

LETTER

The cost of policy simplification in conservation incentive programs

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Abstract

Incentive payments to private landowners provide a common strategy to conserve biodiversity and enhance the supply of goods and services from ecosystems. To deliver cost-effective improvements in biodiversity, payment schemes must trade-off inefficiencies that result from over-simplified policies with the administrative burden of implementing more complex incentive designs. We examine the effectiveness of different payment schemes using field parameterized, ecological economic models of extensive grazing farms. We focus on profit maximising farm management plans and use bird species as a policy-relevant indicator of biodiversity. Common policy simplifications result in a 49–100% loss in biodiversity benefits depending on the conservation target chosen. Failure to differentiate prices for conservation improvements in space is particularly problematic. Additional implementation costs that accompany more complicated policies are worth bearing even when these constitute a substantial proportion (70% or more) of the payments that would otherwise have been given to farmers.

Keywords

Agriculture, agri-environment scheme, biodiversity, cost-effectiveness, ecological economics, grazing, incentive payment, Payments for Environmental Services, trade-off curve.

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INTRODUCTION

Habitat destruction and degradation associated with conversion to and intensification of agricultural land is a leading driver of losses of biodiversity and ecosystem services (Wilcove *et al.* 2000; Millennium Ecosystem Assessment 2005; Donald *et al.* 2006; Venter *et al.* 2006). To limit further losses, governments commonly provide incentives to encourage farmers to adopt production techniques that allow biodiversity to coexist alongside agriculture (Scherr & McNeely 2008). The EU and member states spend on average \$7.2bn per year on incentive payments to farmers that are designed to safeguard environmental benefits including biodiversity, with 22% of the utilised agricultural land area covered by these schemes (Cooper *et al.* 2009). The largest scheme in the US, the Conservation Reserve Program, spends \$1.7bn per year to purchase such benefits on agricultural lands with a combined area the size of North Carolina (United States Department of Agriculture. [USDA] 2010). The most common design of these 'agri-environment schemes' (AES) pays farmers an annual rental fee for the duration of a fixed term contract. In return the farmer agrees to undertake management actions that are thought to provide environmental benefits or to refrain from using environmentally detrimental production tech-

niques. This approach to conservation is one that is now being replicated in Payments for Environmental Services programs throughout the world (Jack *et al.* 2008; Quintero *et al.* 2009; Chen *et al.* 2010; Sommerville *et al.* 2010).

The considerable investment in AES comes despite patchy empirical evidence regarding their effectiveness (Ferraro & Pattanayak 2006). Focusing on biodiversity specifically, assessments of the ecological impact of agri-environment agreements have yielded mixed results (Kleijn & Sutherland 2003; Kleijn *et al.* 2006; Riffell *et al.* 2008; Batary *et al.* 2011). Assessments of whether AES programs improve the plight of biodiversity vary with taxonomic focus (Kleijn *et al.* 2006), including the breadth of that focus (Kleijn & van Zuiljien 2004; Perkins *et al.* 2011); the scale over which the assessment is conducted (Dallimer *et al.* 2010); and the landscape context within which the assessment takes place (Merckx *et al.* 2009; Batary *et al.* 2010). Economic evaluations of the cost-effectiveness of schemes have also taken place (Stoneham *et al.* 2003; Hodge & Reader 2010; Lewis *et al.* 2011), but have been limited by the fact that the cost to a landowner of undertaking particular management actions is private information, not directly measurable by researchers (Schilizzi & Latacz-Lohman 2007).

Concern about AES performance begs the question of what an ecologically and economically effective scheme would look like

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(Whittingham *et al.* 2007; Jack *et al.* 2008; Lewis *et al.* 2011). We built an integrated model of biodiversity change and farm production choices to examine cost-effective AES design. The model examines the ‘production’ of biodiversity on a farm at the same time as examining the production of more traditional agricultural outputs (e.g. numbers of livestock, milk yields). The model allows us to estimate a farmer’s marginal private costs of enhancing some biodiversity target (true supply price of biodiversity). These costs can be represented using trade-off curves relating a given improvement in some biodiversity target to associated foregone farm profits (Osgathorpe *et al.* 2011), providing property-scale counterparts to trade-off curves used to analyse conservation effectiveness at regional scales (Faith *et al.* 1996; Polasky *et al.* 2005). Using the trade-off curves we determine characteristics of optimal policy designs that would purchase the maximum possible improvement in some biodiversity target for a fixed budget. The optimal policies provide a benchmark that allows us to estimate the efficiency cost of policy simplifications common in AES programs.

To parameterize the model, we conducted socioeconomic surveys on extensive livestock farms in the UK and surveyed spatial variation in the bird community on the same properties as an indicator of biodiversity. At the time of the study, improving the abundance of farmland bird species was a target for UK conservation policy (Gregory *et al.* 2004). Synchronous property-scale surveys of biodiversity and economics of land management of this type remain uncommon. Our panel of farms is an interesting case study in which to examine agricultural subsidy designs, because the relatively simple, low-input production system characteristic of agriculture in our study area is one that is replicated all over the world. Additionally, any policy changes would be felt acutely in this area where farms are on the margins of agricultural profitability (Acs *et al.* 2010).

