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Why be wasteful when preserving a valuable resource? – A review article on the cost-effectiveness of European biodiversity conservation policy

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Why be wasteful when preserving a valuable resource?
A review article on the cost-effectiveness of European biodiversity conservation policy

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Abstract:

Issues related to the cost-effectiveness of biodiversity conservation policies have not yet been prominent in European conservation research and policy-making. Nevertheless, there is a small but growing literature which analyses such cost-effectiveness issues on both a conceptual and an applied level. The article reviews this literature, and focuses on reserves and compensation payments for conservation measures as the two most relevant conservation policy instruments in Europe. Progress has been achieved in understanding the cost-effective allocation of conservation measures and reserve sites, and further advances can be expected by integrating knowledge from ecology and the neoclassical analysis of policy instruments. Research on cost-effective monitoring, enforcement and decision-making has addressed selected issues such as designing incentives for farmers to reveal their conservation costs to the regulator. However, issues with high relevance for European conservation policy such as the cost-effectiveness of compensation payments for results and implementation problems related to the network NATURA 2000 have been neglected.

Key words:

Cost-effectiveness, conservation, biodiversity, policy instruments, Europe

1. Introduction[1]

Issues of cost-effectiveness have not yet played a prominent role in European biodiversity conservation research and policy-making. This is astonishing, as some major European biodiversity policies such as the Habitats Directive[2] and compensation payments for conservation measures based on Regulation 2078/92[3] have been in place for more than a decade. Furthermore, these instruments impose substantial opportunity costs on society in terms of foregone consumption and production (cf. Strijker et al. 2000, OECD 2001), suggesting significant room for improvement as far as cost-effectiveness is concerned. Judging by economic research on environmental problems other than biodiversity, it may well be worth addressing the cost-effectiveness of environmental policies. For example, Carlson et al. (2000) estimate that the policy of the US Environmental Protection Agency to reduce SO_2 emissions by using allowance trading may save \$700–800 million per year compared to a command and control programme based on a uniform emission standard.

There may be two main reasons why cost-effectiveness issues have been neglected in biodiversity research and policy-making. The first is that both areas have largely been dominated by natural scientists and have therefore been predominantly concerned with ecological effectiveness and attaining conservation goals.[4] More recently, attention has also been paid to issues of acceptability and conflict resolution, since the implementation of biodiversity policies has frequently encountered some resistance.[5] The second reason why issues of cost-effectiveness have not figured prominently may be that thoroughly analysing the cost-effectiveness of biodiversity policy often requires combining expertise in ecology and economics – an area where, despite all the interdisciplinary rhetoric, not much work has been done.

Despite these obstacles, there is a small but growing body of literature which analyses cost-effectiveness issues in relation to biodiversity conservation policies on both a conceptual and an applied level. The aim of this article is to review this literature and to point out some promising avenues fur further research. The article dwells on the two most relevant conservation policy instruments in Europe: reserves and compensation payments for

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[1] We are grateful to Wanda Born for helpful comments.
[2] Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.
[3] Regulation (EEC) No. 2078/92 of 30 June 1992 on agricultural production methods compatible with the requirements of environmental protection and the maintenance of the countryside.
[4] Examples of research in these fields include Ovenden et al. (1998) and Kleijn et al. (2001).
[5] Research in this field includes O'Riordan and Stoll-Kleemann (2002) and Young et al. (2003).

conservation measures (referred to below simply as 'compensation payments'). On a European level, reserves are particularly important in the context of the Birds and the Habitats Directives and compensation payments in the context of agri-environmental schemes.

The article is structured as follows. Section Two provides some background information on European biodiversity policy. Section Three sketches a framework for the cost-effectiveness analysis of reserves and compensation payments. Based on this framework, the literature on the cost-effectiveness of compensation payments is reviewed in Section Four and on the cost-effectiveness of reserves in Section Five, with selected avenues for further research being identified in each case. Section Six summarises the main results.

2. European biodiversity policy

The purpose of this chapter is to provide some background information about the need for compensation payments and reserves in connection with European biodiversity conservation and to briefly outline the main European legislation relevant to the implementation of these two instruments.

2.1 Agri-environmental schemes targeted at biodiversity conservation

Without human intervention, most parts of Europe would still be covered with forests (Ellenberg 1996, Willmanns 1998). Human influence began when the landscape was first cleared for farming about 6000 years ago. New species immigrated when the first crops were imported from the steppes and semi-deserts of Central Asia and the Mediterranean area (Fukarek 1995). At the same time, the grazing of cattle, sheep, goats and pigs started to thin out the forests. Due to constant grazing, treeless spaces were inhabited by herbs and grasses, which led to the creation of pasture land. Although meadows originated in Europe under Roman influences, most meadows are no more than 1000 years old. Even so, this time was long enough for nearly all meadow plants to have originated in the natural flora, and only very few migrated from East European steppes (Fukarek 1995). Due to the varying types of agricultural land use, until the 19th century biological diversity in terms of habitat types and numbers of species was generally on the increase (Cox et al.1973, Kaule 1996).

This trend was completely reversed during the course of the 20th century when technological developments in agriculture transformed the majority of the centuries-old Middle European cultural landscape into uniform production areas. Intensive fertiliser and pesticide use, the levelling of water levels and the destruction of natural and man-made landscape structures such as wet sinks, hedges and stone walls led and are still leading to the destruction of many habitats (Hampicke 1991). Halting this destruction is vital for European biodiversity as due to the above developments agricultural land use has a key influence on biodiversity in Europe, and much of the biodiversity-rich land in the EU continues to depend on low-intensity farming (Wiseman and Hopkins 2001, Howe and Perkins 2001).

