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How should we incentivize private landowners to 'produce' more biodiversity?

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Abstract Globally, much biodiversity is found on private land. Acting to conserve such biodiversity thus requires the design of policies which influence the decision-making of farmers and foresters. In this paper, we outline the economic characteristics of this problem, before reviewing a number of policy options, such as conservation auctions and conservation easements. We then discuss a number of policy design problems, such as the need for spatial coordination and the choice between paying for outcomes rather than actions, before summarizing what the evidence and theory developed to date tell us about those aspects of biodiversity policy design which need careful attention from policy-makers and environmental regulators.

Key words: biodiversity, economic instruments, payments for ecosystem services, conservation auctions, agglomeration bonus, conservation easements

JEL classification: Q57, Q58, Q24

I. Introduction

Much biodiversity is found on privately owned land. For example, in the UK agricultural land provides important habitats for a wide range of birds and insects (UK NEA, 2011). In the US, privately owned lands contain at least one population of two-thirds of all species listed as being federally endangered (Groves *et al*., 2000). Privately owned forest land in Finland and Poland contains many Natura 2000 sites,[1] a designation which is indicative of high conservation values (Watzold *et al*., 2010).

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[1] In May 1992 European Union governments adopted legislation designed to protect the most seriously threatened habitats and species across Europe. This legislation is called the Habitats Directive, and complements the Birds Directive adopted in 1979. At the heart of implementing these directives is the creation of a network of protected sites known as Natura 2000.

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The way in which private lands are managed therefore has major implications for biodiversity. In Australia, conservation of many endangered native species depends on changing the behaviour of private farmland owners (Reeson *et al*., 2011*a*), while plant species richness in privately owned Austrian hay meadows has been shown to decline with increasing agricultural intensity (Zeckmeister *et al*., 2003). Changes in how agricultural land is managed have had significant impacts historically on a range of biodiversity indicators in the UK (Hanley *et al*., 2009), with the twentieth century, in particular, being associated with declines of many species groups on farmland (Wilson *et al*., 2009). Agricultural land management continues to impact biodiversity. For example, according to the UK National Ecosystem Assessment (NEA) 'recent evidence suggests that about 67% of 333 farmland species (broadleaved plants, butterflies, bumblebees, birds and mammals) were threatened by agricultural intensification in the year 2000' (UK NEA, 2011, p. 65). Globally, habitat destruction and degradation associated with conversion to agricultural land and intensification of agricultural land practices are leading drivers of losses of biodiversity and ecosystem services (MEA, 2005).

These trends of biodiversity loss impose costs on society, since biodiversity plays a key role in sustaining the functioning of ecosystems, and thus in the provision of ecosystem services while individuals have been shown in many studies to be willing to pay for biodiversity conservation (Kontoleon *et al*., 2007). Yet the supply of biodiversity typically goes unrewarded by market forces owing to missing markets: private landowners usually receive no direct financial reward for enhancing or protecting biodiversity, owing to the non-rivalness and non-excludability of these benefits (Hanley *et al*., 2006). Indeed, protecting biodiversity typically comes at an opportunity cost to landowners—for example, if it requires forgoing profitable land conversion or intensification. The market thus generates too little biodiversity conservation effort, and too much biodiversity loss. For this reason, government intervention to promote biodiversity conservation on private land is warranted. Owing to political reluctance to force landowners to produce more biodiversity, and practical issues with extending the planning system to agricultural and forest land management, governments in many countries have introduced a range of schemes whereby landowners and managers can voluntarily opt to take up contracts for changing how they manage land in return for payments. In agriculture, such schemes are known as 'agri-environment schemes', or AES. Spending on AES has been rising as a fraction of total public spending on agriculture, for example in the EU and the US. The EU spends on average US$7.2 billion per year on payments to incentivize farmers to enhance environmental benefits, including biodiversity, and avoid using environmentally detrimental production techniques (Cooper *et al*., 2009). Within the UK, the largest AES has funding of around £400m per year over the period 2007–13 (Dunn, 2011). The largest scheme in the US, the Conservation Reserve Program, spends US$1.7 billion per year (USDA, 2010).

Our focus here is on the design of such agri-environment schemes. To illustrate, we draw on examples drawn from Europe, Australia, and the US. However, it is also important to recognize that AES provide a useful template for informing the design of 'payments for ecosystem service' (PES) programmes more broadly, and many of the issues that we discuss have parallels in debates about designing PES programmes in many other countries (Jack *et al*., 2008; Quintero *et al*., 2009; Chen *et al*., 2010; OECD, 2010; Sommerville *et al*., 2010).

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Economic, socio-demographic, and geographical factors impact participation in AES. Lynch and Lovell (2003) examine participation decisions in Maryland. They find that factors such as farm size and income, the types of crops grown, and whether the children in a family are expected to keep farming when grown up have a positive impact on uptake of AES. Defrancesco *et al*. (2008) present an econometric analysis of the factors affecting AES participation in Italy. They identify ease of adaptation, adequate financial compensation, positive attitudes of neighbouring farmers, as well as attitudes towards environmental conservation as increasing participation. Similar results have been found by Langpap (2004) for AES programmes in Oregon and Washington in the US.

In this paper, we first of all review the economic characteristics of the 'biodiversity policy design problem', before moving to consider a range of policy options, and a series of policy design challenges. We close by offering a classification system by which most policy options for biodiversity conservation on private land can be described in terms of their most important features from an economics viewpoint.

II. The economic characteristics of the 'biodiversity problem'

Landowners often face a cost in taking actions intended to produce biodiversity. This cost can be expected to vary, both across landowners, and for any landowner according to the 'amount' of biodiversity she/he aims to 'produce' (Armsworth *et al*., 2012, Box 1). Variation in this supply price across landowners comes from variations in opportunity costs, which may be due in turn to differences in land productivity, differences in production opportunities, differences in resources, and differences in skills. For example, Hanley *et al*. (1998*a*) found that the opportunity cost for farmers in the Shetland Isles of reducing grazing intensity to improve the ecological quality of moorland varied from £5.70 to £21.87 per sheep removed from grazing moorlands. AES

Box 1: Estimating the supply price of biodiversity improvements

Armsworth *et al*. (2012) attempt to estimate farmers' supply curves for different biodiversity outputs to overcome the hidden information problem. They surveyed 44 extensive grazing farms in the Peak District in England, collecting data on the enterprise mix on farms, input and output prices, and current subsidies being received, as well as on the abundance and richness of different bird species on the properties. The economic data on farm businesses were used to parameterize linear programming (LP) models predicting the enterprise mix that would maximize farm profits for three representative farm types in the region. The ecological data were used to predict the likely responses of different biodiversity indicators (the densities of five single species of conservation concern and two summary indicators looking across the whole bird community) to farm management actions, by fitting nonlinear regressions relating bird responses to farm management practices across the sample farms. The authors then extended the optimization models to include nonlinear constraints

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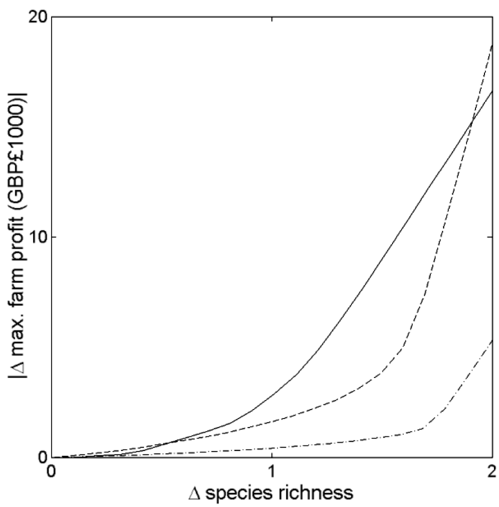
requiring a given level of improvement in each biodiversity indicator. By tracking the decrease in maximum farm profit that resulted, they were able to recover estimates of the 'true' supply price of a given level of improvement in some biodiversity indicator. An example is given in Figure 1.