MATERIALS AND METHODS

An overview of the socioeconomic and ecological surveys and the models is provided here. Additional details are provided in the Supporting Information (see Appendix S1). Parameters for the model are based on field surveys of 44 extensive livestock farms conducted in and around the Peak District National Park in northern England (Fig. 1). The sampling scheme for recruiting farms to the survey was spatially stratified and opportunistic. Locations of study farms were stratified to ensure spatial coverage of the study area, but our sample necessarily reflects the willingness of farms to participate in both the ecological and economic parts of the survey. We divided the study area into three regions for analysis (the Dark Peak, Eastern Moors and Southwest Peak; Fig. 1) based on elevation, wetness and vegetation composition of the landscape and on the prevailing enterprise mix on farms.

Farms in the region focus primarily on production of sheep and dairy or beef cattle. Socioeconomic aspects of farm businesses were surveyed using a detailed, closed form questionnaire (see Appendix S2). Surveys included questions on land area of holding, land type and use (e.g. area of hay or area used for silage - with one, two or three cuts), production activities (e.g. livestock production, crop production, labour use, fertiliser use) and subsidy payments received. To parameterize the models, we averaged these quantities across farms within each of the three regions.

Birds were chosen as a policy-relevant indicator of biodiversity. The density and richness of birds on the 44 farms was surveyed twice in Spring 2007. All surveys were conducted by a single observer (MD),

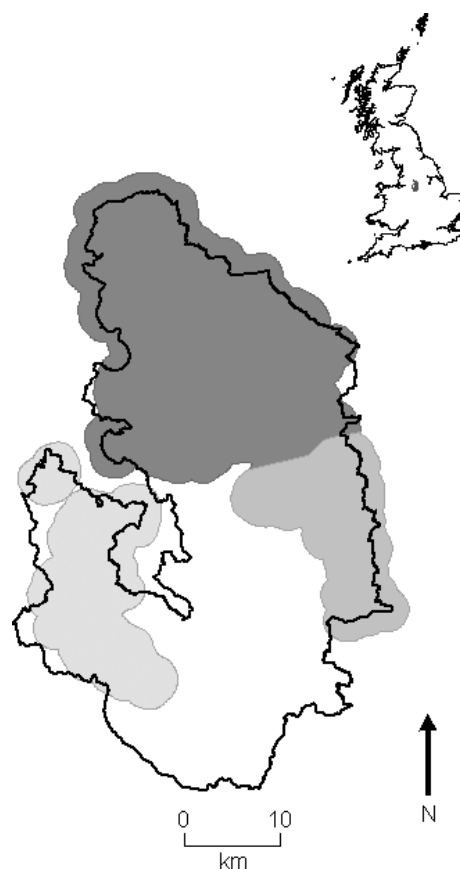


Figure 1 Map of Peak District and its location within Britain, highlighting the three different regions within the study area (Dark Peak – dark grey; Eastern Moors – mid grey; Southwest Peak – light grey).

using standard methodologies (Newson *et al.* 2005). We used distance sampled transect counts. Transects were placed through the inbye portion (common to all survey farms) of the full farm holding. Bird counts were converted to density estimates while controlling for differences in detectability across species using Distance 5.0 release 2. As possible targets for conservation action, we focused on five single species of conservation concern in the UK (Eurasian curlew *Numenius arquata*, northern lapwing *Vanellus vanellus*, linnet *Carduelis cannabina*, song thrush *Turdus philomelos* and skylark *Alauda arvensis*). These species were chosen in part because we anticipated they would show diverse responses to land management actions. In addition, we focused on the total density of all birds (total density) and the total species richness of birds (total species richness) observed on each farm during both field visits as possible whole community targets for conservation action. Summary details of bird survey results are given in Table S1.

For a representative farm in one of the three regions, the model takes the form of a profit maximisation condition

$$\max V = \mathbf{p} \cdot \mathbf{x} \quad (1)$$

subject to linear production constraints

$$A\mathbf{x} \leq \mathbf{b} \quad \text{and} \quad x_i \geq 0 \quad (2)$$

and nonlinear biodiversity constraints

$$c_j(\mathbf{x}) \geq d_j. \quad (3)$$

In the objective equation, eqn 1, V is net farm income (excluding fixed costs), \mathbf{x} is a vector of farming activities (x_i) (number of sheep, number of cattle, tons of fertiliser applied, number of grassland cuts for silage); and \mathbf{p} is a vector of gross margins associated with each activity. In the linear constraint equations, eqn 2, A is a matrix of technical coefficients describing the production system (e.g. to produce each ewe requires a certain amount of feed that can either be supplied through forage produced on farm or purchased); and \mathbf{b} is a vector of resource endowments (e.g. the amount of land on the farm that can be used for forage production). In the nonlinear constraint equations, eqn 3, c_j is a nonlinear function derived by regressing the density or richness of birds against farm management variables; d_j is a biodiversity constraint; and j indexes across the number of different biodiversity targets specified in the given conservation policy.

The inclusion of the biodiversity constraint requires a nonlinear programming approach. We used the CONOPT solver for nonlinear programming in GAMS (General Algebraic Modeling System) version 23.4 to solve this optimisation problem. The full model (eqns 1–3) predicts the farm management plan that provides maximum net farm income (excluding fixed costs) compatible both with the constraints on the available production system and the additional constraints imposed by requiring a given level of biodiversity be supplied on farm.

