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Economic incentives for biodiversity conservation on farmland[1]

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Abstract

In this paper, I review recent advances in the field of economic incentives for biodiversity conservation, focusing on incentives offered to private landowners to change how they manage land. Since profit-maximising land use decisions are rarely consistent with optimal provision of biodiversity, due to market failure, additional financial incentives have been argued to be needed to slow global biodiversity decline and aid biodiversity recovery. The paper organizes recent literature along four thematic lines: paying for results rather than actions, incentives for spatial coordination, collective participation schemes, and biodiversity offset markets.

Jel codes: Q1, Q2, Q5

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1. Introduction

Changing how such land is managed has recently been argued to be crucial in achieving better outcomes for biodiversity (IPBES, 2019). Where land is privately owned, owners often need to be offered economic incentives to implement such changes. These incentive mechanisms are variously categorised as Payment for Ecosystem Service (PES) schemes; and where farmers are the intended participants, as Agri-Environment Schemes (AES). This paper reviews recent work on the design of these incentives from the perspective of economics, informed by insights from ecology on what typically matters most for success in achieving improved biodiversity outcomes. In what follows, I mainly use the term "AES schemes" to describe any economic incentive intended to produce a change in land use or land management which is intended to increase some desirable metric of biodiversity conservation. A key feature of such schemes is that farmers voluntarily chose whether or not to participate. Participation implies accepting some restriction on how land is managed or what is produced, in return for a financial payment.

We typically think of farmers (I will refer to all land managers as "farmers" in this paper) as incurring opportunity costs in changing management to a more biodiversity-productive type. That is, a multi-product output function exists (Dakpo et al, 2016; Ait-Sidhoum et al, 2023) whereby inputs such as fertilizer, land and the farmer's time are used to produce crops and livestock for sale, but which also determine biodiversity outcomes such as species distribution or abundance, overall species richness, and other measures of environmental impact. Typically, we think of increases in commercial output as occurring jointly with decreases in some biodiversity index: there are thus trade-offs between crop/livestock profits and biodiversity production. Absent the correct price signals, farmers choose a crop/livestock output level which is not consistent with socially-optimal levels of biodiversity, due to missing markets for wildlife and their habitats (Hanley, Shogren and White, 2019). Producing an increase in biodiversity typically comes at an opportunity cost of lower profits to the farmer (Armsworth et al, 2012).

There are three features of these opportunity costs which are relevant to the policy design problem. First, for an individual farmer, we expect these costs to be increasing at the margin. As more biodiversity is produced on an individual farm, it gets increasingly expensive to further increase such diversity (evidence for this increasing marginal cost at the individual farm level with respect to a given biodiversity index is scarce, but on average we can expect the farmer to always choose the cheapest option first, even if marginal costs are not continuously rising). Second, these biodiversity supply curves vary across farms, both spatially and by farm type. A given conservation action will have costs which vary across farms due to heterogeneity in land quality, location, and crop/livestock choice (for an example from the US Conservation Reserve, see Hellerstein, 2017). This means that (i) a uniform subsidy for biodiversity conservation will encourage varying levels of participation across farms: some will decide to join the PES scheme, some will not (ii) a cost-effective allocation of conservation actions will involve different degrees of such actions across farms – we prefer the farms with the lowest opportunity costs of "producing" an environmental improvement to undertake the most conservation (Ando et al, 1998; Polasky et al, 2008) and (iii) that a differentiated payment scheme is likely to be more cost-effective than a uniform payment scheme, since each farmer is offered the same price to undertake a given action under a uniform payment, whereas the marginal social cost of buying the environmental improvement varies over farms (Armsworth et al, 2012). Finally, opportunity costs are typically private information for the farmer, such that a regulator can only imperfectly observe their variability. This means that an adverse selection problem can arise, since the regulator offering subsidies for biodiversity conservation cannot easily distinguish between high- and low-cost types (Ferraro, 2008). It may also result in low additionality, as farmers who would have adopted the pro-biodiversity production methods in any case may be paid, rather than those who would not have done so (Wunder et al, 2020). One of the principle attractions of conservation auctions is that they can deal, to some extent, with this hidden or asymmetric information problem (Schilizzi and Latacz-Lohman, 2016), although auctions may reduce participation rates by eligible farmers to rather low levels. For instance, Howard et al (2024) found a proposed auction to improve water quality in Iowa could reduce participation by up to 70% compared to a baseline cost-sharing contract.

A second class of design problem is that of varying ecological potential: that is, the ability of a change in land management at a given location to deliver increases in some biodiversity indicator. Globally, most agri-environment schemes (AES) and payment for ecosystem service schemes (PES) involve paying for actions. Such an action changes the inputs in the multi-product output function such that the biodiversity index is increased. However, evidence from ecology shows that the response of a specific indicator to a change in pressures can be highly site-specific, varying for example with land use in the surrounding landscape, or with the history of site use (Bradfer-Lawrence et al, 2024; Hanley et al, 2008). This implies that the spatial targeting of economic incentives can improve the cost-effectiveness of biodiversity policy, since some locations will show a larger biodiversity response to marginal changes in land use (Fooks et al, 2016). Some existing AES schemes reflect this varying ecological potential through the use of site-based environmental metrics use to weight payments to land managers (eg Iftekar et al 2014, Wallender et al 2018). Moreover, ecological benefits from a specific change in actions at a specific location can vary over time in a partially-stochastic manner, due to the influences of other determinants of biodiversity outcomes, such as weather patterns, invasive species, disease outbreaks and climate change.

Finally, the number of land managers whose behaviour a regulator is trying to influence through offering economic incentives is typically large. For example, in 2023 there were around 103,000 individual farm holdings recorded in England who could participate in an AES scheme (DEFRA, 2024). Observing the behaviour of each of these farmers would be very costly. If farmers accept payments to change their farm management to deliver more biodiversity, where these changes are costly to each farmer (as lost profits), then moral hazard problems occur, since the farmer can accept the payment without undertaking the required conservation effort when the regulator finds it very costly to observe this effort (White and Hanley, 2016).

2. Payment for results, not actions

As noted above, most PES and agri-environment schemes focussed on improving biodiversity outcomes are payments from some regulator or other party (such as an NGO) for changes in the management *actions* undertaken by a farmer. This typifies, for instance, the great majority of agri-environment-climate schemes financed by the Common Agricultural Policy (Hasler et al, 2022). However, offering contracts which pay on the basis of delivered *outcomes* or *results* has a number of advantages (Herzon et al, 2018; Sumrada et al, 2021). Society pays for what it wants – biodiversity improvement- rather than the actions needed to deliver this. Moreover, farmers may have private knowledge on how best to deliver a specific biodiversity outcome on their land, which they can make use of under a payment for results contract (Wuepper and Huber, 2022). Payment for outcome schemes have also been argued to lead to the re-alignment of farmer attitudes and objectives, and potentially to reduce moral hazard problems if outcomes are easier (cheaper) to monitor than actions. Clearly, there will be a great deal of variation in whether outcomes or actions are less costly to observe for the regulator. Finally, paying for results allows the spatial variability of ecological potential to be taken into account in targeting actions: farmers whose land has greater potential to achieve the desired biodiversity outcome will be more likely to enrol than farms with lower ecological potential, ceteris paribus.

