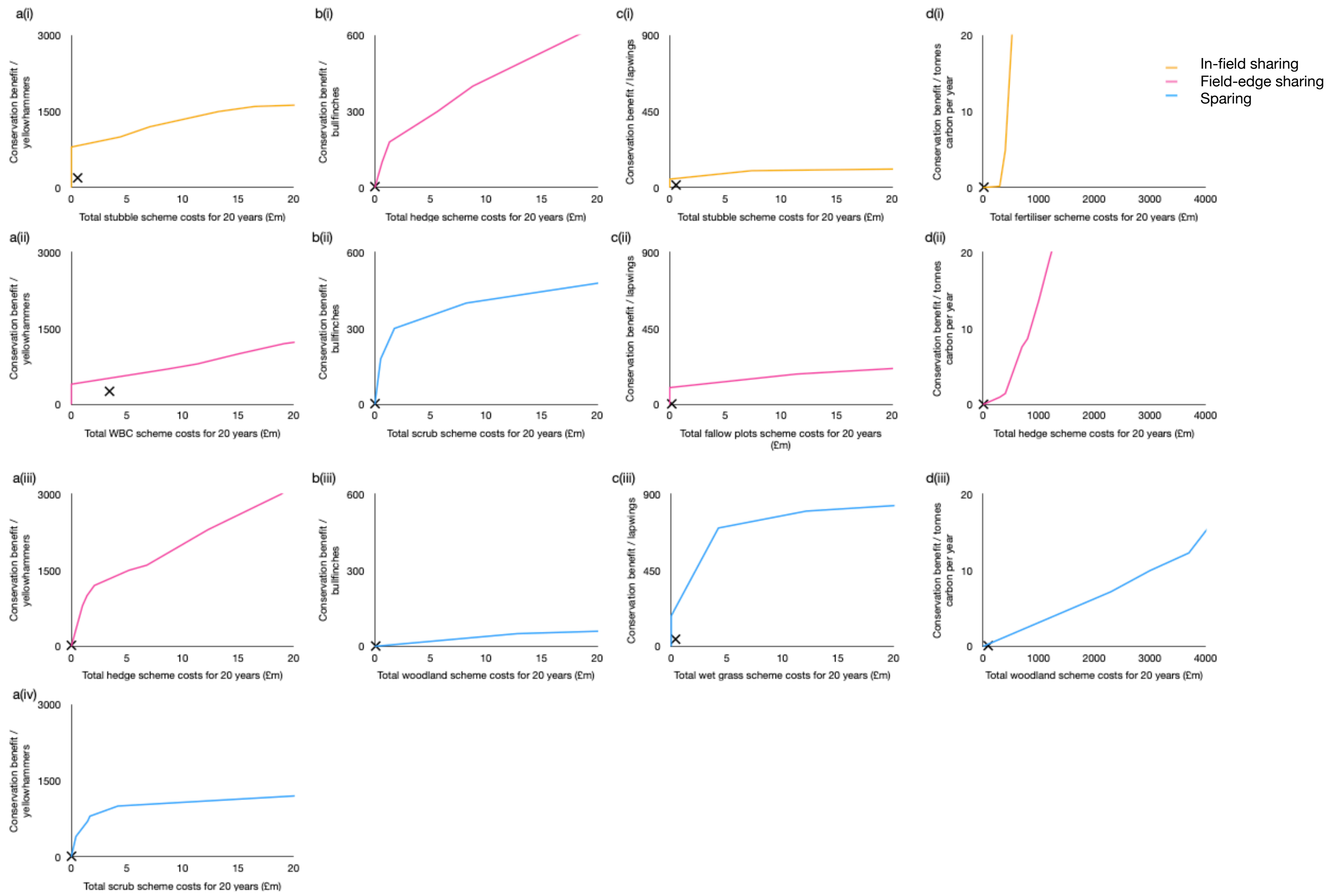

Figure C2. The conservation benefit in terms of (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) tonnes C/y projected, based on choice experiment modelling, to be delivered with a budget up to £20m spent over 20 years on maintenance payments to farmers who implement in-field sharing (orange), field-edge sharing (pink) and sparing agri-environment interventions. Black crosses indicate the benefit delivered by equivalent interventions in the Countryside Stewardship and Environmental Stewardship Schemes and the associated costs, given scheme payment rates for an area of England in 2019 equivalent in size to the land farmed by our study farmers.

2. Reasons participants like and dislike sharing and sparing options

Participants found more reasons to not implement sparing options than sharing options (Figure C3; participants could select multiple reasons for each intervention). The most common concern for sparing was cultivating the land after the contract ended. For field-edge sharing, the time required was most concerning while the reasons stated for not implementing in-field sharing were diverse.

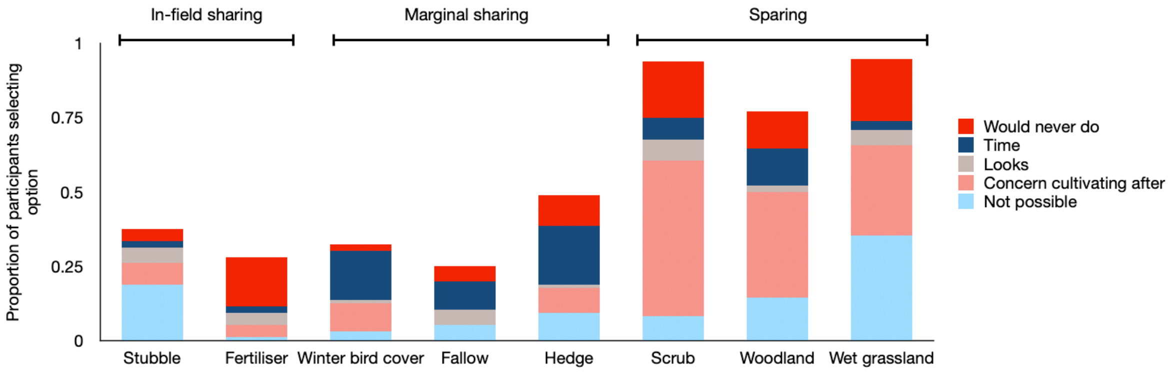

Figure C3. Reasons for which participants would not implement sharing and sparing options in terms of it being something they would never do (red), the time taken (dark blue), changes in how the land would look (grey), concern cultivating the land after the contract ended (pink) and it not being possible on the land farmed (light blue). Participants could select multiple reasons for each option.

In general, participants found more reasons to be discouraged (Figure C4b) than encouraged (Figure C4a) from implementing sharing and sparing options. Of the options studied, the most positive benefits were associated with stubble/spring cropping, namely in terms of weed/pest control and soil health. Far fewer benefits were perceived to be associated with other options. Participants were most commonly discouraged from in-field sharing due to impacts on crop yields. Many were discouraged from field-edge sharing and sparing due to both crop yield and weed/pest control effects. For sparing, impacts on soil health were also concerning.