Parameter estimates in the model primarily come from our field surveys. The gross margins \mathbf{p} , technical coefficients A , and resource endowments \mathbf{b} were all estimated from the farm surveys separately for the three regions. However, where necessary and appropriate survey results were supplemented with parameter estimates obtained from the Farm Management Handbook (Beaton 2007) (e.g. for feeding requirements for ewes and lambs in this region). These additional variables are applied in a spatially uniform manner across farms.

Nonlinear function $c_j(\mathbf{x})$ relates the response of a given biodiversity indicator to the farm management variables and was estimated by regressing the observed patterns in each focal biodiversity indicator against the four key elements of farm management plans (x_1 number of sheep, x_2 number of cattle, x_3 tons of fertiliser, x_4 number of cuts; see Table S2 in Supporting Information). The explanatory power (r^2 or pseudo r^2) of the regressions is relatively low (0.08–0.31, see Table S2). Therefore, we examine the sensitivity of our results to uncertainty in the regression equations governing bird responses to farm management actions (Supporting Information). Including additional covariates describing habitat quality on farms (e.g. proportion of fields covered by wet features) increases these r^2 and pseudo r^2 values to 0.26–0.52. However, these variables do not currently feature in farm management plans for most farmers in our study area. In addition, the economics of having farmers create such habitat features is poorly understood. As such, we chose to focus our main analyses on management variables commonly included in farm management plans and AES designs (e.g. livestock numbers and fertiliser application rates). However, we include further sensitivity tests examining consequences of including additional covariates in the regressions describing bird responses to farm management actions in the Supporting Information.

With our formulation, we model a payment for actions not outcomes (the government agency bears the risk for producing biodiversity rather than the farmer; Zabel & Roe 2009; Gibbons *et al.* 2011), although the combination of actions is estimated on average to produce an increase in a given biodiversity target in this study area. This focus reflects the dominance of payments for management

actions schemes in contrast to payments for environmental outcomes schemes within EU agri-environment policy.

Various modeling efforts have sought to combine a representation of utility or profit-maximising behaviour of land managers with one depicting likely impacts on biodiversity (e.g. Oglethorpe & Sanderson 1999; Roeder *et al.* 2010; Bamiere *et al.* 2011; Lewis *et al.* 2011; Mouysset *et al.* 2011; Osgathorpe *et al.* 2011). The novelty in our approach arises from the combination of: the data resolution used to parameterize our models, the inclusion of nonlinear biodiversity responses to land management actions, the models embedding biodiversity production into farm management plans, the richness of policy scenarios we examine, and the number of conservation targets against which we are able to test these policy scenarios. While some previous studies have included one or two of these elements, ours is the first to integrate all of these factors in a property-scale approach.

RESULTS AND DISCUSSION

Biodiversity-profit trade-off curves

The inefficiency of AES schemes available to our sample farms (Entry Level Stewardship, Higher Level Stewardship, and the Hill Farm Allowance) is suggested by an analysis of the economics of the farms, even before considering resulting biodiversity improvements. Existing payments are intended to compensate farmers for income foregone in undertaking management prescriptions. Were current policies cost-effective, the reduction in the maximum farm income caused by enrolling in these schemes should equal the incentive payment. Using a simpler model that does not account for biodiversity (eqns 1 and 2 only; Acs *et al.* 2010), this comparison suggests only \$0.12 to \$0.46 per dollar of public funds invested compensates for farmers income foregone. The remainder is pure subsidy. However, were the payments only to compensate farmers for income foregone, then a number of farms in our survey would be running at negative farm income and may go out of business. Many of the farms are on the margins of profitability with the surplus provided by existing AES helping to make up the shortfall (Acs *et al.* 2010). The ecological impacts of any land abandonment that might follow are not well-understood (Evans *et al.* 2006; Hanley *et al.* 2008; Amar *et al.* 2011).

The coupled ecological economic models enable us to go further and to predict the maximum farm income compatible (and thereby the payment level required to compensate for income foregone) with supplying a given level of some conservation target. By incrementally increasing the amount of a chosen biodiversity target that farmers are required to produce, we recover trade-off curves relating maximum farm profit to biodiversity change. Examples of these trade-off curves are illustrated in Fig. 2. The figure shows trade-off curves for three farm types representing the regional variation in our study system. The trade-off curves highlight where improvements in biodiversity can be purchased at low cost. For example, low-cost gains are available when aiming to increase densities of *N. arquata* by investing in the Eastern Moors or Southwest Peak. The locations offering low-cost gains in biodiversity vary with the conservation target chosen (compare Figs 2A and 2B or 2D and 2E). Moreover the degree of nonlinearity present in the trade-off curves (the cost of increasing *N. arquata* density by 1% over baseline levels is less than that involved in increasing *N. arquata* density by 10% instead of 9%) varies with region and choice of conservation target.