The drawbacks of payment for actions schemes focus around participation rates: if biodiversity outcomes are partly stochastic from the viewpoint of the farmer, then risk-averse farmers will be less willing to participate in a payment for results scheme than in a payments for action scheme. This means farmers would demand higher payments to participate in a payment for outcomes scheme compared with a payment for actions alternative (Villanueva et al, 2024: although see Granado-Diaz et al (2024) for the opposite result). Moreover, if payment for results are based on an absolute target values (eg 6 flowering plant species out of a list of 30 being counted on your land), then schemes may have low additionality, since those farmers who already have higher species richness will be more likely to participate. For evidence on this point, see Canessa et al (2023). Higher monitoring costs of measuring outcomes rather than actions mean higher transactions costs, and there is a discussion in the literature on who should do the monitoring – farmers, the regulator or some third party (Herzon et al, 2018). In their study of Japanese rice farmers, Tanaka et al (2022) found that farmers preferred that the monitoring of bird species on their farms – the variable used to determine payments – was undertaken by external experts rather than by themselves. Finally, whilst farmers may indeed have private information on how best to deliver biodiversity outcomes on their land, they may equally require training with respect to the relevant actions, and indeed with respect to monitoring outcomes (Sumrada et al, 2021).

Contracts can also be specified in terms of a combination of actions (eg when to mow grassland) and results (number of birds nesting in farm fields). In theory, such mixed contracts can have advantages over payment for results alone, or payment for actions alone (Derissen and Quaas, 2013; White and Hanley, 2016). Canessa et al (2023) use choice modelling to understand Bavarian farmers' willingness to participate in biodiversity-enhancing contracts which mix actions (change in mowing dates, livestock density) with results (plant species presence/absence). They found that "extensive" farms were more likely to enrol in results based schemes, whereas "intensive" farms were more likely to enrol in mixed contracts. See also the paper by Thiermann et al (2023), discussed below in section 4.

Finally, Bartkowski et al (2021) have suggested an interesting modification to payment for results schemes designed to increase participation by reducing risks to the farmer. This is to offer contracts specified in terms of the modelled or predicted impacts on some environmental variable from a change in management: that is, to pay for *modelled results*, rather than actual results. This exploits the variation in ecological potential of actions across space, like paying for actual outcomes. It clearly also reduces payment uncertainty, since the farmer is contracting for a specific change in management – under her control – which is predicted to result in, for example, a specific change in bird populations on their farm. Simpson et al (2023) present a comparison of this payment for modelled results approach with a payment for actions scheme, where the desired outcome is an increase in wader populations on farmland in the UK. The action here is the restoration of grassland. For a fixed budget, they find that the payment for modelled results scheme outperforms the payment for actions alternative in terms of both wader populations and producers' surplus.

3. Incentives for spatial coordination

A large literature has grown up around the incentivisation of spatial coordination in PES schemes, dating back to Parkhurst and Shogren (2007). This literature is reviewed in Nguyen et al. (2022). But why incentivise spatial coordination?

The original notion of Parkhurst and Shogren (2007) was that *"..Protecting biodiversity and ecosystem services on private lands can be facilitated by creating contiguous habitat":* that, for a given quantity of land allocated to conservation, ecological benefits are higher when this land is arranged in spatially-coordinated, contiguous areas, rather than being fragmented. The ecological principles behind such spatial coordination benefits were outlined in Margules and Pressey (2000), and indeed such landscape-level considerations of the importance of how land use varies between neighbouring land parcels is now common in ecology (Terraube et al, 2016). Interestingly, the ecological evidence base shows that spatial coordination benefits vary greatly by species (Bradfer-Lawrence et al, 2024), from essentially zero to large. Even for one taxon (e.g. forest birds), there exist species where the spatial coordination benefits are very high and those for which coordination benefits are very low (Hofmeister et al., 2017). Spatial coordination benefits for birds will not be the same as those for plants or for insects. Whilst much of the economics literature is motivated by non-biodiversity-specific examples, including wetland restoration and nutrient pollution, the idea that creating contiguous patches or conserved corridors in a landscape generates additional benefits is widespread in economics.

Faced with this policy design problem, economists have devised three types of incentives to encourage spatial coordination of conserved parcels. These are (i) the agglomeration bonus; (ii) threshold bonuses , and (iii) spatially-coordinated auctions. Smith and Shogren (2002) and Parkhust and Shogren (2007) suggested the agglomeration bonus (AB), a two-part subsidy where landowners are offered a participation payment independent of the actions of their neighbours, plus a bonus – the AB – if their neighbours enrol land as well as them[2]. Thus, the incentive for any land manager to conserve is increasing in the number of "connected parcels" which others also choose to enrol (typically rewarding between-farm spatial coordination rather than within-farm coordination). This creates a coordination game between landowners which may have multiple equilibria. Coordination failure occurs when land managers do not reach privately-optimal decisions that coincide with the social optimum which maximises the overall net benefits from conservation.

The AB has been extensively analysed in the lab and in lab-in-the-field experiments, and has also been studied using choice experiments (eg Villamayor-Tomas et al, 2019) and simulation models. Few actual AB schemes exist, but Huber et al (2021) study a Swiss scheme which offers a bonus to farmers who are willing to enrol patches which are identified in a spatially-coordinated plan. Other real-world examples of AB schemes can be found in Italy, the Netherlands and France (Bareille et al, 2022).

Summarising the lab evidence on the AB, we find that (1) the AB can achieve a variety of target spatial configurations in the landscape (Parkhurst and Shogren, 2007); (2) can increasingly fail to achieve coordinated conservation over time, and is sensitive to what farmers know about their neighbours' decisions (Banerjee et al, 2014) (3) depends on the number of farmers on the network, and how they communicate which each other (Banerjee et al, 2017); (4) can be successfully combined with a nudge related to the behaviour of players in other networks (Banerjee, 2018),

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[2] An alternative is the agglomeration payment: the subsidy is only positive if at least one neighbor participates. This idea of a one-part payment only offered if spatial coordination is achieved (as distinct from the two-part agglomeration *bonus*) was first suggested by Dreschler et al (2010).

although a nudge can also decrease the performance of the AB through crowding out non-monetary incentives (Kuhfuss et al, 2022). Nguyen et al (2024) show that the sign of the spatial correlation between opportunity costs and environmental benefits across the landscape can determine the ability of an AB mechanism to achieve desired levels of connectivity. Using a lab-in-the-field design, Liu et al (2019) find that the AB when introduced as part of a conservation auction leads to a lowering of bid prices by Chinese farmers, but did not produce an increase in the connectivity of land enrolled in the conservation scheme.

The cost-effectiveness of the AB has been investigated by Bareille et al (2022). They model the formation of coalitions of farmers in a landscape who can agree to cooperate in switching to a conservation land use and earn the AB. The key point in their paper is that many such stable coalitions exist in the landscape, but a "grand coalition" involving all farmers is never stable in any setting. They find that the cost-effectiveness of the AB in delivering biodiversity outcomes depends on the extent to which opportunity costs are spatially correlated in the landscape, and how species disperse in the landscape (that is, the ecological benefits of spatial coordination).