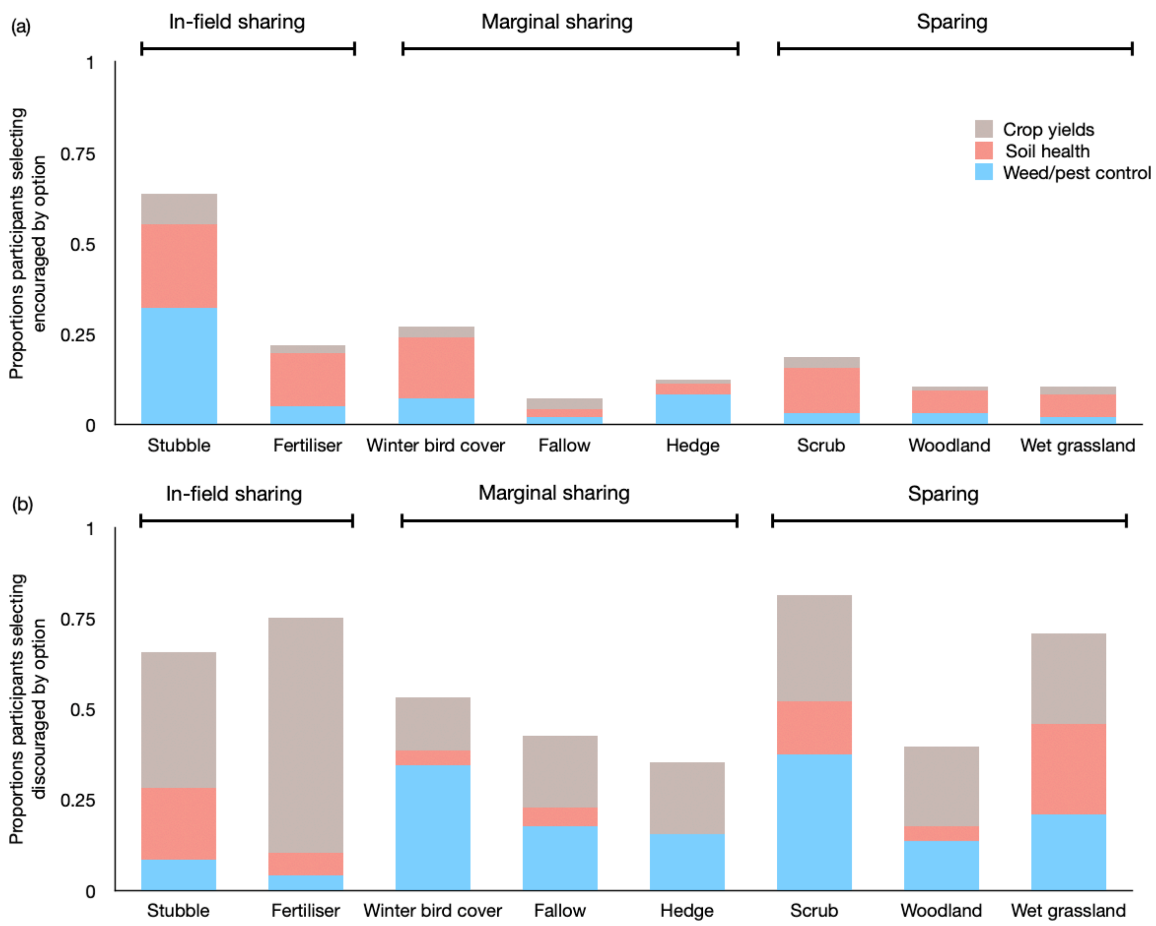

Figure C4. The reasons participants were (a) encouraged and (b) discouraged from participating in sharing and sparing options due to impacts on crop yields (grey), soil health (pink) and weed/pest control (blue).

Appendix D: Chapter 4

Comparison to current schemes of estimated payments required by farmers

Stated preference methods, such as choice experiments, rely on participants' stated intentions rather than actual behaviour; that intentions may not reflect actual behaviour is a clear limitation of this approach. Therefore, it is useful to assess the accuracy of estimates derived from stated preference experiments to preferences revealed by decision-making in the real world (Hanley et al. 2003). We compared our predictions of the benefit, and therefore the participation, delivered by a given spend to the benefit delivered given existing participation and payment rates in current English AES. Here we are considering the cost of maintenance payments to farmers only. We obtained data on the participation rates in current AES in England via Freedom of Information requests to the Rural Payments Agency. We received information on the area enrolled in the closest analogues to our studied interventions in the Countryside Stewardship Scheme (CSS) and Environmental Stewardship Scheme (ESS) in 2019. Based on our estimates of the per unit area benefit delivered by each of our interventions (Appendix D), we estimated the total benefit delivered by the area enrolled in each intervention in the CSS and ESS combined. We calculated the cost of delivering that benefit based on the payment rates offered by the CSS and ESS (Natural England 2013; Defra 2019b) over 20 years using a 3.5% discount rate, following HM Treasury (2018). We then adjusted these cost and benefit figures by 0.017, because our sample of 118 farmers managed ~1.7% of the arable land in England (Defra 2019c). The resulting cost/benefit for each intervention is plotted as a black cross in Figure D1, with our model output shown as a coloured line.

Our model was generally good at predicting participation at the payment rates offered by existing schemes. However, our model poorly predicted the cost of delivering yellowhammers with stubble/spring cropping and winter bird seed plots, which we found to require far less than the compensation currently offered. This suggests that, for these two sharing interventions, we may be overestimating the participation delivered by a given payment rate in our sampled farmers; so sharing may in practice be more costly than implied by our results.

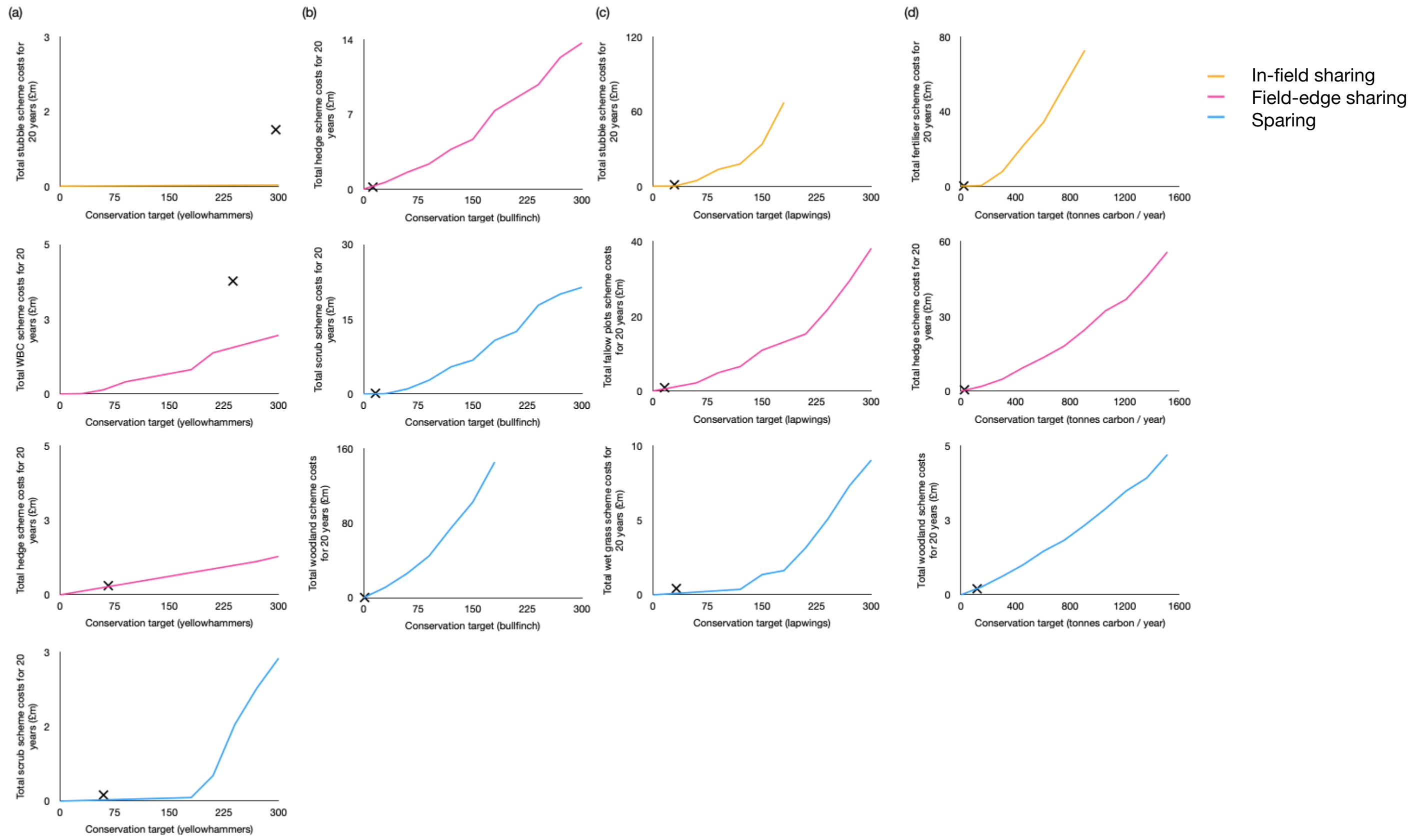

Figure D1. The projected costs based on choice experiment modelling of the maintenance payments made to farmers who participate in 20-year schemes that deliver (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) reduced net carbon emissions with in-field sharing (orange), field-edge sharing (pink) and sparing (blue) interventions. Black crosses show the benefits delivered by equivalent interventions in current Countryside Stewardship and Environmental Stewardship Schemes and associated costs given scheme payment rates for an area of England in 2019 equivalent in size to the land farmed by our surveyed farmers. Some targets could not be delivered even with all the studied farmers enrolled hence lines are not plotted across full range of x-axis: lapwings by stubble/spring cropping (in-field sharing), carbon by reduced fertiliser (in-field sharing) and bullfinches with woodland creation (sparing).