Figure 1: Estimated reduction in maximum farm profit (£'000) when requiring a given improvement in species richness on three representative farm types in the Peak District, UK



The results confirmed many of the policy design challenges discussed here. Opportunity costs of the management actions associated on average with an improvement in some biodiversity target varied across farms. Marginal costs of producing these conservation benefits also increased with the level of biodiversity improvement sought on a particular farm.

Looking across the different candidate conservation targets also emphasized that the specification of the objective in AES was very important. The different species and whole bird community indicators responded in very different ways to changes to the enterprise mix, suggesting little scope for relying on a single 'umbrella' policy target that if improved would enhance biodiversity overall. A similar conclusion arises when using the models to try to understand likely ecological implications of future scenarios of policy and pricing changes (Hanley *et al.*, 2012).

schemes in which payment rates do not vary across landowners will over-compensate all but the marginal farmer if the opportunity costs of taking actions intended to produce a given level of biodiversity improvement differ across farmers. A cost-effective distribution of biodiversity supply effort will involve either the targeting of actions on low opportunity cost sites (Ando *et al.*, 1998), or the use of economic incentives which

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encourage low-cost suppliers to offer to supply biodiversity outputs, rather than high-cost suppliers (Connor *et al.*, 2008).

For a particular landowner, the marginal cost of taking actions intended to produce biodiversity may also be increasing. For example, a farmer will give up the least productive land first for a subsidized wetlands re-creation scheme, before giving up more productive land. The same principle applies to choices individuals will make over the enterprise mix on the farm—lower-cost changes that are compatible with biodiversity improvements will be made first.

Another relevant feature of the problem is that the marginal benefits of actions in terms of biodiversity ‘produced’ may also vary across landowners and respond in non-linear ways to the actions of individual farmers. For example, assume that the action needed to increase abundance of a particular bird species is to reduce livestock grazing intensity, and that this is costly for the farmer. A given reduction in grazing intensity can produce varying responses in terms of bird abundance for reasons to do with the characteristics of an individual site (e.g. its soil type, altitude, or exposure), the characteristics of neighbouring areas (e.g. the presence of woodland within 100 metres), and current grazing intensities already present on the site (Dallimer *et al.*, 2009). For species protection programmes, actions by a given landowner, for example in refraining from the felling of old growth forest, may have marginal pay-offs in terms of species recovery which vary with distance to the nearest existing population of the species. This implies that an efficient policy design would have incentives which vary across space, since the biodiversity pay-off per euro also varies; and/or that the awarding of conservation contracts would partly depend on spatially-varying ecological benefit functions (conservation metrics), such as are used in Australia for scoring bids (Oliver *et al.*, 2005; Connor *et al.*, 2008).

A third feature of the biodiversity problem with economic importance is that of hidden information (Moxey *et al.*, 1999). This is of two types. First, a regulator will typically be unsure about the cost type of individual landowners, in terms of their true marginal supply prices for biodiversity. We have already argued that variations in these supply prices across agents is to be expected. But this information is hard for the government to observe, since it depends on a wide range of landowner and land characteristics, and since there is typically a large number of farmers/landowners who are involved in the supply of biodiversity. Farmers will have private information on these supply prices—whether they are ‘high cost’ or ‘low cost’ type. Farmers also have local knowledge of their land which means they may have more information than the regulator on the likely ecological outcomes of certain actions—for instance, if they know of the existence of bird populations on their land of which the regulator is unaware. Second, AES schemes often involve land managers undertaking ‘actions’ which are hard for the government to monitor accurately. For example, if increasing populations of the bush stone-curlew in Australia requires farmers to engage in predator control, such actions are very hard (and costly) to monitor for the agency paying for these actions by way of conservation contracts. The level of effort which farmers engage in to fulfil the terms of their contracts is not known to the regulator with any precision. If this is so, then given that effort is costly to the farmer, farmers have an incentive to shirk and not undertake the actions for which they are being paid. This in turn means that the expected biodiversity benefits are not forthcoming.

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Hidden information on farmers' cost type, ecological potential, and hidden actions leads to problems of adverse selection and moral hazard, the implications of which are usually analysed within a principal–agent model (Mueller, 1989; Fraser, 2002; Ozanne and White, 2008). Anthon *et al*. (2010) model the effects of these problems on the optimal design of incentive contracts for Natura 2000 forests. In their paper, ecological benefits from landowner actions are unknown before a conservation contract is signed, and only revealed *ex post*, and vary across forests. They show that the regulator should optimally offer forest owners an amount greater than their true supply price in order to induce compliance on high ecological-potential sites. They also conclude that payments should at least partly be linked to observable ecological outcomes, rather than just the cost of actions. However, most AES schemes at present are based on actions, not outcomes.

A final feature of the biodiversity problem which is important for economic analysis is that the biodiversity benefits of a particular set of actions are stochastic from the viewpoint of the individual farmer/forest owner, since they are only partly a function of the actions of this agent. Consider the case of actions designed to increase the population of a bird species that nests and breeds on farmland. Ecologists know that certain actions that farmers can take are likely to contribute to an increase in the overall population size of this species. Such actions might include predator control, creation of small wetlands, and appropriate grassland management. But the abundance of the species on any one farm will be highly variable through time and will also respond to many factors outside the farmer's or regulator's control, including, for example, climatic variations, variations in the abundance of parasite species, variations in abundance of competing species, etc. This means that the outcome which the regulator cares about is only partly under the control of the agent charged with producing it. For risk-averse agents, this means that they face a cost of risk-bearing from non-delivery of the environmental good. This matters if an AES is set up to pay for biodiversity outcomes rather than actions (see section IV below). In such circumstances, it may be necessary to offer farmers a two-part payment, one which depends on actions, and one which depends on outcomes. In this way, the government shares the cost of risk-bearing.

III. Policy design options

(i) Regulation

Governments clearly have the option of compelling landowners to protect biodiversity on their land, for example by refraining from certain potentially damaging operations for specific sites or specific species. The former approach was followed in the Wildlife and Countryside Act in UK, while the latter is exemplified by the US Endangered Species Act. Two problems follow from such legislation. First, legislation often fails to recognize the (opportunity) costs which designation of protected species puts on landowners, and thus creates conflicts (Brown and Shogren, 1998). It also leads to incentives for landowners to take actions which downgrade sites so that they are de-listed, and thus controls removed. Thus a landowner in the US, finding a federally listed species on their land, has an incentive to destroy this species, and thus avoid the restrictions which its public discovery would place on them (Brown and Shogren, 1998). The US Endangered Species

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Act (ESA) has undergone various revisions in a bid to address some of the incentive problems created for private landowners, through, for example, the introduction of the 'no surprises' clause in habitat conservation plans and the introduction of 'safe harbour agreements' (Bean, 2000). The UK Wildlife and Countryside Act recognized that costs would occur as a result of restrictions of 'potentially damaging operations' on Sites of Special Scientific Interest, and offered to pay compensation for profits forgone from such actions. But this led to an incentive for landowners to threaten to undertake such actions, since the only means the Nature Conservancy Council had of stopping them was to offer payments, leading to a problem of moral hazard (Spash and Simpson, 1994).

Moreover, the extension of detailed regulatory control over the actions of private landowners in the countryside with respect to agricultural and forest management has not found political favour in many Western countries, owing to the nature of *de jure* and *de facto* property rights over rural land use. Thus, extensions of the planning system (for example) to cover agricultural land use are uncommon,[2] and, indeed, might be very inefficient due to variations in supply prices for biodiversity across landowners, and the likely magnitude of the administration costs of enforcing such an extension of planning rules.

