Zur Diskussion – For Discussion

Agricultural Conservation Measures – Suggestions for their Improvement

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1 Introduction

Academic scrutiny has recently turned on payments for environmental services (PES). In the European Union (EU), they are embedded in agri-environmental programmes, which have been an important component of the Common Agricultural Policy (CAP) ever since its fundamental reform of 1992. Although agri-environmental programmes amount to just about one-tenth of the EU's main agricultural expenditures (the 'first pillar' of the CAP), they do exert substantial influence. For instance, during recent years they included some 25% of Germany's agricultural acreage.

The present contribution to the debate on PES focuses on a subset of EU agri-environmental programmes which I call agricultural conservation measures (ACMs). They aim explicitly at supporting nature conservation through the continuation or resumption of traditional land-use practices, such as haymaking, low-input pasturage, or low-input cropping along field borders that leaves room for rare weeds. Such measures are vital because the outstanding species richness of Europe's countryside depends on traditional land-use methods (HAMPICKE, 2006, 2013). Other programmes dedicated to issues such as erosion or groundwater control will be remarked on in passing.

Section 2 gives a brief overview of the academic literature on PES. In sections 3 to 6, I describe the current situation with regard to ACMs in the EU, discussing types of ACMs, the related property-rights regimes, and questions of incentives and efficiency. In section 7, I criticize the design of ACMs from a welfare-economic point of view and discuss to what extent the ensuing recommendations can be translated into real policies. Section 8 offers conclusions and economic suggestions for further development, which are supplemented by ethical considerations in section 9.

2 Academic Literature on Payments for Environmental Services

The academic discussion on payments for environmental services (PES) typically takes one of two different approaches. The first approach focuses on the design and implementation of PES from a practical down-to-earth point of view. With regard to European agri-environmental programmes, their success is questioned (see, for example MARGGRAF, 2003; KLEIJN and SUTHERLAND, 2003). The second approach discusses PES at a more abstract and general level in line with the Millenium Ecosystem Assessment (MEA, 2005).

One point of contention in the more abstract debate is the question whether PES can be described as a market-based instrument (VAN HECKEN and BASTIAENSEN, 2010; VATN, 2010) or ought rather to be understood as a Coasian approach (MURADIAN et al., 2010; TACCONI, 2012). Some authors are concerned with the commodification involved in PES, i.e. the reduction of fundamental life-supporting natural systems to tradeable items (VAN HECKEN and BASTIAENSEN, 2010; NORGAARD, 2010). They hold that this reduction is inadequate, given the complexity of ecological systems. Equity and distribution, as well as the role PES plays in income generation and poverty reduction are common features in most of this literature, since many practical examples of PES are found in developing countries (ENGEL et al., 2008; VAN HECKEN and BASTIAENSEN, 2010; ZABEL and ROE, 2009; MURADIAN et al., 2010; PASCUAL et al., 2010). Likewise, many authors discuss the role of property rights, which seem to be fundamental for the success of PES (LASCHEWSKI and PENKER, 2009).

Of special interest for the following discussion is the distinction between input-based and output-based payments (ENGEL et al., 2008) – or, in ZABEL and ROE'S (2009) terms – performance payments. While input-oriented payments reward land users who comply with certain rules of action (e.g. do not use pesticides), output-oriented payments reward those who demonstrate the desired results (e.g. a population of an endangered species on their land). Output-oriented payments have various advantages, although problems of information asymmetry, incentive, and risk need to be addressed (cf. ZABEL and ROE, 2009; MURADIAN et al., 2010; DERISSEN and QUAAS, 2013).

3 Types of Agricultural Conservation Measures (ACMs)

A typical input-based, agricultural conservation measure (ACM) is outlined in Table 1. Although "input-based" might not be the best possible term for every item in the Table, it categorizes the type of instruments reasonably well. Its principal component is a set of rules of action. If a participating farm complies with these rules, it is rewarded with a payment. The payment is based on the average costs and receipts of the process, and its purpose is to indemnify participating farms for the losses they incur by following the rules. Regular inspections are carried out in a sample of all participating farms. A large number of input-oriented ACMs are operating in EU member countries, co-financed by the EU.