Sensitivity tests

We assessed the sensitivity of our overall findings to different land-use approaches and environmental outcomes in five main ways, as follows.

iv. In addition to land sharing and simple (2-compartment sparing), we explored the costs of achieving environmental outcomes via a third strategy: 3-compartment sparing, where high-yield farming sees land spared and managed as low-yield farmland, and other land restored to (semi)-natural habitat (Finch et al. 2019; Feniuk et al. 2019). In the context of our study, 3-compartment sparing used both sharing and sparing interventions to deliver the target outcomes. To do so, we used our existing estimates of the cost of delivering the target outcomes with the studied sharing and sparing interventions to establish the combination of interventions that delivered the outcomes at least cost. We assessed the cost of delivering each incremental increase in the target outcomes with each intervention. We assumed the intervention that cost least to the taxpayer would be used to deliver that increment. Given the contribution of sharing and sparing interventions towards delivering the target outcomes, we estimated the food production lost by the enrolled farmers as detailed elsewhere (Methods; Appendix D)

Combining sharing and sparing interventions in 3-compartment sparing delivered the outcomes at less cost than 2-compartment sparing, whether considering biodiversity outcomes alone (Figure D2a) or including carbon (Figure D2b). Whilst we previously found that delivering all outcomes purely with sharing was more expensive than sparing, there was variation across outcomes when considering each individually: yellowhammers and high numbers of bullfinches were delivered at less expense with sharing compared to sparing (Figure D3a-b). Furthermore, using both sharing and sparing interventions to deliver lapwings offered cost savings; the upper 30% of the lapwing target was delivered at less cost with stubble/spring cropping and fallow plots, compared to sparing (Figure D3c). Sparing alone was least costly for carbon (Figure D3d). Therefore, a combination of sharing and sparing options delivered all our conservation targets (i.e., when proportion of target = 1; Figure D2b) at 61% the cost to the taxpayer of simple 2-compartment sparing (Figure D2). The food production lost under 3-compartment sparing was also lower compared to sparing, though by a smaller margin (Figure D2, red line). Indeed, for all outcomes at high targets, the food lost approaches that of 2-compartment sparing despite 3-compartment sparing continuing to offer savings to the taxpayer.

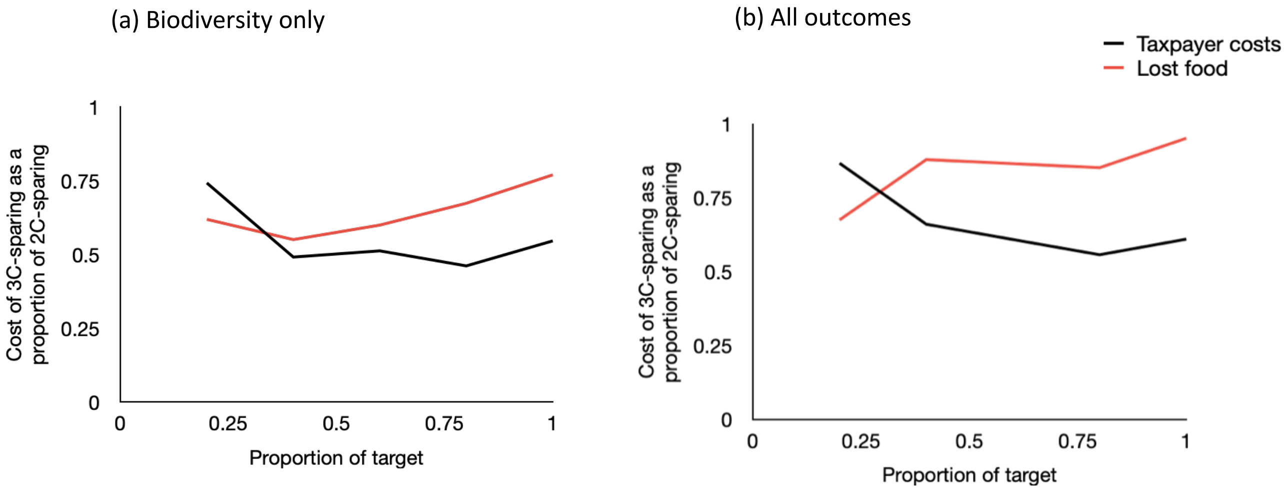

Figure D2. For (a) biodiversity outcomes only and (b) biodiversity and carbon outcomes, the difference in taxpayer costs (black; compensation payments, capital, administration and compliance monitoring), and lost food production (red), of 3-compartment sparing schemes relative to classical (2-compartment) sparing schemes that delivered a range of proportions of the target outcomes.

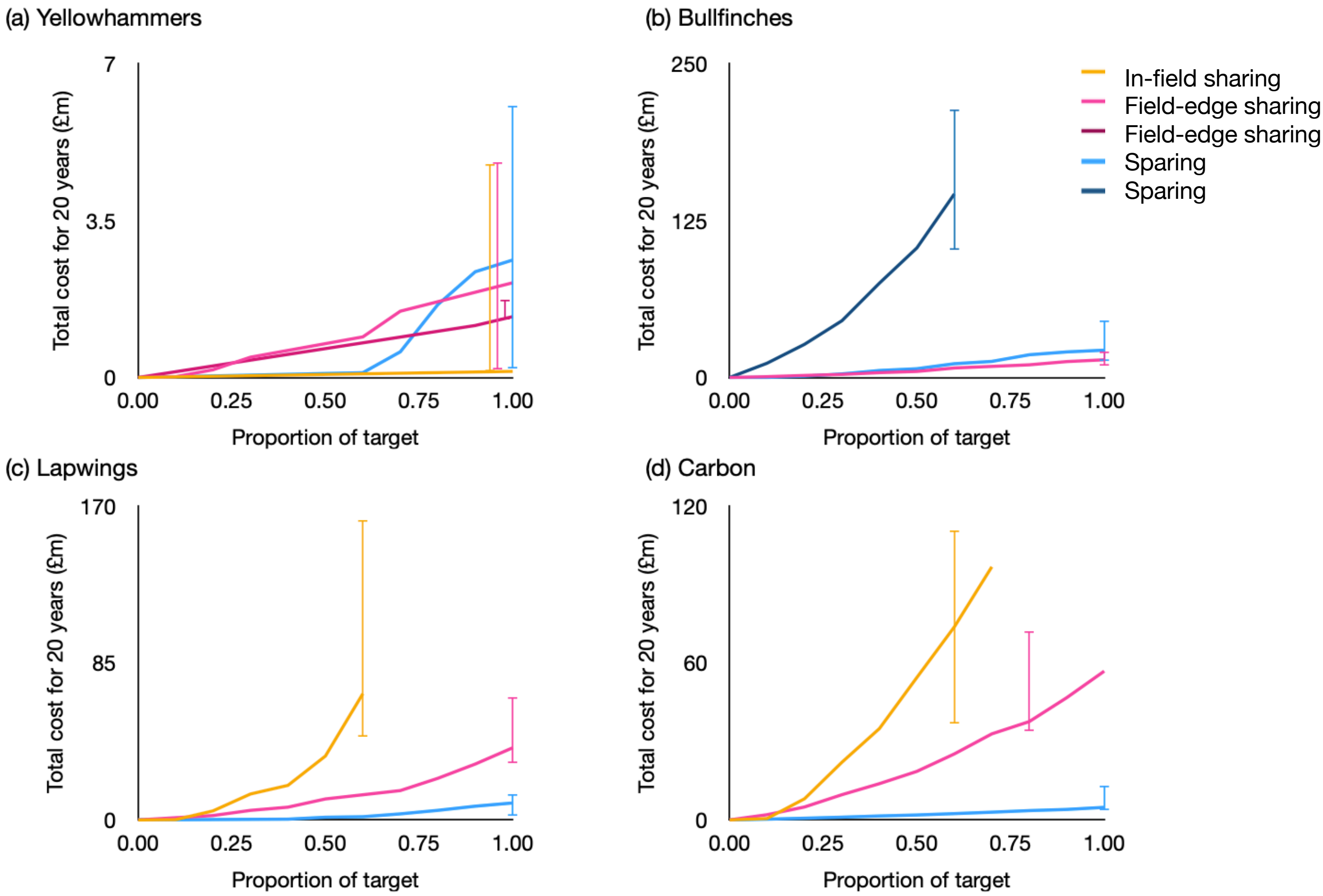

Figure D3. The taxpayer costs for 20 years schemes (including maintenance payments to farmers, capital costs, administration costs and compliance monitoring costs) that deliver varying proportions of each outcome with in-field sharing (orange), field-edge sharing (pink) and sparing (blue) interventions. Two field-edge sharing interventions were studied for yellowhammers (hedgerows, darker pink; winter bird cover, lighter pink) and two sparing interventions were studied for bullfinches (woodland creation, darker blue; scrub creation, lighter blue). Some interventions could not deliver the targets even with all studied farmers enrolled in the scheme hence not all lines plotted across x-axis range.

v. We also explored the consequences for our overarching findings of considering only biodiversity outcomes and excluding action to lower net carbon emissions. We found sparing remained less expensive than sharing but was 77% the cost of sharing (Figure D4) compared to 48% when action to reduce net carbon emissions was included. This demonstrates the disproportionate efficiency offered by delivering carbon, versus biodiversity, with sparing which we explored further in Figure D4.

