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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 need for spatial coordination and the choice between paying for outcomes rather than actions, before summarizing what the evidence and theory developed to date tells us about those aspects of biodiversity policy design which need careful attention from policy makers and environmental regulators.

JEL codes: Q57, Q58, Q24.

Keywords :biodiversity; policy design; conservation auctions; agglomeration bonus.

1. 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, a designation which is indicative of high conservation values (Watzold et al, 2010).

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 behavior of private farmland owners (Reeson et al, 2011), whilst plant species richness in privately-owned Austrian haymeadows 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 20th 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 “...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, p65). Globally, habitat destruction and degradation associated with conversion to agricultural land and intensification of agricultural land practises 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 (Rafaelli, 2006), whilst individuals have been shown in many studies to be willing to pay for biodiversity conservation (Kontoleon, Pascual and Swanson, 2008). Yet the supply of biodiversity typically goes un-rewarded by market forces due to missing markets: private landowners usually receive no direct financial reward for enhancing or protecting biodiversity, due to the non-rivalness and non-excludability of these benefits (Hanley, Shogren and White, 2006). Indeed, protecting biodiversity typically comes at an opportunity cost to landowners, for example if it requires foregoing 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. Due 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 USD\$7.2 billion per year on payments to incentivise 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 £400 million per year over the period 2007-2013 (Dunn, 2011). The largest scheme in the US, the Conservation Reserve Program, spends USD\$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 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) programs more broadly and many of the issues that we discuss have parallels in debates about designing PES programs in many other countries (Jack *et al.* 2008; Quintero *et al.* 2009; Chen *et al.* 2010; Somerville *et al.* 2010).

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.

2. 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) 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 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 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 subsidised 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 nonlinear 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 (eg its soil type, altitude, or exposure), the characteristics of neighbouring areas (eg 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 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; Reeson et al, 2011).

A third feature of the biodiversity problem with economic importance is that of hidden information. 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 are 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) for the agency paying for these actions by way of conservation contracts to monitor. 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.

Box 1: Estimating the supply price of biodiversity improvements

Armsworth et al. (2012) attempt to estimate farmer's 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 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 illustrated in the following figure

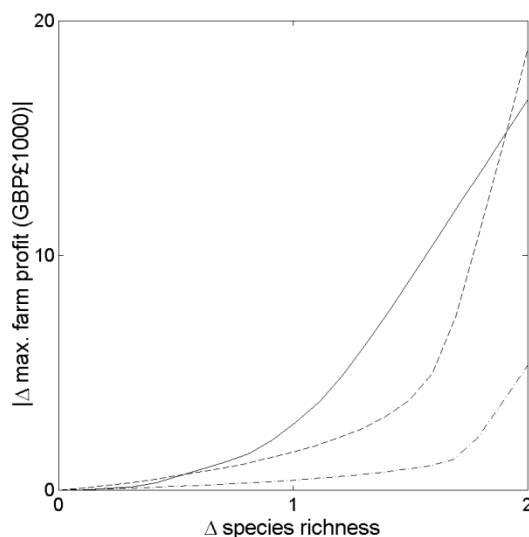


FIGURE CAPTION

Estimated reduction in maximum farm profit (GBP £1000) 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).

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, 2007). 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.

Box 2 – Environmental Stewardship in the UK

Environmental Stewardship (ES) is the flagship agri-environment scheme 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 2005a, 2005b). ES is composed of finite length contracts in which participating landowners are compensated to augment 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 2005a). 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 2005b). One interesting feature of the HLS program 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% of agricultural land in England was under Environmental Stewardship, covering 6.5 million hectares of land (Nature England, 2011).

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

4 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.

3. POLICY DESIGN OPTIONS

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, whilst the latter is exemplified by the US Endangered Species Act. Two problems follow from such legislation. First, legislation often fails to recognise the (opportunity) costs which designation of protected species puts on landowners, and thus creates conflicts (Shogren et al, 1998). It also leads to incentives for landowners to take actions which down-grade 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 Act 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 introduction of “safe harbour agreements” (Bean 2000). The UK Wildlife and Countryside Act recognised 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 foregone 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, 1993).

Moreover, the extension of detailed regulatory control over the actions of private land-owners in the countryside with respect to agricultural and forest management has not found political favour in many Western countries, due 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 un-common¹, 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.

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 (eg the Environmentally Sensitive Areas scheme in the UK that was the predecessor of the ES program described in Box 2); in others they are available country-wide (eg the ES program 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 supply price less than 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; whilst 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 recognise 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

¹ Although aspects of farmer’s activities in UK national parks, for instance, may be regulated by planning procedures: the construction of new agricultural buildings, for instance.

