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Alternative Payment Approaches for Biodiversity Conservation in Agriculture

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Comments are welcome and may be sent to tad.contact@oecd.org.

Abstract

ALTERNATIVE PAYMENT APPROACHES FOR BIODIVERSITY CONSERVATION IN AGRICULTURE

Jussi Lankoski, OECD

Biodiversity conservation and sustainable use policies implemented in OECD countries could be made more environmentally effective and cost-effective. Several policy innovations could change this, however. To test this, a theoretical framework was developed to describe farmers' participation in government payment programmes that enhance semi-natural wildlife habitats on farmland. The types of payments analysed here include: uniform payments; three types of conservation auctions with environmental targeting; uniform payment with environmental targeting; and two types of differentiated payments with environmental targeting. Quantitative results show that uniform payments are less efficient than other payment types, and that auctions with environmental targeting are the most cost-effective option. However, if farmers have knowledge of the environmental value of their offer, the cost-effectiveness of auctions decreases because they tend to increase their bids to benefit from this information rent (overcompensating income forgone). Adding environmental targeting to the uniform payment policy greatly improves the cost-effectiveness of uniform payment. The analysis clearly shows that, when targeted payments are implemented, the gains from environmental targeting are large and exceed the increase in policy-related transaction costs.

Key words: Conservation auction, uniform payment, differentiated payment, policy-related transaction costs, targeting

JEL codes: Q57, Q58

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EXECUTIVE SUMMARY

Biodiversity conservation and sustainable use policies implemented in OECD countries could be made more environmentally effective and cost-effective. Biodiversity objectives can be addressed through several types of policy instruments ranging from environmental regulations to voluntary payment approaches. Countries have offered payments to farmers to promote farmland biodiversity conservation beyond conservation efforts required by environmental regulations or environmental cross-compliance. These voluntary payment programmes usually aim at compensating income foregone for adoption of biodiversity enhancing practices and are based on fixed uniform payments. These payments do not reflect the heterogeneity in farmers' compliance costs and/or the supplied biodiversity benefits, which reduce both environmental effectiveness and the cost-effectiveness of the policy. Indeed, empirical evidence shows that the environmental effectiveness of these payments varies greatly and that many of them have failed to achieve stated biodiversity objectives.

Due to spatial variation in conservation costs and benefits, cost-effectiveness requires spatially differentiated payment mechanisms. The cost-effectiveness gains from *differentiated payments* relative to *uniform payments* increases when spatial heterogeneity increases. There are several payment mechanisms available to address spatial heterogeneity, and thus to improve both environmental effectiveness and cost-effectiveness of the payment programme. These payment mechanisms include spatially differentiated compensation payments and conservation auctions (farmers bid competitively for a limited number of conservation contracts).

However, as the costs of implementing these more effective mechanisms are higher than those of uniform payments, there seems to be a trade-off between gains from spatial targeting and keeping policy-related transaction costs (PRTCs) low. But, potentially higher PRTCs should be put in perspective with the gains from targeting, including better environmental performance and reduced transfers – that is, when the payment level is tailored to match compliance costs and unintended transfers are reduced.

A theoretical framework was developed in order to analyse further in detail what is key to determine the cost-effectiveness of different payment types. The framework analyses farmers' participation in government agri-environmental payment programmes to enhance semi-natural wildlife habitats on farmland through the establishment of biodiversity strips located between field parcel and forest border. Payment types analysed include a uniform payment, three auction design alternatives with environmental targeting, a uniform payment with environmental targeting and two types of differentiated payments with environmental targeting. The theoretical framework was applied to data from Finland.

Quantitative results provide a ranking of payment mechanisms according to cost-effectiveness which is consistent with literature, with uniform payments the least cost-effective and auctions most cost-effective. The results also show that auction performance is affected by the amount of information provided to farmers regarding the environmental score of their offer (for example, environmental benefit index value of their field parcel). Informing farmers about the environmental score of their offer tends to reduce auction performance because farmers tend to increase their bids if informed that the environmental score of their offer is higher than average. Thus, farmers whose offers have higher than average environmental score will use this information to gain information rent (overcompensation of

income forgone). Adding environmental targeting (e.g. by employing environmental benefit index) to the uniform payment policy greatly improves the cost-effectiveness of uniform payments.

Both the base case analysis and sensitivity analysis show that, when targeted payment types are implemented, the gains from environmental targeting are very large and clearly exceed the increase in PRTCs. Cost-effectiveness gains from environmental targeting clearly outweigh increase in PRTCs even when PRTCs are increased by 100%. Thus, concerns about PRTCs of targeted payments may be exaggerated.

A uniform payment with environmental targeting may be a good option in a situation where a government aims to improve environmental effectiveness and cost-effectiveness of the payment programme with relatively small additional expenditure on administrative costs (PRTCs). Introducing environmental targeting provides the highest return on the public transaction costs involved in uniform payments with environmental targeting, in which case one EUR spent on PRTCs for environmental targeting pays back EUR 28 through improved cost-effectiveness.

The empirical application also shows that the cost-effective policy design to address heterogeneous agricultural and environmental conditions requires a combination of differentiated payment levels and environmental targeting. This can be achieved, for example, by employing an auction with environmental targeting. Large efficiency gains can also be achieved by using environmental targeting alone.

It should be noted that when environmental regulations, environmental cross-compliance schemes, and farm income support schemes are already in place, the potential cost-effectiveness gains from targeted and differentiated payments differ from a situation where they would be implemented in isolation. Thus, in addition to the design and implementation details of the targeted payments themselves, the policy package as a whole also matters significantly for their efficiency.

Key policy recommendations:

- With many OECD countries using uniform payments for biodiversity protection and enhancement, environmental targeting would be a relatively straightforward way to improve the environmental effectiveness and budgetary cost-effectiveness of the payment mechanism.
- If reliable environmental data is available, countries could achieve even greater environmental benefits from limited budgets with the use of auction mechanisms combined with environmental targeting, provided that these are well designed, as explained in this document.
- To ensure the efficiency of these payment mechanisms, decision-makers should first consider reforming existing policy measures which interfere with the efficiency of the payments (e.g. agricultural income support payments, such as crop area payments that increase the opportunity costs of allocating land to biodiversity-enhancing uses).

1. Introduction

As a major user of land in many countries agriculture has had, and continues to have, a large impact on biodiversity. Biodiversity and agriculture are closely interconnected. Agriculture both provides and relies upon ecosystem services. Ecological functions and ecosystem services, such as soil structure and fertility maintenance, nutrient cycling, pollination, and the regulation of pests and diseases are vital for crop production. On the other hand, traditional agricultural practices have created diverse semi-natural habitats, such as extensive pastures and meadows, whose species depend on their existence and on the continuation of certain beneficial agricultural practices, such as low intensity grazing and traditional haymaking. But the modernisation of agriculture through intensification, concentration and specialisation can contribute to a loss of both semi-natural habitats and species abundance and richness in agricultural landscapes.

There is potential for agriculture, with appropriate policies, to maintain and even enhance biodiversity, since many semi-natural habitats are dependent on the continuation of certain agricultural land uses and cultivation practices that can be promoted, including through well-designed policy interventions. Agrobiodiversity objectives can be addressed through several types of policy instruments or through the creation of environmental markets. The tool-box of policy instruments includes, for example, environmental regulations, environmental cross-compliance, voluntary agri-environmental payments, payments for environmental services, conservation auctions and biodiversity offsets.

Many OECD countries offer payments to farmers to promote farmland biodiversity conservation beyond conservation efforts required by environmental regulations or environmental cross-compliance. Environmental regulations and environmental cross-compliance often form the basis of environmental protection while voluntary payments are offered to farmers for additional environmental efforts. In most cases the payment is compensation for income forgone and extra costs incurred from environmental practice adoption. Many of these voluntary payment programmes offer a single, fixed payment for compliance with a predetermined set of biodiversity enhancing practices. The obvious problem with such an approach is that the heterogeneity in farmers' opportunity costs and/or the supplied biodiversity benefits is not taken into account in policy design and implementation, which reduces the cost-effectiveness of these payments. Indeed, empirical evidence shows that many of them have failed to achieve stated biodiversity objectives cost-effectively (Kleijn et al., 2001).

Hence, there is a need to identify more cost-effective ways to implement these policies. To this end, several types of payment mechanisms have been proposed in the academic literature including spatially heterogeneous compensation payments, agglomeration payments and results-based payments (see e.g. Wätzold and Drechsler, 2005; Parkhurst et al., 2002; Zabel and Roe, 2009). Similarly, conservation auctions – also called green auctions – have been proposed (and implemented, e.g. the US Conservation Reserve Program) to improve the benefit-cost ratio of environmental efforts and the budgetary cost-effectiveness of government spending on biodiversity conservation and sustainable use.

The main objective of this paper is to provide a review of the various new payment approaches and policy instruments for biodiversity conservation and sustainable use in agriculture[1]. The main focus of the paper is in recent developments of various payment approaches, conservation auctions and

1. Alternative payment and auction designs reviewed in this paper fall under the commonly used term "Payment for environmental services (PES)". On the basis of economic efficiency characteristics the paper makes a distinction between practice-based and results-based payment schemes, even though they are both commonly considered PES. Regarding results-based schemes it is good to note that in many cases payment calculation is based on income foregone and extra costs incurred although the payment is triggered by quantity or quality of the environmental outcome. Social valuation (consumers' willingness to pay) of biodiversity is not reflected in the payment calculation in none of the payment approaches reviewed in this paper.

biodiversity offsets. A quantitative illustration of the environmental effectiveness and cost-effectiveness of various payment approaches and auctions is provided employing data from Finland.

The paper begins with a brief overview of the interactions between agriculture and biodiversity. Section 3 provides a literature review of the various types of payment and auction designs and biodiversity offsets. A theoretical framework is developed and then applied with Finnish data to illustrate environmental effectiveness and cost-effectiveness of the alternative payment types and auctions in Section 4.

2. Interactions between agriculture and biodiversity: A short overview

2.1. Agrobiodiversity and agricultural ecosystems

According to the Convention on Biological Diversity, "*biological diversity* means the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (UNEP, 1992). Biological diversity is commonly considered at three hierarchical levels – genes, species and ecosystems. The term *ecosystem* refers to a community of living organisms with their physical environment interacting as a system in which biotic and abiotic components are linked through processes such as the flow of energy through trophic levels and the cycling of chemical elements and compounds through living and non-living components.

Agricultural biodiversity or agrobiodiversity is a subset of biodiversity. Qualset et al. (1995) define agrobiodiversity as including all crops and livestock and their wild relatives, and all interacting species of pollinators, symbionts, pests, parasites, predators and competitors. Swift and Anderson (1994) divide the biotic components of agroecosystems (from a production viewpoint) into three types: productive, resource and destructive. *Productive* biota includes crops and livestock that provide goods for consumption. *Resource* biota contributes to the productivity of the system and includes e.g. pollinators and soil biota. *Destructive* biota includes weeds, pests and pathogens.