Threshold bonuses have been suggested as a means of improving spatial coordination by the simple device of persuading a greater proportion of land managers in a catchment or landscape to enrol through offering an individual-level bonus payment if the number of farmers enrolling in this catchment/landscape exceeds some threshold (Limbach and Rozan, 2023). The higher the fraction of farmers enrolling in the conservation land use, the more likely it is that neighbours will enrol with neighbours – especially if locations with low opportunity cost parcels are clustered together (Babcock et al, 1997). Kuhfuss et al (2016) point out that offering farmers in a specific landscape or catchment an additional payment if some pre-set percentage of all farmers who could enrol in the biodiversity scheme do indeed participate creates a choice setting similar to the provision of a club good with a minimum contribution threshold. Such a threshold bonus might incentivise farmers to join a conservation programme for three reasons. First, if farmers care about the environmental good which the scheme targets, then establishing a threshold makes it more likely that the public good will be delivered. This increases the utility from participating in the scheme. Second, threshold bonuses could encourage learning and knowledge sharing amongst farmers about how best to promote biodiversity on their own farms at low cost. Third, the threshold can be thought of as a descriptive social norm, and if farmers care about the behaviour of others in their peer group, then simply providing information on how many farmers need to contribute to trigger the bonus could act as a nudge. Kuhfuss et al (2016) test the effects of a threshold bonus on the willingness of wine growers in the south of France to reduce herbicide use, using a stated choice experiment. The threshold bonus would be paid if at least 50% of farmers in each respondent's local area agree to join the low-herbicide scheme within 5 years. The authors found that each euro spent on the threshold bonus had a much bigger effect on the area enrolled in the scheme than the same euro being spent increasing the standard participation payment. For an example of analysis of an actual threshold payment scheme, designed to protect the European hamster (*Cricetus cricetus*) in France, see Limbach and Rozan (2023). We know of no paper which compares a threshold bonus with an agglomeration bonus in an empirical setting.

A third means of achieving spatial coordination in participation in biodiversity-enhancing projects is to make use of environmental benefit scoring rules within a conservation auction (Rolfe et al (2008), Windle et al (2009)). A simple conservation auction to encourage biodiversity conservation would rank bids on the basis of bid price alone. Variations in site-level environmental benefits could be used to weight these bid prices if the policy designer wanted to account for variations in ecological potential across sites – as has been done, for example, in the US Conservation Reserve Policy (Hellerstein, 2017). But the algorithm used to rank bids could also take into account the spatial configuration of offers, not just their site-level environmental benefit. Banerjee et al (2021) refer to these two types of environmental score as *node benefits* – the environmental value of a specific location, which is independent of which other bids are accepted – and the *edge benefits*, which are defined between any two adjacent farms *i* and *j.* The regulator invites bids from farmers to join the conservation programme, and then choses those bids whish maximise environmental benefits – the sum of node and edge benefits across all accepted bids – subject to a budget constraint.

4. Collective participation schemes

In collective participation schemes, contracts are awarded to a *group or collective* and not to *individuals*. Pfaff et al (2019) discuss the potential benefits of such contracts. First, in some countries, property rights over changes to land use may reside with groups of people rather than with individuals. In Mexico for example, 60% of forest land is held under collective title. In such cases, contracting with groups who collectively hold the right to deliver ecosystem services or biodiversity benefits may be the only practical option. Second, the costs of monitoring and negotiating over ecosystem service supply or biodiversity provision will be lower if contracts are with groups of land users, rather than individual land users. Third, it may be necessary or more efficient for collective actions to deliver the desired environmental benefit than for individuals to deliver these benefits. However, Pfaff et al also note that groups of suppliers face issues over the incentives for each member of the group to free ride in terms of team effort in supplying the collective output, creating an "assurance game" which may have multiple equilibria, including one where too little effort is supplied. Such incentive problems also exist for collective buyers of the ecosystem service. In such settings, beliefs about the actions of others and the ability to monitor individual effort and sanction non-compliance with a collective agreement to deliver the environmental output will be important, implying that social capital will be a determinant of which equilibrium emerges. Collective contracting may lead to the accumulation of such social capital within the group of suppliers, but the initial stock of social capital will help determine whether such collective agreements start to work. For indicative, qualitative evidence on this point, see Prager, (2022).

Villamayor et al (2021) identify three dimensions of collective agreement schemes, which they refer to as "..public good provision, coordinated implementation and externality internalization". Public good provision refers to the coordination problem about decisions of each member of the collective of whether or not to participate in the joint contract; coordinated implementation refers to spatial coordination amongst those who do decide to participate; whilst internalizing externalities refers to the impacts of the actions taken each individual on their neighbours. They find that the preferences of Swiss farmers asked about their participation in a hypothetical agri-environment scheme which requires spatial coordination between neighbours was influenced by their beliefs about the likely behaviours of their neighbours with respect to their ability to overcome social dilemmas around coordination. Overall, though, the survey found that requiring farmers to cooperate with neighbours in order to receive a PES payment was a dis-incentive to participate.

Since 2016, all agri-environment schemes in the Netherlands are open only to collectives of farmers as participants (Westerink et al, 2017, Barghusen, 2021). Collectives reach agreements with the state regulator over what kind of AES contract to sign, and then reach agreement with individual farmers over how to deliver these outcomes and how to share payments. Bouma et al (2019) note that such schemes face challenges in terms of participation rates due to the likely variation in farmer types and scale, leading to heterogeneity in opportunity costs. If payments to the collective are equally divided, then this means some members will not choose to participate if their opportunity cost exceeds this uniform share. In an experimental study developed with Dutch farmers, Bouma et al study the role of thresholds in determining whether payments are made to the group, noting that this incentive mechanism is a type of threshold public good game with multiple Nash equilibria involving different levels of coordination between members of the collective. They test whether the success of such collective action schemes is influenced by whether collective members get to decide on what the threshold payment should be (the environmental outcome used to define payments by the state to the collective), and the principle by which funds are then allocated to individual members of the collective. One interesting result is that setting a higher threshold improves the performance of the scheme.

Thiermann et al (2023) use a stated choice experiment undertaken with Dutch farmers to investigate preferences for a hypothetical scheme to improve the conservation of meadow birds on agricultural grasslands. They include a payment attribute which is specified as the number of meadow birds on land managed by farmers within a specific collective, relative to the mean number of birds counted across all collectives. This attribute combines a results-based aspect with a collective provision of the biodiversity outcome. Sampling was limited to farmers who were already part of an action-based collective participation AES for meadow birds. Contracts were specified in terms of a mix of biodiversity outcomes and actions such as changes to mowing dates. Results showed that farmers were positively influenced to join the AES scheme by the presence of a collective performance bonus set at a high level.

Another question we might ask is whether making a given policy offer open to collective participation *in addition to* individual participation improves the economic and ecological performance of that policy. If collective participation would appear to make more likely the achievement of spatial coordination for example, and if spatial coordination is a key determinant of ecological benefits, then this hypothesis is worth testing. Banerjee et al (2021) undertake such a test using the idea of joint bidding in a conservation auction. Their argument is: if environmental benefits are increasing in spatial coordination, and if joint bidding with a neighbour improves such coordination, then will the cost effectiveness of the auction be better when farmers have the option to bid jointly with their neighbours? The authors note that this is not obviously the case: joint bidding can reduce the number of bids offered relative to individual-only bidding, and thus make the auction less competitive. This could increase bid prices and information rents. Moreover, deciding whether and how to bid jointly with a neighbour incurs transactions costs for participants. To offset these costs, the auction design offered a “bonus” for accepted joint bids.

Arranging “farmers” in a circular landscape – so that each individual has two direct neighbours she can decide whether or not to bid jointly with – the authors ran a lab experiment, using students as subjects. Environmental benefit scores were based both on the node environmental benefits of each individual bidding and the edge benefits between neighbours, to simulate spatial coordination gains. Subjects were allocated “farms” which varied in their opportunity costs and environmental scores. Depending on treatment, an individual Agglomeration Bonus (AB) was offered to individual winning bids. Results showed that with such an AB on offer, allowing joint bidding gave no improvement to either the environmental performance of the auction or its cost-effectiveness. Without such an individual AB, joint bidding improved the environmental performance of the auction but resulted in a loss of cost-effectiveness. The provision of a financial bonus for joint bids reduced how much participants marked up their opportunity costs in submitting a bid, with higher joint bidding bonuses leading to lower mark-ups. But with a fixed budget, paying these joint bidding bonuses reduced the number of winning contracts the regulator could award.