Changing agricultural land-use practices to make them more conducive to biodiversity conservation is usually costly due to foregone agricultural production. Since the political will often exists that land-owners and in particular farmers should not be made to shoulder the burden of these costs, compensation has to be paid (Bromley and Hodge 1990, Hanley et al. 1998, Hanley and Oglethorpe 1999). This reasoning has led to the development of ‘agri-environmental schemes’, in which farmers are paid to adapt the management of (parts of) their farms to benefit biodiversity, the environment or the landscape while allowing a wide range of measures depending on the aim, country or region (Kleijn et al. 2001). Throughout the EU, agri-environmental schemes are mostly voluntary management agreements taking the form of state-farmer contracts (Van Huyelenbroek and Whitby 1999).

Agri-environmental schemes were implemented on a large scale in Europe following the enactment of Regulation 2078/92, in which they figure as part of the “flanking measures”, which also include early retirement and afforestation. The regulation provided for the co-financing of compensation payments to farmers providing environmental services by up to 75% from the EU (European Commission 2000). Although the regulation gave some general guidelines, the details of the compensation payments were left up to the individual Member States (European Commission 1992, Meyer-Marquart 2000). Regulation 2078/92 expired in 1999; agri-environmental schemes are now based on Regulation 1257/99,[6] which was developed in connection with Agenda 2000 and amalgamates several rural development programmes. Being part of the ‘second pillar’ of the Common Agricultural Policy has bestowed a higher priority on agri-environmental schemes. Further changes are expected to follow the Agenda 2000 mid-term review, the results of which had not been published at the time of writing.

To ensure that the requirements for financial support are met, the European Commission enacted Regulation 445/2002[7] specifying administrative and on-the-spot checks hat are supposed to encompass at least 5% of all beneficiaries each year.

[6] Regulation (EC) 1257/1999 of 17 May 1999 on support for rural development from the European Agricultural Guidance and Guarantee Fund (EAGGF) and amending and repealing certain Regulations.

[7] Regulation (EC) No. 445/2002 of 26 February 2002 laying down detailed rules for the application of Council Regulation (EC) No. 1257/1999 on support for rural development from the European Agricultural Guidance and Guarantee Fund (EAGGF).

2.2 Birds Directive, Habitats Directive and NATURA 2000

Whereas many endangered habitats and species in Europe require certain land-use measures to be carried out on a regular basis in order to survive, others need reserves for their protection where human influence is limited to the extent necessary depending on the habitat or species to be conserved. For example, the habitat quality of raised bog is inversely related to human disturbance, while the black stork (*Ciconia niger*) requires undisturbed forest patches to breed. However, in many cases the designation of reserves alone is not sufficient to protect endangered species or habitats; frequently additional management activities are required, which may be initiated by the instrument of compensation payments (for an example see Dubgaard et al. 1994, p. 39). While in reality the two instruments – reserves and compensation payments – often complement each other, in the remainder of this paper we make a clear distinction between them for sake of analytical clarity.

When it comes to protecting Europe's natural habitats and endangered species, the Habitats Directive is, along with the Birds Directive,[8] the most important instrument. The Birds Directive focuses solely on the protection of birds and their natural habitats, and calls for the establishment of protected areas to ensure the conservation of all European bird species, especially the species specified in Annex I, whose populations need special protection measures. The aim of the Habitats Directive is to cover European natural habitats and endangered species as a whole. The natural habitat types to be conserved are listed in Annex I of the Habitats Directive, while the animal and plant species are listed in Annex II. Given the threat to which they are exposed, certain species are classified in Annex IV as priority species. Taken together, the two directives are designed to create a coherent network of protected areas known as NATURA 2000.

The Birds and the Habitats Directives had to be transposed into national law and implemented by each member state. The criteria for the selection of protected areas are based solely on ecological aspects, although Article 2 of the Birds Directive and Article 2(3) of the Habitats Directive state that measures taken pursuant to these Directives must take into account economic, social and cultural requirements.

3. A framework for analysing the cost-effectiveness of biodiversity policy instruments

Analysing the cost-effectiveness of biodiversity policy instruments has several dimensions, each of which is typically addressed separately in the literature. In Section 3.1 a framework is

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[8] Council Directive 79/409/EEC of 2 April 1979 on the conservation of wild birds.

presented which allows the various dimensions to be systematically gauged by distinguishing between different cost categories. Each of these categories (production costs, implementation costs and decision-making costs) is considered in more detail in Sections 3.2, 3.3 and 3.4 respectively.

3.1 Overview of the framework

We present a framework based on Birner and Wittmer (2004), who originally developed it for the cost-effectiveness analysis of governance structures for natural resource management in developing countries. We chose Birner and Wittmer's approach because it systematically covers all the relevant aspects of the cost-effectiveness of policy instruments and can be easily adapted to the analysis of European conservation policy. Within the framework the following three cost categories are distinguished: production costs, implementation costs and decision-making costs.