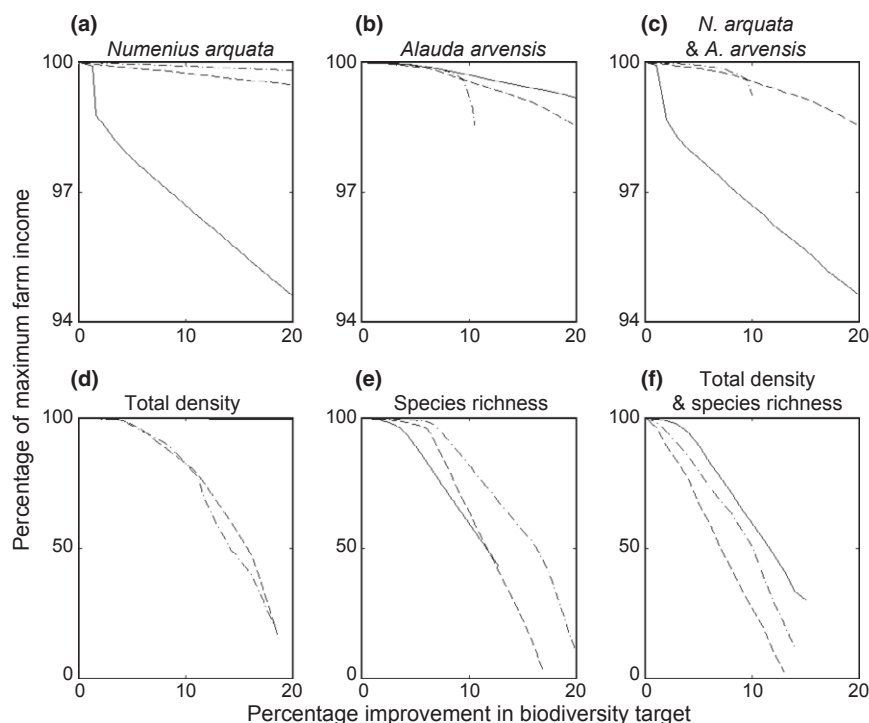


Figure 2 Trade-off curves relating the percentage reduction in maximum farm income that results from requiring a given percentage increase in a focal biodiversity target. Examples of trade-off curves shown for (a) *Numenius arquata*, Eurasian curlew; (b) *Alauda arvensis*, skylark; (c) when requiring simultaneous enhancements in densities of both *N. arquata* and *A. arvensis*; (d) total density of birds; (e) total richness of birds; and (f) when requiring simultaneous enhancements in both the total density and total richness of birds. Three curves illustrate representative farm types in three study regions (solid – Dark Peak, dashed – Eastern Moors, dot-dashed Southwest Peak). For some targets and some regions there is a maximum possible improvement in a given biodiversity target compatible with existing farm plans, indicated by curves that end before a 20% improvement in the focal biodiversity target is reached.

Elasticities are calculated as a proportional change in maximum net farm income divided by the corresponding proportional change in some conservation target. A comparison of elasticities across levels of desired biodiversity improvement, regions and conservation targets illustrates the variation present within and between the trade-off curves. Table 1 shows elasticities evaluated at a 5 and 10% improvement in the five single species targets and two community level targets in each of the three regions. All elasticities are negative reflecting the trade-off between maximum farm income and biodiversity improvement. Larger magnitude elasticities indicate regions, biodiversity targets and biodiversity levels for which improvements in biodiversity are more costly. Fig. 2 only illustrated trade-off curves for a subset of the conservation targets that we consider. The elasticities for the fuller set of conservation targets shown in Table 2 indicate that the variation in locations offering low-cost gains and in the nonlinearity represented in trade-off curves illustrated in Fig. 2 is also found when seeking improvements in other conservation targets.

In addition, the trade-off curves and relative magnitude of elasticities make clear that opportunities for low-cost gains are more limited when aiming for community-based conservation targets than single species targets (Fig. 2, Table 1). This provides an interesting complement to observations that AES narrowly focused on increasing the abundance of individual bird species have often been more successful than those focused on improving the plight of birds more broadly (Peach *et al.* 2001; Kleijn & Sutherland 2003; Kleijn & van Zuiljien 2004; Perkins *et al.* 2011).

We also calculate biodiversity-profit trade-off curves and accompanying elasticities when simultaneously requiring an improvement in

multiple conservation targets. Fig. 2 and Table 1 show the results for two species (*N. arquata* and *A. arvensis*) that have positively correlated responses to land management actions and for the two community targets for which responses are less tightly correlated (see Table S3). Requiring simultaneous improvement in two conservation targets is no more costly than requiring improvements in the most expensive single target when the responses of conservation targets to land management actions are closely correlated (Fig. 2C, second to last row in Table 1). However, requiring improvements in both conservation targets simultaneously becomes more expensive where responses to land management actions are less well-correlated, because opportunities for low-cost gains become more limited (Fig. 2F, last row in Table 1).