In agricultural systems, biodiversity supports ecosystem goods and services beyond the production of food, fiber and fuel, including recycling of nutrients, control of local microclimate, regulation of hydrological processes, regulation of undesirable organisms, and detoxification of noxious chemicals (Altieri and Nicholls, 1999). The type and abundance of biodiversity differs across agroecosystems depending on their age, diversity, structure and management (Altieri and Nicholls, 1999). Biological diversity generally depends on four main characteristics of agroecosystems (Southwood and Way, 1970): (i) the diversity of vegetation within and around agroecosystems, (ii) the permanence of the various crops within the agroecosystems, (iii) the intensity of management, and (iv) the extent of the isolation of the agroecosystem from natural vegetation. Agroecosystems that are more diverse, permanent, isolated, and managed with low intensity benefit from ecological processes associated with biodiversity more than highly simplified, external input-driven and disturbed systems (Altieri, 1995).

Vandermeer and Perfecto (1995) distinguish between planned and associated biodiversity. Planned biodiversity relates to crops and livestock and is managed by the farmer through choices regarding input use, land use and technology. Associated biodiversity includes all soil flora and fauna, herbivores, carnivores, decomposers etc. that contribute to agroecosystem functioning depending on its management and structure.

2.2. Impacts of agricultural production practices on biodiversity

Agriculture and biodiversity are closely connected with multiple interactions. Agricultural production depends on biodiversity through ecosystem services such as pollination, pest control, and nutrient cycling. Agricultural land use and production practices have both beneficial and harmful impacts on biodiversity. Low-intensity agricultural practices, such as grazing and traditional haymaking, have created diverse semi-natural habitats and landscape structures with novel species communities that are dependent on the continuation of low-intensity production practices. However, the modernisation of agriculture through intensification (e.g. increased use of fertilizers and pesticides), specialisation (reduced crop rotations and smaller amount of mixed crop-livestock farms), and rationalisation (removal of hedges, edges and other boundary habitats) has had a major impact on these semi-natural habitats and their associated species richness and abundance (Poláková et al., 2011).

The ecological processes and species associated with agricultural semi-natural habitats are affected by landscape scale factors including (Poláková et al., 2011):

- i) the spatial scale of field parcels and farming systems,
- ii) the presence and ecological quality of field boundary habitats (e.g. hedges and ditches) and other non-farmed features (e.g. trees and bonds), and
- iii) overall landscape diversity, including composition, structure and interactions with other habitats (e.g. forests and wetlands).

The spatial scale of field parcels is important as some species have requirements regarding the habitat area and its spatial configuration. For example, some birds and mammals require a large interconnected habitat area for foraging, some species rely on hedgerows and similar field boundary features for food and shelter, while some species favour open habitats without enclosing tall hedgerows or trees (Poláková et al., 2011).

Modern intensively cultivated landscapes dominated by croplands have low local diversity due to intensive tillage practices and high fertiliser and pesticide use as well as low habitat diversity due to crop specialisation and reduced crop rotations (Poláková et al., 2011). Semi-natural traditional agricultural landscapes often entail a high level of local species diversity due to low intensity disturbances from grazing and mowing. Mixed farming system landscapes may have lower local diversity (species richness) than semi-natural landscapes but higher habitat diversity due to agricultural improvements, such as the introduction of hedgerows and the diversification of crops and agricultural practices. As a result, overall landscape diversity may sometimes be higher in mixed farming system landscapes than in traditional semi-natural landscapes.

Agricultural production practices have differential impacts on biodiversity depending on the type of habitat, the intensity of the given practice, as well as local factors such as soil type, microclimate and historical management practices (Poláková et al., 2011):

Beneficial impacts

- Low intensity grazing and haymaking have beneficial impacts for most semi-natural grasslands. In mixed farming system landscapes diversified land use patterns, including crop rotations and fallowing, have beneficial impacts on habitat diversity and species richness. Also field boundary habitats – e.g. hedgerows, field-forest margins, field edges, and ditches – provide important habitat patches for some farm wildlife species. These patches provide food, shelter and connectivity in agriculture-dominated landscapes (Poláková et al., 2011).

Harmful impacts

- Some agricultural production practices, such as fertiliser and pesticide application, nearly always have detrimental impacts on species richness and abundance, thus leading to biodiversity loss (Poláková et al., 2011). The application of chemical fertilizers and manure increases biomass production through denser and taller swards and changes also species composition, resulting in grass dominance and a reduction of species diversity (Kleijn et al., 2009). Increase in plant biomass and decrease in plant diversity has indirect effects on those invertebrates that favour shorter and more sparse vegetation (Nagy, 2009). On the other hand, earthworms and other decomposers can benefit from moderate manure applications (Vickery et al., 2001).

- Insecticides and herbicides have direct impacts on species diversity due to their intended toxicity to pests and weeds, but they also affect non-target species diversity and abundance, and thus have substantial knock-on effects on food webs, competitors and parasites in ecosystems of intensively cultivated agricultural landscapes (Poláková et al., 2011).
- Ploughing grassland and reseeding it with new cultivars, with associated fertiliser and herbicide use, results in species-poor grasslands that provide less food resources for invertebrates (McCracken and Tallowin, 2004).
- Crop specialisation and decreased use of crop rotation reduces structural heterogeneity and ecological connectivity between habitat patches which reduces food resources and breeding options (Poláková et al., 2011)[2].
- Enlargement of field parcels through removal of boundary and other non-farmed habitats – hedges, ditches, trees – reduces habitat diversity and connectivity between non-farmed habitat patches in agriculture-dominated landscapes.

3. Payment approaches for biodiversity conservation and sustainable use in agriculture: Review of the literature

Governments' strategies for biodiversity conservation and sustainable use are based on a wide range of policy instruments ranging from direct regulation, economic instruments, to voluntary approaches. Economic instruments have gained increased policy interest in recent years. These include government agri-environmental payments, conservation auctions, biodiversity offsets, and so called results-based schemes. The chapter begins with a brief description of policy challenges for biodiversity conservation and sustainable use and with a short overview of criteria for policy instrument choice and evaluation before alternative policy instruments are reviewed.

3.1. Policy challenges for biodiversity conservation

Young and Cunningham (1997) discuss the complexities facing biodiversity policy:

- *Heterogeneity:* heterogeneity of biodiversity and ecosystems complicates policy design and implementation as targeted and tailored policy instruments and instrument mixes are required to address the variation of both opportunity costs and environmental benefits of policy instruments.
- *Irreversibility:* effects of agricultural practices on biodiversity can sometimes go beyond certain thresholds (or tipping points) and these impacts may be irreversible and cause species extinction or ecosystem collapse. Irreversibility requires appropriate policies that account for precautionary principle, safe minimum standards, and adaptive management (OECD, 1999).
- *Impact accumulation:* small biodiversity impacts over a long time may accumulate and create large losses with irreversible outcomes in the long run.
- *Information gaps and asymmetry:* due to the inherent complexity of interactions between agricultural practices, agrobiodiversity and agroecosystems, considerable information gaps exist; for example, regarding the role of agrobiodiversity for ecosystem services in agroecosystems – especially for supporting (e.g. nutrient cycling and soil formation) and regulating (e.g. climate and disease regulation) services (Jackson et al., 2007). These information gaps mean that policy

2. This is especially the case for intensive arable systems that have very low in field diversity and low habitat diversity due to lack of diversified cropping patterns as they are dominated by one or two cultivated crops. These intensively farmed landscapes also lack boundary and other non-farmed habitats (Poláková et al., 2011).

decisions have to be taken under uncertainty. Moreover, asymmetric information between a farmer and a policy maker creates a challenge as farmers have informational advantage regarding their compliance costs of adopting environmental practices. Some policy instruments, such as auctions, do better job than others of getting farmers to reveal privately held information, which can then be used to design more efficient policies.

- *Mix of values*: agrobiodiversity and agroecosystems provide a vast range of goods and services and their use and non-use values contribute to the well-being of humans. Some goods have *direct use* values, while many ecosystem services provide *indirect use* values. In addition, *option* and *quasi-option* values – option to make choices in an uncertain future – are linked to both direct and non-direct uses. As regards non-use values, *existence* value refers to the fact that humans value ecosystems and biodiversity for their pure existence, and *bequest* value for the possibility of maintaining them for future generations (OECD, 1999).

Monetary valuation of biodiversity is a difficult task, since in many cases economic valuation of biodiversity does not capture all elements and services provided by biodiversity – in particular, ecological functions and regulating services – and thus, the estimated value may constitute only a small part of the total value of biodiversity. Nunes and van den Bergh (2001) review and evaluate empirical literature on monetary valuation of biodiversity and conclude that the empirical literature fails to account for the entire range of biodiversity benefits and that therefore available economic valuation estimates should generally be considered to provide "at best lower bound estimates to the unknown value of biodiversity changes". Nevertheless these lower bound estimates provide information that is necessary for informed decision-making processes and the selection of appropriate policy instruments, as well as for defining their ambition, and the need for instrument mixes (OECD, 1999). Failing to account for the value of ecosystems and biodiversity can lead to misinformed policy choices and thus understanding and capturing the value of ecosystems and biodiversity is essential for efficient policy responses (OECD, 1999, TEEB, 2009).

In addition to the economic valuation of biodiversity, also the measurement of biodiversity remains challenging (due to the complexity and multidimensionality of biodiversity) although numerous surrogates and correlates have been proposed to quantify biodiversity (Gaston, 1996). Species richness of one single group or a small set of organisms will not necessarily provide sufficient information for: (i) comparing habitats or ecosystems, (ii) analysing species diversity functions in ecosystems, and (iii) analysing the impact of environmental factors on biodiversity (Jeanneret et al., 2003). Furthermore, traditional biodiversity measures (e.g. species richness) that have the meaning "the more, the better" or "the more, the more sustainable" do not necessarily provide correct information for the selection of effective target elements for functional biodiversity in agroecosystems (Moonen and Bárberi, 2008). Information on the functional roles of different species groups would be essential for agroecosystem management as it could help to focus conservation efforts on those species whose functional traits influence the agroecosystem processes of interest.

A further challenge for voluntary biodiversity enhancing policy programmes is farmers' motivation to participate and farmers' attitudes to biodiversity conservation and sustainable use. Studies focusing on the adoption of agri-environmental measures (e.g. Wynn et al., 2001, Wanslembrouk et al., 2002 and Siebert et al., 2006) have revealed factors influencing farmers' participation decision. Wynn et al. (2001) and Defrancesco et al., (2008) propose the following classification: (i) farm's physical factors, (ii) farmer characteristics, (iii) business factors, and (iv) situational factors. Farm physical factors include farm size and farm type while farmer characteristics include farmer's age and education. Business factors include land tenure and off-farm income, while situational factors contain the amount of information received and the neighbours' participation as well as the interface between farmers and the policy. Although empirical evidence does not lend unanimous support, farmers' participation decision is generally positively influenced by farm size, educational level, and farmer's interest in environmental conservation, while it is negatively related to farmer's age.

Naturally agri-environmental scheme factors, such as the level of payment and the length of the contract influence farmer's participation decision. Ruto and Garrod (2009) employs choice experiment method to investigate the role of agri-environmental programme design on encouraging participation. Their results show that farmers require higher compensation payments to enter programmes that involve longer contracts, offer less flexibility with regard to management practices, and greater amount of paperwork. Moreover, good relationship between the government and farmers affect participation rate as the trust in the government increases participation.