A second, much less common model of ACM rewards the delivery of defined 'ecological goods'. For instance, in a programme in the German state of Baden-Württemberg, participating farms receive € 50 per hectare per year if they can demonstrate at least four species of flowers, out of a catalogue of 28, to grow in their grassland (MELRBW 1999). A similar but more refined programme is operating in Switzerland (OPPERMANN and GUJER, 2003; GUJER, 2005, 2006). Output-oriented ACMs appear preferable as they target results and should therefore be implemented more widely (GEROWITT et al., 2003). However, this result hinges on some underlying issues that I will address in the following sections.

Table 1. Example of an input-oriented Agricultural Conservation Measure

Programme: Grassland Management Conducive to Nature Conservation	
Financing:	European Union 75%, State Mecklenburg-Vorpommern, Germany 25%
Reward:	€ 205 per hectare per year
Term:	five years
Obligations:	
- no mineral fertilizing	
- no organic manuring	
- no sewage sludge manuring	
- no pesticide spraying	
- no maintenance works (rolling, levelling) between 1 April and 31 May	
- no usage (mowing, pasturage) between 1 December and 30 April	
- stocking capacity not exceeding 1,7 large animal units	
- no irrigation, no drainage	
- toleration of temporary waterlogging	

Source: Directive of January 29, 2003. AMTSBLATT (Official Gazette) MECKLENBURG-VORPOMMERN (2003): 113

4 Property Rights

According to new institutional economics, property rests upon an agreement among citizens (OSTROM, 1990; OSTROM and SCHLAGER, 1995; BROMLEY, 1997). Someone granted a property is, in other words, given the right to dispose of an asset, although this right may be restricted. Only the dominium type of property rights gives the owner unlimited power of disposal; the owner may even destroy the asset. In contrast, if property is granted as a patrimonium, the owner is free to utilize the asset (*usus* and *usus fructus*) but must not damage or destroy it (*abusus*).

In practice, there are countless fine-graded variants of dominium and patrimonium. Specifically, property rights in the European countryside are not always explicit and clear, but rather implicit and open to interpretation. For instance, Article 14(2) of the German Constitution demands that private property be used in a way that promotes the public well being, but it is not immediately clear what this means for a farmer. Moreover, property rights are not stipulated once and forever – rather, the bundle of rights granted to a specific owner may change with time. Something that used to be a dominium may be turned into a patrimonium by a change in law or regulations.

For example, there seems to be a growing implicit agreement in the EU that soil and water resources ought to be regarded as a patrimonium. Over time rules and regulations define what becomes part of 'good agricultural practice' – the accepted professional standard among farmers. Once farmers demand payments for treating soil and water properly we are on a slippery slope that parallels motorists who demand payments for stopping at the red light. Just as we do not pay the motorists, so the argument goes, we should not pay the farmers. Notwithstanding this concern, a large number of agro-environmental schemes still offer payments for erosion control, reduction of nitrogen pollution, and other restrictions. This may reflect the ongoing 'revalorisation of rural property objects', as discussed in detail by LASCHEWSKI and PENKER (2009). However, I will not dwell further on these delicate questions here.

In contrast to soil and water, the plant diversity in rural fields, meadows, and pastures is still regarded as a dominium, the only exception being woody structures interspersed in farmland. Outside protected areas, farmers are free to intensify cultivation to the point that all flowers in the grassland and all weeds in the crop land are eliminated – as long as they respect water and soil integrity and use only admissible means (specified in terms of permitted herbicides, dosage, machinery, and timing). Farmers may freely choose to participate in a programme for, say, weed protection. In this situation, it is fully legitimate for those who do participate to ask for payment. Their demand would cease to be legitimate as soon as property rights were defined so that weeds had to be tolerated on farmland, turning them, too, from a dominium into a patrimonium.

Many practitioners would protest the last remark because they regard the current property-rights regime as self-evident, as a necessary result of physical facts. This is a misconception. The rules of a society can and do change along with the prevailing value judgements they express. Society is in principle free to redefine property rights – although with regard to their consequences, some regimes may be better than others.

Along this line of reasoning, some authors (e.g. VAN HECKEN and BASTIAENSEN, 2010) have doubted the legitimacy of vesting property rights in land users so that they are entitled to receive PES. Such doubts appear justified in many cases. However, with regard to ACMs in Central Europe, the current arrangement is in fact legitimate, for three reasons.