(ii) Uniform payment schemes

Uniform payment schemes dominate agri-environmental policy. Farmers are offered a payment for a set of management actions which are thought to increase biodiversity. In many cases, such payments are only available within certain geographic regions of a country (e.g. the Environmentally Sensitive Areas scheme in the UK, that was the predecessor of the ES programme described in Box 2); in others they are available countrywide (e.g. the ES programme that replaced it). Uniform payments have a number of advantages. They are relatively simple to set up and to administer, and may be perceived as 'fair' since every landowner is offered the same price for undertaking a given action. Uniform payments are more cost-effective than regulation, since only those farmers with a supply price less than the subsidy will sign up. However, such schemes ignore many of the features of 'the economic problem', in that they over-reward all but the marginal producer; while usually no recognition is made of spatial variation in the supply price or variation along supply curve for a given farmer. Payment rates may or may not recognize variations in ecological potential of sites, depending on the basis on which they are calculated—for example, a calculation of average opportunity costs of complying with a set of management measures would not reflect variations in ecological potential. An improvement would be to allow spatial targeting of farms where payments are to be offered which could still be combined with uniform payments for contracts. For an illustration of how the benefits of making policy design more complex to reflect differences in supply prices and ecological potential, see Box 3.

(iii) Conservation auctions

Conservation auctions are reverse or procurement auctions, where the auctioneer—the policy-maker—procures environmental benefits such as biodiversity improvements from

[2] Although aspects of farmers' activities in UK national parks, for instance, may be regulated by planning procedures—the construction of new agricultural buildings, for instance.

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Box 2: Environmental stewardship in the UK

Environmental stewardship (ES) is the flagship AES in England. Launched in 2005, the objectives of ES include: wildlife conservation; natural resource protection; prevention of erosion and water pollution; and promotion of public access to and understanding of the countryside (DEFRA, 2005*a,b*). ES is composed of finite-length contracts in which participating landowners are compensated for augmenting their land practices to meet programme objectives. The scheme has two contract levels: Entry Level Stewardship (ELS) and Higher Level Stewardship (HLS). ELS is non-targeted and all landowners who agree to a minimum level of stewardship will be accepted. Contracts last for 5 years and landowners receive a flat-rate payment of £30 per hectare per year for all eligible land, irrespective of the management actions undertaken (DEFRA, 2005*a*). The HLS, on the other hand, is highly targeted to deliver environmental benefits to priority areas. HLS contracts run for 10 years with landowners receiving payments dependent upon the management actions undertaken (DEFRA, 2005*b*). One interesting feature of the HLS programme is that while it pays a fixed price per contract, farmers compete to secure these contracts by offering to undertake varying numbers of conservation activities. In effect, the payment per activity can vary even if the payment per contract is fixed. By 2011, approximately 70 per cent of agricultural land in England was under ES, covering 6.5m hectares of land (Natural England, 2011).

Box 3: Consequences of simplifying payment schemes

Using the estimated supply curves discussed in Box 1, Armsworth *et al.* (2012) examine the improvements in each biodiversity target offered by different simplified payment designs. They compare the performance of these simplified policies to the biodiversity enhancements offered by the optimal policy design from the perspective of the regulator, which pays farms at their opportunity costs (no producer surplus) and targets contracts towards those who can supply conservation improvements most cost effectively. They considered different policy simplifications individually and in combination, including offering open enrolment and failing to target contracts towards the most cost-effective producers; paying a fixed price for biodiversity improvements across three different regions; and purchasing improvements at the cost of the last and most expensive unit in each location. All of these policy simplifications are common in AES policies.

When acting together these different policy simplifications resulted in most of the expenditure on AES programmes simply increasing producer surplus. The reduction in biodiversity gain offered when compared to the optimal policy ranged from 49 to 100 per cent across seven different conservation targets. Of the policy simplifications examined, a failure to differentiate pricing in space was particularly problematic, a conclusion that was robust to idiosyncratic responses of different conservation targets to farm management changes and considerable uncertainty in the regressions relating bird responses to farm management actions. A failure to target contracts

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towards cost-effective producers or to reflect changing marginal costs of producing biodiversity improvements within farms only occasionally yielded comparable reductions in the biodiversity gains available.

Of course, AES programmes that avoid such policy simplifications will likely be more expensive to administer. By reversing the question to ask how much a government agency would have to pay farmers with the optimal policy to secure the same level of improvement in some biodiversity target as is offered by each simplified policy, the authors sought to place an upper bound on the extra transaction costs it would be worth taking on to implement more complicated policies. For example, looking across the seven different biodiversity indicators, it would have been worth spending 70 per cent or more of the funds that otherwise would have been given to farmers to discriminate pricing in space and to target contracts towards cost-effective producers. Importantly, achieving this type of cost effectiveness in payment rates and contract allocations is the goal of some AES designs, such as reverse auctions, currently being advocated in the literature.

a selected set of landowners. These landowners are chosen on the basis of their submitted bids which reflect their supply price. These bids are anchored from below by the opportunity costs of changing land-use management and may have institutionally fixed upper limits or 'bid caps'. An auction fosters competition between bidders to minimize the 'information rents' or profits earned by landowners and maximize the amount of ecosystem services procured for a given budget, since lower bids have more chance of being accepted. Given a fixed budget for contracts, farmers have an incentive to moderate bids if they wish to be awarded such an agreement (Stoneham *et al.*, 2003; Rolfe *et al.*, 2009).

Perhaps the most prominent conservation auction is the Conservation Reserve Program (CRP) which was started in 1985 by the US Department of Agriculture (USDA, 2011, Box 4). Under the CRP landowners' bids are ranked in descending order on the basis of a benefit–cost index, termed the Environmental Benefit Index. The benefit element of the ratio is the ecological value of the environmental benefits supplied by the project and the cost is the bid submitted (monetary values for these benefits are not computed). Use of this benefit–cost ratio discourages landowners from marking up their bids too high as this reduces chances of selection. A range of auction mechanisms, such as the BushTender (Stoneham *et al.*, 2003), Catchment Care Australia (Connor *et al.*, 2008), and the Auction for Landscape Recovery pilot (Gole *et al.*, 2005) have been employed in Australia. Brown *et al.* (2011) describe an auction in the Canadian Prairies linked to conservation easement payments.

Several design options exist for conservation auctions (Schilizzi and Latacz-Lohmann, 2007). For example, a government needs to decide whether to use a uniform price design (all successful bidders receive the same payment) or a discriminating price design (successful bidders receive their bid price). Uniform price designs can do a better job of revealing true opportunity costs, since if the price is set equal to the highest losing bid, then an individual farmer's bid only determines the chances of winning a contract, not the value of the contract. However, uniform price designs may deter participation (Brown *et al.*, 2011). In multi-round iterative auctions, participants can submit bids repeatedly in multiple rounds. In these auctions, bidders get the opportunity to revise their bids. Thus losing bidders have a chance of lowering their bids and getting accepted

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Box 4: The Conservation Reserve Programme

The Conservation Reserve Program (CRP) is the largest land retirement programme in the USA (Khanna and Ando, 2009). The programme consists of 10–15-year contracts in which eligible landowners are compensated by the state for removing sensitive cropland from production. The CRP was established in 1985 with the initial objective of reducing rates of agricultural soil erosion, but has since been expanded to focus additionally on wildlife conservation and water and air quality (Hansen and Hellerstein, 2006). In 2010, the CRP had set aside 31.2m acres of sensitive agricultural land (Hellerstein and Malcolm, 2011). The CRP operates as a discriminatory-price auction in which landowners submit bids to retire a proportion of their land. Studies of the programme suggest that after the first few rounds of the auction, the regional bid cap becomes known, because landowners' bids tend to the maximum value (Shoemaker, 1989; Khanna and Ando, 2009). This results in landowners receiving economic rents. Despite the informational problems that exist within the scheme, the CRP has been responsible for many environmental benefits. For example, the CRP has reduced soil erosion by an estimated 450m tons per year and has increased the abundance of Prairie Pothole ducks by approximately 2m per year (USDA, 2011).

in latter rounds. Schilizzi and Latacz-Lohmann (2007), Cason *et al.* (2003), Cason and Gangadharan (2004), and Rolfe *et al.* (2009) indicate that inter-temporal learning in general reduces the cost efficiency of the auctions relative to a subsidy irrespective of the ecological goal, or that there is only a very modest improvement of performance over time (Cason and Gangadharan, 2005).