Using these categories and depending on the political circumstances, two equivalent definitions of cost-effectiveness can be formulated. Firstly, a conservation policy x can be considered more cost-effective than a conservation policy y if the sum of the production, implementation and decision-making costs for policy x is lower than for policy y to achieve a given conservation goal. This definition is useful in a situation with a given conservation aim such as ensuring a certain survival probability of an endangered species and we want to find out how this goal can be achieved as inexpensively as possible. The second definition is that a conservation policy x can be considered more cost-effective than a conservation policy y if it generates a higher level of conservation for a given amount of production, implementation and decision-making costs. This definition is useful for a situation where society is willing to devote a certain amount of financial resources for conservation and we want to maximise the conservation output. For simplicity's sake, in the remainder of this section we only refer to the first definition of cost-effectiveness.

3.2 Production costs: allocation of conservation measures in space and time

The category of production costs refers to the costs of the actual conservation activities that have to be performed to achieve the aim of conservation. Cost-effectiveness analysis related to production costs has its roots in the traditional neoclassical analysis of policy instruments for pollution control. Here, the analysis of cost-effectiveness is focused on the allocation of abatement activities and starts from the observation that the costs and benefits of pollution abatement activities vary from one source to the next. Based on this observation, various instruments (primarily standards, taxes and tradable permits) are analysed in terms of their capacity to initiate the least-cost combination of pollution abatement activities to achieve a given pollution reduction aim (for an overview see Baumol and Oates 1988).

The criterion of cost-effectiveness with respect to production costs for biodiversity conservation requires compensation payments to initiate the combination of individual conservation measures to be carried out or the combination of individual areas to be designated reserves such that a certain conservation aim is achieved at least cost. The reason why the costs of achieving a certain conservation aim may vary is that the costs and benefits of individual conservation measures and areas may be subject to spatiotemporal variation. For instance, mowing a meadow at location x_1 at point in time y_1 may well be more or less costly and have a different effect on the survival probability of a species than mowing a meadow at location x_2 at point in time y_2. Usually, achieving a certain conservation aim entails carrying out a number of individual conservation measures or designating several areas as reserves. Hence, various combinations of conservation measures or areas may exist that are all able to achieve the conservation aim, albeit at different costs.

One important reason for spatial cost differences of conservation measures and of areas to be designated reserves is different opportunity costs for the land required for both conservation activities. Such differences stem for instance from the varying suitability of the land for economic development. Other reasons for spatial variations of cost that may be relevant with respect to conservation measures include differences in opportunity costs for labour and in the availability of equipment to carry out conservation measures. Benefit differentiation of conservation measures and areas may be caused by the relevant land's varying suitability to achieve the desired conservation aim (e.g. because of differences in habitat quality). The spatial scale of cost and benefit differentiation may differ depending on the conservation problem and range from regional differentiation to differentiation on a much smaller scale, for instance between adjacent plots of land no more than 1 ha in size.

The costs and benefits of conservation measures and areas to be designated reserves may be subject to not only spatial variation but also variation over time. Temporal variations in costs may be caused by changing opportunity costs for the land needed for both types of conservation instruments. For example, the foregone benefits of not developing land for commercial purposes may vary over time. Other reasons for temporal variations of costs for conservation measures may consist in differences in the opportunity costs for labour and in the economic losses accompanying a conservation measure. An example of such an economic loss caused by a conservation activity is the reduction in the quality of hay as fodder if the mowing date is switched from the time most suitable for the farmer to that best for conservation. The temporal scale of cost and benefit differentiation may range from a matter of days to a few years depending on the particular conservation measure concerned.

3.3 Implementation costs: monitoring and enforcement

Implementation costs occur because compliance with environmental legislation cannot be taken for granted. Realising that many environmental policies fail due to a lack of proper implementation, much attention has been devoted to the cost-effective design of monitoring and enforcement activities.

The groundwork for the economic literature on the monitoring and enforcement of legislation was done by Becker (1968). He assumed that individuals' decisions concerning legal compliance follow the same cost-benefit rationality as decisions throughout the rest of the economic sphere: individuals will violate the law if it maximises their utility, i.e. if for a given input of resources the expected net utility of an offence is higher than the expected net utility of any other legal activity. The expected net utility of an offence is subject to a variety of parameters including the expected punishment (which depends on the likelihood of getting caught and the expected fine). Governments basically have two options to reduce crime: they can step up monitoring activities and increase punishment. In terms of cost-effectiveness, the latter option is often preferable: intensifying monitoring activities is usually expensive for society, whereas increasing punishment need not be (e.g. if offenders are required to pay higher fines).

Becker's approach was soon applied to the implementation of environmental legislation (see Downing and Watson 1974 and Harford 1978 as two early examples) and nowadays there is a large body of theoretical and empirical literature addressing this issue with the aim of both explaining existing monitoring and enforcement activities and giving recommendations on how such activities can be made more cost-effective (see Cohen 1999 for an overview). If this literature is applied to the implementation of the two biodiversity policy instruments under review, it should be borne in mind that theoretical insights into the cost-effective design of monitoring and enforcement developed against the background of pollution control standards are usually also relevant for reserves. There is no difference in the structure of these instruments as both impose certain restrictions which individuals or firms have to comply with. By contrast, compensation payments are commonly voluntary and therefore differ in an important respect from the policy instruments traditionally addressed in the monitoring and enforcement literature (standards, taxes and tradable permits). The voluntary nature of compensation payments implies that the design of monitoring and enforcement activities may have repercussions on the decision of land-users to participate in a compensation scheme.