One of the most important factors affecting farmer's participation decision to agri-environmental programmes is private transaction costs caused by application and implementation procedures of the programmes. These private transaction costs are in turn affected by farm, farmer, institutional, and programme specific factors. Public and private transaction costs are discussed in more detail in the next section dealing with criteria for policy instrument choice and evaluation.

3.2. Criteria for policy instrument choice and evaluation

The main policy choice and evaluation criteria used are environmental effectiveness and cost-effectiveness. However, there are other important criteria that contribute to the success and acceptability of policy instruments, including administrative costs and feasibility, and so-called ancillary benefits and costs (OECD, 2010*a*).

Environmental effectiveness

Environmental and conservation effectiveness refers to the capacity of the policy instrument to achieve the stated environmental and conservation objectives. Thus, an explicit statement of the objectives and selection of measurable indicators or practices that empirically have been shown to lead to the desired environmental and conservation outcomes is required. Potential indicators include *pressure* indicators, such as applications of pesticides and fertilisers, soil tillage practices, and land use choices like green fallowing. *State* indicators measure the condition of agroecosystems that are affected by pressures. Examples of state indicators are species diversity and abundance and the area of semi-natural habitats. Environmental objectives are typically expressed in terms of state indicators while indicators used for agricultural and environmental practices are typically pressure indicators. Essentially, state indicators express ends, while management practice indicators express means (OECD, 2010*a*).

Additionality of environmental practice adoption is a key factor affecting environmental effectiveness and budgetary cost-effectiveness of voluntary payment programmes. Environmental practice adoption is additional if it would not have been adopted without the payment. Payments for practices that would have been adopted even without the payment represent wasted programme resources without environmental benefits and thus resulting low budgetary cost-effectiveness. Claassen et al. (2014) have analysed additionality in U.S. agricultural conservation programmes. Their analysis shows that additionality varies by conservation practice and is high (80-82%) for structural practices (e.g. buffer practices) that require substantial upfront investment and provide small on-farm benefits. For management practices that may also provide on-farm benefits additionality is much lower, for example 54% in the case of conservation tillage. Additionality is also very important in the context of environmental credit and biodiversity offset mechanisms because non-additional credits and offsets compromise environmental targets of these mechanisms.

Cost-effectiveness

The cost-effective policy instrument is the one that minimises the farmer's compliance costs while achieving the environmental target, thus maximising cost-effectiveness (OECD, 2010a). Farm-level cost-effectiveness means that farm-level environmental and conservation outcomes have been attained at least cost. Cost-effectiveness at the landscape level means that landscape-level outcomes have been attained at least cost. It deals with the allocation of environmental and conservation efforts across individual farms within a region. Differences in both opportunity costs and environmental benefits of conservation efforts – spatial variation in costs and impacts – imply that the cost-effective achievement of landscape-scale environmental and conservation targets will generally entail differential levels of environmental and conservation effort across farms (OECD, 2010*a*).

Transaction costs and policy-related transaction costs

Environmental policy efficiency can be affected by the demands policy measures have on the management capacities of public agencies, and the associated public sector transaction costs for design, implementation, monitoring and enforcement. Transaction costs can be defined as the cost of resources used to define, establish, maintain and transfer property rights (McCann et al. 2005)[3]. Furubotn and Richter suggest that transaction costs can be divided into institutional, managerial and market transaction costs. The policy-related transaction costs (PRTCs) belong to the class of institutional transaction costs (OECD, 2007). Accounting for PRTCs is important for several reasons: (i) it improves comparison among and screening of alternative policy instruments, (ii) it can help the effective design and implementation of policy instruments to achieve policy objectives, (iii) it improves the evaluation of policy instruments, and (iv) it helps track budgetary costs of policy instruments over their whole life cycle (McCann et al. 2005).

Despite the importance of PRTCs for policy choice, PRTCs have seldom been formally included in agricultural and agri-environmental policy making (OECD 2007). Moreover, there are only a few studies that provide empirical estimates of PRTCs of agricultural and agri-environmental policies. These include notably Falconer and Whitby (1999), Falconer et al. (2001), Falconer and Saunders (2002), Mann 2000, McCann and Easter (1999), McCann and Easter (2000), Vatn et al. (2002), Rørstad et al. (2007), Ollikainen et al. (2008) and Mettepenningen et al. (2009). The common way to define PRTCs in the empirical literature has been to express them as a percentage of transfers.

Overall, empirical literature shows that there is huge variation in PRTCs between different agricultural and agri-environmental policy instruments. Studies demonstrate that policy instruments applied to existing commodity market transactions, such as pesticides and fertilizers, imply low PRTCs (0.1-1.1% of tax revenue). Also arable area payments and payments based on livestock numbers have low PRTCs, ranging from 1% to 7% (Vatn et al. 2002, Ollikainen et al., 2008, McCann and Easter 1999, Falconer et al. 2001). On the other hand individually tailored agri-environmental management agreements have the highest PRTCs (25-66% of transfer) because of their high asset specificity and the low frequency of transactions (McCann and Easter 1999, Vatn et al., 2002, Rørstad et al., 2007, Ollikainen et al., 2008). These studies also demonstrate that the number of transactions (or number of contracts) decreases PRTCs (Falconer et al. 2001). Moreover, some studies suggest that there is a significant learning effect, which exerts downward pressure on PRTCs in the longer run (Falconer and Whitby, 1999) as an indication of improved efficiency of administration.

Some recent policy developments towards increased spatial targeting and tailoring of policies assume that there should be a trade-off between improving the precision of policies and increasing PRTCs. Increasing precision usually increases policy-related transaction costs (Vatn, 2002, OECD, 2010a). Therefore, a good grasp of PRTCs is required to find a policy instrument that has a good balance between improved precision and increased transaction costs (Ollikainen et al., 2008). The trade-off between the precision of policies and increasing PRTCs and their implications for cost-effectiveness

3. McCann et al. (2005) developed a typology of transaction costs for environmental and resource policies. In their typology transaction costs are categorized into: (i) research, information collection, and analysis, (ii) enactment of enabling legislation, (iii) design and implementation of the policy, (iv) support and administration of the policy, (v) costs of contracting, (vi) monitoring of the compliance and environmental outcome, and (vii) enforcement. McCann et al. (2005) emphasise the importance of capturing transaction costs in all stages of policy from early development (research and information) to established policy (*ex post* evaluation).

of policies will be analysed in empirical application of the theoretical framework developed in Chapter 4.

Studies on PRTCs seldom consider whether analysed policy instruments have achieved their stated environmental objectives. Information on both environmental effectiveness and PRTCs is fundamental for comparing targeted and non-targeted policy instruments. Fährmann and Grajewski (2013) developed a cost-impact synopsis (CIS) framework for analysing the relationship between policy effectiveness and PRTCs and applied it to rural development programmes in five German States. Their empirical results give some support for hypothesis that the probability of the policy measure to be environmentally effective is higher when its relative PRTCs are higher. Thus, increase in relative PRTCs increases the effectiveness of the policy (in this case agri-environmental measure).

While most of the existing literature focuses on public administration PRCTs, a few studies have estimated farmers' private transaction costs of programme or agri-environmental measure participation (Falconer 2000, Falconer and Saunders 2002, Vatn et al. 2002, Rørstad et al., 2007, Mettepenningen et al. 2009). Rørstad et al. (2007) results from Norway show that these private transaction costs range from 2.3% (payment for reduced tillage) to 9.1% (payment for preserving cattle breeds) of the compensation payment. Mettepenningen et al. (2009) employ data collected from ten different European regions to estimate farmers' private transaction costs in the context of European Agri-environmental Schemes. Their results indicate that total private transaction costs are on average EUR 40.2 per ha, they account 14.3% of total costs of policy for a farmer (other cost items include e.g. income forgone and investment annuity) and represent 25.4% of the agri-environmental payment.

Ancillary benefits and costs

Ancillary benefits (also called co-benefits) and costs refer, for example, to a situation where a policy driven adoption of a given environmental practice has effects on multiple environmental goods and services. For example, conversion of cropland to perennial grasses improves wildlife habitat while it also increases soil carbon sequestration and improves water quality. Environmental impacts that are ancillary with respect to one policy objective may be the primary target of another. When this is the case, policy coordination to achieve the multiple environmental objectives is important, particularly if the pursuit of one objective conflicts with the pursuit of another. Policy coordination that simultaneously manages for multiple objectives can realise the gains from potential synergies and improve overall efficiency.

Other policy choice considerations

In the context of biodiversity policies it is also important to employ "precaution criteria" so that a policy instrument avoids the chance of irreversible outcomes, when there is scientific uncertainty about the outcome (Young and Cunningham, 1997). Both the "precautionary approach" and "safe minimum standards" can be used to complement other policy choice criteria. The precautionary approach suggests that when there is likelihood for irreversible outcome, lack of scientific evidence should not be used as a reason for postponing the implementation of conservation measures. The safe minimum standard approach suggests that biodiversity losses should be avoided unless the opportunity costs of doing so are very high, that is, the benefit-cost ratio of any project that incidentally causes biodiversity loss should be very high (OECD, 1996).

3.3. Review of policy instruments

Payments for biodiversity-enhancing practices and land-uses

Uniform versus non-uniform payments

The majority of OECD countries offer payments to farmers to encourage them, on a voluntary basis, to implement more environmentally friendly farming practices going beyond those required by regulations or environmental cross-compliance. Most of these agri-environmental payment programmes offer a fixed-rate payment for compliance with a predetermined set of environmental requirements, such as limits on the intensity and timing of fertiliser, manure and pesticide applications (OECD, 2010a)[4].

Both the costs (farmers' forgone profit) and the benefits of biodiversity conservation measures may vary significantly over space. The spatial scale of cost and benefit variation may differ from a field scale to a regional one depending on the particular conservation problem (Wätzold and Schwerdtner, 2005). In addition to spatial variation, the costs and benefits of biodiversity conservation measures may also vary over time, for example due to temporal variation in farmers' profit forgone, which is dependent on market prices for agricultural inputs and outputs as well as government support payments for agriculture.

Given the spatial differences in costs and benefits, the problem with the aforementioned fixed-rate payment (uniform payment) is that spatial heterogeneity neither in costs nor benefits is reflected in policy design and implementation. As a result, the cost-effectiveness (amount of conservation benefit for a certain amount of conservation budget) of uniform payments is reduced. Hence, due to spatial variation in costs and benefits the cost-effectiveness of biodiversity conservation requires spatially differentiated practices and payments.

However, spatial differentiation of payments and practices is likely to increase both the administrative costs of the payment scheme and the difficulty of assessing the criteria on which to base payment variations, while there may be potential objections based on equity (uniform payment for all farmers irrespective of their profit forgone) concerns. Hence, a cost-effectiveness advantage of differentiated payments needs to be weighed up against increased administrative costs as well as equity concerns (Wätzold and Schwerdtner, 2005).