The relative effectiveness of different conservation auction designs in any context hinges on political, economic, and ecological considerations. Connor *et al.* (2008) note that assessments of the performance of auctions relative to uniform payments can be undertaken by fixing either the total cost of the scheme, or the environmental success (e.g. acres enrolled), with rather different conclusions emerging about comparative performance. Moreover, changes in the design of auctions could have differing impacts on alternative criteria of policy performance. For instance, increasing information about the spatial location and characteristics of other bidders might improve ecological outcomes at the expense of higher pay-outs to farmers (Cason *et al.*, 2003). Finally, the design of the conservation metric with which bids are weighted is crucial to determining the success of auctions (Connor *et al.*, 2008).

(iv) Conservation easements

Conservation easements provide a popular policy option for securing conservation improvements on private land in many parts of the world (Environmental Law Institute, 2003; Merenlender *et al.*, 2004; Land Trust Alliance, 2011). Easements are voluntary, market-based agreements between landowners and conservation agencies in which the landowner receives a direct payment and/or tax rebate in recompense for ceding particular land rights. However, the landowner retains overall fee title to the property. Some

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easements have been used to limit development, but many others place restrictions on grazing activities, timber operations, etc. Moreover, while a popular tool for land trusts and other non-profit organizations, easements are also commonly used by government agencies to secure conservation gains. Taken together, these aspects make easements often very comparable to AES. Unsurprisingly then, when designing easements, policy-makers face many of the same challenges that are present in designing AES. For example, hidden information about the true supply price of conservation benefits on a given property make it challenging for regulators to avoid overcompensating all but the marginal landowner (Armsworth and Sanchirico, 2008). Similarly, government agencies face challenges in monitoring compliance with easement terms.

One obvious difference between easements that restrict agricultural uses and timber extraction on a property and AES is that the exchange of property rights in an easement is commonly made 'in perpetuity', whereas AES typically offer fixed-duration contracts. In general, the advantages and disadvantages of operating conservation contracts of different durations is not a well-studied area, something that we return to below. Also, often large numbers of AES contracts are being issued simultaneously in scheduled (re-)enrolment rounds. In contrast, easement transactions often occur in a more piecemeal fashion, proceeding on a deal-by-deal basis, which limits scope for relying on competitive allocation mechanisms to overcome limitations of hidden information. Despite these differences in how the two instruments are being applied, we believe that much could be learned from comparative studies contrasting experiences with AES and easements.

(v) Creating markets for biodiversity

One aspect of the 'biodiversity problem' outlined in section II is that of missing markets. Since many of the benefits which biodiversity conservation provides are non-rival and non-excludable, markets may not emerge in which buyers and sellers trade. However, government agencies sometimes enable such markets to form. For example, the US Fish and Wildlife Service has sometimes allowed trading in endangered species and their habitats under the US ESA (Bean and Dwyer, 2000; Fox and Nina-Murcia, 2005) following a cap-and-trade-type approach. Under this model, a landowner who plans to undertake land management actions that may harm individuals of a federally listed species is required to undertake compensatory mitigation to improve the plight of the species elsewhere. This could involve purchasing species conservation credits from a third-party mitigation bank that specializes in creating and restoring habitat for the species on a different site. The potential economic benefits from such a scheme are realized through the gains from trade made possible by introducing flexibility into the command-and-control regulation. Ecological benefits could also result by allowing otherwise disparate conservation actions on the landscape to be aggregated in space. Also, some species require proactive management of habitats, such as fire management, something that can be incentivized with this approach but otherwise is not covered by the US ESA. As originally framed, the US ESA prohibited private landowners from taking certain actions that would harm listed species but did not require them to undertake conservation management that would aid these species. Despite the proposed benefits of such trading schemes, it should be emphasized that designing and implementing conservation banking programmes in such a way that promised economic

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and ecological benefits are realized is a formidable policy challenge in its own right (Salzman and Ruhl, 2000).

Markets for biodiversity can also arise in the absence of a regulatory cap, for instance if private buyers can capture some of the benefits of conservation. Conservation organizations can offer conservation contracts to farmland owners, with their members benefitting from resultant conservation outcomes (more birds), an example being the Ducks Unlimited Canada scheme in prairie habitats of Alberta, Manitoba, and Saskatchewan, which offers payments to farmers for wildfowl-friendly farming practices (Banack and Hvenegaard, 2010). Numerous voluntary markets are also starting to emerge where buyers pay for the delivery of specified ecosystem services—for instance, water companies paying farmers to reduce run-off of water pollutants by changing how they manage livestock, or paying landowners for peatland restoration as a way of reducing downstream water treatment costs (Dunn, 2011). Such voluntary markets are much less abundant for biodiversity conservation, presumably because the private benefits of increases in biodiversity are lower and dispersed across greatly more beneficiaries than, say, the increase in profits to a single water company from a reduction in water treatment costs. Biodiversity improvements could be purchased by bundling them with water quality improvements. Government's role in such emerging markets may be as a facilitator, as in the setting of codes of practice (although these can also emerge from the sector without intervention), and as a regulator of trades.

IV. Policy design challenges

In this section we identify some challenges for conservation policy design.

(i) Paying for outcomes not actions

Since the objective of biodiversity policy is to increase the supply of biodiversity, an obvious question is whether payments should be targeted at outcomes (more bird species, higher species density) rather than at the management actions thought to lead to such outcomes. Most agri-environmental policy is, indeed, targeted at management actions, typically because these are thought to be easier to observe, and because the 'output' of biodiversity from a given area of land is determined by a wide range of factors, only some of which are under the control of the landowner. This means that outcome-based contracts are riskier for the landowner than action-based contracts (Whitten *et al.*, 2007). Moreover, it may be more expensive for the regulator to monitor conservation outcomes (e.g. counting birds) than management actions (e.g. whether a farmer has drained a wetland or not). However, outcome-based payments have other advantages (Gibbons *et al.*, 2011). If some of the management actions which are crucial to achieving a biodiversity target are hidden (very expensive for the government to observe), then paying for outcomes may be more efficient. Moreover, landowners and managers quite likely hold information on the best areas of land within their properties for promoting target species populations, and may have alternative options for encouraging such increases in species. Outcome-based payments encourage land managers to make use of this information to generate biodiversity conservation more efficiently than payment for actions.

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Whitten *et al*. (2007) consider the case of promoting conservation of ground-nesting birds in the Murray Catchment in Australia. From the perspective of the regulator, enhancing populations of birds such as the bush stone-curlew and brolga requires a combination of observable actions (e.g. stocking levels) and hard-to-observe actions, such as predator control and the day-to-day movement of stock. Moreover, landowners are likely to have private information on where on their land it is best to promote population increases of these birds. The authors present a theoretical model which combines an auctioned up-front payment for management actions with an *ex post* payment for conservation outcomes. They find that setting the outcome payment relatively high compared to the up-front payment is desirable, since it induces landowners with high ecological potential to enrol and to supply higher levels of conservation effort, although this is at the expense of fewer participants for a fixed budget. Whitten *et al*. then run a trial of the combined scheme with farmers in the area. Seventeen farmers made bids for contracts, with outcome-based contracts being preferred to action-based contracts. The costs of securing a given area of land enrolled was lower with outcome-based contacts, with a cost saving of around 30 per cent. Crucially, the researchers had developed a metric for measuring conservation outcomes in a relatively low-cost manner. In a similar vein, White and Sadler (2011) use a simulation modelling approach to investigate the design of a payment-for-outcomes scheme for native vegetation conservation in south-west Australia, which is based on an observable species metric for outcomes, along with observable conservation actions (fencing) which can also be rewarded.