The idea is further tested in Liu et al (2024), this time using actual farmers in China as the participants. Farmers from the Huanshang area were organised into local networks of n=6 players. Each player was told the opportunity costs, node and edge environmental benefits of their “farm” on this network. These environmental benefits were associated with farmers joining a hypothetical PES programme to reduce chemical pesticide use. Farmers either submitted individual or joint bids, with and without (i) an individual-level AB and (ii) a bonus for joint bidding. The baseline treatment was no joint bidding bonus and no individual-level AB, with only individual bidding allowed. The goal of the regulator was to maximise environmental benefits on the network subject to a budget constraint. Results showed that joint bidding resulted in no improvement in either total environmental benefits or cost effectiveness. This comes about partly because of the cost of paying the joint bidding bonuses. Farmers with higher edge benefits were more likely to chose to bid jointly, potentially since such joint bids could results in greater rent capture owing to the higher environmental scores (and thus a more competitive bid) which such a joint bid could result in.

5. Markets for biodiversity offsets

For many of the policy ideas discussed above, funding typically comes from the public sector. For example, Villanueva et al (2023) find that only 14 out of 93 schemes they review are funded by the private sector. Whilst NGOs certainly play a major role in funding PES programmes worldwide, and whilst user groups can also be the buyers of ecosystem services (Smith et al, 2019), in many cases the state acts as the buyer on behalf of multiple beneficiaries within society. This is perhaps especially the case for biodiversity, which lacks the privately-capturable benefit characteristics of some benefits provided by ecosystem services – such as water quality enhancements. *Markets in biodiversity offsets* are a means of bringing private sector funding into incentives for biodiversity conservation on private land. Such markets can be regulatory or voluntary. In regulatory markets, buyers – say house builders – are required by law to obtain sufficient credits from suppliers to offset some measurable negative impacts of their actions on biodiversity. Voluntary markets emerge from the desire of buyers to be seen to be paying for conservation, rather than a regulatory imperative. In what follows, we focus on regulatory offset markets.

An offset is an entitlement to a measured gain in some metric (eg hectares of wetland, or hectares of quality-adjusted wetland) which is supplied by a land manager who changes their land management from some baseline to increase the value of this metric, eg by creating new wetlands (Needham et al, 2019). In some offset markets (eg in New South Wales), credits can also be claimed for avoided, measurable damage to some metric, such as not draining an existing wetland. Offset credits can be created by one party, say a farmer in Kent, and then sold to a second party, say a housebuilder in Surrey. Those agents whose behaviour leads to the creation of offset credits constitute the supply side of the market. Buyers constitute the demand side. Trades between buyers and sellers can be bilaterial, or can be moderated by an offset bank, which collects buy and sell offers, and then matches buyers to sellers. The offset bank, or a regulator, can act both as an intermediary between buyers and sellers, but in a regulated market also through validating and monitoring the creation and purchase of credits. For suppliers, their willingness to supply credits depends on the (opportunity) costs of creating such credits via changing land use. If such opportunity costs are heterogenous across space, we expect to find an upward sloping supply curve in the market which reflects these increasing marginal costs. Buyers' Willingness to Pay depends on the value they create by developing land: the offset market demand is thus derived from the demand for whatever the end point of the development is (eg housing). If the value of development varies across space, then again we can expect to observe a downward sloping demand curve which reflects the ranked willingness to pay of potential buyers (house builders) for credits (Simpson et al, 2021b).

In equilibrium, development is encouraged to take place in those locations where biodiversity loss is expected to be lowest as measured by the metric, since these locations are where the number of credits the developer needs to buy are lowest (ceteris paribus). Increases in biodiversity due to the creation of credits is encouraged where the potential gain is greatest, since here land managers can earn the biggest number of credits from a given restoration action. The market thus directs development away from the highest ecologically-valued land (where more credits are needed to allow development), and pushes conservation actions to locations with higher biodiversity potential (where more credits can be earned). Depending on the choice of metric, the creation of credits can produce coordinated, large-scale restoration and the emergence of conserved corridors. The existence of the market means that (i) farmers have an on-going financial incentive to invest in conservation whilst (ii) developers face an on-going cost from actions which deplete biodiversity. In an important sense, the missing market is replaced and in principal, the market failure corrected.

However, this very much depends on how the offset market is designed, in terms of its metric and trading rules, spatial scale, conservation objective and how the market is monitored. Needham et al (2019) reviewed lessons learned from tradeable pollution permit markets for the design of emerging biodiversity offset markets, and concluded that there were four design parameters that mattered:

- Policy targets and exchange currencies;
- Trading ratios;
- Market scale; and
- Market regulation.

Policy targets relate to whether the ambition is no net loss of some measure of biodiversity, or a net gain. Current UK policy on biodiversity offsetting mandates a 10% gain from each trade, but typically we think of the target as the equivalent of the cap in a pollution trading market. Simpson et al (2021b) investigate the ecological and economic implications of changing this target in an ecological-economic model of farmland use. They show that, in theory, the implications of a tightening of the target (eg from no net loss to a 10% gain) on the equilibrium price and quantity of offsets traded is ambiguous, but can be shown to depend on the price elasticity of demand for offsets. As the offset target is tightened, developers need to acquire more credits for a given level of development, but this same toughening of the target reduces the likelihood that the marginal development will go ahead. These two effects work against each other in terms of equilibrium price and quantity. Their empirical model relates changes in land use to predicted impacts on the abundance and distribution of lapwings (*Vanellus vanellus).* Building houses at a location decreases lapwing numbers. Creating low-intensity grassland increases lapwing and thus earns a number of offset credits depending on how big the modelled increase in lapwing is. Using this model combined with Willingness To Pay and Willingness to Accept functions (demand and supply functions) derived from house prices and farm profits, the authors find that as the net gain target is increased, the equilibrium price and quality of offset credits traded decreases, as the price elasticity of demand at the no net loss equilibrium is highly elastic. Importantly, as the target for lapwing is increased (made tougher), this will have impacts on other birds which depend on low intensity grassland. Setting a given quantitative target (eg no net loss) with regard to one species baseline will also produce different results for the target being set for no net loss for a different species sharing similar habitat requirements, such as the curlew (*Numenius Arquata)*. Moreover, as the net gain target is increased, we see changes in both overall lapwing numbers but also in where in the landscape gains and losses occur (Simpson et al, 2021a).

Exchange currencies are the metrics in which offsets are denominated. In the example above, the metric is predicted numbers of lapwings on farmland in a specific area. However, biodiversity is a multi-faceted concept, and multiple potential metrics exist. Using the same modelling structure as that outlined above, Simpson et al (2022) evaluate the impacts of changing metrics in an offset market. They consider the effects of (i) metrics defined in terms of predicted impacts on a specific species, such as lapwing or curlew (ii) a metric defined in terms of hectares of a specific land cover. Overall, they show that a trading metric defined in terms of habitat may lead to unintended losses in species-based metrics; whilst a species-based metric will have impacts on how many hectares of the focus habitat are lost or gained in aggregate. Setting the metric as numbers of curlew results in trades, and thus land use changes, which impact on the numbers and distribution of lapwings. These patterns of gains and losses in species and habitats depend on the underlying spatial distributions of species, farmland profits and housing values.