Wätzold and Drechsler (2005) develop a conceptual framework for assessing the cost-effectiveness losses of uniform payments relative to differentiated payments under different types of ecological benefit functions and cost functions. Their results show that the cost-effectiveness of uniform payments relative to differentiated payments varies significantly depending on the spatial variation of the costs and benefits (from 0% to almost 100% of the cost-effectiveness of differentiated payments). Overall their results indicate that the cost-effectiveness of uniform payments may be low.

Payments for habitat heterogeneity and habitat networks

According to Ohl et al. (2008) agri-environmental payments for biodiversity enhancing practices in many cases: (i) fail to cover all relevant species in a given area; (ii) neglect the fact that conservation practices need to be spatially and temporally differentiated due to time dependent habitat quality; (iii) do not consider the fact that successful conservation of certain species may require heterogeneous measures due to multiple resource use of species; and (iv) lack sufficient knowledge about the effects of conservation measures on species. To address these weaknesses a spatially and temporally differentiated portfolio of measures that leads to habitat heterogeneity is proposed.

Ohl et al. (2008) note that it may be difficult (and sometimes even impossible) to design and implement a payment mechanism that incentivises a group of farmers to adopt a certain portfolio of measures that provides desired habitat heterogeneity in a given region. The reason for this is that it is difficult to find a correct payment levels for measures in the portfolio so that the adoption rate of each measure is the desired one – not too high and not too low. One way to address this problem is to take a "closed membership approach" so that farmers get payments for a certain measure on a first-come-first-served principle until the target for this measure is reached, whereafter the rest of the farmers need to select between remaining measures in the portfolio. The drawback of this approach is that it does not guarantee a cost-effective allocation of conservation measures as farmers with high opportunity costs

4. There are naturally some payment programmes that employ regional targeting (e.g. criteria related to floristic composition of meadows).

may apply first and be selected, and thus the budget may not be sufficient to cover the desired habitat area (Ohl et al., 2008).

The contribution of a given habitat to a conservation objective depends on its spatial extent and its location relative to other habitats. In spatially structured landscapes species populations exist as so-called metapopulations for which it is beneficial if the species can move between habitat patches, and thus for a given total habitat area connected habitats are more valuable than isolated ones (Hanski, 1999). Drechsler (2011) develops a conceptual ecological-economic model on the basis of metapopulation theory and investigates cost-effective habitat networks when conservation costs vary spatially and over time. His analysis focuses on trade-offs between the amount and spatial connectivity of habitat patches, and between amount and temporal continuity (habitat turnover). His analysis shows that for a given conservation budget, the cost-effective network radius lies between the radius that would maximise connectivity (selecting relatively costly field parcels) and the one that would maximise habitat area (selecting only the least-cost field parcels). As regards the trade-off between amount and continuity of habitat, the cost-effective turnover rate (the fraction of conserved sites that are destroyed and recreated elsewhere) of habitat network was analysed. For a given conservation budget a high (low) turnover rate allows protecting more (less) field parcels and so the average cost is lower (higher). But a high turnover rate increases the local extinction risk of the species and increases the extinction risk of the metapopulation in the habitat network, especially if habitat restoration takes time (Drechsler, 2011).

For many species connected habitats are ecologically more valuable than isolated ones and in fragmented landscapes the improved spatial connectivity of habitats is needed. Because of this, payment design needs to provide incentives for farmers to conserve spatially connected habitats. Parkhurst et al. (2002) proposed the “agglomeration bonus” as an incentive to landowners to generate ecologically valuable configurations of habitat patches. A bonus payment is paid for landowners if managed habitats are arranged so that a desired spatial configuration is achieved. A payment system based on an agglomeration bonus likely leads to a beneficial ecological outcome but it may also be a costly system as it may be necessary to include high opportunity cost patches in order to achieve the desired spatial configuration.

Drechsler et al. (2010) compare the cost-effectiveness of an agglomeration bonus payment relative to a spatially homogenous payment conceptually, numerically and empirically. Their analysis shows that the cost-effectiveness of the agglomeration payment is determined by the interaction of three effects: (i) the connectivity effect; (ii) the patch restriction effect; and (iii) the surplus transfer effect. The bonus payment increases ecological connectivity, which in turn leads to higher survival rates of endangered species and improves the ecological effectiveness of the habitat network. The patch restriction effect arises because under the bonus system patch selection is restricted to smaller compartments of the landscape (enabling agglomeration). The patch selection therefore does not systematically go for the least cost patches from whole landscape and as a result the relative cost-effectiveness of the bonus payment is reduced. The surplus transfer effect relates to a situation where a desired spatial configuration requires the participation of those landowners who would incur income losses from participation. Therefore their participation requires side payments from other landowners, whose surpluses from participation in the bonus system are thus reduced. Illustrative numerical results show that the patch restriction effect never dominates the other two effects, and thus the bonus payment scheme is always more cost-effective than a homogenous payment scheme. This result is confirmed in the case study focusing on the protection of an endangered butterfly (Large Blue, *Maculinea teleius*) in a region around the city of Landau in Germany. The case study results show 30-70% cost savings with an agglomeration payment scheme relative to a homogenous payment scheme (comparison includes transaction costs) (Drechsler et al., 2010).

Results-based payment schemes

Insufficient performance of *practice-based* (also called *action-oriented*) agri-environmental payment programmes in enhancing biodiversity conservation (see e.g. Kleijn et al., 2001; 2004) has generated increased interest in *results-based* (also called *payment-by-results* and *outcome-based*) payment schemes. Experiments of results-based schemes have been conducted, for example, in Germany and in the Netherlands. In Germany the focus has been on the conservation of species-rich grasslands and meadows (in Brandenburg, in Baden-Wuerttemburg and in Lower Saxony), while in the Netherlands results-based schemes have been employed in the conservation of meadow birds (Schwartz et al., 2008)[5].

In the results-based scheme the payment is directly linked to the environmental outcome, such as the number of indicator species in species-rich grassland. Thus, the payment is linked to the quantity or quality of the environmental outcome, but the payment calculation may still be based on income foregone and extra costs incurred (Schwartz et al., 2008).

Schwartz et al. (2008) define key characteristics of results-based schemes as follows:

- Direct link of the payment to the environmental outcome
- Payment level is differentiated according to the level of the environmental outcome
- The farmer is free to choose management practices to achieve the environmental outcome.

Results-based payment schemes have been argued to provide greater flexibility and promote innovation in biodiversity conservation, which allows cost-effectiveness gains over practice-based payments (Zabel and Roe, 2009)[6].

Matzdorf and Lorenz (2010) discuss various advantages and disadvantages of the results-based approach and provide empirical evidence on the cost-effectiveness of a results-based scheme in Germany. Their study employs data from Baden-Wuerttemberg where a results-based scheme started in 2000. The objective of this scheme is to support the protection and maintenance of species-rich grasslands that are cultivated extensively and traditionally. Over 9 100 farmers, representing an area of 65 959 hectares (ha) participated in the scheme in 2005. The scheme area covered almost 12% of the total grassland area of the region of Baden-Wuerttemberg.

The empirical data is based on interviews conducted with farmers and officials of agricultural administrations. The interview questionnaire addressed various advantages and disadvantages considered in the context of results-based schemes. Potential advantages include: (i) the farmer enjoys greater flexibility in choosing practices for achieving the environmental outcome, (ii) results-based schemes increase innovation, intrinsic motivation and interest for biodiversity conservation, and (iii) results-based schemes promote cooperation between farmers. Potential disadvantages include: (i) increased financial risk to farmers, and (ii) transaction costs related to the complexity of control.

The results of Matzdorf and Lorenz (2010) support the hypothesis that the results-based scheme provides greater flexibility to farmers and contributes positively to farmers' attitudes towards biodiversity conservation. Farmers and officials of agricultural administrations assessed transaction costs to be relatively low and farmers considered that the financial risk is not so relevant in this case as the scheme focuses on maintaining the quality of grassland. Finally, the study results provide support for the argument that results-based schemes increase cost-effectiveness.

5. The former Meadow Bird Agreement was discontinued 2004, but a number of agri-environment cooperatives in the Netherlands operate results-based schemes focusing on protection of grassland bird populations. These schemes focus on nest protection through marking and avoiding nests during the field operations. The result indicator is the number of bird nests and the payment per nest varies according to rarity of bird species. Cooperative pools the share of Common Agricultural Policy agri-environment payments paid to member farmers and then re-distributes according to the results achieved by different farmers.

6. Results-based schemes are ideal for innovation. In input-based schemes flexibility and innovation can be improved by allowing a large menu of practices.

Burton and Schwarz (2013) discuss two key concerns related to results-based schemes, which are the increased risk to farmers and the difficulties in developing and monitoring of indicators.

Results-based schemes may increase the risk to farmers due to the impact of exogenous factors (factors outside farmers' control), such as climate and weather conditions, the behaviour of neighbouring farmers, and uncertainty related to species migration, breeding, feeding etc. A number of means have been proposed to mitigate these risks in results-based schemes: (i) the combination of a base payment for actions and a bonus payment for results, (ii) making the payment dependent on both farmer's actions and a weather variable, and (iii) the use of multiple performance indicators that spread the risk that any particular species is absent in one year (Burton and Schwarz, 2013; Zabel and Roe, 2009). While these means mitigate the risk to farmers, they may also increase transaction costs of the scheme.

Results-based schemes require indicators that represent the desired outcomes. Burton and Schwarz (2013) present the characteristics of good indicators in the context of results-based schemes: (i) indicator species should be easily identifiable as in the majority of the schemes farmers have a role in monitoring them (i.e. self-monitoring), (ii) indicators should be consistent with ecological goals and should not conflict with agricultural goals, (iii) indicators should reflect the effort of the farmer. If the objective of the scheme is to promote biodiversity in given region, multiple indicators are needed in order to reflect the diversity of habitats and species groups (Burton and Schwarz, 2013).

Burton and Schwarz (2013) develop a conceptual framework for describing whether results-based schemes are strongly or weakly result-oriented. The framework builds on the following three dimensions: (i) the proportion of payment derived from outcomes, (ii) the sensitivity of the payment level with respect to level of outcome, and (iii) the temporal extent of the contracts and schemes. A strongly result-oriented scheme would be one where the proportion of result-oriented payment is high, the payment increases with increases in the environmental outcome, and the duration of the contract and the scheme is long enough to reward farmers' investments in skills and knowledge, as well as to allow time lags between changes in practices and environmental outcomes.

Based on the literature review and on Schwarz et al. (2008) review of several results-based schemes we can derive the following conclusions:

- First, results-based schemes can improve the environmental targeting and cost-effectiveness of agri-environmental schemes compared to practice-based payments since they provide greater flexibility and promote innovation.
- Second, results-based schemes may increase the risk of farmers and it is important that the payment design considers this risk higher exposure and uncertainty through, for example, a combination of a fixed basic payment and a bonus payment, or by using flexible payment rates to address the impact of external factors (such as extreme weather events).
- Third, the availability and quality of indicators representing the desired outcomes is key to the effectiveness of results-based schemes.
- Fourth, results-based schemes may perform best when targeting local environmental issues in specific areas.