(ii) Determining contract length and other dynamic considerations

In AES, contracts with landowners are generally finite but span a variety of durations across different programmes (Lennox and Armsworth, 2011). Perpetual easements covering agricultural land uses can to some degree be thought of as an extreme case. Contract duration discussions are particularly salient given that ecological and economic conditions relevant to AES design vary through time and future predictions about these conditions are subject to considerable uncertainty. Contract expiry can result in the loss of some or all of the ecological benefits supplied during the lifetime of the contract (Whitby, 2000).

Several theoretical studies are relevant to discussions of contract duration. Ando and Chen (2011) investigated the optimal length of conservation contracts in an analysis that incorporates enrolment and re-enrolment issues. They found that while longer contracts increase conservation benefits from any single landowner, they lead to fewer landowners being willing to re/enrol in the programme. The authors also show that contracts should be longer when the ecological benefits mature slowly and that it may be optimal not to contract at all when uncertainty surrounds likely ecological outcomes. Finally, the authors show that non-ecological characteristics are also central to optimal length of a conservation contract; optimal contracts are longer where the turnover rate of parcels enrolled in conservation programmes is high and where the average private land income is low. Lennox and Armsworth (2011) also investigated how uncertainty regarding future ecological benefits of contracts and regarding a landowner's willingness to re-enrol on contract completion interact to determine optimal contract lengths. They find that uncertainty over future re-enrolment exerts more influence on the optimal choice of contract duration and they also emphasize conditions under which a

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portfolio of contract lengths can outperform employing uniform length contracts. Finally, in related work, Gulati and Vercammen (2006) examine a different dynamic aspect of conservation contracting and consider the potential benefits of offering time-varying payment schedules to recognize the changing incentive faced by landowners as a contract progresses and ecological conditions on the property improve.

(iii) Spatial coordination

Some elements of biodiversity (e.g. species with home ranges spanning multiple properties) can be more efficiently conserved if protection is targeted towards spatially adjacent parcels. Conservation agencies are thus sometimes interested in concentrating similar land uses on spatially connected parcels rather than dispersing them at different locations on the landscape. Conservation policies intended to achieve this spatial coordination have focused on combining uniform subsidy payments, which pay for the land-use changes, with top up with bonuses when neighbouring participants have similar land uses or have connections between patches which contain biodiversity friendly habitats. Examples of such subsidies include those under the Conservation Reserve Enhancement Program (CREP) in the state of Oregon in the US, and subsidies with network bonuses in Switzerland (Mann, 2010). These polices have their economic foundations in the Agglomeration Bonus (AB), proposed by Parkhurst *et al*. (2002), Parkhurst and Shogren (2007), and Warziniack *et al*. (2007), which can incentivize spatial coordination. Communication between neighbours can produce the ecologically desirable outcomes, but may also imply a lower level of cost-effectiveness.

Given these issues, attention has been devoted to implementing spatially connected auctions which give greater weight to bids which are spatially adjacent to each other. Reeson *et al*. (2011*a*) and Windle *et al*. (2009) have experimented with such auctions, where spatial connectivity is one metric used to rank bids. A challenging proposition in the domain of spatial conservation auctions is to reduce intensified rent seeking by participants at strategic locations on the landscape. As budgets are limited, if players at strategic positions exploit their locational advantage and submit very high bids, then too few projects may be procured and spatial patterns may not be attained at all. Thus the auction achieves neither economic efficiency nor ecological effectiveness.

(iv) Transactions costs

Transaction costs faced by landowners seeking to enrol in AES have been found to deter participation. The transactions costs incurred by participants can be classified into search, negotiation, administrative, monitoring, and enforcement costs (Dahlman, 1979; Hobbs, 2004). Of these, search, negotiation, and administrative costs are *ex ante* costs incurred prior to participation (Mettepenningen *et al*., 2009). The magnitude of these costs can play an important role in influencing farmer participation. McCann and Easter (1999) and Mettepenningen *et al*. (2009, 2011) estimate the transactions costs for water pollution reducing programmes in the Minnesota River in the US and for farmers and public agencies for AES participation in different parts of the EU. A study on AES participation in the EU highlights reduced participation of farmers in Sweden and Germany owing to such costs (Falconer, 2000). Moreover complex conservation contracts with

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complicated ecological goals also increase transactions costs and discourage participation (Ollikainen *et al.*, 2008). *Ex ante* costs, such as costs of filling up forms, going to workshops, negotiation, and joint planning between neighbours, are germane to the evaluation of the Agglomeration Bonus. Parkhurst and Shogren (2007) have analysed spatial coordination of neighbours as a coordination game. In their study, non-participation is a strictly dominated strategy since the pay-offs from the Agglomeration Bonus scheme are greater than the pay-offs from business-as-usual agricultural land use. This scenario may, however, change in the presence of transactions costs of participation. Agglomeration pay-offs can be obtained if neighbours participate and choose the same action as the player. Yet if the transaction costs of participation are high enough, eligible participants may opt not to participate at all. Additionally, if farmers reason that owing to high transactions costs, their neighbours will not participate, they may not participate either.

V. Discussion and conclusions

The main policy attributes which are important in the design of any AES which emerge from the above material are summarized in Table 1. Since the government is seen as contracting with private landowners for the supply of environmental goods, we describe these in terms of the nature of the contract arrived at. These attributes are: (i) the allocation mechanism—who the potential suppliers of the good are, and how they will be chosen; (ii) contract stipulation—what is to be supplied (e.g. hectares of wetland restored; reductions in stocking rates; or an output measure such as a density increase in a species of conservation concern); (iii) contract duration—how long the contract is for; and (iv) price—what payment is offered to the farmer, and how this is determined. We also highlight two possible responsibilities for fulfilling each of these aspects of mechanism design, according to whether responsibility lies with the principal—the government or its regulatory bodies—or the agents, namely the farmers or land owners. Thus in entry-level Environmental Stewardship, a government agency sets payment levels, but the overall allocation of contracts arrived at is determined entirely by which farmers choose to enrol. In contrast, with a conservation auction, farmers take responsibility for deciding what price they will receive for their actions when

Table 1: Schematic of policy design attributes and responsibility for actions

	Principal decides	Agent decides
Allocation mechanism: who receives contracts	Conservation auctions, targeted enrolment programmes (higher-level ES in the UK)	Open enrolment programmes (e.g. entry-level ES in the UK)
Contract stipulation: what they must provide	E.g. nitrate sensitive areas in the UK	Higher-level ES in the UK where farmers choose what to offer from a menu of options
Contract duration	CRP in the US, ES in the UK, etc.	
Price	Entry-level ES in the UK. Higher-level ES pays a fixed price per contract, but farmers may offer differing numbers of actions	Reverse auctions such as CRP in the US, the BushTender programme in Australia

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formulating their bids, and the principal chooses among bids to determined which of those contracts to accept. The more responsibility that the principal takes on for setting these design parameters, then the greater the burden of information acquisition, for instance on farmers' costs and ecological benefits, it must bear. This framework also highlights one obvious gap in the current panoply of programme designs: namely, a scheme in which farmers compete in part by offering to commit to contracts of varying durations. Finally, we note that an important feature of all schemes is the state of knowledge about the ecological production function linking landowner actions to biodiversity outcomes; landowners and the government may know different things about such functions.

As is well known, government intervention in agriculture has often resulted in increases in rents for owners of the factor of production the supply of which is least elastic, namely land. Enrolment of land into conservation programmes can increase both the value of farmland as well as non-enrolled farmland (Shoemaker, 1989; Wu and Lin, 2010). The rise in enrolled land values can be attributed to the land rents (over and above the actual value of the land) accruing to participants from participation in AES. Also, since many of these conservation programmes entail retirement of land out of crop production, a rise in commodity prices following reduced agricultural supply increases the values of non-enrolled farmland as well. Finally, increased amenity benefits from enrolled lands can improve the values of adjoining developed parcels. Vukina and Wossink (2000) estimate the impact of the Dutch Nutrient Quota system and other environmental policies on land values and find them to be increasing as well.