3.4 Decision-making costs

The analysis of the cost-effective allocation of abatement activities and the design of monitoring and enforcement activities is an established field in the (neoclassical) economic analysis of environmental policy instruments. By contrast, only recently has analysis of the costs of decision-making received attention in the context of new institutional economics (cf. e.g. Williamson 1998 and 1999), and despite the fact that some aspects related to decision-making costs are understood quite well, a comprehensive framework for the analysis of these costs is still lacking.

The decision-making costs arising in the context of a particular conservation policy instrument consist of the costs of acquiring the information necessary to make appropriate decisions, including scientific knowledge on natural resources, information on preferences in the case of conflicting goals and information on production costs (Birner and Wittmer 2004). The need for information on production costs arises for instance if the producer is to be compensated for conservation costs. The economic literature on pollution abatement has drawn attention to the fact that the polluter (producer) is often better informed about abatement (production) costs than the regulator, and has an incentive to lie about these costs. For example, if the regulator wants to determine an emission standard, the polluter has an incentive to overstate abatement costs, as this may lead the regulator to adopt a lower standard than would have been the case if the true costs had been known.

Decision-making costs also include the costs of co-ordinating decision-making if different individuals or groups are involved. They include the resources spent on meetings and resolving conflicts as well as costs arising due to delayed decisions. To take into account a potential trade-off between the quality of a decision and the costs arising for the decision, the loss caused by suboptimum decisions can be integrated into the proposed framework under the category “decision-failure costs” (Birner and Wittmer 2004).

The costs of decision-making are obviously relevant for the instrument of reserves as the selection of appropriate areas requires a substantial amount of information. The costs of decision-making are similarly relevant for compensation payments as their cost-effective design also requires substantial information. Furthermore, participation in payment schemes is usually voluntary and thus the farmer has to make a decision about participation which will be based on (expensive) information about the scheme.

4. Compensation payments

Based on the framework developed in the previous section, we will now review the literature on compensation payments with respect to the cost-effective allocation of conservation measures, cost-effective monitoring and enforcement activities, and cost-effective decision-making. Finally, challenges for further research are identified.

4.1 Cost-effective allocation of conservation measures

On a conceptual level, a substantial amount of research has addressed the cost-effective allocation of conservation measures in space and time. With respect to how such measures are financed, research has been partly limited to conservation funds and has partly explicitly assumed that such funds are allocated through the instrument of compensation payments. However, research referring to conservation funds is also relevant for the development of cost-effective compensation payments as it improves our understanding of what the relevant factors are that determine the cost-effective allocation of conservation measures and how they interact.

Wu and Bogess (1999) analyse how the existence of thresholds in the ecological benefit function (i.e. the presence of cumulative effects) influences the spatial allocation of conservation funds. They find that if thresholds exist, the optimum spatial allocation of limited conservation funds is such that funds should be concentrated in one region in order to exceed the threshold instead of being distributed evenly among regions. In an extension of their analysis, they integrate interaction and correlation among environmental benefits. Interaction refers to the causal relationship between direct and indirect environmental benefits, e.g. improved water quality enhances fish habitat. Its existence influences the spatial allocation of funds such that if the direct environmental benefit increases the indirect benefit at an increasing (decreasing) rate, ignoring the indirect benefit will over (under) fund regions with a lower direct benefit. Correlation refers to a situation where two environmental benefits (benefit 1 and benefit 2) are jointly produced by the same action, e.g. retiring land from crop production may improve wildlife habitat and groundwater quality. The spatial allocation of funds should then be orientated such that (I) the marginal benefit of a conservation expenditure is identical in both regions, and (II) the marginal rate of substitution between the two benefits (which measures the amount of benefit 1 a region is willing to give up for a unit of benefit 2) and the marginal rate of trade-off between the two benefits (which measures the amount of benefit 1 a region must give up in order to gain enough resources for the production of benefit 2) are identical in each region.

Drechsler and Wätzold (2001) systematically examine how the budget size, the shape of the cost functions and the shape of the benefit functions affect the cost-effective allocation of a conservation fund between two regions. Their results include that if in both regions marginal costs only increase weakly and if the benefit function increases slightly more than proportionally with habitat area, a small increase in conservation funds may require significant reallocation between areas. If the total area that can be allocated is below a certain threshold, it should all be allocated to the region with the larger initial amount of habitat area, but as soon as one more small unit of land can be transformed into habitat, even allocation becomes cost-effective.

If concerns of cost-effectiveness require a spatial differentiation of conservation measures, and if such measures are to be induced by compensation payments, such payments have to be spatially differentiated. However, when deciding whether to implement uniform or spatially differentiated compensation payments, the regulator has to balance uniform payments' disadvantages in terms of cost-effectiveness with the possible disadvantages of spatially differentiated payments, such as high administrative and information costs as well as equity concerns. To help resolve this issue, Wätzold and Drechsler (2004) provide a conceptual framework that allows the losses in terms of cost-effectiveness associated with uniform compensation payments for different types of benefit and cost functions to be assessed. Their results show that losses can vary from 0 to almost 100%. This as well as the other research cited suggests that the spatial differentiation of conservation measures and, hence, of compensation payments is important to improve cost-effectiveness.