Biodiversity conservation auctions

Experiences from biodiversity conservation auctions (trials and pilots) show that auctions improve cost-effectiveness relative to fixed uniform payment schemes while the cost-effectiveness gains from auctions vary significantly.

Auctions and competitive bidding have been recently applied to environmental protection and biodiversity conservation in agriculture (Latacz-Lohmann and Hamsvoort, 1997, Stoneham et al., 2003, Vukina et al., 2006). The basic idea in conservation auctions is that farmers bid competitively for a limited number of environmental conservation contracts. When making a bid a farmer faces a trade-off between net payoffs and acceptance probability so that a higher bid increases the net payoff but reduces the probability of getting a bid accepted (OECD, 2010*a*). Thus, competitive bidding can push farmers to reveal – at least to some extent – their compliance costs. As a result competitive bidding could improve the budgetary cost-effectiveness of biodiversity conservation as farmers' information rents (i.e. the difference between payment level and farmer's compliance cost) are reduced. The primary reasons for implementing conservation auctions are to improve both allocative efficiency (bids with the highest benefit-cost ratio are selected for the programme) and budgetary cost-effectiveness (maximising environmental benefits with a given fixed budget) (OECD, 2010*a*)[7].

Latacz-Lohmann and Schilizzi (2005) provide detailed discussion and guidance on how to design and implement conservation auctions. In the *discriminatory payment* format each bidder is paid according to his winning bid. In the *uniform-price* auction all those who are successful receive the same cut-off price, which is either the highest accepted or the lowest rejected bid. Different payment formats affect farmers' bidding behaviour. Under the discriminatory format the farmer's bid not only depends on the farmer's compliance cost but also on his best guess with regard to the highest acceptable bid. Thus, there is an incentive for a farmer to bid above his compliance costs and thus secure an information rent. This incentive is higher for those farmers with low compliance costs than those with high compliance costs. Under uniform pricing the farmer's dominant strategy is to bid his estimated compliance costs, because the bid only determines his chance of getting into the conservation programme but not the payment level.

Another important auction design parameter is the so called *reserve price*, which is the upper limit of payment per unit of conservation benefit, which can be pre-announced or not. A reserve price increases bidding competition (as it increases the risk that bidders might lose an auction by bidding too high) and thus reduces farmers' information rents, but also provides the signal of maximum willingness to pay for farmers' provision of biodiversity conservation. Pre-announcing a reserve price may create problems in repeated auctions where farmers learn the level of the reserve price and offer their bids at the reserve price (Reichelderfer and Boggess, 1998; Latacz-Lohmann and Schilizzi, 2005). Hellerstein et al. (2015) analyse various auction design alternatives to improve conservation auction performance. They find that bid caps (i.e. estimate of field parcel's agricultural rental value) that are employed in the USDA Conservation Reserve Program (CRP) to prevent excessive information rents can be improved. Although bid caps reduce information rents to low cost bidders, an improperly set bid cap can reduce program participation of higher cost bidders with high environmental benefits and may also discourage adoption of environmental practices that improve environmental benefits of farmers' offers. Auction design alternatives, such as quota auction and reference price auction can reduce program costs and improve performance (Hellerstein et al. 2015). In a reference price auction each field parcel has a reference price that reflects its agricultural value and bids are ranked relative to this reference price so that the lower the bid is relative to the reference price, the higher the bid will be scored. Reference price auction should incentivise low-cost (that is, low reference price) bidders to bid closer to their opportunity cost (Hellerstein et al. 2015). In a quota auction, bids from a group or a region sharing similar characteristics, such as low agricultural productivity, are limited in order to induce competition within this group and thus lower bids and information rents (Hellerstein et al. 2015). Hellerstein et al. (2015) employ economic experiments performed in classroom laboratories and find that programme costs can be reduced by 18% using a reference price auction and by 14% using a quota auction relative to using a standard open auction.

Conservation auctions can be implemented with a fixed budget or with a fixed target. A fixed-target conservation auction sets, for example, an acreage target for a semi-natural habitat conserved and

7. Note that not all programmes use benefit/cost ratios as scores, but also other mathematical relationships. Ratio rules are superior regarding maximisation of benefits with a given expenditure. However, some other rules may be better in cases where benefits are difficult to measure or minimum standards across space are valued

bids are accepted from farmers until the target is reached. This means that the budget of the programme is not known before the auction is completed. The fixed budget conservation auction is a common form in which bids are accepted on the basis of conservation benefit to conservation cost (farmer's bid) ratios until the predetermined fixed conservation budget is exhausted (OECD, 2010*a*).

Bidder learning can be a problem with repeated conservation auctions and thus information about highest or average acceptable bids or the distribution of bids received in the previous bidding rounds should not be published. Potential problems related to bidder learning can be reduced by changing the rules of the conservation auction in each bidding round in order to create uncertainty among bidders (Latacz-Lohmann and Schilizzi, 2005).

Strategic bidding behaviour, as well as collusion, reduces the potential cost-effectiveness gains of auctions. Conservation auctions may not be suitable for small scale, local biodiversity conservation projects, since the smaller the number of potential bidders the lower is the bidding competition and the higher is the risk of collusion and strategic bidding (OECD, 2007c). Moreover, single round bidding is preferred to multiple rounds, since cost-effectiveness gains from auctions are eroded when bidders learn the reserve or cut-off price (e.g. Stoneham et al., 2004; Latacz-Lohmann and Schilizzi, 2005; Hailu and Schilizzi, 2004). Changing the weights of multi-criteria bid scoring between different bidding rounds, however, should reduce such strategic bidding behavior.[8]

Cost-effectiveness gains from auctions relative to fixed uniform payment schemes have been reported in many studies. However, these cost-effectiveness gains vary significantly between different studies.

In Victoria, Australia, the BushTender programme addressed biodiversity conservation through improved bush management. The BushTender auction in 2001 was a sealed bid, discriminatory price auction to manage vegetation areas in two regions of Victoria. A Biodiversity Benefits Index – which describes a benefit to cost ratio for the government – was used to rank applications from highest value bid down until the budget constraint was hit (Stoneham et al., 2003). Stoneham et al. (2003) report the first-round cost-effectiveness gains from the auction to be 700% – that is, the biodiversity conservation auction provided seven times more biodiversity benefits than a fixed price scheme with the same budget. The transaction costs of the conservation auction were estimated to be 50-60% of the expenditure in the first round.[9]

The World Wildlife Fund auction (Auction for Landscape Recovery) in Australia was part of a larger trial of market-based policy instruments and focused on multiple environmental benefits from land management improvements, including biodiversity enhancement and salinity control. This auction was a sealed bid, discriminatory price auction and employed a regional metric of biodiversity complementarity within a systematic conservation planning framework (Latacz-Lohmann and Schilizzi, 2005). White and Burton (2005) report the cost-effectiveness gains from this conservation auction relative to a fixed price scheme to be between 207% and 315% in the first round and between 165% and 186% in the second round, depending on whether the fixed price scheme is input-based or output-based.

8. If farmers know the weights given for each attribute then they will use this information to overbid and earn information rent. Changing the weights between bidding rounds creates uncertainty to the bidders and as a result they will bid closer to their opportunity costs. These weights reflect relative values (preferences) of an agency for the different attributes or environmental objectives. Changing the weights means that the relative value of different environmental objectives or cost (bid) would change as well (Latacz-Lohmann and Schilizzi, 2005).

9. There have been multiple interpretations of the results of Stoneham et al. (2003). The OECD has interpreted the results to imply that the biodiversity auctions provided seven times more biodiversity benefits than a fixed price scheme with the same budget. Another interpretation is that a fixed price scheme required a budget seven times bigger than the biodiversity conservation auction to achieve the same biodiversity benefits.

Connor et al. (2008) use data from the Catchment Care auction (in the Onkaparinga catchment in South Australia), which was a sealed bid, discriminatory price auction, and simulate various alternative auctions, differentiated payments and fixed payment policies. Their results show that with the same budget a uniform fixed payment achieves 56% of the estimated environmental benefit obtained with auctions. They also show that the cost-effectiveness gains from auctions come through the Environmental Benefit Index (EBI) bid prioritisation rather than through the reduction of information rents.

The efficiency of the various instruments described above depends on the actual variations of opportunity costs and environmental quality as illustrated in Table 1. Uniform payment works well when both opportunity costs and environmental quality are homogenous. When opportunity costs vary but environmental quality is homogenous then differentiated payment on the basis of costs would perform better and be fairer than uniform payment. In this case an auction system would also perform well, but differentiated payment would probably be an easier and more flexible system when opportunity costs are reasonably well known. When environmental quality varies, the added value of auction systems and other targeting mechanisms (results-based or differentiated payments) increase. In these cases, auctions work well when the number of potential participants (bidders) is large, and results-based payment would be best suited for situations where the number of participants is low. When environmental quality varies efficiency requires that auction and other mechanisms employ an environmental scoring system to address environmental heterogeneity, e.g. use of environmental benefit index. Also, policy-related transaction costs affect the efficiency of alternative payment types and thus auctions may be preferred to results-based schemes when potential pool of participants is large.

Table 1. Efficiency of biodiversity payment instruments depending on opportunity costs and environmental quality

Environmental quality	Opportunity costs	
	Homogenous	*Heterogeneous*
Homogenous	Uniform payment (N, n)	Differentiated payment-cc (N, n)
Heterogeneous	Differentiated payment-eb (N,n) Auction-eb (N) Results-based payment-eb (n)	Differentiated payment–cc and eb (N,n) Auction–cc and eb (N) Results-based payment–cc and eb (n)

Note: N = works well with large number of participants; n = suitable for small number of participants; cc = differentiated on the basis of compliance costs; eb = differentiated on the basis of environmental benefits.

Source: modified from Iho et al. (2011).

Biodiversity offsets

Biodiversity offsets are defined as: "measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development and persisting after appropriate prevention and mitigation measures have been implemented" (BBOP, 2009). The goal of biodiversity offsets is usually to achieve no net loss of biodiversity with respect to species composition, habitat structure, and ecosystem services. They are intended to be used during the final step of mitigation hierarchy (avoid, minimise, restore and offset), and thus they represent "a last resort" and should be applied to compensate for the residual adverse impacts on biodiversity after preceding mitigation hierarchy stages have been addressed properly.

According to TBC (The Biodiversity Consultancy) (2013) at least 56 countries have laws or policies that require biodiversity offsets (or some form of compensatory biodiversity conservation) to compensate biodiversity losses related to development projects. As regards the finance mobilised by biodiversity offset programmes, estimates indicate the total global market turnover to be over

USD 3 billion per year (90% are compliance-based and 10% voluntary schemes) (OECD, 2016; Ecosystem Marketplace, 2013).

According to OECD (2013, 2016) biodiversity offsets can be implemented through three different approaches, including one-off offsets, biobanking and in-lieu programmes. One-off offsets are undertaken by the developer or by a third party acting on behalf of the developer. Under biobanking approach (also called habitat banking or conservation banking) a developer can buy offsets directly from biobank, which can be a public biobank or a private one. Under in-lieu fee approach a developer pays to regulatory agency a fee that reflects the costs of residual biodiversity loss to society and this fee will be used for compensatory biodiversity conservation.