All of the policy options discussed in this paper are based on the notion that landowners are primarily motivated by profit maximization, and that monetary incentives are required to encourage them to supply costly biodiversity benefits. In a competitive industry (such as farming), where the great majority of producers are price-takers and sell undifferentiated products, profit maximization is the strategy most likely to be consistent with long-term economic viability. We thus think profit maximization is a reasonable assumption to make for the representative farmer's motivation. This is not to dispute that other motivations are important, as summarized in the recent paper by Sheeder and Lynne (2011). They ask whether profit-maximizing is a reasonable assumption in describing the environmental behaviour of farmers when deciding whether to take up AES payments. They give examples of empirical studies showing that the assumption is reasonable, for example for conservation auctions in Australia and participation in soil conservation programmes in Maryland, but also cases where the assumption did not predict well (e.g. Chouinard *et al.* (2008) in the Pacific Northwest). We also note that several schemes operate on the basis that farmers can be persuaded to adopt conservation-friendly behaviour if simply provided with information on, for instance, grassland management techniques which promote the survival of ground-nesting birds (Beedell and Rehman, 1999, 2000). If farmers are, indeed, willing to engage in conservation-friendly action voluntarily, then it is possible that offering monetary incentives may crowd out behaviour which is so motivated.

Another variant on the basic model of offering payments to individual farmers for conservation actions is to offer payments for teams or groups of land managers to sign up, which can encourage spatial coordination of actions as well. This approach is epitomized in the Netherlands. Policy-makers there have developed schemes in which farmers work in collaboration with each other and with local, regional, and national

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agencies. By 2004, cooperative agreements existed between 10 per cent of all farmers in the Netherlands, covering 40 per cent of all agricultural land (Cooper *et al*., 2009). On a much smaller scale, the UK’s Higher Level Stewardship Scheme offers a financial incentive for group applications for a single management option, although up-take seems to be rather limited (Franks, 2011).

Moreover, the policy options considered here are frequently implemented in a decidedly second-best world. Agricultural activity continues to be heavily subsidized in many countries, and such subsidies have in the past been argued to result in an intensification of production and an expansion of the area of land under farming which resulted in species declines for fauna and flora (Bowers and Cheshire, 1983; see also references in Dallimer *et al*., 2009). Reducing or removing agricultural subsidies might thus result in an improvement in the conservation status of many farmland species, although such impacts are not likely to be uniform in direction or extent across species (Hanley *et al*., 2012). Farming also results in a range of negative externalities such as non-point nutrient pollution, which can reduce aquatic biodiversity (Dodds *et al*., 2009). Correcting such negative externalities should also be part of the portfolio of policies considered.

Finally, a cost–benefit consideration of policies to promote biodiversity conservation on private land is also helpful in thinking about both the economic efficiency of schemes such as the Conservation Reserve Progamme and Environmental Stewardship Scheme, and the design of such schemes in terms of which aspects of biodiversity (and landscape) conservation they target. Early studies showed that, on the whole, the benefits of agri-environmental schemes in the UK exceeded the costs (Hanley *et al*., 1999), but that taxpayers’ preferences for the design of such schemes were at odds with the distribution of spending (Hanley *et al*., 1998*b*). A benefits assessment for the Environmental Stewardship Scheme in England in 2010 also showed benefits to exceed costs (Natural England, 2011). Valuation studies have also focused on how the benefits of such policy interventions can be transferred across space (Colombo and Hanley, 2008), and on the sensitivity of benefits estimates to econometric procedures (Campbell *et al*., 2008).

References

Ando, A., and Chen, X. (2011), ‘Optimal Contract Lengths for Voluntary Ecosystem Service Provision with Varied Dynamic Benefit Functions’, *Conservation Letters*, **4**, 207–18.

— Camm, J., Polasky, S., and Solow, A. (1998), ‘Species Distributions, Land Values and Efficient Conservation’, *Science*, **279**, 2126–8.

Anthon, S., Garcia, S., and Stenger, A. (2010), ‘Incentive Contracts for Natura 2000 Implementation in Forest Areas’, *Environmental and Resource Economics*, **46**(3), 281–302.

Armsworth, P. R., and Sanchirico, J. N. (2008), ‘The Effectiveness of Buying Easements as a Conservation Strategy’, *Conservation Letters*, **1**, 182–9.

— Acs, S., Dallimer, M., Gaston, K. J., Hanley, N. D., and Wilson, P. (2012), ‘The Cost of Policy Simplification in Conservation Incentive Programs’, *Ecology Letters*, **15**(5), 406–14.

Banack, S. A., and Hvenegaard, G. (2010), ‘Biodiversity-friendly Farming Practices in Alberta’s Central Parkland Region’, *Human Dimensions of Wildlife*, **15**, 67–9.

Bean, M. J. (2000), ‘Strategies for Biodiversity Protection’, in B. A. Stein, L. S.Kutner, and J. S. Adams (eds), *Precious Heritage: The Status of Biodiversity in the United States*, Oxford, Oxford University Press, 255–73.

Downloaded from http://oxrep.oxfordjournals.org/ at University of Stirling Library on August 12, 2014

Bean, M. J., and Dwyer, L. E. (2000), 'Mitigation Banking as an Endangered Species Conservation Tool', *Environmental Law Reporter*, **7**, 10537–56.
Beedell, J., and Rehman, T. (1999), 'Explaining Farmers Conservation Behavior: Why do Farmers Behave as They Do?', *Journal of Environmental Management*, **57**(3), 165–76.
— (2000), 'Using Social-psychology Models to Understand Farmers' Conservation Behavior', *Journal of Rural Studies*, **16**(1), 117–27.
Bowers, J. K., and Cheshire, P. (1983), *Agriculture, the Countryside and Land Use: An Economic Critique*, London, Methuen.
Brown, G. M., and Shogren, J. (1998), 'Economics of the Endangered Species Act', *Journal of Economic Perspectives*, **12**(3), 3–20.
Brown, L., Troutt, E., Edwards, C., Gray, B., and Hu, W. (2011), 'A Uniform Price Auction for Conservation Easements in the Canadian Prairies', *Environmental and Resource Economics*, **50**, 49–60.
Campbell, D., Hutchinson, W. G., and Scarpa, R. (2008), 'Incorporating Dis-continuous Preferences into the Analysis of Discrete Choice Experiments', *Environmental and Resource Economics*, **41**, 401–17.
Cary, J. W., and Wilkinson, R. L. (1997), 'Perceived Profitability and Farmers' Conservation Behaviour', *Journal of Agricultural Economics*, **48**, 13–21.
Cason, T., and Gangadharan, L. (2004), 'Auction Design for Voluntary Conservation Programmes', *American Journal of Agricultural Economics*, **86**, 1211.
— — (2005), 'A Laboratory Comparison of Uniform and Discriminative Price Auctions for Reducing Non-point Source Pollution', *Land Economics*, **81**(1), 51.
— — Duke, C. (2003), 'A Laboratory Study of Auctions for Reducing Non-point Source Pollution', *Journal of Environmental Economics and Management*, **46**(3), 446–71.
Chen, X., Lupi, F., Vina, A., He, G. and Liu, J. (2010), 'Using Cost-effective Targeting to Enhance the Efficiency of Conservation Investments in Payments for Ecosystem Services', *Conservation Biology*, **24**, 1469–78.
Chouinard, H., Paterson, T., Wandschneider, P., and Ohler, A. (2008), 'Will Farmers Trade Profits for Stewardship?', *Land Economics*, **84**(1), 66–82.
Colombo, S., and Hanley, N. (2008), 'How Can We Reduce the Errors from Benefits Transfer? An Investigation Using the Choice Experiment Method', *Land Economics*, **84**(1), 128–47.
Connor, J., Ward, J., and Bryan, B. (2008), 'Exploring the Cost-effectiveness of Land Conservation Auctions and Payment Policies', *Australian Journal of Agricultural and Resource Economics*, **51**, 303–19.
Cooper, T., Hart, K., and Baldock, D. (2009), 'The Provision of Public Goods through Agriculture in the European Union', report prepared for DG Agriculture and Rural Development, Contract No. 30-CE-0233091/00-28, London, Institute for European Environmental Policy.
Dahlman, C. J. (1979), 'The Problem of Externality', *Journal of Law and Economics*, **22**, 141.
Dallimer, M., Tinch, D., Acs, S., Hanley, N., Southall, H. R., Gaston, K. J., and Armsworth, P. R. (2009), '100 Years of Change: Examining Agricultural Trends, Habitat Change and Stakeholder Perceptions through the 20th Century', *Journal of Applied Ecology*, **46**(2), 334–43.
DEFRA (2005*a*), *Entry Level Stewardship Handbook*, Department for Environment, Food and Rural Affairs.
— (2005*b*), *Higher Level Stewardship Handbook*, Department for Environment, Food and Rural Affairs.
Defrancesco, E., Gatto, P., Runge, F., and Trestini, S. (2008), 'Factors Affecting Farmers' Participation in Agri-environmental Measures: A Northern Italian Perspective', *Journal of Agricultural Economics*, **59**(1), 114–31.
Dodds, W. K., Bouska, W. W., Eitzmann, J. L., Pilger, T. J., Pitts, K. L., Riley, A. J., Schloesser, J. T., Thornbrugh, D. J. (2009), 'Eutrophication of US Freshwaters: Analysis of Potential Economic Damages', *Environmental Science and Technology*, **43**, 12–19.
Dunn, H. (2011), 'Payments for Ecosystem Services', *Evidence and Analysis Series Paper 4*, London, Department for Environment, Food and Rural Affairs.
Environmental Law Institute (2003), *Legal Tools and Incentives for Private Lands Conservation in Latin America*, Washington, DC, Environmental Law Institute.