Theoretical research on the cost-effective differentiation of conservation measures over time has received less attention. Drechsler and Wätzold (2003) analyse a situation where the survival of an endangered species depends on carrying out certain types of conservation measures on a regular basis, yet due to uncertain political commitment or an economic downturn the periodical availability of a budget to compensate landowners for such measures cannot be guaranteed. To insure against future underfunding, money may be saved for conservation in later periods. To maximise the long-term survival of the endangered species, it has to be decided in each period whether the money available should be spent now or saved for future use. The findings indicate that the survival probability of the endangered species is maximised if the money is spent as equally as possible in each period, which requires a certain amount of 'precautionary saving' to hedge against any future underfunding.

The conceptual research described above provides valuable insights into the interaction of ecological and economic factors relevant to the cost-effectiveness of compensation payments and the relative importance of the various parameters on an abstract level. However, the design of actual compensation schemes requires research which is more applied.

The first question to be addressed when determining cost-effective compensation payments is how to generate and integrate quantitative knowledge of the economic costs and ecological benefits of certain specific conservation measures. This question was tackled by Oglethorpe and Sanderson (1999), who combine an ecological and an economic model to assess the conservation benefits and costs of agri-environmental policies. The economic model is able to calculate the costs for farmers of various policies aimed at less intensive production methods, and the ecological model determines the effect of these policies on various plant communities depending on site-specific characteristics such as soil type, rainfall, gradient and altitude. While the combined ecological-economic model allows the benefits of certain conservation measures and their costs (and thus the necessary compensation payments) to be simultaneously evaluated, it does not address the issue of how to allocate conservation measures cost-effectively.

Another step towards integrating quantitative ecological and economic expertise is the research by Hanley et al. (1998), who calculate compensation payments to protect Scottish heather moorland from degeneration into grassland through overgrazing by sheep. An ecological model determines the maximum stocking rate which is compatible with moorland conservation for a selection of farms in the Shetland Islands. To calculate the opportunity costs of the necessary stocking rate reductions on a farm-by-farm basis and thus the compensation payments, a linear programming model is constructed. One important result is that necessary compensation payments significantly differ among individual farms, suggesting that the spatial differentiation of compensation payments may be important in terms of allocative cost-effectiveness. Another result draws attention to the possible distortionary effects of existing policies in the agricultural sector. For example, a reduction in overgrazing can simply be achieved by cutting CAP support payment in the form of headage payments received per ewe.

A procedure to quantitatively determine cost-effective compensation payments for species protection which are differentiated in space and time is developed by Johst et al. (2002). How the method works is illustrated through the development of a compensation payment scheme to protect the white stork (*Ciconia ciconia*). The effects of conservation measures (mowing of meadows on certain pre-specified dates) on the white stork population are calculated using an ecological simulation model and the costs are assessed via a survey carried out among farmers. The costs and benefits are input into a numerical optimisation procedure which calculates the cost-effective compensation payments in a hypothetical landscape for a range of conservation budgets.

The extent to which conservation funds may be misallocated when threshold effects, correlated benefits or indirect benefits are ignored is illustrated in a case study by Wu and Skelton-Groth (2002). Their example is investment in riparian vegetation that reduces the stream temperature necessary for salmon restoration in the Pacific Northwest. They find that significant environmental benefits may be lost if conservation funds are distributed equally and are not primarily directed at those streams where thresholds exist, i.e. where a small reduction of stream temperatures already leads to a suitable habitat for the cold-water fish targeted. The importance of not neglecting correlated benefits is shown by the fact that a decrease in stream temperature is beneficial to cold-water fish but leads to a decrease in the number of warm-water fish, which may be of relevance if such fish are valued for angling or are endangered. Taking into account indirect benefits is important as due to local differences in habitat quality an improvement in the water temperature (direct benefit) has varying effects on the salmon population (indirect benefit).

4.2 Monitoring and enforcement

Economic research on the implementation of compensation payments has addressed the issue that compliance with some conservation measures such as the reduction of fertiliser and pesticides in farming activities is often tricky for the regulator to monitor. Choe and Fraser (1998) analyse the impacts of imperfect monitoring on the cost-effective design of compensation payments. In their model, imperfect monitoring means the regulator wrongly assuming with a certain probability that farmers are failing to comply whereas in fact they are, and vice versa. They find that cost-effective compensation payments for a desired conservation level are higher under imperfect monitoring than under perfect monitoring. The reason is that farmers have to be compensated for the possibility of being misjudged and receiving no adequate payment for their efforts. This implies that for a given conservation budget, a higher level of conservation can be achieved when monitoring is straightforward rather than complicated. The results of Choe and Fraser (1998) also underline the importance of improving monitoring techniques as a way of reducing payments.

4.3 Costs of decision-making

The common starting point of research on cost-effective decision-making is the observation that each farmer should be compensated according to their individual conservation costs. If this is not the case and farmers receive payments that exceed their costs, i.e. farmers are able to earn a producer surplus, the financial budget for achieving a given level of conservation is higher than if compensation payments are based on true costs. A higher budget, in turn, leads to a welfare loss as the taxation required to finance public funds has a distortionary effect on consumption or production (Laffont and Tirole 1993). While this welfare loss provides an argument for tailoring compensation payments according to each farmer's costs for carrying out conservation measures, it may be difficult for the regulator to obtain information about these costs. The reason is that farmers are often better informed about their conservation costs than the regulator, and they know that the compensation they receive depends on the (estimated) conservation costs. Therefore, farmers have an incentive to claim higher costs than those actually incurred. This implies that to gain the information necessary for tailored compensation payments based on each farmer's conservation costs, the regulator may encounter significant information and negotiation (transaction) costs.