Trading ratios determine how many credits from a conservation site *j* need to be purchased to offset the predicted impacts of development in location *i.* A considerable literature exists on the impacts and specification of trading ratios for tradeable pollution permits, which shows that too complex a trading ratio can increase the transactions costs of trading, and thus reduce the efficiency of the market, but can also result in unintended consequences on different measures of pollution (eg ambient quality levels versus aggregate emissions). In biodiversity offsets, a largely ecological literature discusses issues such as the incorporation of recovery risks and time lags in trading ratios (eg Laitala et al, 2014; Kangas and Ollikainen, 2019). Bush et al (2023) explore whether the concept of “irreplaceability” in the systematic conservation planning literature can be used to inform trading ratios for offset markets. Irreplaceability refers to the ability to exchange one conservation site for another and still achieve target outcomes, which are usually related to species abundance or extinction risks. Sites which are more unique in their characteristics will have higher irreplaceability scores than those which display more widely found attributes, defined in terms of their ability to move the planner closer to hitting species conservation targets. Many such targets can be simultaneously considered. Using what they refer to as “summed alpha irreplaceability” forming the trading ratio between any two locations (Baisero et al, 2021), they show that trading on this basis results in more cost effective outcomes from the offset market than using trading ratios based on more common metrics.

Market scale refers to issues such as geographic scope. The new biodiversity net gain offsets market in the UK limits trades to the same council planning area (eg county) which the development occurs in. Bigger geographic scales mean more choice for developers in terms of who they purchase their offsets from, and a wider range of buy-sell offers. However, ecologists may worry about the ability of distant conservation sites to offset damages at more local sites. Simpson et al (2021a) compare the same biodiversity offset market at two different spatial scales. In the “full market scenario” the entire case study area constitutes the geographic scale. An ecological model predicts the trading ratios between pairs of sites across this whole area. A “three service area scenario” then divided the case study area into 3 smaller zones. Within-zone trades are allowed, but no inter-zone trading is permitted. All trades occur in a metric defined in terms of predicted impacts on oystercatchers (*Haematopus ostralegus)*. Results showed that the full market scenario led to no net loss in oystercatchers, but bigger negative impacts on two other wader species (lapwings and curlew) compared to the three service area scenario. A market clearing price of £21,000 per oystercatcher occurs in the full market scenario, but the equilibrium prices in the three service area scenarios varied from £771 to £39,000. This implies that moving towards a single geographic market would lead to gains from trade compared to the divided markets equilibria. Interestingly, these gains from trade in moving to a single market favoured developers: suppliers (farmers) do better in the divided markets scenario.

Finally, how an offset market is regulated will help determine its ecological and economic consequences. Relevant aspects of regulation include monitoring and enforcement actions (eg to make sure that predicted conservation outcomes are achieved in offset supply sites), and the time scales over which offset contracts are agreed. Another key aspect of regulation will be whether the regulator and/or offset banks facilitate trades by providing information on offers and demands from sellers and buyers, and whether a bank of already-validated credits is made available to prospective buyers. Ecological evidence on the effectiveness of offset schemes is currently rather pessimistic in its assessment of outcomes (zu Ermgassen et al, 2019, 2021), so that much progress is still needed in improving the design of this incentive mechanism.

6. Conclusions

In this paper, we considered the main characteristics of the policy design problem, given an objective of incentivising more biodiversity conservation on privately-owned land. These characteristics were heterogeneous and partly-hidden opportunity costs of conservation, spatial variation in the capacity of land to deliver more of a biodiversity metric, and the costs of observing farmers' actions or efforts in conservation, or the outcomes of these actions. We then reviewed recent research which has tried to cast light on how best to solve these design problems, under the headings of paying for results rather than actions, spatial coordination, collective participation and biodiversity offset markets.

Many other avenues of research are relevant to incentivising biodiversity conservation on private land which are not dealt with here. These include (i) the use of nudges to change farmer's behaviours (eg Massfeller et al, 2022; Thomas et al, 2019); (ii) the fairness or otherwise of economic incentives for conservation (Loft et al, 2020; Qambemeda et al, 2024), (iii) the extent to which PES payments crowd out environmental motivations (Vorlaufer et al, 2023), and the (iv) optimal balance between regulatory approaches and voluntary opt-in PES-type measures (eg Barreiro-Hurle et al, 2023). There has also been increasing interest in understanding what motivates farmers to participate in voluntary PES-type schemes, broadening these drivers beyond profit to include non-monetary factors such as environmental preferences, and social pressures such as perceptions of what it means to be a "good farmer" (McGuire et al, 2013; Sulemana and James, 2014; Dessart et at, 2019). Stated preference choice experiments have proved to be a useful tool to examine this wider set of drivers, as have lab-in-the-field approaches (Cortes-Capano et al, 2021, Lefebvre et al, 2021; Schulze et al, 2024). Finally, I note that there are a number of existing reviews in the literature evaluating the performance of PES-type schemes for conservation: see for example Engel (2016) and Wunder et al (2020).

A summary of the impacts of research on actual policy in these areas would be to say that it has been small as far as one can judge. Most PES policies concerned with farmland use and biodiversity still involve fairly simple payment-for-action schemes which do not account for any of this recent research (Hasler et al, 2022). Some progress has occurred, in terms of the spread of payment-for-results schemes (for example, CAP eco-schemes in Germany and Holland), greater uptake of collective contracting, and the new (2024) UK market in biodiversity net gain offsets: but these changes are at the margin of policy design. Clearly, economists still have a lot of work to do to better communicate their ideas to biodiversity policy designers.

References

Ait Sidhoum, Amer, Philipp Mennig, Johannes Sauer, Do agri-environment measures help improve environmental and economic efficiency? Evidence from Bavarian dairy farmers (2023), *European Review of Agricultural Economics*, Volume 50, Issue 3, Pages 918–953.

Ando, A.W. et al. (1998). Species distributions, land values, and efficient conservation. *Science* **279:** 2126–2128.

Armsworth P, Acs S, Dallimer M, Gaston K, Hanley N and Wilson P (2012) "The costs of simplification in conservation programmes" *Ecology Letters,* Volume 15, Issue 5, Pages: 406–414.

Babcock B., Lakshminarayan P., Wu J. and Zilberman D., 1997. Targeting tools for the purchase of environmental amenities" *Land Economics*, 73 (2), 325-339.

Baird, J., Ken W. Belcher & Michael Quinn (2014) Context and capacity: The potential for performance-based agricultural water quality policy, *Canadian Water Resources Journal*, 39:4, 421-436, DOI: 10.1080/07011784.2014.965033

Baisero D., Schuster R. and Plumptre A. (2021) "Redefining and mapping global irreplaceability" *Conservation Biology,* 36, e13806.

Banerjee, S., De Vries, F. P., Hanley, N., & van Soest, D. P. (2014). The impact of information provision on agglomeration bonus performance: an experimental study on local networks. *American Journal of Agricultural Economics, 96*(4), 1009-1029.

Banerjee, S. (2018). Improving spatial coordination rates under the agglomeration bonus scheme: A laboratory experiment with a pecuniary and a non-pecuniary mechanism (NUDGE). *American Journal of Agricultural Economics, 100*(1), 172-197.

Banerjee, S., Cason, T. N., de Vries, F. P., & Hanley, N. (2017). Transaction costs, communication and spatial coordination in Payment for Ecosystem Services Schemes. *Journal of environmental economics and management, 83*, 68-89.

Banerjee S, Cason T., de Vries F. and Hanley N. (2021) “Spatial Coordination and Joint Bidding in Conservation Auctions” *Journal of the Association of Environmental and Resource Economists*, 8 (5), 1013-1045. https://www.journals.uchicago.edu/doi/10.1086/714601.