Downloaded from http://oxrep.oxfordjournals.org/ at University of Stirling Library on August 12, 2014

Falconer, K. (2000), ‘Farm Level Constraints on Agri-environmental Scheme Participation: A Transactional Perspective’, *Journal of Rural Studies*, **16**(3), 379.

Fox, J., and Nino-Murcia, A. (2005), ‘Status of Species Conservation Banking in the United States’, *Conservation Biology*, **19**, 996–1007.

Franks, J. R. (2011), ‘The Collective Provision of Environmental Goods: A Discussion of Contractual Issues’, *Journal of Environmental Planning and Management*, **54**, 637–60.

Fraser, I. (2002), ‘Moral Hazard and Risk Management in Agri-environmental Policy’, *Journal of Agricultural Economics*, **53**(3), 475–88.

Gibbons, J., Nicholson, E., Milner-Gulland, E. J., and Jones, J. (2011), ‘Should Payments for Biodiversity Conservation be Based on Action or Results?’, *Journal of Applied Ecology*, **48**, 1218–26.

Gole, C., Burton, M., Williams, K.J., Clayton, F., Faith, D. P., White, B., Huggett, A., and Margules, C. (2005), *Auction for Landscape Recovery: ID 21 Final Report, September 2005*, Commonwealth Market Based Instruments program, WWF Australia, available from http://www.napswq.gov.au/mbi/round1/project21.html (accessed 18 August 2006).

Groves, C. R., Kutner, L. S., Stoms, D. M., Murray, M. P., Scott, J. M., Schafale, M., Weakley, A. S., and Pressey, R. L. (2000), ‘Owning Up to Our Responsibilities: Who Owns Lands Important for Biodiversity?’, in B. A.Stein, L. S. Kutner, and J. S. Adams (eds),*Precious Heritage: The Status of Biodiversity in the United States*, Oxford, Oxford University Press, 275–300.

Gulati, S., and Vercammen, J. (2006), ‘Time Inconsistent Resource Conservation Contracts’, *Journal of Environmental Economics and Management*, **52**, 454–68.

Hanley, N., Shogren, J., and White, B. (2006), *Environmental Economics in Theory and Practice*, 2nd edn, Basingstoke, Palgrave MacMillan.

— Whitby, M., and Simpson, I. (1999), ‘Assessing the Success of Agri-environmental Policy in the UK’, *Land Use Policy*, **16**(2), 67–80.

— Kirkpatrick, H., Oglethorpe, D., and Simpson, I. (1998*a*), ‘Paying for Public Goods from Agriculture: An Application of the Provider Gets Principle to Moorland Conservation in Shetland’, *Land Economics*, **74**(1), 102–13.

— MacMillan, D., Wright, R. E., Bullock, C., Simpson, I., Parsisson, D., and Crabtree, B. (1998*b*), ‘Contingent Valuation versus Choice Experiments: Estimating the Benefits of Environmentally Sensitive Areas in Scotland’, *Journal of Agricultural Economics*, **49**(1), 1–15.

— Tinch, D., Angelopoulos, K., Davies, A., Barbier, E., and Watson, F. (2009), ‘What Drives Long-run Biodiversity Change? New Insights from Combining Economics, Paleoecology and Environmental History’, *Journal of Environmental Economics and Management*, **57**(1), 5–20.

— Acs, S., Dallimer, M., Graves, A., Morris, J., Gaston, K. J., and Armsworth, P. R. (2012), ‘Farm-scale Ecological and Economic Impacts of Agricultural Change in the Uplands’, *Land Use Policy*, **29**, 587–97.

Hansen, L., and Hellerstein, D. (2006), *Better Targeting, Better Outcomes, Economic Brief Number 2*, Washington, DC, US Department of Agriculture, Economic Research Service.

Hellerstein, D., and Malcolm, S. (2011), *The Influence of Rising Commodity Prices on the Conservation Reserve Program, ERR-110*, US Department of Agriculture, Economic Research Service, February.

Hobbs, J. (2004), ‘Markets in Metamorphosis: The Rise and Fall of Policy Institutions’, in G. Van Huylenbroeck, W. Verbeke, and L. Lauwers (eds), *Role of Institutions in Rural Policies and Agricultural Markets*, Amsterdam, Elsevier.

Jack, B., Kousky, C., and Sims, K. (2008), ‘Designing Payments for Ecosystem Services: Lessons from Previous Experience with Incentive-based Mechanisms’, *Proceedings of the National Academy of Sciences of the United States of America*, **105**, 9465–70.

Khanna, M., and Ando, A. W. (2009), ‘Science, Economics and the Design of Agriculture Programmes in the US’, *Journal of Environmental Planning and Management*, **52**, 575–92.

Kontoleon, A., Pascual, U., and Swanson, T. (2007), *Biodiversity Economics*, Cambridge, Cambridge University Press.

Land Trust Alliance (2011), *2010 National Land Trust Census Report: A Look at Voluntary Land Conservation in America*, Washington, DC, Land Trust Alliance.

Langpap, C. (2004), ‘Conservation Incentives Programs for Endangered Species: An Analysis of Landowner Participation’, *Land Economics*, **80**(3), 375–88.

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Lennox, G. D., and Armsworth, P. R. (2011), 'The Suitability of Short or Long Conservation Contracts under Ecological and Socio-economic Uncertainty', *Ecological Modelling*, **222**, 2856–66.

Lynch, L., and Lovell, S. J. (2003), 'Combining Spatial and Survey Data to Explain Participation in Agricultural Land Preservation Programs', *Land Economics*, **79**(2), 259–76.

McCann, L., and Easter, K. W. (1999), 'Transaction Costs of Policies to Reduce Agricultural Phosphorous Pollution in the Minnesota River', *Land Economics*, **75**(3), 402.

Mann, S. (2010), 'Eine Schwachstellenanalyse der Öko-Qualitätsverordnung', *Agrarforschung Schweiz*, **1**(1), 24–9.