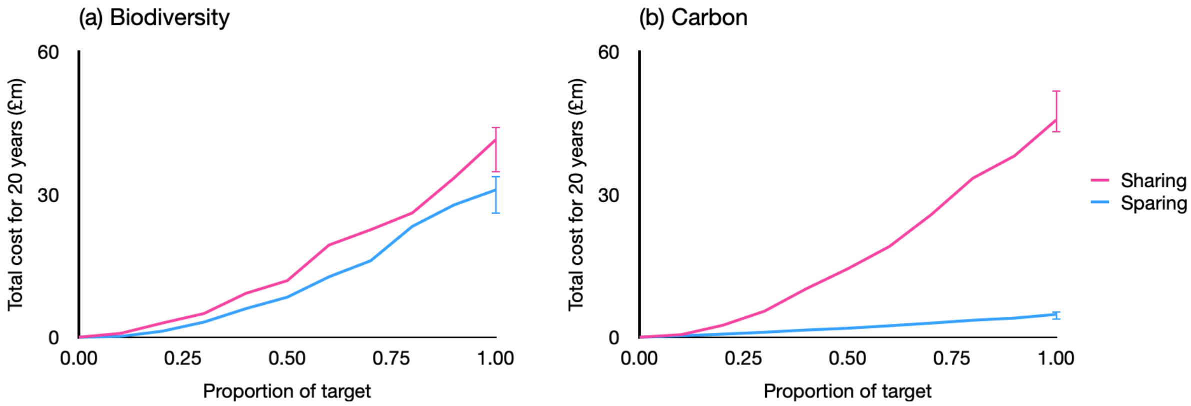

Figure D4. The taxpayer costs (compensation payments to farmers, plus capital, admin and compliance monitoring costs) of 20-year schemes that deliver varying proportions of (a) the target biodiversity outcomes and (b) the carbon emissions reduction target. 95% bootstrapped confidence intervals reflect uncertainty in compensation payments to farmers.

vi. We explored the consequences of varying the size of the carbon target. Figure D5 shows the taxpayer cost of delivering the combined biodiversity and reduced net carbon emissions outcomes when the carbon target was varied between 0 and the original target (1557tC/y; set such that the spend on biodiversity and carbon outcomes was equal under sharing) whilst the biodiversity targets were held constant at 300 bullfinches, lapwings and yellowhammers. For all targets, sparing was less expensive than sharing, though the margin by which sparing was less expensive increased at higher carbon targets. We also explored scheme costs when increasing the carbon target to 12x the original target given the scale of action required to reach net-zero emissions, assuming payments for any intervention could not exceed £3000/ha/y (Figure D5b). The cost of delivering targets with sharing interventions increased rapidly at higher targets; it was not possible to deliver >2x the original target without exceeding payments of £3000/ha/y, which is far more than current payment rates, whilst sparing could deliver up to 12x the original target.

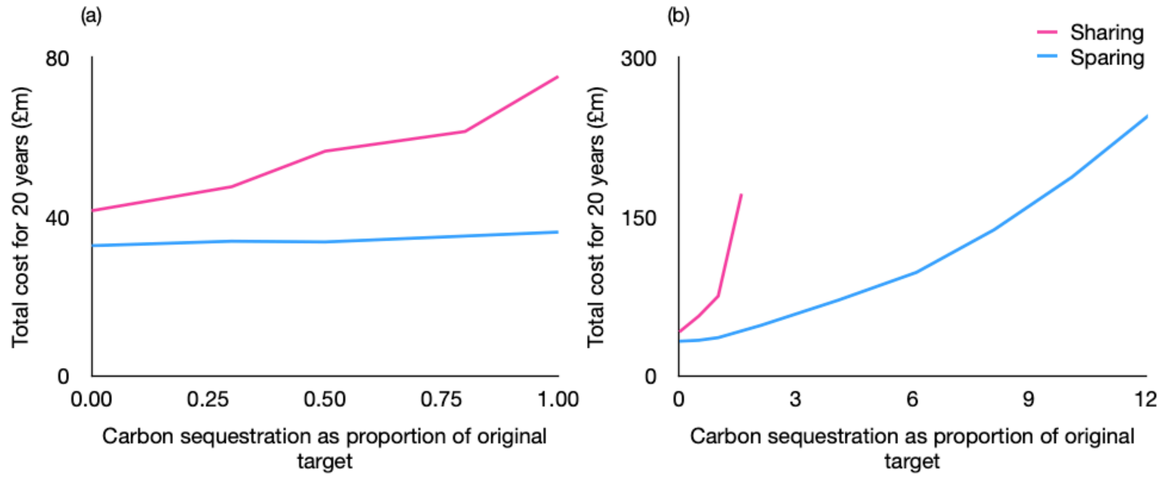

Figure D5. The combined taxpayer cost (compensation payments to farmers, plus capital, admin and compliance monitoring costs) of 20-year schemes that deliver biodiversity and carbon whilst biodiversity target was held at 300 bullfinches, lapwings and yellowhammers and the carbon target was varied as a proportion of the original target of 1557tC/y between (a) 0 and 1 and (b) between 0 and 12 where payments for any intervention could not exceed £3000/ha/y.

vii. Given the uncertainty in our estimates of the per-area benefit delivered by the studied interventions, we explored the extent to which the benefit delivered by sparing could be degraded before sharing became the cheaper strategy. We found sharing cost the taxpayer less in delivering the target biodiversity and carbon outcomes only when the benefit delivered by sparing was reduced by more than a third (Figure D6).

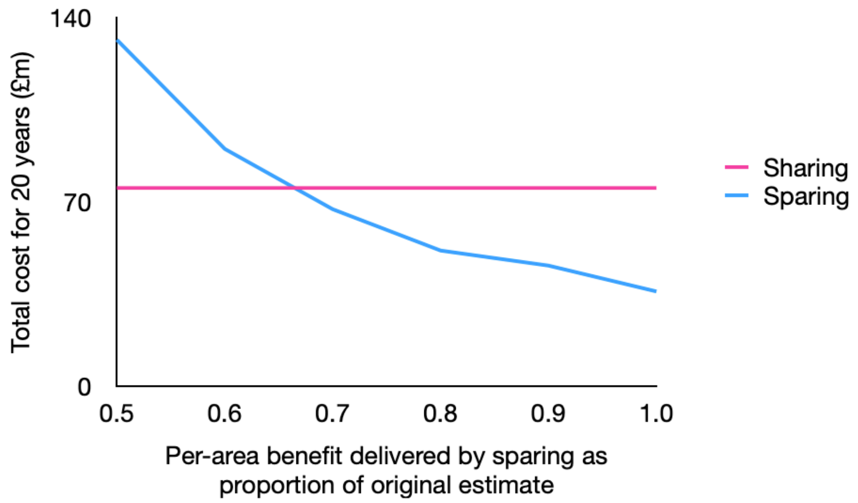

Figure D6. The taxpayer costs (compensation payments to farmers, plus capital, admin and compliance monitoring costs) of 20-year schemes that deliver the target biodiversity and carbon outcomes by sharing and sparing where the benefit assumed to be delivered by sparing is degraded by varying amounts.

viii. To confirm whether the difference between sharing and sparing schemes was significant, we found the 95% confidence interval of the difference following the bootstrap method (Figure D7). The difference is significant whether considering all outcomes or only biodiversity outcomes.

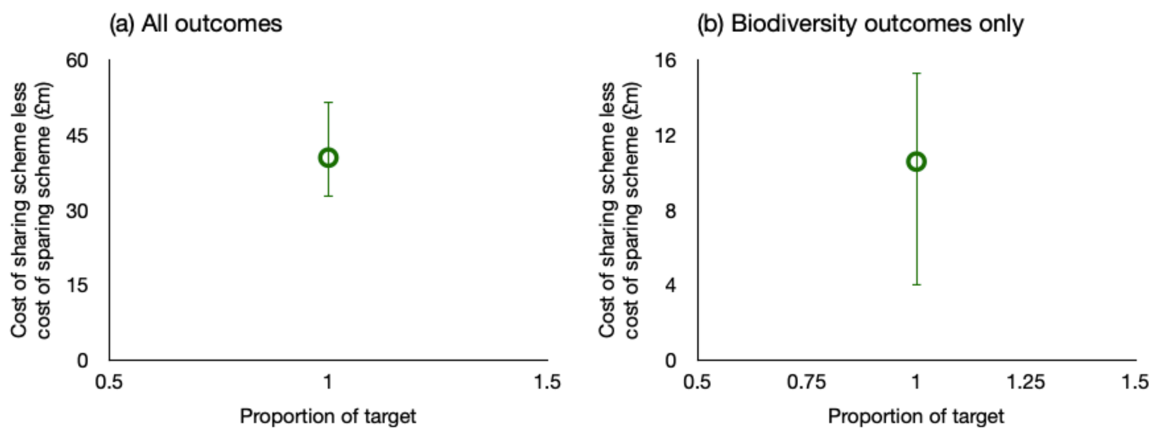

Figure D7. The cost of delivering (a) all outcomes and (b) biodiversity outcomes only with sharing less the cost of sparing. 95% bootstrapped confidence intervals were calculated from finding the difference across 1000 replicates and excluding the top and bottom 2.5%.