Barghusen R., Sttler C., Matzdorf B. (2021) “Motivations of farmers to participate in collective agri-environment schemes” *Ecosystems and People*, 17 (1), 539-555.

Bareille F., Zavalloni M. and Viaggi D. (2022) “Agglomeration bonus and endogenous coalition formation” *Amer.Jnl. Ag. Econ.* 105 (1), 76-98.

Barreiro-Hurle J., Francois J. Dessart, Jens Rommel, Mikołaj Czajkowski, Maria Espinosa-Goded, Macario Rodriguez-Entrena, Fabian Thomas, Katarzyna Zagorska (2023) “Willing or complying? The delicate interplay between voluntary and mandatory interventions to promote farmers' environmental behavior” *Food Policy,* Volume 120, 102481, https://doi.org/10.1016/j.foodpol.2023.102481.

Bartkowski, B., Droste, N., Ließ, M., Sidemo-Holm, W., Weller, U., Brady, M.V., 2021. Payments by modelled results: A novel design for agri-environmental schemes. *Land Use Policy*, 102, 105230.

Bouma J., Nguyen T., van der Heijden E. and Dijk, J. (2020) “Analysing group contract design using a threshold public goods experiment” *European Review of Agricultural Economics,* 47 (3), 1250-1275.

Bradfer-Lawrence, T., Dobson, A., Finch, T., Fuentes-Montemayor, E., Hanley, N., Matthiopoulos, J., Nthambi, M., Simpson, K., Watts, K., Whytock, R. and Park, K. (2024). Spillovers and legacies of land management on temperate woodland biodiversity. Research Square: https://www.researchsquare.com/article/rs-4109245/v1.

Bush, A., Simpson, K. H., & Hanley, N. (2023). Systematic nature positive markets. *Conservation Biology*, 00, 00–00. https://doi.org/10.1111/cobi.14216

Canessa, C., Terese E. Venus, Miriam Wiesmeier, Philipp Mennig, Johannes Sauer, (2023) Incentives, Rewards or Both in Payments for Ecosystem Services: Drawing a Link Between Farmers' Preferences and Biodiversity Levels, *Ecological Economics,* Volume 213, 107954,

Cortés-Capano Gonzalo, Nick Hanley, Oleg Sheremet, Anna Hausmann, Tuuli Toivonen, Gustavo Garibotto-Carton, Alvaro Soutullo, Enrico Di Minin (2021) "Assessing landowners' preferences to inform voluntary private land conservation: The role of non-monetary incentives," *Land Use Policy,* Volume 109,105626, https://doi.org/10.1016/j.landusepol.2021.105626.

Dakpo, K. Hervé, Philippe Jeanneaux, and Laure Latruffe. (2016) "Modelling pollution-generating technologies in performance benchmarking: Recent developments, limits and future prospects in the nonparametric framework." *European Journal of Operational Research* 250.2 : 347-359.

Department for Environment, Food and Rural Affairs (DEFRA) (2024), https://www.gov.uk/government/statistics/england-biodiversity-indicators. Accessed 10 January 2025.

Derissen, S, and Quaas, M (2013) Combining performance-based and action-based payments to provide environmental goods under uncertainty. *Ecological Economics* 85: 77-84

Dessart, F., Barreiro-Hurle, J., van Bavel, R. (2019). Behavioural factors affecting the adoption of sustainable farming practices: a policy-oriented review. *Eur. Rev. Agric. Econ.* 46 (3), 417–471

Dreschler M., Watzold F., Johst K., Shogren J. (2010) "An agglomeration payment for cost-effective biodiversity conservation" *Resource and Energy Economics,* 32 (2), 261-275.

Engel S. (2016), "The Devil in the Detail: A Practical Guide on Designing Payments for Environmental Services", International Review of Environmental and Resource Economics: Vol. 9: No. 1–2, pp 131-177. http://dx.doi.org/10.1561/101.00000076

zu Ermgassen, S.O.S.E., Baker, J., Griffiths, R.A., Strange, N., Struebig, M.J. and Bull, J.W., (2019). The ecological outcomes of biodiversity offsets under “no net loss” policies: A global review. *Conservation Letters*, 12(6), p.e12664.

zu Ermgassen, S. O. S. E., Marsh, S., Ryland, K., Church, E., Marsh, R., Bull, J. W. (2021). Exploring the ecological outcomes of mandatory biodiversity net gain using evidence from early-adopter jurisdictions in England. *Conservation Letters*. 14: e12820. https://doi-org.ezproxy1.lib.gla.ac.uk/10.1111/conl.12820

Ferraro, P.J. (2008) Asymmetric information and contract design for payments for environmental services, *Ecological Economics*, Volume 65, Issue 4, Pages 810-821.

Fooks, J. R., Higgins, N., Messer, K. D., Duke, J. M., Hellerstein, D., & Lynch, L. (2016). Conserving spatially explicit benefits in ecosystem service markets: experimental tests of network bonuses and spatial targeting. *American Journal of Agricultural Economics*, 98(2), 468-488.

Granado-Díaz, R., Villanueva, A. J., & Colombo, S. (2024). Land manager preferences for outcome-based payments for environmental services in oak savannahs. *Ecological Economics*, *220*, 108158. https://doi.org/10.1016/j.ecolecon.2024.108158

Hanley N, Davies A, Angelopoulos K., Hamilton A., Ross A., Tinch D., and Watson F. (2008) “Economic determinants of biodiversity change over a 400-year period in the Scottish uplands” *Journal of Applied Ecology,* 45, 1557–1565

Hanley N, Banerjee S., Lennox G, and Armsworth P. (2012) "How should we incentivise private landowners to "produce" more biodiversity?" *Oxford Review of Economic Policy*, 28 (1), 93-113.

Hanley N., Shogren J. and White B. (2019) *Introduction to Environmental Economics.* Oxford: Oxford University Press.

Hellerstein D. (2017) "The US Conservation Reserve Program: the evolution of an enrollment mechanism" *Land Use Policy*, 63, 601-610.

Hasler, B., Termansen, M., Nielsen, H. Ø., Daugbjerg, C., Wunder, S., & Latacz-Lohmann, U. (2022). "European Agri-environmental Policy: Evolution, Effectiveness, and Challenges". *Review of Environmental Economics and Policy, 16*(1), 105-125. doi:10.1086/718212.

Herzon, I., Birge, T., Allen, B., Povellato, A., Vanni, F., Hart, K., Radley, G., Tucker, G., Keenleyside, C., Oppermann, R., Underwood, E., Poux, X., Beaufoy, G. and Pražan, J. (2018). Time to look for evidence: Results-based approach to biodiversity conservation on farmland in Europe. *Land Use Policy*, 71, pp.347–354

Hofmeister, J., Hošek, J., Brabec, M., & Kočvara, R. (2017). Spatial distribution of bird communities in small forest fragments in central Europe in relation to distance to the forest edge, fragment size and type of forest. *Forest Ecology and Management*, 401, 255–263.

Howard, Gregory, Wendong Zhang, Adriana Valcu-Lisman, and Philip W. Gassman (2024).
" Evaluating the Tradeoff between Cost Effectiveness and Participation in Agricultural Conservation

Programs.” *American Journal of Agricultural Economics* 106(2): 712–738. https://doi.org/10.1111/ajae.12397

Huber, Robert, Astrid Zabel, Mirjam Schleiffer, Willemijn Vroege, Julia Brändle, and Robert Finger. (2021) “Conservation Costs Drive Enrolment in Agglomeration Bonus Scheme.” *Ecological Economics* 186: 107064.