MEA (2005), *Ecosystems and Human Well-being: Synthesis, Millennium Ecosystem Assessment*, Washington, DC, Island Press.

Merenlender, A. M., Huntsinger L., Guit, G., Fairfax, S. K. (2004), 'Land Trusts and Conservation Easements: Who is Conserving What for Whom?', *Conservation Biology*, **18**, 65–75.

Mettepenningen, E., Beckmann, V., and Eggers, J. (2011), 'Public Transaction Costs of Agri-environmental Schemes and their Determinants—Analyzing Stakeholder Involvement and Perceptions', *Ecological Economics*, **70**.

— Verspecht, A., and Huylenbroeck, G. V. (2009), 'Measuring the Private Transaction Costs of European Agri-environmental Schemes', *Journal of Environmental Planning and Management*, **52**(5), 649.

Moxey, A., White, B., and Ozanne, A. (1999), 'Efficient Contract Design for Agri-environment Policy', *Journal of Agricultural Economics*, **50**, 187–202.

Mueller, D. (1989), *Public Choice*, Cambridge, Cambridge University Press.

Natural England (2011), *Look After Your Land with Environmental Stewardship*, accessed at http://naturalengland.etraderstores.com/NaturalEnglandShop/NE124

OECD (2010), *Paying for Biodiversity: Enhancing the Cost-effectiveness of Payments for Ecosystem Services*, Paris, Organization for Economic Cooperation and Development.

Oliver, I., Ede, A., Hawes, W., and Grieve, A. (2005), 'The NSW Environmental Services Scheme: Results for the Biodiversity Benefits Index, Lessons Learned, and the Way Forward', *Ecological Management and Restoration*, **6**, 197–205.

Ollikainen, M., Lanoski, J., and Nuutinen, S. (2008), 'Policy Related Transaction Costs of Agricultural Policies in Finland', *Agricultural and Food Science*, **17**, 193.

Ozanne, A., and White, B. (2008), 'Hidden Action, Risk Aversion and Variable Fines in Agri-environmental Schemes', *Australian Journal of Agricultural and Resource Economics*, **52**, 203–12.

Parkhurst, G. M., and Shogren, J. F. (2007), 'Spatial Incentives to Coordinate Contiguous Habitat', *Ecological Economics*, **64**(2), 344.

— Bastian, C., Kivi, P., Donner, J., and Smith, R. B. W. (2002), 'Agglomeration Bonus: An Incentive Mechanism to Reunite Fragmented Habitat for Biodiversity Conservation', *Ecological Economics*, **41**(2), 305–28.

Quintero, M., Wunder, S., and Estrada, R. D. (2009), 'For Services Rendered? Modeling Hydrology and Livelihoods in Andean Payments for Environmental Services Schemes', *Forest Ecology and Management*, **258**, 1871–80.

Reeson, A., Rodriguez, L., Whitten, S., Williams, K., Nolles, K., Windle, J., and Rolfe, J. (2011*a*), 'Adapting Auctions for the Provision of Ecosystem Services', *Ecological Economics*, **70**, 1621–7.

— Williams, K., and Whitten, S. (2011*b*), 'Targetting Enhanced Spatial Configuration in Biodiversity Conservation Incentive Payment Programs', paper presented to BioEcon conference, Geneva, September.

Rolfe, J., Windle, J., and McCosker, J. (2009), 'Testing and Implementing the Use of Multiple Bidding Rounds in Conservation Auctions: A Case Study Application', *Canadian Journal of Agricultural Economics*, **57**(3), 287–303.

Salzman, J., and Ruhl, J. B. (2000), 'Currencies and Commodification of Environmental Law', *Stanford Law Review*, **53**, 607–94.

Schilizzi, S., and Latacz-Lohmann, U. (2007), 'Assessing the Performance of Conservation Auctions: An Experimental Study', *Land Economics*, **83**(4), 497.

Sheeder, R. J., and Lynne, G. D. (2011), 'Empathy-conditioned Conservation', *Land Economics*, **87**(3), 433–52.

Downloaded from http://oxrep.oxfordjournals.org/ at University of Stirling Library on August 12, 2014

Shoemaker, R. (1989), ‘Agricultural Land Values and Rents Under the Conservation Reserve Program’, *Land Economics*, **65**(2).

Sommerville, M., Jones, J. P. G., Rahajaharison, M., and Milner-Gulland, E. J. (2010), ‘The Role of Fairness and Benefit Distribution in Community-based Payment for Environmental Services Interventions: A Case Study from Menabe, Madagascar’, *Ecological Economics*, **69**, 1262–71.

Spash, C., and Simpson, I. (1994), ‘Utilitarian and Rights-based Motives for Protected Sites of Special Scientific Interest’, *Journal of Agricultural Economics*, **45**, 15–26.

Stoneham, G., Chaudhri, V., Ha, A., and Strappazzon, L. (2003), ‘Auctions for Conservation Contracts: An Empirical Examination of Victoria’s BushTender Trial’, *Australian Journal of Agricultural and Resource Economics*, **47**, 477–500.

UK NEA (2011), *The UK National Ecosystems Assessment*, available at http://uknea.unep-wcmc.org/, last accessed 16 February 2012.

USDA (2010), ‘Monthly Active Conservation Reserve Program Contract Reports’, US Department of Agriculture, available at https://arcticocean.sc.egov.usda.gov/CRPReport/ monthly_report.do?method=selectMonthlyReport&report=November-2010

— (2011), ‘Conservation Reserve Program’, US Department of Agriculture, Farm Service Agency, available at http://www.fsa.usda.gov/FSA/webapp?area=home&subject=copr&topic=crp

Vukina, T., and Wossink, A. (2000), ‘Environmental Policies and Agricultural Land Values: Evidence from the Dutch Nutrient Quota System’, *Land Economics*, **76**(3), 413–29.

Warziniack, T., Shogren, J. F., and Parkhurst, G. (2007), ‘Creating Contiguous Forest Habitat: An Experimental Examination on Incentives and Communication’, *Journal of Forest Economics*, **13**(2–3), 191–207.

Watzold, F., Mewes, M., Apeldoorn, R., Varjopuro, R., Chmielewski, T., Veeneklaas, F., and Kosloa, M. (2010), ‘Cost-effectiveness of Managing Natura 2000 Sites: An Exploratory Study for Finland, Germany, the Netherlands and Poland’, *Biodiversity Conservation*, **19**(7), 2053–69.

Whitby, M. (2000), ‘Challenges and Options for the Agri-environment: Presidential Address’, *Journal of Agricultural Economics* **51**, 317–332.

White, B., and Sadler, R. (2011), ‘Optimal Conservation Investment for a Biodiversity-rich Agricultural Landscape’, *Australian Journal of Agricultural and Resource Economics*, **56**, 1–21.

Whitten, S., Goddard, R., Knight, A., Reeson, A., and Stevens, D. (2007), ‘Designing and Testing an Outcome Focused Conservation Auction: Evidence from a Field Trial Targeting Ground Nesting Birds’, Discussion Paper, CSIRO Sustainable Ecosystems Group.

Wilson, J. D., Evans, A. D., and Grice, P. V. (2009), *Bird Conservation and Agriculture*, Cambridge, Cambridge University Press.

Windle, J., Rolfe, J., McCosker, J., and Lingard, A. (2009), ‘A Conservation Auction for Landscape Linkage in the Southern Desert Uplands, Queensland’, *The Rangeland Journal*, **31**, 127–35.

Wu, J., and Lin, H. (2010), ‘The Effect of the Conservation Reserve Program on Land Values’, *Land Economics*, **86**(1), 1–21.

Zeckmeister, H., Scmitzberger, I., Steurer, B., Peterseil, J., and Wrbka, T. (2003), ‘The Influence of Land Use Practices and Economics on Plant Species Richness in Meadows’, *Biological Conservation*, **114**, 165–77.

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