However, if policies are intended to deliver multiple conservation outcomes, it is still much more effective to include these in the scheme design explicitly rather than rely on an ‘umbrella’ approach, which specifies one conservation target with the hope that it will deliver ancillary benefits for others. Typically, a reliance on an umbrella approach would fail to deliver required benefits for the second target, because individual species and whole community indices differ in their responses to land management. Indeed, in our study, actions that would improve the status of one conservation target were as likely to be detrimental to an ancillary biodiversity indicator as they were to enhance it (see Table S3). The variation in ecological responses should be expected. For example, the management actions one would advocate in a bid to enhance *N. arquata* (an extensive upland specialist) densities on a property, such as reducing stocking rates and fertiliser application rates, likely would be similar to those one would recommend to improve densities of

Table 1 Elasticities of farm income with respect to biodiversity for five single species targets (*Numenius arquata*, Eurasian curlew CU, *Vanellus vanellus*, northern lapwing, *Carduelis cannabina*, linnet, *Turdus philomelos*, song thrush, and *Alauda arvensis*, skylark S), 2 community-level targets [total density (TD) and total richness (TR) of birds], and 2 sets of simultaneous targets (increase CU and S simultaneously and TD and TR simultaneously). Elasticities summarise curvature in the type of trade-off curves illustrated in Fig. 2 and are calculated as a proportional change in maximum net farm income divided by the corresponding proportional change in the conservation target. Elasticities are calculated at a 5 and 10% improvement in each conservation target in each of the three regions. All elasticities are negative reflecting the trade-off between maximum farm income and biodiversity improvement. Larger magnitude elasticities indicate regions, conservation targets and biodiversity levels for which improvements in biodiversity are more costly

	Dark Peak		Eastern Moors		Southwest Peak	
	5%	10%	5%	10%	5%	10%
<i>N. arquata</i> (CU)	-0.25*	-0.24*	-0.025	-0.031	-0.0094	-0.0098
<i>V. vanellus</i>	-0.020	-0.020	-0.0068	-0.011	-0.19†	-0.32†
<i>C. cannabina</i>	-0.00029	-0.00030	-0.0057	-0.0084	-0.0022	-0.0026
<i>T. philomelos</i>	-0.032	-0.035	-0.12	-0.16	-0.0085	-0.0088
<i>A. arvensis</i> (S)	-0.036	-0.056	-0.038	-0.088	-0.025	-0.80
TD	-0.0035	-0.027	-2.30	-4.28	-2.30	-5.98
TR	-6.77	-10.84	-1.07	-14.31	-0.34	-6.74
CU & S‡	-0.24	-0.24	-0.039	-0.088	-0.025	-1.36
TD & TR‡	-6.83	-10.85	-12.96	-37.67	-7.88	-19.35

*The trade-off curve for *N. arquata* in the Dark Peak is approximately a straight line across a range that includes a 5 and 10% increase in density. The elasticities here are effectively equal if carrying more significant figures.

†Elasticities for *V. vanellus* in SW peak calculated at an absolute density corresponding to 0.1–0.2 birds / farm on inbye land, because exceptionally low densities are predicted for the unconstrained optimal farm management plan.

‡When requiring an improvement in two conservation targets simultaneously, elasticities are always computed on the binding constraint if only one biodiversity constraint is binding.

A. arvensis. However, they would be quite different from the management actions that you would recommend to improve densities of *T. philomelos* (primarily a lowland species but one also of conservation concern), such as increasing cutting rates and fertiliser application rates.

In those instances where an umbrella target approach can achieve multiple conservation goals, it can prove much more expensive. To illustrate, we consider the case where policy-makers require a 5% improvement in both the total density of birds and total richness of bird species in one region. If they developed a policy that required

only an increase in richness in the Dark Peak, it would deliver both outcomes at no additional cost. A policy focused on only increasing the total density of birds in the Southwest Peak could also deliver the required improvement in species richness but for 3.8 times the cost of a policy that included both conservation targets to begin with. Meanwhile setting policies based on the four remaining combinations of locations and single targets (total density in the Southwest Peak or Eastern Moors or total richness in the Dark Peak or Eastern Moors) would fail to deliver on the combined conservation objectives.

Optimal policy design

Next we analysed the policy that would purchase the maximum improvement in biodiversity for a given budget. This optimal policy design involves preferentially allocating conservation contracts to regions where biodiversity enhancements can be provided relatively cheaply. The optimal allocations that result contrast strongly for different conservation targets (e.g. optimal allocations for *V. vanellus*, *C. cannabina* and *T. philomelos* in Table 2) reflecting the contrasting patterns of spatial variation in the trade-off curves.

The optimal policy also involves setting payment rates that vary across regions and with the amount of biodiversity produced on a farm to recognise the nonlinear nature of the trade-off curves involved. Specifically, within each region, the optimal policy pays farmers just enough that they would be indifferent between continuing with current practices or joining the scheme (i.e. the payment rate exactly equals their income foregone by implementing management practices that improved biodiversity) after accounting for endogenous changes in management. Because our models are parameterized for farms that are already enrolled in government subsidy schemes, we focused on the design of an additional, new scheme that would provide either \$184k (= GBP £100k) to the study area to improve the plight of a single species or \$921k (= GBP £500k) to improve the state of a whole community indicator. These totals are equivalent to an additional \$921 (= GBP £500) or \$4606 (= GBP £2500) per farm per year on average.