Methods: Figures and tables cited in the main text

Table D1. The attributes and levels studied in the choice experiment. In the main text, 'sharing' incorporates both in-field options which affect food-producing practices across the whole field, and field-edge which involve addition of an intervention outside the area used to produce food, often the field margin.

Attribute	Levels							
Type of intervention	In-field sharing		Field-edge sharing			Sparing		
Within type intervention	Stubble, spring cropping	50% N fertiliser reduction	Winter bird cover	Fallow plots	Hedge[1]	Scrub	Woodland	Wet grass
[2]**Area/ha**	10, 20, 50	10, 20, 50	5, 10, 20	5, 10, 20	2, 4, 8	10, 20, 50	10, 20, 50	10, 20, 50
Duration/ years	5, 10, 20	5, 10, 20	5, 10, 20	5, 10, 20	10, 20, 50	10, 20, 50	10, 20, 50	10, 20, 50
Payment rates £/ha	40, 80, 120, 160, 240	130, 260, 400, 550, 800	175, 350, 525, 700, 1050	170, 340, 500, 700, 1000	459, 918, 1360, 1700, 2720	300, 600, 900, 1100, 1700	300, 600, 900, 1100, 1700	300, 600, 900, 1100, 1700

[1]For hedgerow creation, areas and payment rates were presented per km length hedgerow.

[2]50ha area requirements were not presented to participants farming <100ha.

Table D2. Fit for a range of mixed logit models: those that included all participants, and that excluded participants who opted out of every choice; and those assuming either a normal, log-normal (as presented in the main text), distribution or fixed value for the payment parameter.

	All participants		Opt-outs excluded		
	Normally distributed payment	Log-normally distributed payment	Normally distributed payment	Log-normally distributed payment	Fixed payment
Log-likelihood	-1147	-1121	-1125	-1129	-1132
R^2	0.29	0.29	0.28	0.26	0.26
AIC	2339	2264	2294	2303	2305
BIC	2452	2374	2405	2414	2411

Capital costs

Capital costs are one-off expenses incurred when creating a habitat. Participants were told that capital costs would be paid in-full, in addition to the annual compensation payments. The options studied that involved capital costs were the creation of hedgerows, wet grassland, scrub and woodland. Our estimates of capital costs are as follows:

i. Hedgerow creation

The capital costs of hedgerow creation were estimated at £9.40/m following the rate paid by the CSS in 2019 (Defra 2019a). For the sparing options, we considered capital costs to be those associated with planning, infrastructure and labour.

ii. Wet grassland creation

The costs of planning, infrastructure and labour for wet grassland creation were estimated at £1454/ha (2019 GBP) based on Ausden and Hirons (2002) who explored the costs associated with creating three wet grassland sites.

iii. Scrub creation

Scrub is typically created by allowing natural regeneration (RSPB 2020b). The costs associated with planning scrub creation were not documented in the literature, so they have been taken from Ausden and Hirons (2002) who estimate the planning costs for three wet grassland sites as £278/ha. We assumed there to be no infrastructure or labour costs because scrub regeneration is assumed to occur naturally. Therefore, the capital costs assumed associated with scrub creation were £278/ha.

iv. Woodland creation

We assumed woodland creation requires planting since natural regeneration is slow and can only be achieved with exclusion of grazers. Again, planning costs were not well documented so the £278/ha from Ausden and Hirons (2002) was again used. We estimated infrastructure costs according to the Countryside Stewardship Woodland Creation Grants Scheme (Natural England 2018). The costs of trees, planting, weeding and guards (but no fencing), plus replacements following mortality, gave an estimated infrastructure cost of £1613/ha. These costs do not appear to include the associated labour. CJC Consulting (2014) estimated the labour costs of woodland in Wales at £423/ha and at £1530/ha for a broadleaved woodland managed for game/biodiversity in south-west England. We took the mid-point of these estimates. Allowing for inflation, and combining these costs, the capital costs of woodland creation were estimated at £2976/ha.

Compliance monitoring

The costs of compliance monitoring were studied together with spend on compensation payments since with less monitoring, non-compliance increases, and more participants must be paid to enrol in the scheme to make up the benefit lost to non-compliance (Ozanne et al. 2001). In summary, our approach (detailed below) used utility theory to assess the non-compliance arising at given 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 this rate of monitoring using cost estimates from current schemes.

Following Ozanne et al. (2001), for a given sharing or sparing intervention, we assumed farmer *n* will participate in a scheme that costs c_n (£/y) and is compensated at rate *P* (£/y):

$$P - c_n > 0 \quad \textbf{[1]}$$

Here, we assumed the cost, c_n, to a participant of enrolling in the scheme was their willingness to accept payment as calculated in the choice experiment. Participants could enrol varying areas; we assumed they would enrol the maximum area given the payment offered, subject to Equation [1] holding true.

To determine the level of non-compliance at any given monitoring and payment rate, we assumed the utility of non-compliance is the payoff given it is undetected, less the cost if caught and fined; and farmers will cheat when this exceeds the utility of complying (as in Ozanne et al. (2001)), i.e.:

$$(P - xc_n)(1 - l) - Pfl > P - c_n \quad \textbf{[2]}$$

where x is the extent to which cheating reduces the cost of compliance, l is the likelihood that non-compliance is fined, and f is the fine as a proportion of the payment rate. Following Gómez-Limón et al. (2019), we explored different degrees of non-compliance: total (where farmers incur 0% of the cost and deliver 0% of the benefit) and minor (where farmers pay 70% of the cost and deliver 70% of the benefit). Minor non-compliance may be deliberate or accidental (e.g. arising from not fully studying the agreement terms (Finn et al. 2009)). The 70% bound is arbitrary but was set according to the extent to which we judged it possible to accidentally not comply (P. Carey, *pers. obs.*).

Parameterising these equations

We parameterised equations [1] and [2] using the best available parameter estimates, as follows.

Fines: We set fines at 1x and 2x the annual payment rate for minor and major non-compliance respectively. This approximately follows the fines applied when participants overstate the area enrolled in an option (Defra 2019b).

Probability non-compliance is fined: We assumed that majorly non-compliant participants would always be fined when monitored (following Gómez-Limón et al. 2019)). For in-field sharing and sparing, we assumed minor non-compliance (e.g., removing stubble early, over-grazing of grassland, etc.) would only be detected in 50% of cases. Minor non-compliance of field-edge sharing options was considered detectable in 100% of cases (since it would most probably involve interventions being implemented over less than the required area, which should be detected during in-person visits).

Remote monitoring: Given recent developments in remote sensing, we assumed major non-compliance could be detected remotely (as well as in-person) for field-edge sharing and sparing but assumed detection of minor non-compliance required in-person checks. For in-field sharing options, we assumed remote sensing could not detect non-compliance at all (J. Griffin, *pers. comm.*).

Monitoring costs: We used cost estimates of current CSS monitoring to estimate the costs of monitoring participants (P. Carey; *pers. obs*.). The time and mileage of professional surveyors travelling to the site and conducting the inspection was valued at £240. Analysis of habitats was estimated at £700 per km^2; this was adjusted according to the area of the option in question. Finally, aerial analysis was assumed to be used to quantify the area under in-field sharing and sparing options, at the cost of £60. Remote monitoring was assumed to cost £180/site, regardless of the option in question (approximately in line with the Rural Payments Agency (RPA 2012).

Identifying the most efficient payment and monitoring rate

Based on this parameterisation, we simulated the costs of 20-year schemes which delivered the target outcomes assuming a 3.5% discount rate (following HM Treasury (HM Treasury 2018)); all costs were adjusted to 2019, using a UK GDP deflator index (Bank of England 2021). We explored costs for a range of monitoring rates and varied the proportion of monitoring that was conducted remotely vs in-person. We calculated total costs as the sum of compensation payments, capital costs, administration costs, and compliance monitoring costs, less fines. Having explored a range of payment and monitoring rates, we identified that which delivered the target outcomes at least cost. This payment rate informs Figure 4.2a, the cost of compensation payments to farmers, with the cost of delivering the required monitoring rate then detailed in Figure 4.2d.