Whitby and Saunders (1996) explicitly addressed this trade-off between the transaction costs required to identify the lowest possible compensation payments for each farmer and farmers' producer surpluses that arise through uniform payments for all farmers in a case study with two compensation payment schemes in England. In Environmentally Sensitive Areas (ESAs) an equal amount of compensation is paid to all land-users for a package of prescribed conservation management practices whereas for management agreements on Sites of Specific Scientific Interest (SSSI) payments are negotiated with individual land-users based on their costs for conservation measures. Not surprisingly, their results show that transaction costs are higher in the SSSI scheme whereas financial transfers are higher in the ESAs scheme. On comparing both schemes, they find that the lower transaction costs are not sufficient to offset the higher transfers, and SSSI payments therefore require overall less public expenditure.

Moxey et al. (1999) explore options to design contracts for agri-environmental payments in such a way that farmers reveal hidden information about their conservation costs, and thus their producer surplus can be reduced. Using a principal-agent-model they show that it is possible (under certain parameter constellations) to design a menu of contract options which does not only induce farmers to participate voluntarily, but also to disclose some of their information on costs. This can be achieved by structuring the payments in such a way that makes it not worthwhile for farmers to apply for a "wrong" contract. However, compared to an optimal scheme where each farmer is compensated according to his conservation costs a "price" has to be paid. Designing payments according to the aim of truth revelation implies that other aims such as achieving a certain level of conservation cannot be reached to the desired extent. Still, designing payments in such a way that farmers have an incentive to reveal hidden information may have a higher cost-effectiveness compared to an undifferentiated scheme, where all farmers receive the same payments.

Latacz-Lohmann and Van der Hamsvoort (1997) analyse auctions as another possible option that encourages farmers to reveal their true conservation costs. An auction scheme exists in the US Conservation Reserve Programme where land retirement contracts are awarded on the basis of a competitive bidding mechanism. If such a mechanism were applied to compensation payments for conservation measures, farmers would indicate in their bids the level of compensation required to carry out a certain measure and, if selected, would be paid accordingly. As farmers risk their bids not being accepted if they ask for more than the minimum compensation they actually want, there is a certain incentive for each farmer to reveal the true costs. This in turn reduces the regulator's overall payments compared to a predetermined fixed-rate payment where each farmer receives the same amount.

However, allocating conservation measures through auctions may also have disadvantages limiting their usefulness. Holm-Müller et al. (2002) point out that auctions may lead to substantial transaction costs and that there is the possibility of price-fixing in markets with only a small number of bidders. Another problem arises when the benefits of conservation measures spatially differ. In this case, the cost-effective allocation of conservation measures requires this differentiation to be integrated into the bidding mechanism, for instance through the spatial differentiation of bidding markets. This, however, may result in the number of bidders being too small for a competitive market.

4.4 Avenues for further research

Although the research presented in the previous sections is fairly comprehensive with respect to some aspects of the cost-effectiveness of compensation payments, it lacks some important dimensions and certain areas remain practically untouched. A few selected avenues for further research are identified below which in our opinion are promising.

The essential innovation when analysing the cost-effective spatial and temporal allocation of conservation measures was to combine the traditional neoclassical analysis of policy instruments with ecological knowledge. This is a common feature of all the research described in Section 4.1. The integration of ecological expertise is essential as it provides us with an understanding of the ecological effects of conservation measures, which is necessary to adequately address the problem of the cost-effective allocation of conservation measures. In this way, our understanding of how best to allocate conservation measures has significantly improved in many respects, and continuing this approach seems promising.[9] For instance, although the metapopulation concept has been analysed thoroughly in ecology (see e.g. Hanski 1999 and Drechsler et al. 2003), we know only little of how it affects the cost-effective allocation of conservation measures. It remains to be explored what further ecological findings ought to be integrated with the traditional neoclassical analysis of policy instruments to increase our understanding of the cost-effective allocation of conservation measures in space and time.

Comparatively little research has been carried out on monitoring and enforcing compensation payments. It may have been overlooked that the knowledge gained from the traditional analysis of monitoring and enforcement of environmental legislation cannot be simply applied to compensation payments. For example, the policy recommendation already made by Becker (see Section 3.3) that a cost-effective way of increasing compliance is to increase the penalty will, if applied to (voluntary) compensation payments, act as a deterrent to participation in the scheme as of a certain level. The reason is that the regulator is often unable to determine with certainty whether a farmer is failing to comply, and hence there is always the danger that farmers are given penalties even if they comply with the scheme's requirements. A lower participation rate in turn will have a negative impact on conservation and may thus outweigh the positive benefits of compliance. Although Choe and Fraser (see Section 4.2) analysed some implications of the facts that participation in payment schemes is voluntary and monitoring imperfect, an in-depth analysis of the repercussions of the voluntary nature of compensation payments on the cost-effective design of compensation payments as well as of monitoring and enforcement measures is still missing.