Cost of policy simplifications

We compared the maximum biodiversity gain possible with the optimal policy to the biodiversity gain expected when employing simpler, but more readily implemented, schemes (Fig. 3). All of the policy simplifications we study are common in AES design (Cooper

Table 2 Proportion of the overall budget allocated to farms in each region corresponding to the optimal policy (Opt.) that maximises the improvement in a focal conservation target and to the best possible allocation subject to a restriction that payment levels cannot vary with the amount of biodiversity produced on farms within a region (fixed cost, F. cost). Results shown for 5 single species targets (*Numenius arquata*, Eurasian curlew, *Vanellus vanellus*, northern lapwing, *Carduelis cannabina*, linnet, *Turdus philomelos*, song thrush, and *Alauda arvensis*, skylark) and 2 community-level targets [total density (TD) and total richness (TR) of birds]. Also shown (final column) is the allocation that results when there is no spatial targeting of conservation contracts. Contracts are then allocated in proportion to the area of farms in each region for conservation targets related to species densities (first value) and in proportion to the number of farms in each region when species richness provides the conservation target (second value)

	Focal biodiversity target														
	<i>N. arquata</i>		<i>V. vanellus</i>		<i>C. cannabina</i>		<i>T. philomelos</i>		<i>A. arvensis</i>		TD		TR		Aspatial
	Opt.	F. cost	Opt.	F. cost	Opt.	F. cost	Opt.	F. cost	Opt.	F. cost	Opt.	F. cost	Opt.	F. cost	
DP	0.039	0.042	1	1	0.56	0.55	0	0	0.017	0.14	0.80	0.85	0.44	0.32	0.26/0.34
EM	0.74	0.50	0	0	0	0.01	0.66	0.64	0.88	0.70	0.13	0.048	0.33	0.30	0.40/0.22
SW	0.23	0.45	0	0	0.44	0.45	0.34	0.36	0.099	0.16	0.065	0.099	0.23	0.38	0.34/0.44

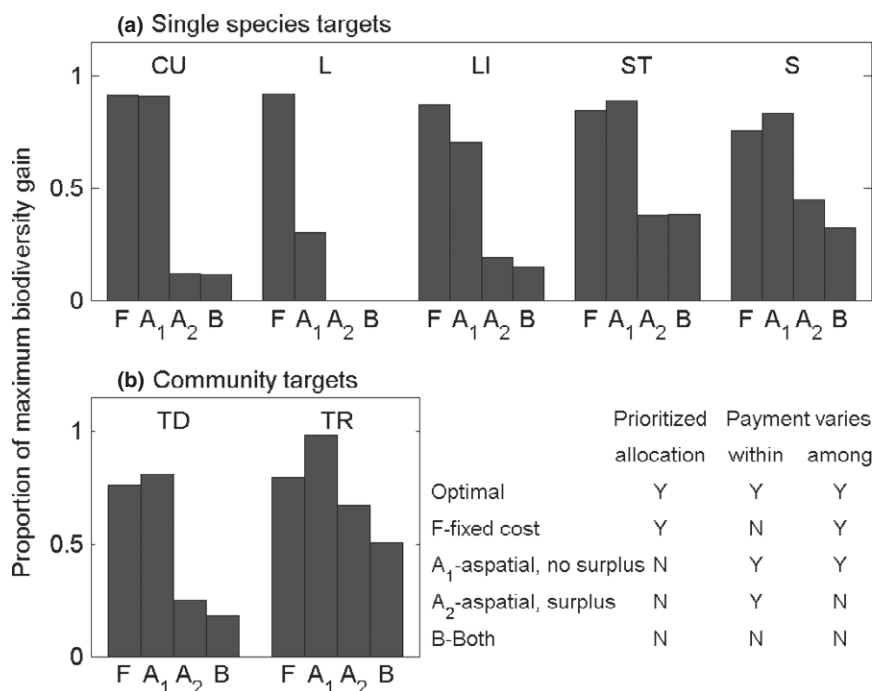


Figure 3 Ecological cost of common policy simplifications shown as the proportion of the maximum biodiversity gain available for a given budget with the optimal policy that is achieved with each simplified policy. Results shown (panel a) for five single species biodiversity targets (CU – *Numenius arquata*, Eurasian curlew, L – *Vanellus vanellus*, northern lapwing, LI – *Carduelis cannabina*, linnet, ST – *Turdus philomelos*, song thrush, and S – *Alauda arvensis*, skylark) and (panel b) for two whole community biodiversity targets (TD – total density and TR – total richness of birds). The optimal policy prioritizes the allocation of conservation contracts across regions and involves payment rates that vary among regions and with the amount of biodiversity produced within regions. Policy simplifications: F (fixed cost) – payment rate does not vary with the amount of biodiversity produced within each region; A₁ (aspatial, no surplus) – contracts are not preferentially targeted towards some regions but are allocated in proportion to farm area for density measures or farm number for species richness; A₂ (aspatial, surplus) – contracts are allocated in proportion to farm area or farm number and payment rates do not vary among regions; B (both simplifications) both the fixed cost and aspatial policy simplifications apply.

et al. 2009; Hodge & Reader 2010), although some programs (e.g. reverse auctions or schemes that pay for outcomes rather than actions) have sought to resolve variation in the private costs of supplying conservation enhancements more fully (Verhulst *et al.* 2007; Claassen *et al.* 2008; Quillerou & Fraser 2010). First, we evaluated schemes that failed to reflect the nonlinear costs faced by farmers in supplying biodiversity benefits and instead assumes that the price paid to a farmer within each region per unit biodiversity produced equals the cost to her/him of producing the final and most expensive unit of biodiversity. This fixed cost scheme generates some surplus to farmers when compared to a scheme that provides a sliding payment rate with the number of units of biodiversity supplied (as embedded in the optimal policy), but is able to respond to spatial variation by targeting contract allocations (Table 2).