Estimating lost food production and lost gross margin

Next, we estimated the food energy and gross margin lost by the surveyed group of farmers in delivering out target environmental outcomes. We asked farmers about the crop/livestock types they produced as well as the associated areas, yields, selling prices and variable costs for the 2018 harvest year. Participants provided an amalgamated value of variable costs which included fertiliser, crop protection, seed, water and haulage (where relevant), as well as reporting fertiliser use and fertiliser costs separately. Any missing information was completed using averages from the Farm Business Survey (FBS 2020). We used this information to estimate the food energy and gross margin lost by farmers participating in our schemes. Methods differ for each intervention type. In summary, we found the tonnes of production and converted this to food-energy using standard conversion ratios (see below). We then found the value of this lost production by multiplying by participants' selling prices and subtracting their associated variable costs (following FBS 2020). In estimating lost production we assumed that yields vary within and between fields and, where possible, that farmers would implement interventions on the least profitable parts of the farm (see below). We also recognised that crops/livestock are rotated such that interventions cannot only take land away from the least profitable crop/livestock type, unless the interventions can also be moved each year. Where relevant, we also added any implementation costs to our lost gross margin estimates; here we included only the cost of materials, and not the associated labour. The full methods for each intervention are described in detail below; in explaining our approach to estimating lost food-energy, we provide the method for estimating the tonnes of each crop/livestock type lost for each intervention before explaining afterwards, for all interventions together, how this was converted to lost food-energy.

i. In-field sharing: Stubble retention followed by spring cropping

We specified to farmers that, in line with expert guidance, this intervention can only be applied to wheat, barley and/or oats (RSPB 2022). We assumed the area would be met by taking area, in order, from the barley, oats and then wheat grown by the farmer (based on the order in which least gross margin is lost, on average, in data reported in the FBS (2020)). To estimate lost food production, we found the average spring-sown yield as a proportion of the winter-sown yield of the same crop and assumed lost production was initial production (i.e. area x winter-sown yield) multiplied by this proportion. The method for converting this lost production to lost food-energy is provided later.

To estimate lost gross margin, we estimated the gross margins of winter-sowing these crops according to the standard method of multiplying the participant's reported production (yield x area) by selling price and subtracting their variable costs (FBS 2020). Then, using the FBS (2020), we found the average spring-sown gross margin for each crop type as a proportion of the average winter-sown gross margin and assumed that each participant's winter-sown gross margin would be reduced by this proportion. In this way, we estimated lost gross margin as the difference between gross margins for winter- and spring-sown crops.

ii. In-field sharing: 50% reduction in use of N fertiliser

In the choice experiment this intervention required farmers to reduce their current use of inorganic nitrogen fertiliser by 50% on wheat, barley, oil seed rape, sugar beet and/or potatoes. Participants were asked to state their current fertiliser use, either per tonne of product or per unit area, in the survey. Since this option can be rotated each year, we assumed farmers would choose to enrol their crop(s) with the lowest gross margin. We established the relationship between fertiliser use and yield using Kindred et al. (2008). This paper studied only winter wheat, and average fertiliser rates differ across crops, but no comparable studies were available for other crops. To allow us to make inferences for the other relevant crops, we first took the relationship for winter wheat from Kindred et al. (2008; Figure D7a) and found the fertiliser application rates as a proportion of the FBS (2020) mean application rate for winter wheat. We plotted this against the yield reported in Kindred et al. (2008) when fertiliser application was reduced 50% as a proportion of the yield associated with the initial fertiliser application rate (Figure D7b). Second, for each participant and crop type, we found their stated fertiliser application rate as a proportion of the mean reported by the FBS (2020) for that crop. Third, given this fertiliser use, we used Figure D7b to predict their new yield, as a proportion of their initial yield, given a 50% reduction in their fertiliser use. We multiplied their initial yield by this proportion to find their new yield at the 50% lower application rate, and multiplied this by the area enrolled in our scheme to find lost production. Then, we calculated lost gross margin by multiplying the difference in yield by the area enrolled in our scheme and their selling price, assuming it was unchanged, before subtracting the variable costs assumed changed only by the 50% lower cost of fertiliser.

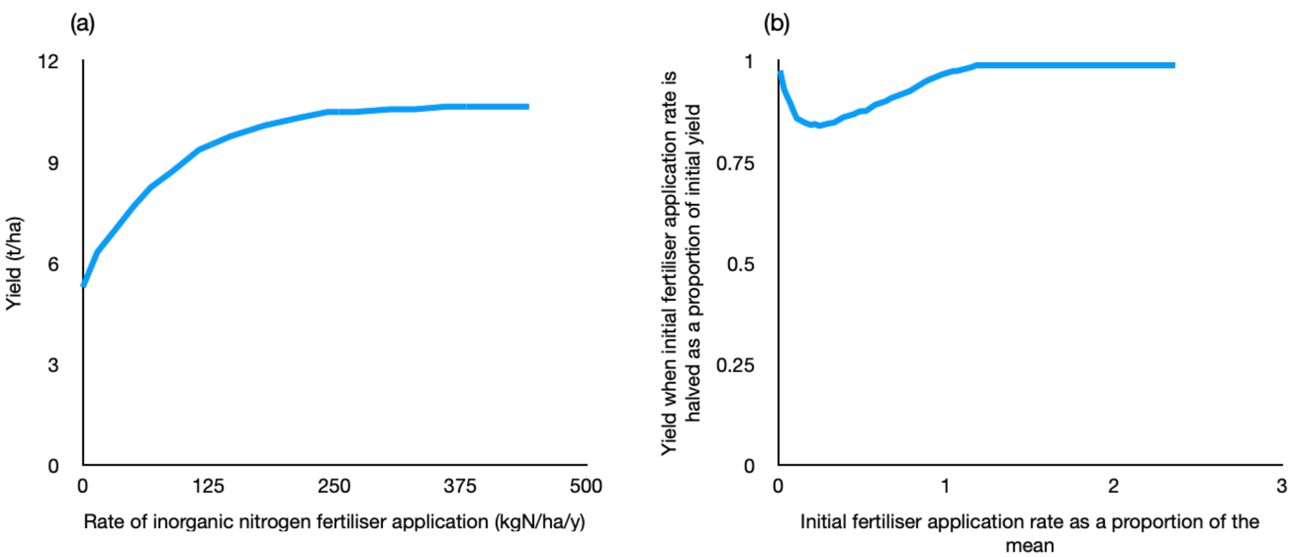

Figure D7. (a) From Kindred et al. (2008), the yield associated with varying rates of inorganic N-fertiliser application to winter wheat and, derived from (a), in (b) we plotted the winter wheat yield (as a proportion of the initial yield) when the fertiliser application rate is halved against the initial fertiliser application rate (as a proportion of the 2018 mean fertiliser application rate reported by FBS (2020) for winter wheat).

iii. Field-edge sharing: Winter bird seed and fallow plots

The choice experiment stated that fallow and winter bird seed plots must be created in arable areas. Since these plots can be moved each year, we assumed they would only be implemented in the crops with the lowest gross margins, though to ensure that plots are well spaced out, we assumed no more than a quarter of each crop type could be plots (as recommended by the RSPB 2021). Because we therefore assumed plots would be located in the least profitable crops as they get rotated around the farm each year, we did not consider between-field variation to be important (as we do below for hedgerows and sparing below). It made sense to allocate plots to the least profitable crops, rather than to the least profitable fields, because our data suggested that crops varied more in their gross margins than fields. We did, however, assume the plots would be allocated to the least profitable parts of fields, as described below.

We assumed farmers would allocate plots to the lowest-yielding, and therefore least profitable, parts of fields first, and that the amount of this more marginal land would scale with the size of the farm. We first considered the cumulative production, as a percentage of the total field production, that would be lost by taking incrementally larger areas of the field out of production, assuming yield was constant across the field. Then, in Figure D8, we plotted this relationship according to the yields measured in 5x5m patches across a field in Muhammed et al. (2016), plotting the lowest-yielding 5x5m patches first, and the highest-yielding patches last (blue line). This showed that cumulative production is not directly proportional to field area: some parts of a field yield far less than others. However, we assumed that not all the lowest-yielding 5x5m patches would be adjacent to each other such that, due to blocking constraints, in implementing a fallow or winter bird seed plot, some higher-yielding land would unavoidably be taken out of production. We assumed that 2/3rds of the area for the plot would be the lowest-yielding patches, and 1/3 would yield the average field-wide yield: we again plotted this relationship in Figure D8 (green line). Then, to estimate the production lost by participants, we calculated the area of plots they implemented as a percentage of the area of the least profitable crop(s) in which we assumed they were established. We used the green-line relationship in Figure D8 to establish the production lost on the crop area taken out by the plots, as a percentage of the total field's production. We multiplied this by farmer's reported yield, and the area of plots, to find the lost production from the area enrolled in our schemes. We assumed that all yield was lost where plots were established, with no yield lost in surrounding areas. We estimated the value of this lost production using farmers' reported selling prices and variable costs, assuming they did not vary across the field. Last, we allowed allocation to move to the next least profitable crop if less gross margin was lost by adding to the worst parts of those fields, rather than continuing to use more land in the fields growing the least profitable crop. We estimated lost gross margin in this way for fallow plots and for winter bird seed plots where we also added the cost of seed which Nix (2018) reported at £48/ha/yr.