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.

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 programs simply increasing producer surplus. The reduction in biodiversity gain offered when compared to the optimal policy ranged from 49-100% 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 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 programs 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% 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, like reverse auctions, currently being advocated in the literature.

Conservation Auctions

Conservation auctions are reverse or procurement auctions, where the auctioneer - the policy maker - procures environmental benefits like biodiversity improvements from 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 like the Bush Tender (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. (2010) provide details of a reverse auction for forest conservation in Tasmania, whilst Brown et al (2011) describe an auction in the Canadian Prairies linked to conservation easement payments.

Several design options exist for conservation auctions (Latacz-Lohmann and Schillizzi, 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 in latter rounds. Schillizzi 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).

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 to 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 31.2 million 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 program suggest that after the first few rounds of the auction, the regional bid cap becomes known, because landowners' bids tend to the maximum value (Khanna and Ando, 2009; Shoemaker, 1989). 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 450 million tons per year and has increased the abundance of prairie pothole ducks by approximately 2 million per year (USDA FSA 2011).

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 (eg 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).

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 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 nonprofits, 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.

Creating markets for biodiversity

One aspect of the “biodiversity problem” outlined in section 2 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 have 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 programs in such a way that promised economic 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 organisations 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.

4. POLICY DESIGN CHALLENGES

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

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 (eg counting birds) than management actions (eg 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.

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 require a combination of observable actions (eg 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 enroll 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. 17 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 contracts, with a cost saving of around 30%. Crucially, the researchers had developed a metric for measuring conservation outcomes in a relatively low cost manner.

Determining contract length and other dynamic considerations

In AES, contracts with landowners are generally finite but span a variety of durations across different programs (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.

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 reenrolment 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 programs 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 landowners' willingness to reenrol 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 also emphasize conditions under which a 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.

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 neighboring 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 policies have their economic foundations in the Agglomeration Bonus (AB) proposed by Parkhurst et al. (2002, 2007) and Warziniack et al. (2007) which can incentivize spatial coordination. Communication between neighbors can produce the ecologically desirable outcomes, but may also imply a lower level of cost-effectiveness.

Given these issues, attention has been devoted to implement spatially connected auctions which give greater weight to bids which are spatially adjacent to each other. Reeson et al. (2011) and Rolfe 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.

Transactions Costs

Transaction costs faced by landowners seeking to enrol in AES has 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 (Mettenpenningen 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 programs 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 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 payoffs from the Agglomeration Bonus scheme are greater than the payoffs from business as usual agricultural land use. This scenario may however change in the presence of transactions costs of participation. Agglomeration payoffs 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 to not participate at all. Additionally if farmers reason that owing to high transactions costs, their neighbours will not participate, they may not participate either.

5.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 are the potential suppliers of the good, and how will they be chosen (ii) contract stipulation – what is to be supplied (eg hectares of wetland restored; reductions in stocking rates; or an output measure like a density increase in a species of conservation concern); (iii) contract duration – how long is the contract for; and (iv) price – what payment is offered to the farmer, and how is this 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 whether it lies with 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 what farmers choose to enroll. In contrast, with a conservation auction, farmers take responsibility for deciding what price they will receive for their actions when formulating their bids, and the Principal chooses among bids to determine which of those contracts to accept. The more responsibility that the Principle 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 program 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.

Table 1 – schematic of policy design attributes and responsibility for actions

	Principle decides	Agent decides
Allocation mechanism: who receives contracts	Conservation auctions, targeted enrolment programs (higher level ES in the UK)	Open enrolment programs (e.g. Entry Level ES in the UK)
Contract stipulation: what they must provide	Eg Nitrate Sensitive Areas in the UK	Higher Level ES in the UK where farmers choose from among a menu of options what to offer.
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 like CRP in the US, the BushTender program in Australia

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 programs 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 programs 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 un-differentiated 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. This is not to dispute that other motivations are important, as summarized in the recent paper by Sheeder and Lynn (2011). They ask whether profit maximizing is a reasonable assumption in describing the environmental behaviour of farmers, in 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 (eg Chouinard et al, 2008 in the Pacific NW). We also note that several schemes operate on the basis that farmers can be persuaded to adopt conservation-friendly behavior 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 behavior 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 agencies. By 2004, cooperative agreements existed between 10 per cent of all farmers in the Netherlands, cover 40% 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 very 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 Programme 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, 1998b). A benefits assessment for the Environmental Stewardship Scheme in England in 2010 also showed benefits to exceed costs (Natural England, 2010). 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, 2009).

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