We also evaluated schemes that do not reflect spatial variability in the costs of enhancing biodiversity by not prioritizing some regions for investment over others (final column in Table 2). Instead we assumed contracts are allocated uniformly across the three regions based on their areal extent. For conservation targets tied to species density, contracts are allocated across the three regions in proportion to the overall area of farms. For those tied to species richness, they are allocated in proportion to the number of farms. We compared two types of aspatial policy (Fig. 3). One allows spatially differentiated payment rates in each region. The other pays farmers at the rate of the most expensive region in which biodiversity is being produced. In both cases, to examine the effects of space *per se*, we assume a sliding

payment rate with the number of units of biodiversity supplied as is present in the optimal policy. In the case of spatially differentiated pricing, no surplus accrues to any of the farmers and any inefficiencies are solely a consequence of the failure to allocate contracts preferentially to regions that can produce biodiversity improvements cheaply. In the case with spatially uniform pricing, surplus accrues to farmers in the two regions where biodiversity can be produced most cost-effectively, adding an additional type of inefficiency. Finally, we compared the optimal policy to one that included all of these simplifications (no sliding payment rate within regions, no price variation between regions and no preferential allocation of conservation contracts to some regions over others).

Including all policy simplifications results in a 49–100% reduction in the amount of biodiversity provided for a given level of investment relative to the maximum improvement that is possible through the use of the optimal policy (Fig. 3 difference between 1 and bar (B) representing both aspatial and fixed cost simplifications). Comparing the relative contribution of each policy simplification indicates that a failure to exploit spatial variation in the cost of producing biodiversity enhancements is particularly problematic (in that smaller biodiversity gains are possible for a fixed budget; see also Wunscher *et al.* 2008; Chen *et al.* 2010). By comparing the two variants of aspatial policies, we find that this loss of efficiency stems from not discriminating pricing spatially rather than a failure to prioritize regions for conservation investment (Fig. 3). Purchasing conservation improvements at a fixed cost rather than employing a sliding payment rate only incurs a comparable loss of

efficiency when targeting enhancements in species richness (comparing Fixed Cost bar and dark Aspatial bar in Fig. 3), reflecting the relative variation across the trade-off curves for the three regions vs. within the curves for each region. Failing to prioritize regions for investment only incurs comparable efficiency costs when seeking enhancements in *V. vanellus* density, reflecting the concentrated spatial allocation strategy that is optimal for this species (Table 2).

We tested the sensitivity of the rankings of policy simplifications shown in Fig. 3 to uncertainty in the regressions relating responses of different conservation targets to farm management actions. We assumed that policy-makers based payment scheme designs on our estimated regressions (Table S2) and farmers responded to these policy specifications. However, we assumed that changes in the focal conservation target were determined by a different regression equation. In one set of sensitivity tests, we restricted attention to the same set of predictor variables describing responses to farm management actions and focused on uncertainty across estimated regression coefficients. In a second set of sensitivity tests, we examined sensitivity to missing covariates describing habitat conditions on farm and landscape features surrounding survey farms. The prediction that a failure to discriminate pricing for conservation improvements spatially was particularly problematic appeared robust to uncertainty in the regression equations and the inclusion of additional covariates (Tables S4, S5). In the subset of runs where sensitivities did arise regarding this prediction, they were caused by the

optimal policy itself being ineffective when judged against the second regression equation.

Implementation costs obviously would be higher for more elaborate policy designs. By reversing the question to examine what payment levels would be needed under the optimal policy to achieve the same improvement in biodiversity delivered by each of the simplified policies, we can calculate an upper bound on the additional implementation cost that it would be worth bearing to avoid each policy simplification (Fig. 4). It is often worth pursuing a more complicated policy even if the added implementation costs of doing so would constitute a substantial proportion of the overall program budget. For example, irrespective of the focal biodiversity target, it would be worth spending 70% or more of the funds that would otherwise be given to farmers to implement policies that recognise regional variation in costs of enhancing biodiversity and that allocate incentive contracts accordingly [subtracting A_2 (aspatial, surplus) bar in Fig. 4 from 1 gives a range of values from 0.7 for *T. philomelos* to 1 for *V. vanellus*].