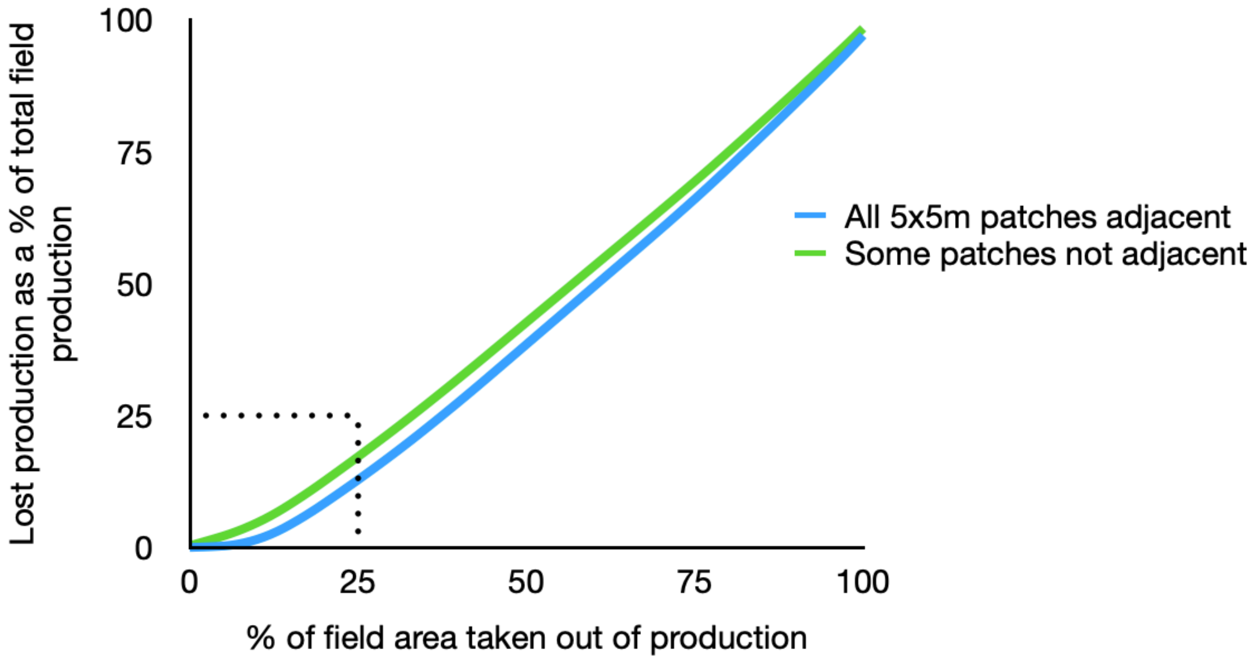

Figure D8. The cumulative production lost by taking incrementally larger proportions of a field out of production when assuming (a) that yields vary according to Muhammed et al. (2016) and blocks of only the lowest-yielding patches can be taken out of production (blue line) and (b) that only two-thirds of blocks can be established in the lowest-yielding patches, and one-third established in areas supporting the field-average yield (green line). We assumed interventions could only take-up one quarter of field area, so the dotted box shows the part of the graph relevant to our calculations.

iv. Field-edge sharing: Hedgerow creation

In the choice experiment farmers were told that hedgerows could only be created in arable areas (since this affected the carbon sequestered; IPCC 2019). In creating hedgerows, we assumed all production was lost in the area occupied by the hedge and some production was lost in the area surrounding the hedge due to shading (Raatz et al. 2019), dealing with each loss in turn as set out below.

We estimated the production lost to the hedge based on it being 6m wide. To allow for crop rotation, and given the hedge cannot be moved each year, we assumed this area would come from all crop types, weighted by their relative areas. To minimise lost gross margin, we assumed farmers would create hedges in the lowest-yielding fields. We established how yields vary between fields using data from Muhammed et al. (2016) who reported wheat yields for a number of fields across one farm. We plotted the production across the whole farm relative to farm area, assuming that fields vary in their production potential according to Muhammed et al. (2016; Figure D9). In applying Figure D9 to our data, first we assumed all crops/livestock were rotated around all the fields on the farm, where the relative areas of each crop/livestock remained the same as in the year we collected data. Second, we calculated the area of fields that would not be bordered by hedgerows, assuming fields were 9ha in area (Marshall et al. 2006) and square. Third, we calculated this area as a percentage of the total farm area and used Figure D9 to estimate the percentage of the production that would be lost assuming the hedgerows were allocated to the lowest-yielding fields. In addition, we assumed the edges of fields, where hedges were located, would yield less than field centres. As for fallow and winter bird seed plots, we used Figure D8 to estimate the within-field difference in yield between where the hedge would be implemented and the field-average yield. We thus assumed that the production lost was from the area occupied by the hedge from all crops, weighted by their relative areas, and adjusted for both between-field and within-field yield variation. We used farmers' reported crop-specific selling prices and variable costs to estimate the gross margin associated with this lost production, assuming crop-specific selling prices and variable costs did not vary within or between fields.

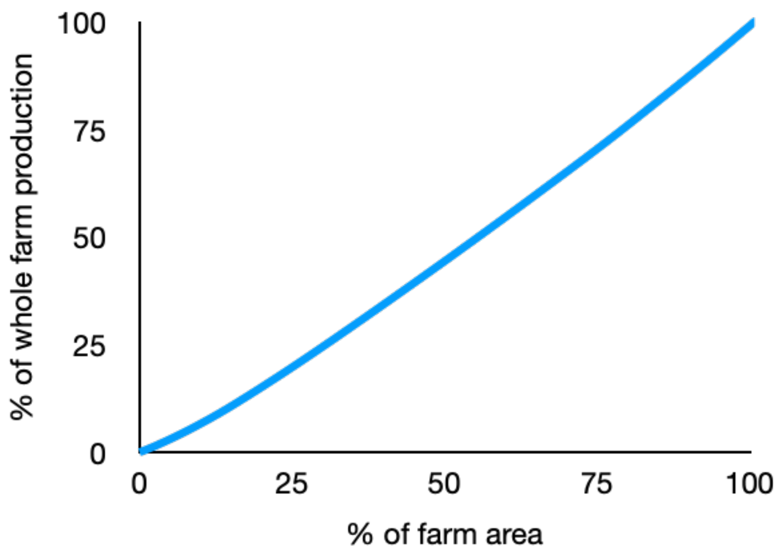

Figure D9. Cumulative production, as a percentage of total production, across the whole farm plotted for incrementally larger proportions of the total farmed area assuming that yields vary between fields. In plotting this, we averaged across the rotation assuming all crops/livestock were rotated around all fields (derived from Muhammed et al. (2016)). Only between-field, and not within-field, variation is included here.

We then calculated the production lost from hedgerow shading based on Raatz et al. (2019) who found yields adjacent to hedgerows were 14.5% lower, on average, than the in-field yield up to 17.85m from the field edge. For a boundary that didn't contain a hedgerow, 7.8% of the yield was estimated to be lost up to 6.93m into the field. However, this was based on full-size hedgerows. Therefore, we assumed that the full shading effect would not be realised until year 7 (based on a growth rate of 0.6m/y and that the full effect occurs when the hedges are 4m high (Raatz et al. 2019)) with yield lost to shading increasing linearly in years 1-6. Furthermore, we assumed half of hedges would border one field, and half of the hedges created would border two fields and thereby shade twice as much crop. We estimated the production lost by shading by multiplying these anticipated yield losses by the area affected and averaging across each year of the 20-year scheme. We estimated lost food production by adding the production lost to shading to the production lost on the area occupied by the hedge. To estimate the value of this lost production, we multiplied lost production by farmers' reported selling prices and subtracted costs for each relevant crop/livestock type. Finally, we estimated lost gross margin by adding this value of lost production to the average annual maintenance costs of £40/ha/y (Nix 2018).

v. Sparing

Sparing results in total loss of production and gross margin on the land taken out of production. Given rotation, we assumed land for sparing would come from all crops/livestock, weighted by their relative areas. In this way, we may overestimate the food production lost to sparing since farmers may be able to spare land without affecting the more profitable parts of the rotation. We also assumed that fields vary in their profitability, and that the least productive fields would be spared first; within-field variation was not relevant here since the whole field is taken out of production. Here, we may again be overestimating food production lost to sparing since we assumed that all crops/livestock were rotated around all fields when, in reality, the most profitable aspects of the rotation may be less often grown on the fields with the least production potential. We used the extent of yield variation across fields again based on Figure D9 (derived from Muhammed et al. (2016)). Using this relationship, we adjusted the yield of the relevant crop/livestock types lost according to the proportion of the farm that was spared. To estimate lost food production, we used this, and farmers' reported yields, to estimate the crop/livestock-specific tonnes of product lost.