(1) ACMs contribute to the fair distribution of conservation costs. Throughout Central Europe, farmers constitute only a small fraction of the total population (in Germany 3%). Non-farmers, i.e. the overwhelming majority, are just as responsible for the preservation of biodiversity as are farmers. ACMs spread conservation costs over the whole population rather than imposing them on a small minority. As a result, the cost per contributor is reduced to a trifle.

(2) For individual farms, maintenance of traditional land-use systems is prohibitively costly; Table 2 gives a typical example. To demand such efforts without adequate payments would be to drive many farmers into bankruptcy.

(3) 'PES are never established in an institutional vacuum' (VATN, 2010: 1247). Suppose that, for the past 200 years, farmers had been granted their property as a patrimonium subject to the duty of preserving biodiversity. Then, many mechanical and chemical implements of modern agriculture would never have been widely applied. Species living in fields, meadows, and pastures would not be endangered today, but food would be much more expensive. Perhaps the conventional agriculture of this alternative history would resemble the organic agriculture as we know it. But of course, history has taken a different course, and to ignore it, in a context that is as strongly path-dependent as social arrangements, would be to ask for unnecessary trouble. Against this background, paying farmers for nature protection, e.g. through ACMs, seems to be a wise policy.

Table 2. Cost calculation of suckler cow keeping with high value for biodiversity

	€/ha. yr
Variable costs (restocking, feed concentrate, - mineral feed, stud fee, veterinarian and - medicines, insurance, energy, water, fuel, - bedding, others, interest on working capital	300,00
Summer and winter fodder costs	367,12
Labour costs	115,00
Fixed costs	42,88
Total costs	**825,00**
Market performance	**351,51**
Deficit	**473,49**

Source and more details on suckler cows: RÜHS and HAMPICKE (2010): 356

More calculations on land-use conducive to nature conservation in HAMPICKE (2013), GEISBAUER and HAMPICKE (2012).

5 Incentives

Some farmers cooperate in ACMs out of their personal environmental convictions, but financial reasons are believed to be most important. In other words, they follow incentives. It is, however, important to distinguish between the incentive to participate and the incentive to meet a policy objective like increased biodiversity.

5.1 Incentives to Participate

The effectiveness of any supposed incentive depends on how the costs and benefits of participation in an

ACM compare to those of alternative courses of action. This is illustrated by two common cases.

(1) In agriculturally low-productive regions, e.g. in hilly or sandy country, common practice (like low stocking rates in grassland farming) already resembles what is demanded by input-oriented ACMs. Hence, farmers willingly cooperate to secure their income.

(2) In high-productive regions, the ACM payments are unattractive as farmers earn more by conventional high-input cropping. Consequently, they have little interest to accept such contracts, registering only the odd low-productive plot in the programmes.

5.2 Incentives to Achieve Success

Both input- and output-oriented ACMs offer incentives to participate, but only output-oriented ACMs offer an incentive for participating farmers to achieve an ecological success.

Farmers in input-oriented programmes sometimes frankly admit that they do not care about the success. Their attitude towards their ecological output thus stands in striking contrast to the high quality standards applied today to agricultural commodities. All that these farmers do is to execute instructions (see Table 1) to avoid sanctions. Given this lamentable state of affairs, the ecological success of input-oriented ACMs depends strictly and entirely on the design of the instructions.

Output-oriented ACMs differ in this respect by design. Because participants receive payments only if they can demonstrate the contractually agreed output, e.g. if certain plant species are present in a meadow, they are forced to take a direct interest in the success of their efforts. There are also important secondary effects, as observed in recent programmes (OPPERMANN and GUJER, 2003). Farmers who wish to participate need to acquire (often, reacquire) the ability to identify wild plant species and learn about their ecological needs to understand and adhere to the performance criteria of the ACM. This engagement changes farmers' valuations. They no longer regard meadow flowers as useless or unwanted but appreciate them as valuable. In recent years, competitions for the title of the region's most beautiful meadow have been organized in parts of Germany – events that, one or two decades earlier, would have been ridiculed by farmers.

6 Efficiency

Three types of 'efficiency' need to be distinguished: physical effectiveness, fiscal efficiency, and efficiency in welfare-economic or benefit-cost terms.