Assumptions and extensions

We compared different policy simplifications on cost-effectiveness grounds. The inefficiencies of the different policy simplifications stems from their failure to resolve heterogeneity within and across farms in the private costs of producing biodiversity benefits. Our

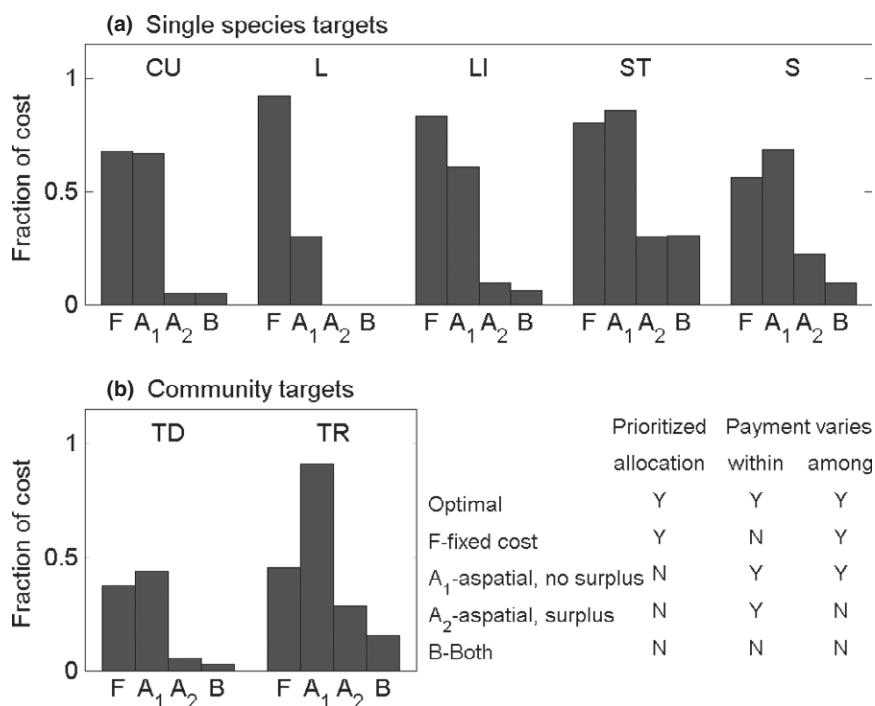


Figure 4 Payment levels (expressed as a fraction) that would be needed under the optimal policy to achieve the same improvement in biodiversity as delivered by each of the simplified policies for a given overall budget level. The difference between each bar and one provides an upper bound on the additional implementation cost (expressed as a proportion of the program budget paid to farmers) that it would be worth bearing in order to implement the more complicated policy. Result shown (panel a) for five single species biodiversity targets (CU – *Numenius arquata*, Eurasian curlew, L – *Vanellus vanellus*, northern lapwing, LI – *Carduelis cannabina*, linnet, ST – *Turdus philomelos*, song thrush, and S – *Alauda arvensis*, skylark) and (panel b) for two whole community biodiversity targets (TD – total density and TR – total richness of birds). The optimal policy prioritizes the allocation of conservation contracts across regions and involves payment rates that vary among regions and with the amount of biodiversity produced within regions. Policy simplifications: F (fixed cost) – payment rate does not vary with the amount of biodiversity produced within each region; A_1 (aspatial, no surplus) – contracts are not preferentially targeted towards some regions but are allocated in proportion to farm area for density measures or farm number for species richness; A_2 (aspatial, surplus) – contracts are allocated in proportion to farm area or farm number and payment rates do not vary among regions; B (both simplifications) both the fixed cost and aspatial policy simplifications apply.

study design resolves these heterogeneities much more finely than existing policies. However, because we aggregated heterogeneity across farms found within each of the three regions, we may, if anything, be underestimating how inefficient different policy simplifications can be (Appendix S1).

We compared the effectiveness of different simplified policies to the policy that would compensate farmers for their income foregone in delivering required enhancements in biodiversity leaving them indifferent between entering the scheme or not. To encourage farmers to enroll, some premium over this amount may need to be paid, especially if there are transaction costs associated with enrolment. Indeed, some AES payment programs include bonus payments at sign-up (e.g. signing incentive payments in USDA Conservation Reserve Program schemes).

While we focus on the cost effectiveness of different policy simplifications, it is important to recognise that these different payment schemes will have different distributional impacts. Under most simplified policies and also under *status quo* management today, farmers receive some surplus, but who receives the surplus and how much they receive varies across policies. For payment schemes that are narrowly targeted in space, distributional impacts will be particularly obvious. Evaluation of different policy designs should also consider these distributional impacts. Indeed, distributional impacts could in turn influence the cost-effectiveness of different policies, if they result in differences in transaction costs being associated with implementing different policies.

CONCLUSION

Because incentive payment programs account for a large percentage of overall investments in biodiversity conservation (Lerner *et al.* 2007), it is imperative that these are designed to deliver biodiversity benefits in a cost-effective manner. Our results highlight that the lower administrative burdens that accompany commonly employed, simple program designs offer false economies. Instead, the additional implementation costs that accompany policies that account more fully for variation in the costs faced by landowners in producing biodiversity benefits would be worth bearing even when these constitute a substantial proportion (70% or more) of the payments that would otherwise have been given to farmers. Spatially differentiating the prices paid to farmers for biodiversity enhancements appears to be particularly important. These conclusions generalise across conservation targets, ranging from enhancing densities of individual species of conservation concern to enhancing whole community indicators such as species richness, despite idiosyncratic differences in how individual conservation targets respond to land management actions.

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AUTHORSHIP

All authors contributed to study design and wrote the article. P.R.A. and S.A. analysed the integrated models. M.D. conducted the ecological surveys and analysed the ecological data. P.W. coordinated the farm surveys.

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