To estimate lost gross margin, we found forgone output by multiplying lost production by participants' reported selling prices and subtracting variable costs. We assumed variable costs did not differ across fields, and therefore used the reported mean variable costs for the relevant crop/livestock types. In estimating lost gross margin, to the value of lost production we also added the annual maintenance costs associated with materials for the spared habitats in our estimates of lost gross margin. These were estimated, in 2018 GBP, at £0/ha/y for scrub (given restoration typically involves natural regeneration; RSPB 2020) , £50/ha/y for woodland (based on Nix 2018) and £50/ha/y for wet grassland (based on Ausden and Hirons 2002).

Converting lost tonnes to lost food energy

We followed the method of Finch et al. (2019) to convert our estimates of lost tonnes of harvested products to lost food energy. For crops, this involved using fixed, crop-specific estimates of the proportion of the total harvested production used for human consumption versus animal feed and adjusting for the proportion of the crop that is edible. Then, estimates of the energy content of specific crops, incorporating feed-conversion ratios where crops are fed to livestock, were used to estimate lost food energy. For livestock products, we converted estimates of tonnes product to food energy by multiplying by the edible fraction and the estimated energy content of each product.

Responses of unstudied species to sharing and sparing

We estimated the proportion of bird species that occur in the UK but not on land farmed at any yield according to the densities reported in Lamb et al. (2019). For every species with a breeding population above 50 pairs, Lamb et al. (2019) presented species densities in a range of habitats calculated from data collected by both the RSPB and the Breeding Bird Survey (BBS). We used the RSPB dataset since this included 16 species for which BBS-based density estimates could not be obtained. We excluded 8 species with breeding season distributions that did not include lowland areas in England/Wales; we assumed these species would not be found on the land of the farms studied, regardless of whether it was managed as farmland or (semi-)natural habitat. For the remaining species, we estimated the proportion not found on farmed land. We considered a species not to be present in a given habitat when their density was <0.1 birds/km^2. Lamb et al. (2019) studied a range of farmland habitats: arable, improved grassland and rough grassland. Of the 114 species considered, 23% were not found on any of these farmland habitat types whilst 36% were not found on arable land.

Appendix E: Chapter 5

Comparison to current schemes of estimated payments required by farmers

Stated preference methods, such as choice experiments, rely on participants' stated intentions rather than actual behaviour; that intentions may not reflect actual behaviour is a clear limitation of this approach. Therefore, it is useful to assess the accuracy of estimates derived from stated preference experiments to preferences revealed by decision-making in the real world (Hanley et al. 2003). We compared our predictions of the benefit, and therefore the participation, delivered by a given spend to the benefit delivered given existing participation and payment rates in current English AES. Here we are considering the cost of maintenance payments to farmers only. We obtained data on the participation rates in current AES in England via Freedom of Information requests to the Rural Payments Agency. We received information on the area enrolled in the closest analogues to our studied interventions in the Countryside Stewardship Scheme (CSS) and Environmental Stewardship Scheme (ESS) in 2019. Based on our estimates of the per unit area benefit delivered by each of our interventions (Appendix D), we estimated the total benefit delivered by the area enrolled in each intervention in the CSS and ESS combined. We calculated the cost of delivering that benefit based on the payment rates offered by the CSS and ESS (Natural England 2013; Defra 2019b) over 20 years using a 3.5% discount rate, following HM Treasury (2018). We then adjusted these cost and benefit figures by 0.017, because our sample of 118 farmers managed ~1.7% of the arable land in England (Defra 2019c). The resulting cost/benefit for each intervention is plotted as a black cross in Figure D1, with our model output shown as a coloured line.

Our model was generally good at predicting participation at the payment rates offered by existing schemes. However, our model poorly predicted the cost of delivering yellowhammers with stubble/spring cropping and winter bird seed plots, which we found to require far less than the compensation currently offered. This suggests that, for these two sharing interventions, we may be overestimating the participation delivered by a given payment rate in our sampled farmers; so sharing may in practice be more costly than implied by our results.

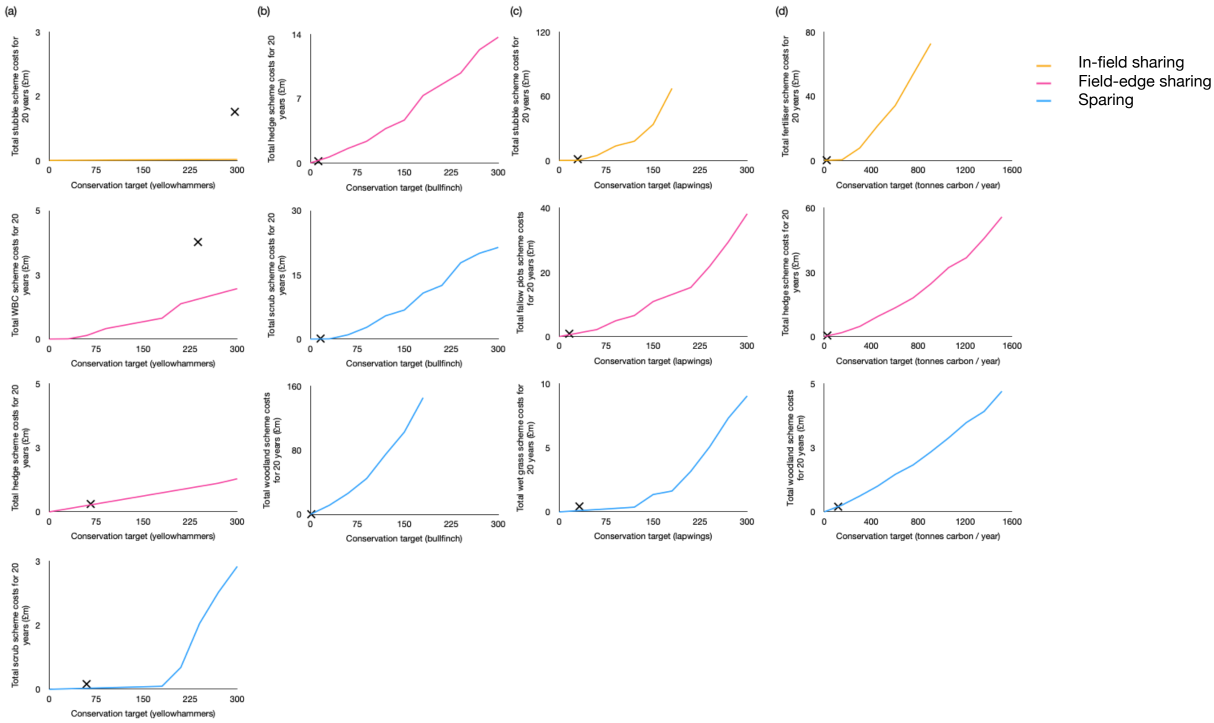

Figure E1. The projected costs based on choice experiment modelling of the maintenance payments made to farmers who participate in 20-year schemes that deliver (a) yellowhammers, (b) bullfinches, (c) lapwings and (d) reduced net carbon emissions with in-field sharing (orange), field-edge sharing (pink) and sparing (blue) interventions. Black crosses show the benefits delivered by equivalent interventions in current Countryside Stewardship and Environmental Stewardship Schemes and associated costs given scheme payment rates for an area of England in 2019 equivalent in size to the land farmed by our surveyed farmers. Some targets could not be delivered even with all the studied farmers enrolled hence lines are not plotted across full range of x-axis: lapwings by stubble/spring cropping (in-field sharing), carbon by reduced fertiliser (in-field sharing) and bullfinches with woodland creation (sparing).

Appendix F: Chapter 6

Choice Experiment Summary

Here, we based our estimates of the payments required by farmers to participate in our habitat creation/maintenance schemes on the results of a discrete choice experiment which is detailed in full in Chapter 4 but summarised here. The experiment was conducted with 118 arable farmers in England and bordering areas of Wales who farmed 1.7% of English arable farmland. Participants were recruited through a variety of channels, including through farming organisations, newsletters, Twitter and online forums. The experiment was designed to compare farmers' willingness to accept (WTA) payment to participate in a broader range of interventions; here we use only the results for the creation of areas up to 50ha in size of scrub, woodland and wet grassland.

The experiment was conducted online using Qualtrics and asked participants to make 12 choices between options that differed in terms of the intervention involved, the area and duration over which it must be implemented and the offered payment rate. Participants were able to choose not to accept any of the options presented. We analysed the results using mixed logit modelling which allows for participants to vary in their preferences. We converted the preferences produced by mixed logit modelling to WTA estimates (Chapter 4) and this gave a distribution of farmers' WTA payment to participate in scrub, woodland and wet grassland schemes across varying areas and durations.

In the analysis presented in the main paper, we extracted the appropriate WTA estimates based on the timescale considered, which was varied between 10-100 years. We assumed participants enrolled the maximum area they were willing given the offered payment rate and scheme duration.