The research reviewed addressed decision-making costs exclusively with respect to how farmers can be induced to reveal their conservation costs to the regulator. Other types of decision-making costs such as the costs incurred by farmers in order to find out about compensation schemes, and administrative costs, including those for choosing the best location for conservation and for selecting adequate conservation measures, have not been addressed in a similar fashion, despite the fact that the Member States differ in their institutional arrangements and administrative procedures to address issues related to these costs. According to the Agriculture Working Group, IUCN (2000), four different types may be distinguished: (I) The Sweden Type, where farmers are responsible for all aspects of land designation, the selection of measures and the completion of an application form. Extensive training is provided, but the administrative effort is low due to the fact that there are no farm visits or detailed discussions. (II) The Greece/Ireland Type, where reliance is placed on experts who draw up agreements for a fee or a percentage of payments. (III) The Italy Type, in which the farming union is involved with promotion, agreements, measures and even as an agent for payments. (IV) The United Kingdom Type, in which expert officials to agree contracts with each farmer on an individual basis, and experts also provide technical advice, causing high administrative costs.

It would make sense to analyse the advantages and disadvantages of these various types in order to identify “best practice” in terms of cost-effectiveness, taking into account that different institutional arrangements, administrative procedures and decision-making processes have repercussions on the expected level of compliance as well as on the allocation of conservation measures.

The underlying assumption of the research on compensation payments reviewed in Section Four is that compensation is paid when certain *measures* for conservation are carried out. However, there also exist a few compensation schemes in Europe (e.g. the MEKA II programme in Baden-Württemberg, Germany, and several programmes in Swiss cantons[10]) where compensation is paid for *results*. In these cases, for instance, compensation is paid to

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[9] This supports the notion of authors who argue in favour of more integrated research between ecologists and economists to improve our understanding of the interaction between ecological and economic systems (see e.g. White 2000 and Perrings 2002).
[10] See Oppermann and Briemle (2002).

farmers not for creating suitable habitat for an endangered plant but for the actual presence of the plant on their fields. Payments for results prevent losses due to the non cost-effective spatial allocation of conservation measures because they lead those farmers to produce the desired ecological results that can do so at least cost. The disadvantages of compensation payments for results include high administrative costs for verifying whether the ecological results have actually been achieved (e.g. whether an endangered plant is actually growing) and that farmers may demand a risk premium on top of costs for conservation measures, as payment depends not only on farmers' efforts (as in the case of payments for measures) but also on fluctuating exogenous influences such as weather conditions (Wätzold and Drechsler 2003). The arguments reported here need to be analysed in depth and augmented by a comprehensive analysis of the cost-effectiveness of the two alternatives. This would allow a thorough evaluation of payments for results as a promising alternative to payments for measures.

5. Reserves

Following the same structure as in the last section, the literature on reserves will be reviewed in terms of the various cost categories. However, as research on the cost-effectiveness of monitoring and enforcement activities and of decision-making in the context of reserves is very rare, both types of cost categories will be considered together. Finally, challenges for further research will be identified.

5.1 Cost-effective allocation of reserve sites

In conservation biology, one typical approach to selecting reserves is to cover the maximum number of species given a constraint on the total number of sites that can be included (see e.g. Willis et al. 1996). Implicit in this 'site-constrained' reserve site selection procedure is the assumption that establishing a reserve at each potential reserve site costs the same. In reality, of course, there are often large cost differences among sites. Taking into account cost differences Ando et al. (1998) solved a budget-constrained reserve site selection problem using data on the locations of endangered species and average land value by county for the United States. Under a budget-constrained reserve approach, a set of selected sites is only a feasible reserve system, irrespective of the number of selected sites, if the sum of the costs of the selected sites (in this case the land value) is less than or equal to the conservation budget. Ando et al. (1998) compared the costs of covering the same number of species under both the budget-constrained and the site-constrained approaches and found that the costs of achieving a given level of species coverage were far lower with the budget-constrained approach. For example, the cost of covering approximately half of the 911 endangered species in the database under the budget-constrained approach was less than a third of the cost of the site-constrained approach.

The conceptual framework put forward by Ando et al. (1998) to integrate economic and ecological aspects in the selection of reserves was applied by Polasky et al. (2001) to the problem of terrestrial vertebrate conservation in Oregon, United States. Focusing on an individual state rather than a nationwide level enables more detailed data to be gathered and patterns analysed at a finer geographic scale, which in turn allows a more realistic picture of possible cost-savings than with the approach proposed by Ando et al. (1998). The results support the notion that significant cost-savings can be achieved by applying the budget-constraint rather than the site-constrained approach. For example, for a certain range of species coverage, cost savings of up to 90% may be realised with the budget-constraint compared to the site-constrained approach.

5.2 Cost-effective monitoring and enforcement and decision-making

To our knowledge, no studies have analysed the cost-effectiveness of monitoring and enforcement activities with direct reference to reserves, while only one (Pouta et al. 2002) explicitly addresses issues of cost-effective decision-making related to reserve site selection of relevance in the European context.[11]

This lack of research into decision-making costs contrasts with the fact that decisions about the designation of areas as reserves in the course of the implementation of NATURA 2000 have been highly controversial in recent years and are often met with stiff public opposition. To illustrate this controversy, we briefly sketch the situation in Finland, where the implementation of NATURA 2000 was particularly contentious, with almost 15,000 letters of complaint being received by the national environment authorities (Hilden et al. 1997). An exceptionally strong protest took place in the rural district of Karvia in south-western Finland, where four landowners went on hunger strike to voice their objection to the proposed network. Landowners from other parts of Finland came out in support of the protesters in Karvia. As a result of these objections, nearly half of the areas in Karvia were withdrawn from the proposed network. Hiedanpää (2002) argues that one reason for the sharp protest was

[11] There are, however, many studies that analyse decisions related to the selection of reserve sites in Europe under other aspects than cost-effectiveness (see e.g. Stoll-Kleemann (2001) for a study focusing on the aspect of participation). Some research addresses issues of cost-effective decision-making related to reserve site selection, albeit without relevance to the European context (e.g. by concentrating on problems relevant to the USA (Costello and Polasky 2004) and Kenya (Mburu and Birner 2002)).

that people in Karvia had not been allowed to actively participate in the decision-making process, merely being invited to express their opinion regarding the proposed sites.