Different approaches have different implications for potential temporal loss of biodiversity due to the development. Problem with the one-off offsets is that loss of biodiversity in the development site and generation of offsets at offset site happen around the same time and due to time lags for offset benefits to mature there may be a temporal loss of biodiversity. This is not the problem with biobanking approach as biodiversity outcome from offset is known before it can be used as an offset (OECD, 2016; Wissel and Wätzold, 2010).

Wissel and Wätzold (2010) and OECD (2013, 2016) discuss the general characteristics and potential cost-effectiveness of biodiversity offsets. Because the costs and benefits of land development and biodiversity conservation vary spatially, the flexibility allowed through offsetting can reduce the compliance costs and transaction costs of firms from a regulation[10]. Under one-off offset and biobanking schemes the prices of offsets are determined by supply and demand while with in-lieu schemes a fixed price is set by the regulatory agency that should, in theory, reflect the external costs of the biodiversity loss. Thus, an in-lieu scheme closely resembles an environmental tax (OECD, 2016).

OECD (2013, 2016) provides extensive discussion of the key design and implementation features of biodiversity offset schemes. These features include: (i) thresholds and coverage; (ii) equivalence; (iii) additionality; (iv) permanence; (v) monitoring, reporting, and verification; (vi) compliance and enforcement; and (vii) transaction costs. From these features equivalence is one that is a defining issue for biodiversity offsets, since offset scheme needs to ensure that biodiversity gain at the offset site equals the loss at the development site (OECD, 2016). In their analysis Wissel and Wätzold (2010) review the key challenges in applying offsets to biodiversity conservation. Destroyed and created habitats differ in three important dimensions: (i) type (e.g. different habitats have different functional values for different species and their life stages), (ii) space (e.g. destroyed habitat may be part of contiguous habitats while created habitat may be isolated and thus have different biodiversity value), and (iii) time (e.g. time lags between destruction of habitat and maturation of created habitat may increase species extinction risk). The differences in destroyed and created habitats can be reflected in trading-ratio (also called mitigation-ratio) between destroyed and created habitats. Trading ratio reflects the scarcity of different habitat types and determines how many credit units of offset are needed to compensate for one unit loss at the development site (e.g. 3 to 1 ratio).[11] Trading ratios can be tailored to account various factors, such as equivalence, location, temporal lags between impacts and offset maturity (McKenney and Kiesecker, 2010).

Factors influencing trading activity are, for example, opportunity cost differences, regional size of the market, and transaction costs of market exchange (Wissel and Wätzold, 2010). Other things being equal, situations with large differences in opportunity costs, large regional size of potential offset

10. The objective of biodiversity compensation as practiced in France is not to reduce compliance costs to developers for compliance with the regulations, but only to reduce transaction costs through pooling done by special service providers on the behalf of developers.

11. It is important to note that very rare habitats need to be protected and thus development impacts cannot be authorised in these cases. Overall, careful consideration is needed in the application of trading-ratio, even in the case of more typical habitats.

market (supply and demand), and lower transaction costs promote higher trading activity and therefore more efficient biodiversity offset schemes.

Agricultural biodiversity offsets

As regards the role of agriculture in biodiversity offset systems, the primary example is mitigation banking in the United States. A national goal of *no net loss of wetlands* is set and mitigation hierarchy is followed so that after avoidance and minimisation, the unavoided residual impacts to wetlands have to be compensated (this is called compensatory mitigation in the US). In wetland banking impacts can be offset through the purchase of credits that are created through restoration, creation, enhancement, and preservation of wetlands.

There is a specific agricultural wetland bank in the state of Minnesota. It is a subset of the State's wetland bank and is dedicated to offset impacts to wetlands in cultivated agricultural fields. It targets to replace wetlands functions commonly lost to agricultural impacts, including wildlife habitat, water quality protection and floodwater attenuation.

Eligibility for the agriculture bank is as follows: the site must contain a wetland that has been drained, filled or degraded by cropping, that can now be restored. Only restored wetlands are allowed (credits from creation, enhancement, and preservation can be deposited for general wetland bank) and restoration takes place through tile breaks, ditch plugs and establishment of native vegetation. Wetlands restored under conservation programs (such as CRP) can be eligible, but the contract must expire prior to credits deposits in the bank and several other conditions apply. The restored wetland should provide an increase in wetland function over the impacted farmed wetlands.

Bank service areas are watershed-based geographic areas in which a bank can be expected to provide replacement for impacted wetlands. Offset close to the impact site are preferred.

Payments for environmental services

Most of the payment types analysed here fall in the category of "Payments for environmental services" (PES), which are defined as "a voluntary, conditional agreement between at least one *seller* and one *buyer* over a well-defined environmental service or a land use presumed to produce that service" (Wunder, 2005). In the context of public sector finance a PES scheme can be defined as "a transparent system for the additional provision of environmental services through conditional payments to voluntary providers" (Tacconi, 2012). PES are financed directly by the user or beneficiary of the environmental service (*user-financed*) or by a third party acting on behalf of the beneficiaries, such as a government, an NGO, or an international agency (Engel et al., 2008).

PES programmes vary in the type of environmental service addressed, the type of practice financed, the performance measure used, the source of finance, and the payment amount and type (Engel et al., 2008; OECD, 2010*b*). Most of the programmes implemented worldwide are used to address biodiversity, carbon sequestration and watershed services (Blackman and Woodward, 2010). In most of the cases PES are paid for specific land uses or management practices that are thought to generate the desired environmental service (input-based payment) rather than for the actual delivery of an environmental service (output-based payment or performance-based payment) due to the difficulties in directly observing the level of environmental service or due to time lags between practice adoption and environmental service provision (Engel et al., 2008; OECD, 2010*b*).

4. Theoretical framework, data and empirical illustration of environmental effectiveness and cost-effectiveness of different payment approaches

4.1. Theoretical framework

In this section a simple theoretical framework is developed to simulate a farmer's decision to participate in a government agri-environmental payment programme, as well as, analyse government's selection of participants for a programme.

Uniform payment for biodiversity strip

The starting point of the conceptual framework is a heterogeneous land quality model with different soil types and land productivities. Following Lichtenberg (1989) and Lankoski and Ollikainen (2003), land quality differs over field parcels and it is ranked by a scalar measure, *q*, with the scale chosen so that minimal quality is zero and maximal quality is one, i.e. $0 \leq q \leq 1$. Crop yield per hectare, *y*, is a function of land quality *q* and fertilizer application rate *l*, that is, $y = f(l;q)$. In the absence of government agri-environmental payment programme farmers' profits are defined as $\pi_0 = pf(l;q) - cl - K$, where *p* and *c* represent the respective prices of crop and fertilizer and *K* other cultivation costs per hectare. Agriculture can contribute to species diversity and abundance by providing semi-natural habitats. The focus here is on a special type of field margin, a biodiversity strip, which can locate between crop fields or between a crop field and the forest. Biodiversity strips are uncultivated, managed and covered by perennial grasses. Let *m(q)* denote the share of field parcel of quality *q* allocated to crop production that is retained as a biodiversity strip. Heterogeneous productivity of land implies that the establishment of biodiversity strip *m* results in differential opportunity costs in different field parcels. For the ease of implementation of this policy, the government can set a fixed width of biodiversity strip, $\hat{m}$, (e.g. 5 or 10 meters) and uniform payment for biodiversity strip $\hat{b}$. In this case farmers' profits are defined as, $\pi_1 = (1-\hat{m})\left[pf(l;q) - cl - K\right] + (\hat{b} - h)\hat{m} - \Omega$, where *h* denotes the annualised establishment and management costs of biodiversity strip and Ω denotes farmer's private transaction costs of participating in agri-environmental programme. A farmer will participate in this programme if his profits under the programme, π_1, are higher than his reservation profits, π_0.

Conservation auctions

In the case of conservation auction farmers competitively bid for a limited amount of agri-environmental contracts. Iho *et al.* (2014) and Glebe (2013) modified Latacz-Lohmann and van der Hamsvoort (1997) model and included environmental benefit index (EBI) in the bidding. Following Iho et al. (2014) farmers make expectations on bid/EBI ratios when they participate in bidding. EBI values are denoted by *e* and the upper limit of the bidder's expectation about the maximum expected bid/EBI is denoted by $\overline{\beta} = b''/e''$. By assumption bidders' expectations about this implicit bid cap are uniformly distributed in the range $\left[\underline{\beta},\ \overline{\beta}\right]$, where the lower bar represents the minimum (defined by $\underline{\beta} = b'/e'$) and upper bar the maximum expected bid cap. The probability that the bid is accepted is given by

$$P(\theta \leq \overline{\beta}) = \int_{\theta}^{\overline{\beta}} f(\theta)d\theta = 1 - F(\theta) \tag{1}$$

Where $\theta = \frac{b}{e}$, $f(\theta)$is density function and $F(\theta)$ distribution function. The expected net payoff of the risk-neutral farmer from bidding is a product of the revenue from winning the bid and the acceptance probability:

$$(\pi_1 + b - \pi_0)(1 - F(\theta)), \tag{2}$$

where π_0 denotes the profit under no participation and π_1 is profit under the secured conservation contract. The farmer chooses the bid, *b*, and thereby the ratio *b*/EBI, according to:

$$b^* = \pi_0 - \pi_1 + \frac{(1 - F(\theta))e}{f(\theta)}, \tag{3}$$

where *f(θ)* is the probability density function associated with $F(\theta)$ and *e* is the field parcel's EBI-value.

The difference $\pi_0 - \pi_1$ in equation (3) represents the income forgone and the establishment and management costs of biodiversity strip and farmer's private transaction costs of participation. The additional term, $(1 - F(\theta))e / f(\theta)$, is the information rent.

The optimal bid in the presence of EBI is determined by

$$b^* = \frac{\pi_0 - \pi_1 + e\overline{\beta}}{2}. \tag{4}$$

Hence, when EBI matters for participation in an auction, the optimal bid depends on the conservation costs and the expected cap multiplied by the bidder's own EBI value ($e\overline{\beta} = e(b''/e'')$). The higher is the EBI of the submitted field parcel, the higher is the bid ($\frac{db}{de} = \frac{\overline{\beta}}{2} > 0$). Glebe (2013) also shows that farmer's optimal bid changes when he receives information about the environmental score of the field parcel and farmer's bid increases (decreases) when informed that his environmental score is greater (smaller) than average score.

When farmers expect similar environmental performances across farmers, the optimal bid is the same as under the auction without EBI, that is, $b^* = \frac{\pi_0 - \pi_1 + b''}{2}$, (see Iho *et al*., 2014). Hence, in this case farmers' expectations are formed only on the basis of income forgone for adopting environmental practice and not on bid/EBI ratios.

Government selection of participants under various payment types

Following Connor *et al*. (2008) the bid selection problem for the government in the case of *auction* employing EBI can be written as: $\max \sum_i L_i EBI_i$ subject to $\sum_i L_i\, b_i \leq AEB$, where L_i is a set of binary choice variables taking a value of 1 for each bid that is selected and 0 for those that are not selected, *AEB* denotes the agri-environmental budget.