Iftekhar M.S., Tisdell J.G.. and L. Gilfedder (2014) “Private lands for biodiversity conservation: review of conservation coventanting programmes in Tasmania” *Biological Conservation*, 169, 176-184.

IPBES (2019): Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. S. Díaz, J. Settele, E. S. Brondízio E.S., H. T. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. A. Brauman, S. H. M. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S. M. Subramanian, G. F. Midgley, P. Miloslavich, Z. Molnár, D. Obura, A. Pfaff, S. Polasky, A. Purvis, J. Razzaque, B. Reyers, R. Roy Chowdhury, Y. J. Shin, I. J. Visseren-Hamakers, K. J. Willis, and C. N. Zayas (eds.). IPBES secretariat, Bonn, Germany. 56 pages.

Kangas, J. and Markku Ollikainen, (2019) Economic Insights in Ecological Compensations: Market Analysis With an Empirical Application to the Finnish Economy, *Ecological Economics,*

Volume 159,Pages 54-67,https://doi.org/10.1016/j.ecolecon.2019.01.003.

Kreft, C., Huber, R., Wuepper, D., & Finger, R. (2021). The role of non-cognitive skills in farmers’ adoption of climate change mitigation measures. *Ecological Economics*, *189*, 107169. https://doi.org/10.1016/J.ECOLECON.2021.107169

Kuhfuss, L, Preget, R, Thoyer, S & Hanley, N (2016), 'Nudging farmers to enrol land into agri-environmental schemes: the role of a collective bonus' *European Review of Agricultural Economics*, vol 43, no. 4, pp. 609-636.

Kuhfuss, Laure, Raphaële Préget, Sophie Thoyer, Frans P. de Vries, Nick Hanley (2022) "Enhancing spatial coordination in payment for ecosystem services schemes with non-pecuniary preferences" *Ecological Economics*, Volume 192, 107271, https://doi.org/10.1016/j.ecolecon.2021.107271.

Laney, R. (2023). Delivering wildlife habitat on productive agricultural lands cost-effectively: The case of migratory shorebirds on California rice lands. *Conservation Science and Practice*, 5(7), e12955. https://doi.org/10.1111/csp2.12955

Laitila, J., Moilanen, A. & Pouzols, F.M. (2014) A method for calculating minimum biodiversity offset multipliers accounting for time discounting, additionality and permanence. *Methods in Ecology and Evolution,* **5,** 1247-1254

Lefebvre, M., Barreiro-Hurlé, J., Blanchflower, C., Colen, L., Kuhfuss, L., Rommel, J., Šumrada, T., Thomas, F. and Thoyer, S. (2021), Can Economic Experiments Contribute to a More Effective CAP?. *EuroChoices*, 20: 42-49. https://doi.org/10.1111/1746-692X.12324

Limbach K. and Rozan A. (2023) "Coordinating farms in collective agri-environment schemes" *European Review of Agricultural Economics,* 50 (5), 1715-1753.

Liu, Zhaoyang, Jintao Xu, Xiaojun Yang, Qin Tu, Nick Hanley, and Andreas Kontoleon. (2019). "Performance of Agglomeration Bonuses in Conservation Auctions: Lessons from a Framed Field Experiment." *Environmental and Resource Economics* 73: 843–69.

Liu, Zhaoyang, Simanti Banerjee, Timothy N. Cason, Nick Hanley, Qi Liu, Jintao Xu, and Andreas Kontoleon. (2024). " Spatially Coordinated Conservation Auctions: A Framed Field Experiment Focusing on Farmland Wildlife Conservation in China." *American Journal of Agricultural Economics* 1–26. https://doi.org/10.1111/ajae.12447

Loft L., S. Gehrig, C. Salk, J. Rommel, (2020) Fair payments for effective environmental conservation, Proc. Natl. Acad. Sci. U.S.A. 117 (25) 14094-14101, https://doi.org/10.1073/pnas.1919783117

McGuire, J., Morton, L.W. & Cast, A.D. (2013) Reconstructing the good farmer identity: shifts in farmer identities and farm management practices to improve water quality. *Agric Hum Values* **30**, 57–69 https://doi.org/10.1007/s10460-012-9381-y

Margules, C., Pressey, R. Systematic conservation planning. *Nature* **405**, 243–253 (2000). https://doi.org/10.1038/35012251

Massfeller Anna, Manuela Meraner, Silke Hüttel, Reinhard Uehleke, (2022) "Farmers' acceptance of results-based agri-environmental schemes: A German perspective," *Land Use Policy*, Volume 120, 106281, https://doi.org/10.1016/j.landusepol.2022.106281.

Moran, J., Byrne, D., Carlier, J., Dunford, B., Finn, J. A., Ó hUallacháin, D., & Sullivan, C. A. (2021). Management of high nature value farmland in the Republic of Ireland: 25 years evolving toward locally adapted results-orientated solutions and payments. *Ecology and Society*, *26*(1). https://doi.org/10.5751/ES-12180-260120

Needham K., F. de Vries, P. Armsworth and Hanley N. (2019) 'Designing Markets for Biodiversity Offsets: Lessons from Tradable Pollution Permits' *Journal of Applied Ecology*, 56:1429–1435. https://doi.org/10.1111/1365-2664.13372

Nguyen C., Latacz-Lohman U., Hanley N., Schilizzi S., S. Iftekhar, (2022) Spatial Coordination Incentives for Landscape-scale Environmental Management: A Systematic Review *Land Use Policy,* Volume 114, 105936, doi.org/10.1016/j.landusepol.2021.105936.

Nguyen C., Hanley N., Latacz-Lohman U. (2024) “"Landscape-level determinants of the performance of an agglomeration bonus in conservation auctions." *Journal of Agricultural Economics*, 00, doi.org/10.1111/1477-9552.12576

Palm-Forster, L. H., Swinton, S. M., Redder, T. M., DePinto, J. V., & Boles, C. M. W. (2016). Using conservation auctions informed by environmental performance models to reduce agricultural nutrient flows into Lake Erie. *Journal of Great Lakes Research*, *42*(6), 1357–1371. https://doi.org/10.1016/j.jglr.2016.08.003

Parkhurst, G. M., & Shogren, J. F. (2007). Spatial incentives to coordinate contiguous habitat. *Ecological economics, 64*(2), 344-355.

Pfaff, A., Rodriguez, L. A. and Shapiro-Garza, E. (2019) 'Collective Local Payments for ecosystem services: New local PES between groups, sanctions, and prior watershed trust in Mexico', *Water Resources and Economics*, 28, p. 100136. doi: 10.1016/j.wre.2019.01.002.

Polasky, S. et al. (2008). Where to put things? Spatial land management to sustain biodiversity and economic returns. *Biol. Conserv.* **141:** 1505–1524.

Prager, K. (2022) "Implementing policy interventions to support farmer cooperation for environmental benefits" *Land Use Policy,* 119, 106182.

Qambemeda M. Nyanghura, Lisa Biber-Freudenberger, Jan Börner, (2024) Incentives for biodiversity conservation under asymmetric land ownership, *Ecological Economics*, Volume 219, 108152, https://doi.org/10.1016/j.ecolecon.2024.108152

Rolfe J., McCosker J. and Windle J. (2008) "Identifying the incentives that graziers in central-western Queensland need to manage land for conservation outcomes" *The Rangeland Journal,* 30, 1-7.