This argument is supported by Pouta et al. (2002), who assess the willingness to pay for an increase in nature conservation areas among Finnish citizens depending on the planning process. Their results show that respondents would be willing to pay more for nature conservation if they could participate in the planning process than with current Finnish planning procedures, which are perceived as bureaucratic. Pouta et al. conclude that a socially acceptable process of planning nature conservation is an attribute of the conservation programme itself which also has its own value. Neglecting this could lead to a negative welfare effect, which in the terminology of Birner and Wittmer may be coined "decision-failure costs" (see Section 3.4).

5.3 Avenues for further research

So far, research on the various aspects of cost-effectiveness related to reserve site selection is limited. What issues need to be addressed?

Rather like the analysis of the cost-effective allocation of conservation measures, the innovative step in analysing the cost-effective allocation of reserve sites was to integrate economic knowledge into an ecological approach to selecting reserves. This was done with respect to the allocation of reserves in space, and it may be useful to also do so with respect to the allocation of reserves over time. As explained in Section 3.2, the opportunity costs of reserves may change over time, implying that selection that was cost-effective at one point in time may not necessarily be cost-effective later on. It would be interesting to explore whether a mechanism such as a tradable habitat system can be found that allows the potential cost-savings to be exploited when opportunity costs of reserves change over time. Again, this can only be done by ecologists collaborating with economists, as economists do not know when reserves are of equal value in terms of conservation and can thus be traded. A starting-point for fruitful integration may be the concept of dynamic ecological networks, which has been developed in ecology. Ecological networks are a spatially arranged set of ecosystems of the same type which are linked within a spatially coherent system. The key feature of the networks is that they can have different spatial configurations and still serve the same conservation goal (Opdam et al. 2004).

The lack of research on cost-effective monitoring, enforcement and decision-making that explicitly addresses reserves may be explained by the fact that theoretical research on these issues which is relevant to pollution standards is also relevant to reserves. Both instruments have the same structure, i.e. they impose certain restrictions on individuals or firms. Hence, while the lack of theoretical research does not appear to be a problem, this is different for empirical research on cost-effective monitoring, enforcement and decision-making. The difficulties described above about the implementation of NATURA 2000 in Finland suggest that it is important to improve our understanding of what creates these difficulties in practice and what can be done to reduce 'decision failure costs'. Furthermore, the strong opposition of landowners to the designation of reserves suggests that even if a certain area has been designated a reserve, compliance with this decision cannot be taken for granted. To ensure that reserves are a successful instrument for conservation in Europe, we need to know whether the necessary restrictions for land use are being complied with – and, if not, how compliance can be improved.

6. Summary and conclusions

What motivated this review article was the observation that as far as biodiversity is concerned, policy issues of ecological effectiveness and social acceptability have received far more attention in research and policy-making than cost-effectiveness. This is unfortunate as ignoring issues of cost-effectiveness implies that scarce financial resources are being wasted which could be used by society for other purposes (including more conservation). Furthermore, it might boost public acceptance of conservation policies – and hence ultimately their ecological effectiveness – if such policies were considered economically sound and not wasteful (cf. Shogren et al. 1999).

For both the instruments reviewed in this paper, most research on cost-effectiveness has been carried out with respect to production costs, i.e. regarding the cost-effective allocation of conservation measures and reserve sites. The innovative aspect (and a common feature of all this research) is the combination of ecological expertise with the traditional neoclassical analysis of policy instruments. Proceeding along this route appears to be the best way forward. Two possible options would be to combine the metapopulation concept with analysis of the cost-effective allocation of conservation measures, and to integrate the concept of dynamic ecological networks with the cost-effectiveness analysis of reserve site selection.

Although some research has been carried out on the cost-effective monitoring and enforcement of compensation payments, more research is needed to better understand the implications of the voluntary nature of payment schemes on the cost-effective design of compensation payments as well as on monitoring and enforcement measures. Cost-effective decision-making has been analysed in particular with respect to how to induce farmers to reveal their true conservation costs. However, the fact that institutional arrangements for decision-making and related administrative procedures differ among European countries suggests that much can be learned from comparing these arrangements and identifying best practices in terms of cost-effectiveness. Taking into account all three types of costs, an analysis of the cost-effectiveness of compensation payments for results compared with compensation payments for measures is of high relevance as it would allow the extent to which payments for results represent an encouraging alternative to the existing model of payments for measures to be determined. So far very little research has addressed cost-effective monitoring, enforcement or decision-making in relation to reserve site selection. This contrasts with the severe difficulties encountered in some countries when implementing NATURA 2000 and suggests that the decision-making processes need to be better understood to avoid decision failure costs equating to less conservation.

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