In the case of *uniform payment policy* $\hat{b}$ without environmental targeting through EBI, all farmers for whom the sum of the income forgone, the establishment and management costs of biodiversity strips, and private transaction costs is less than the uniform payment level are assumed to participate in the programme. From this subpopulation of farmers the programme participants are selected as a random draw up to the same budget limit as in the auction, *AEB*.

In the case of *uniform payment $\hat{b}$ policy with EBI targeting*, all farmers for whom the sum of the income forgone, the establishment and management costs of biodiversity strips, and private transaction costs is less than the uniform payment level are assumed to participate in the programme. From this subpopulation of farmers the programme participants are selected from highest to lowest ratio of EBI/payment until budget limit *AEB* is reached.

In the case of *environmentally differentiated payment policy with EBI targeting*, payment level is differentiated between soil types to reflect differential biodiversity significance of soil types[12]. All farmers for whom the sum of the income forgone, the establishment and management costs of biodiversity strips, and private transaction costs is less than differentiated payment level for a given soil type are assumed to participate in the programme. From this subpopulation of farmers the programme participants are selected from highest to lowest ratio of EBI/payment until budget limit *AEB* is reached.

In the case of *differentiated payment policy on the basis of compliance costs with EBI targeting*, payment level is differentiated between soil types to reflect differential income forgone of establishing biodiversity strip. All farmers for whom the sum of the income forgone, the establishment and management costs of biodiversity strips, and private transaction costs is less than differentiated payment level for a given soil type are assumed to participate in the programme. From this subpopulation of farmers the programme participants are selected from highest to lowest ratio of EBI/payment until budget limit *AEB* is reached.

Three types of auction design alternatives are analysed in the empirical section:

- *One-dimensional bid-scoring index auction*: farmers' expectations regarding bid cap are formed on the basis of compliance costs and it is assumed that farmers have identical beliefs regarding variation in compliance costs (variation by 20% around the mean).
- *Two-dimensional bid-scoring index auction*: farmers' expectations regarding bid cap are formed on the basis of the ratio of bid to EBI (with assumed variation by 20% around the mean).
- *Group specific one-dimensional bid-scoring index auction*: farmers' expectations regarding bid cap are formed on the basis of compliance costs and it is assumed that farmers in a certain group have identical beliefs regarding variation in compliance costs (variation by 20% around the mean). In group specific auction, assumed low compliance costs farmers (those from lowest productivity soil type, that is, sandy soils) form expectations on the basis of heterogeneity within their group and without observable heterogeneity of higher cost bidders in other soil types. This type of auction should increase competition within low cost bidders and reduce their information rents.

In addition to various payment types a theoretical cost-effectiveness benchmark is calculated in the empirical section. The cost-effectiveness (C-E) benchmark describes a theoretical situation where the Government (buyer of the environmental good or service) knows each participant's compliance cost and offers a payment that exactly compensates compliance costs. In this case farmers' information rents (difference between payment level and compliance costs) are zero. Farmers are selected to the programme on the basis of the ratio EBI/payment from highest to lowest until the budget limit is reached. This C-E benchmark maximises budgetary cost-effectiveness.

4.2. Data

For a quantitative illustration of the theoretical framework Finnish data is employed. A 5-meter wide biodiversity strip on a field-forest border was selected as a biodiversity enhancing measure.

Following Miettinen et al. (2014), the biodiversity strip is sown with grass seed mixture (*Trifolium pratense, Phleum pratense, and Festuca pratensis*). Grass seed mixture costs EUR 77 per ha. In

12. See Biodiversity benefit index in Table 2.

addition to seed costs also tilling and sowing costs, as well as, potential harvest value of dry hay are taken into account. Total establishment costs are EUR 288/ha and annual management and labour costs are EUR 39/ha and EUR 182/ha, respectively. Profit margin (total revenue including decoupled agricultural support payments and revenue from dry hay minus variable costs and labour costs) for establishment and management of biodiversity strip is EUR 234/ha.

Spring barley (*Hordeum vulgare*) is selected as a representative crop for this application as it is the most common crop covering about half of the field parcels under crop cultivation in Finland. The soil type under cultivation is assumed to be one of the three most common soil types; clay, sandy and organic soil. Clay and silt soils cover over half of the cropland area in Finland, organic about 14% and sandy soils 35% (Puustinen et al., 2010).

Per hectare crop yield is modelled as a function of nitrogen fertilizer application. Mitscherlich yield function is applied for spring barley to define the optimal fertiliser application and yield level.

$$f(l;q) = \varphi(1-\sigma \exp(-\rho l)) \tag{5}$$

Where l is nitrogen application rate and φ, σ and ρ are parameters. The yields depend on soil type; clay > organic > sandy as well as soil productivity. Soil productivity differences are incorporated through maximum yield parameter, φ.

Production cost data for barley cultivation is based on Tuottopehtori (2010).

Biodiversity benefits from a field-forest border biodiversity strip are based on biodiversity benefit index developed in Iho *et al*. (2011) for the purposes of Finnish agri-environmental auctions.

Table 2. Biodiversity benefit index

Soil type	North-West and North-East			East and West			South-West and South-East		
	Field slope			Field slope			Field slope		
	<1.5%	1.5-6.0%	>6%	<1.5%	1.5-6.0%	>6%	<1.5%	1.5-6.0%	>6%
Organic	13	19	27	19	28	40	24	34	49
Clay	19	27	39	28	40	57	34	49	70
Sandy	27	39	55	40	57	82	49	70	100

Source: Iho et al. (2011).

In this index three characteristics describe biodiversity benefit of a biodiversity strip on a field-forest border: soil type, field slope and opening direction of the forest border. Forest borders opening to the South have usually higher plant and insect species richness and abundance than forest borders opening to other compass directions (Bäckman et al., 2004). Species richness of rare meadow plant species is usually highest in steep slope meadows opening to the South (Luoto, 2000 and Pykälä et al., 2005). Also butterfly species richness correlates with the slope of the field edge (Kuussaari et al., 2007). Soil type (soil textural class) and its chemical composition have large effect on plant species richness (and through that on diversity of fauna). On sandy soils plant and butterfly species richness is usually higher than in other soil types (Kivinen et al., 2006). Generally, plant species richness increases when soil nutrient level decreases, especially in the case of phosphorus.

4.3. Results

One hundred differential soil type/land productivity combinations were analysed. Table 3 provides a summary of basic results (minimum, mean and maximum) regarding nitrogen application, barley yield, profits and profit forgone.

Table 3. Nitrogen application, barley yield, profits, and profit forgone for establishing a 5-meter wide biodiversity strip (minimum, mean, and maximum)

	Nitrogen application kg/ha	Barley yield kg/ha	Profits EUR/ha	Profit forgone EUR/ha
Minimum	95	3 452	482	45
Mean	109	4 586	604	51
Maximum	121	5 666	722	57

Farmers' profit forgone from establishing a 5-meter wide biodiversity strip varied between EUR 45 and EUR 57 per hectare.

The agri-environmental budget (*AEB*) was set at EUR 2000 and government selection of participants for various payment types followed the procedure described in Section 4.1. Uniform payment level is fixed at EUR 52/ha. Environmentally differentiated payment level varies from EUR 48/ha to EUR 54/ha depending on soil type. It is highest for sandy soils and lowest for organic soils on the basis of biodiversity benefit index in Table 2. Differentiated payment on the basis of compliance costs varies from EUR 48/ha to EUR 54/ha. It is highest for organic soils and lowest for sandy soils.

Policy-related transaction costs (PRTCs) for administration and farmers' private transaction costs were calculated for different payment types on the basis of Finnish studies (Ollikainen at al., 2008; Lankoski et al., 2010, Iho et al., 2011). The share of PRTCs of total payment transfers are reported in Table 4.

Table 4 provides main results regarding PRTCS, the number of selected participants under different payment types, total budget costs, total EBI points, the cost-effectiveness and targeting gains ratio.

As shown by earlier studies identified in the literature, the uniform payment policy (which does not employ systematic selection of participants on the basis of the cost-effectiveness) performs less efficiently than other payment types. This is indicated by its high cost per EBI point and the low performance relative to C-E benchmark. The one-dimensional bid scoring auction performs much better than the uniform payment as farmers are selected on the basis of benefit/cost ratio[13]. The two-dimensional bid scoring auction performs only slightly better than uniform payment and clearly worse than one-dimensional bid scoring auction. The reason for this was shown in the theoretical framework, where farmers have access to EBI information, higher EBI increases bids and thus information rent for those farmers who are selected. This is confirmed in the first and second column of the Table 5 which shows that relative to one-dimensional bid scoring auction farmers' information rents are twice as large under two-dimensional bid scoring auction. Increase in information rents clearly decreases the performance of two-dimensional bid scoring auction. Employing group specific one-dimensional bid scoring auction for low cost bidders reduces bidders' information rents relative to one-dimensional bid scoring auction without group specific competition. Overall performance of these two different one-dimensional bid scoring auctions is quite similar. Using EBI targeting as a part of uniform payment policy greatly improves the cost-effectiveness of uniform payment. In fact, it is ranked third in cost-

13. Relative to uniform payment the cost-effectiveness gain from the one-dimensional bid scoring auction is 16%. This cost-effectiveness gain is smaller than figures presented in the review of literature (paragraphs 64-66) and is mainly explained by small variation of income forgone for the measure in this application.

effectiveness. Differentiated payments, either on the basis of compliance costs or environmental benefits, perform slightly better than uniform payment without EBI targeting but clearly less efficiently than one-dimensional bid scoring auctions (with and without group specific competition) and uniform payment with EBI targeting.

Table 4. Share of policy related transaction costs of total payment, number of selected participants for different payment schemes, total budget cost, total EBI points, cost-effectiveness and targeting gains ratio

Payment	Share of PRTCs of total payment transfer, %	Number of participants selected	Budget cost, EUR	EBI points	Cost-effectiveness, EUR/EBI point	C-E relative to C-E benchmark %	Targeting gains ratio[1]
C-E benchmark	6.1	37	1 992	2 885	0.73	100%	25
Uniform payment	5.0	38	1 976	2 192	0.95	77%	-
One-dimensional bid scoring auction with EBI targeting	7.0	35	1 952	2 599	0.80	91%	8
Two-dimensional bid scoring auction with EBI targeting	7.0	31	1 964	2 337	0.90	81%	3
Group specific one-dimensional bid scoring auction with EBI targeting	7.7	36	1 986	2 660	0.80	91%	7
Uniform payment with EBI targeting	5.4	38	1 976	2 476	0.84	87%	28
Environmentally differentiated payment with EBI targeting	6.1	37	1 982	2 254	0.93	78%	2
Differentiated payment on the basis of compliance costs with EBI targeting	6.1	36	1 986	2 660	0.90	81%	5

1. C-E gains from EBI increase relative to the increase in PRTCs with uniform payment as a benchmark.

The results also highlight that the potential cost-effectiveness gains achieved using auction systems can be uncertain. They depend on detailed auction design and farmers' information and assumptions regarding selection criteria (e.g. criteria used for ranking and selecting bids and information provided regarding maximum acceptable bids and environmental scores). Auctioning can be the most cost-effective payment system if farmers bid based on compliance costs assumptions without any knowledge of EBI. But auction systems can also perform below uniform payment with EBI targeting if farmers have knowledge about EBI levels and their use in bid selections. This means that (i) there is uncertainty on these cost-effectiveness gains, and (ii) these gains can diminish over time when farmers' knowledge of selection criteria increases.