Schulze, C., Zagórska, K., Häfner, K., Markiewicz, O., Czajkowski, M. & Matzdorf, B. (2024) Using farmers' ex ante preferences to design agri-environmental contracts: A systematic review. *Journal of Agricultural Economics*, 75, 44–83. doi.org/10.1111/1477-9552.12570

Schilizzi, S. and Latacz-Lohman, U. (2016) "Incentivizing and Tendering Conservation Contracts: The Trade-off between Participation and Effort Provision" *Land Economics* 16, 92 (2) 273-291; **DOI:** 10.3368/le.92.2.273

Sheremet, O., Ruokamo, E., Juutinen, A., Svento, R. and Hanley, N. (2018) Incentivising participation and spatial coordination in payment for ecosystem service schemes: forest disease control programs in Finland. *Ecological Economics*, 151, 260-272.

Shimada H (2020) “Do monetary rewards for spatial coordination enhance participation in a forest conservation program?” *Ecological Economics,* 177,

Simpson, K. H. , de Vries, F. , Dallimer, M. , Armsworth, P. R. and Hanley,. (2021a) “Understanding the performance of biodiversity offset markets: evidence from an integrated ecological - economic model”. *Land Economics* , (doi: 10.3368/le.97.4.030420-0032R)

Simpson K, Hanley N, Armsworth P, de Vries F and Dallimer M (2021b) “Incentivising biodiversity net gain with an offset market” *Q Open*, https://doi.org/10.1093/qopen/qoab004

Simpson K., de Vries F., Dallimer M, Armsworth P and Hanley Nick (2022) “Ecological and Economic Implications of Alternative Metrics in Biodiversity Offset Markets” *Conservation Biology,* 00, e13906. https://doi.org/10.1111/cobi.13906

Simpson K., Paul R. Armsworth, Martin Dallimer, Mary Nthambi, Frans P. de Vries, Nick Hanley, (2023) Improving the ecological and economic performance of agri-environment schemes: Payment by modelled results versus payment for actions, *Land Use Policy*, Volume 130, 2023, 106688, https://doi.org/10.1016/j.landusepol.2023.106688.

Smith, G., Day, B. & Binner, A. Multiple-Purchaser Payments for Ecosystem Services: An Exploration Using Spatial Simulation Modelling. *Environ Resource Econ* **74**, 421–447 (2019). https://doi.org/10.1007/s10640-019-00324-z

Smith, R. B., & Shogren, J. F. (2002). Voluntary incentive design for endangered species protection. *Journal of environmental economics and management, 43*(2), 169-187

Sulemana I., Harvey S. James, (2014) “Farmer identity, ethical attitudes and environmental practices,” *Ecological Economics*, Volume 98, Pages 49-61, https://doi.org/10.1016/j.ecolecon.2013.12.011

Šumrada, T., Vreš, B., Čelik, T., Šilc, U., Rac, I., Udovč, A., and Erjavec, E. (2021). Are result-based schemes a superior approach to the conservation of High Nature Value grasslands? Evidence from Slovenia. *Land Use Policy*, 105749.

Tanaka, K., Hanley, N. & Kuhfuss, L. (2022) Farmers' preferences toward an outcome-based payment for ecosystem service scheme in Japan. *Journal of Agricultural Economics*, 73 (3), 720-738. https://doi.org/10.1111/1477-9552.12478

Terraube, J., Archaux, F., Deconchat, M., van Halder, I., Jactel, H., & Barbaro, L. (2016). Forest edges have high conservation value for bird communities in mosaic landscapes. *Ecology and Evolution*, 6(15), 5178–5189.

Thiermann Insa, Brechtje Silvius, Melody Splinter, Liesbeth Dries, (2023) Making bird numbers count: Would Dutch farmers accept a result-based meadow bird conservation scheme? *Ecological Economics*, Volume 214, 107999.

Thomas, Fabian, Estelle Midler, Marianne Lefebvre, Stefanie Engel, Greening the common agricultural policy: a behavioural perspective and lab-in-the-field experiment in Germany (2019), *European Review of Agricultural Economics*, Volume 46, Issue 3, 367–392, https://doi.org/10.1093/erae/jbz014

Tyllianakis, E., Martin-Ortega, J., Ziv, G., Chapman, P.J., Holden, J., Cardwell, M., Fyfe, D., (2023). A window into land managers' preferences for new forms of agri-environmental schemes: evidence from a post-Brexit analysis. Land Use Policy 129, 106627. https://doi.org/10.1016/j.landusepol.2023.106627

Villamayor-Tomas, Sergio, Julian Sagebiel, Roland Olschewski, (2019) "Bringing the neighbors in: A choice experiment on the influence of coordination and social norms on farmers' willingness to accept agro-environmental schemes across Europe" *Land Use Policy*, Volume 84, Pages 200-215, doi.org/10.1016/j.landusepol.2019.03.006.

Villamayor-Tomas Sergio, Julian Sagebiel, Jens Rommel, Roland Olschewski, (2021) "Types of collective action problem and farmers' willingness to accept agri-environmental schemes in Switzerland" *Ecosystem Services*, Volume 50, 101304, doi.org/10.1016/j.ecoser.2021.101304.

Villanueva A., Gomez-Leon J., Arriaza M. and Rodriguez-Entrena M (2015) "The design of agri-environment schemes: farmers' preferences in Southern Spain" *Land Use Policy,* 46, 142-154.

Villanueva, A.J.; Barreiro-Hurle, J.; Perni-Llorente, A. (2023). "A meta-analysis on choice experiments focusing on land managers' willingness to accept for environmental scheme participation". 5th Annual REECAP Meeting, Warsaw 26-28 September

Villanueva, A. J., Granado-Díaz, R., & Colombo, S. (2024). Comparing practice- and results-based agri-environmental schemes controlled by remote sensing: An application to olive groves in Spain. *Journal of Agricultural Economics*. https://doi.org/10.1111/1477-9552.12573

Vorlaufer T., S. Engel, J. de Laat, B. Vollan, (2023) Payments for ecosystem services did not crowd out pro-environmental behavior: Long-term experimental evidence from Uganda, *Proc. Natl. Acad. Sci. U.S.A.* 120 (18) e2215465120, https://doi.org/10.1073/pnas.2215465120 .

Wallander, Steven & Hellerstein, Daniel M. & Johnsen, Reid, (2018). "Cost Effectiveness of Conservation Auctions Revisited: The Benefits of Information Rents," 2018 Annual Meeting, August 5-7, Washington, D.C. 274457, Agricultural and Applied Economics Association.

Westerink, J., Jongeneel, R., Polman, N., Prager, K., Franks, J., Dupraz, P., & Mettepenningen, E. (2017). Collaborative governance arrangements to deliver spatially coordinated agri-environmental management. *Land Use Policy*, 69, 176-192.

White, B & Hanley, ND (2016), 'Should we pay for ecosystem service outputs, inputs or both?' *Environmental and Resource Economics*, vol 63, no. 4, pp. 765-787., **10.1007/s10640-016-0002-x**.

Windle J., Rolfe J., McCosker J. and Lingard A. (2009) "A conservation auction for landscape linkage" *The Rangeland Journal,* 31, 127-135.

Wuepper, David, and Robert Huber. (2022). “Comparing effectiveness and return on investment of action- and results-based agri-environmental payments in Switzerland.” *American Journal of Agricultural Economics* 104(5): 1585–1604. https://doi.org/10.1111/ajae.12284

Wunder S., Borner J., Ezzine-de-Blas D, Feder S., and Pagiola S. (2020) “Payments for Environmental Services: Past Performance and Pending Potentials” *Annual Review of Resource Economics* Vol. 12:209-234 https://doi.org/10.1146/annurev-resource-100518-094206