The last column of Table 4 shows the targeting gains ratio of different payment types. The targeting gains ratio identifies the cost-effectiveness gains from EBI targeting relative to the increase in PRTCs. Uniform payment without EBI targeting is a benchmark for calculating this ratio (i.e. the ratio of the difference in the value of EBI points and PRTCs for a given payment type and uniform payment). Targeting gains ratio varies greatly between different payment types. It is highest for uniform payment with EBI targeting in which case one EUR spent on PRTCs pays back EUR 28 through cost-effectiveness gains from improved environmental targeting. Hence, uniform payment with environmental targeting may be a good option in a situation where a government aims to improve environmental effectiveness and cost-effectiveness of the payment programme with relatively small additional expenditure on PRTCs[14]. One-dimensional auctions also perform well with regard to targeting gains ratio while this ratio is lowest for environmentally differentiated payment with EBI targeting.

Due to relatively small differences in PRTCs between different payment types the cost-effectiveness ranking of the payment types is only slightly affected by inclusion or exclusion of PRTCs (see Annex A: Table A.1.)

Table 5 provides results regarding information rent.

Table 5. Information rents under various payment types

Payment	Information rent		
	EUR/ha	%	Total
C-E benchmark	-	-	-
Uniform payment	4.5	8.6	169
One-dimensional bid scoring auction with EBI targeting	5.9	10.7	208
Two-dimensional bid scoring auction with EBI targeting	14.2	21.6	439
Group specific one-dimensional bid scoring auction with EBI targeting	5.2	9.5	189
Uniform payment with EBI targeting	3.8	7.3	145
Environmentally differentiated payment with EBI targeting	4.2	7.8	154
Differentiated payment on the basis of compliance costs with EBI targeting	2.8	5.5	109

Figure 1 shows the development of farmers' information rents in different auction formats. As indicated by theory and empirical results two-dimensional bid scoring auction results in highest information rents. As regards one-dimensional bid scoring auctions the group specific auction slightly reduces farmers' information rents relative to open one as it increases competition within the lowest cost sellers. Reduction of farmer's information rent (that is, the difference between the payment received for participating and true cost of participation for farmers) is important if the government's objective is to maximise budgetary cost-effectiveness of the program and to avoid overpayment of certain farmers.

The data used in the results above portray a relatively limited variability in profits forgone for the establishment and management of biodiversity strip, which favours payment types that do not address so well heterogeneity in compliance costs. This is particularly the case for uniform payment with EBI targeting. Its relatively good performance could well be linked to this limited variability. In order to test the influence of the variability of income forgone a sensitivity analysis is conducted. In this sensitivity analysis the variation in income forgone is increased by 35%.

Table 6 provides the results from sensitivity analysis (variation of income forgone increased by 35%).

14. Uniform payment with environmental targeting is environmentally effective if farmers enrol in the programme. This depends on the level of payment relative to the distribution of opportunity costs. This is naturally true for all voluntary approaches.

Figure 1. Information rent in different auction formats

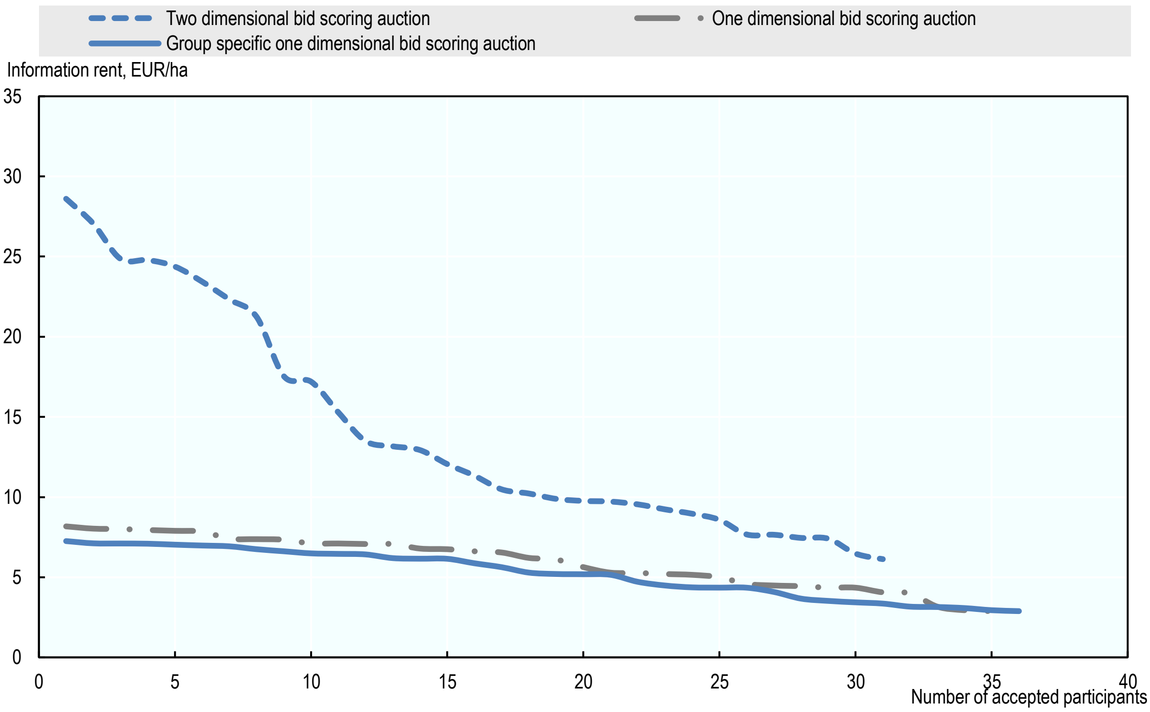

Table 6. Sensitivity analysis (variation of income forgone increased by 35%)

Payment	Budget cost	EBI points	Information rent	Cost-effectiveness EUR/EBI point	C-E relative to C-E benchmark	Targeting gains ratio[1]
	EUR	#	EUR/ha	EUR	%	
C-E benchmark	1964	2719	-	0.72	100%	24
Uniform payment	1976	2069	8.5	1.00	72%	-
One-dimensional bid scoring auction with EBI targeting	1954	2586	5.8	0.76	95%	10
Two-dimensional bid scoring auction with EBI targeting	1975	2350	14.0	0.90	80%	6
Group specific one-dimensional bid scoring auction with EBI targeting	1942	2586	5.4	0.81	89%	8
Uniform payment with EBI targeting	1976	2340	7.2	0.89	81%	27
Environmentally differentiated payment with EBI targeting	1951	2424	6.5	0.85	85%	7
Differentiated payment on the basis of compliance costs with EBI targeting	1988	2423	5.7	0.87	83%	12

With a higher variability of income forgone, the relative cost-effectiveness of uniform payment without EBI targeting is reduced. The relative performance of one-dimensional bid scoring auction is further increased (it remains the most cost-effective option). Group specific one-dimensional bid scoring auction performs relatively less well with higher variability of income forgone. Relative cost-effectiveness of uniform payment with EBI targeting is reduced and both differentiated payments perform slightly better than it does. Overall, cost-effectiveness gains from targeting modalities are increased. However, with larger variations of income forgone, the variability of and the uncertainty on auction performance also increases.

If the focus is put on less uncertain instruments (in terms of efficiency gains) then uniform payments with EBI targeting appear to be the best "secure" option in the base case analysis, while larger variation on income forgone seem to favour environmentally differentiated payment with EBI targeting.

Additional sensitivity analysis was conducted regarding PRTCs. Robustness of results regarding gains from targeting versus PRTCs were analysed by increasing PRTCs by 100%. Results are reported in Annex A: Table A.2. These results show that even with 100% increase in PRTCs gains from targeting clearly outweigh increase in PRTCs for most of the payment types.

While the empirical application conducted in the study is illustrative and focuses only on one type of biodiversity enhancing measure, some general conclusions and policy implications can be drawn:

- Both the base case analysis and the sensitivity analysis show that – on the basis of Finnish data – the gains from environmental targeting are potentially very large and clearly outweigh the increase in PRTCs when targeted payment types are implemented. This confirms results from the literature review.
- The empirical application also shows that the cost-effective policy design to address heterogeneous agricultural and environmental conditions requires the combination of differentiated payment level and environmental targeting, for example by employing one-dimensional bid scoring auction with EBI targeting or differentiated payment with EBI targeting. It was shown, in the case of uniform payment, that large efficiency gains can be rapidly achieved at low PRTC by using environmental targeting alone, but the most cost-effective policy design and implementation requires addressing also differences in compliance costs.
- Targeted payments can be used as effective and cost-effective mechanisms to complement environmental regulations and environmental cross-compliance schemes that provide base level biodiversity protection in agriculture.
- When government objective is to maximise budgetary cost-effectiveness of these targeted payments (biodiversity protection and enhancement with given budget) then environmental targeting and tailoring of payment level in accordance with farmer's compliance cost are key factors for success of policy.

New payment approaches reviewed in this paper, including agglomeration payment, results-based payment and conservation auction require sufficient testing or piloting in the field before extensive implementation can be considered. Before this piloting phase, key policy design and implementation features can be analysed and tested using theoretical modelling, policy simulations and economic experiments (laboratory experiments). Moreover, both biodiversity indicators selected for results-based payments and environmental benefit indices used for environmental targeting need supporting ecological data (and other data as well) in order to be consistent with policy objectives and be measurable at reasonable cost.

ANNEX A

Table A.1. Cost-effectiveness with and without PRTCs

	With PRTCs EUR/EBI point	Without PRTCs EUR/EBI point
	EUR	EUR
C-E benchmark	0.73	0.69
Uniform payment	0.95	0.90
One-dimensional bid scoring auction with EBI targeting	0.80	0.75
Two-dimensional bid scoring auction with EBI targeting	0.90	0.84
Group specific one-dimensional bid scoring auction with EBI targeting	0.80	0.75
Uniform payment with EBI targeting	0.84	0.80
Environmentally differentiated payment with EBI targeting	0.93	0.88
Differentiated payment on the basis of compliance costs with EBI targeting	0.90	0.85

Table A.2. 100% increase in PRTCs

	With PRTCs EUR/EBI point	Targeting gains ratio
	EUR	
C-E benchmark	0.77	13
Uniform payment	0.99	-
One-dimensional bid scoring auction with EBI targeting	0.86	4
Two-dimensional bid scoring auction with EBI targeting	0.96	2
Group specific one-dimensional bid scoring auction with EBI targeting	0.86	3
Uniform payment with EBI targeting	0.88	15
Environmentally differentiated payment with EBI targeting	0.99	1
Differentiated payment on the basis of compliance costs with EBI targeting	0.95	3

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