6.1 Physical Effectiveness

It is true that, given how much money has been spent on ACMs since 1992, their scholarly evaluation has been deficient (MARGGRAF, 2003). Among the existing studies, a number attempt to evaluate the physical effectiveness of input-oriented ACMs, covering most of the actual programmes. Some studies directly compare the performances of participating and non-participating farms (FEEHAN et al., 2005), others compile such evaluations, in many cases justly complaining of a lack of methodological rigour (KLEIJN and SUTHERLAND, 2003). The conclusions are mixed. Studies that describe input-oriented programmes as an unequivocal success (SCHUMACHER, 2007) are the exception. Rather, it seems that there have been failures as well as successes, and success seems to depend on the existence of a set of favourable factors, including the continuity of measures over decades and the presence of key persons who have both ecological expertise and the farmers' respect and trust.

In earlier years, complications arose from the fact that agri-environmental measures under EU Regulation 2078/1992 had three quite different objectives: nature conservation, market stabilization, and income aid. The authorities who designed specific programmes often focussed on the second and third objectives. As a result, not only was ecological output poor, but advocates of free trade who opposed any kind of subsidies also had an opportunity to denounce agri-environmental measures as plain old subsidies in a green disguise. They were not altogether wrong.

EU Regulation 1257/1999 put an end to this practice. Ever since, the objectives of market stabilization and income aid have been pursued through other instruments. This has allowed agri-environmental programmes to focus exclusively on ecological objectives. Therefore, the aspects of income generation and poverty reduction, much discussed in the international literature on PES (see section 2, above), are irrelevant for ACMs in Central Europe.

The physical effectiveness of ACMs is difficult to evaluate, and authors who are less familiar with ecological details tend to underestimate these methodological problems. One problem is the diversity of specific objectives. Even though ACMs now have an exclusively ecological purpose, different programmes serve different ecological objectives: They may aim to preserve, safeguard, and foster existing items of ecological value, e.g. populations of rare plant species. Or they may aim to reestablish such items. Ecologists recommend that priority be given to stopping the loss of biodiversity and preserving such items of ecological value as remain, because such remnants are extremely valuable as starting points for future re-enhancement. Another problem is that it is usually very difficult to prove that an ACM has preserved what would otherwise have been lost. In general, however, one might argue that ACMs have at least curbed further losses.

As for the re-establishment of certain species, ecologists are not surprised that the record has been unimpressive so far. The reason is time. Weeds do reappear promptly when herbicide spraying is abandoned, as long as the seed bank in the soil is active (LITTERSKI et al., 2005). Depending on their mobility, some animals may also reappear soon. But examples of slow recovery are much more common, especially among plants. Numerous experiments have demonstrated that the reappearance of plant species displaced from meadows and pasturages takes many years (BRIEMLE et al., 1991), often longer than most ACMs have been operating. In this context, it is rarely sufficiently appreciated that age is a fundamental quality feature of ecological structures.

In contrast to input-oriented payments, output-oriented ones are physically effective by definition, because payments are made only upon demonstration of the desired physical result.

6.2 Fiscal Efficiency

The aim of fiscal efficiency is to spend as little money as possible for a given good, and this is a very pressing public concern for obvious reasons. Public authorities, therefore, seek the cheapest way to obtain ecological services.

Economists have suggested that conservation contracts be auctioned (LATACZ-LOHMANN and VAN DER HAMSVOORT, 1997) to whoever offers to fulfil them at the least cost. In theory, it is true that competition improves both fiscal and overall economic efficiency. While there are very few practical examples to judge from, I suspect, for several reasons, that the theoretical advantages of open competition may fade away in practice (for a discussion on this see CONNOR et al., 2008). For one, economists tend to underestimate how much intricate knowledge and experience it requires from a land user to meet a well-defined ecological objective. With few exceptions, when the contract may be awarded either to a cattle breeder who has both known his pasturage for decades and been personally engaged in conservation or to some firm unfamiliar with the site, the prudent choice in terms of success will be the local farmer, even if the firm makes the cheaper offer.

Among practitioners, the debate on fiscal efficiency focuses on two questions. For one, should farmers receive incentive payments, i.e. payments in addition to the reimbursement of extra costs or forgone receipts? EU Regulations 2078 and 1257 once granted supplements of up to 20% of the true costs. Following demands by the World Trade Organization (WTO, 1994), much to the relief of treasurers, supplements were abandoned in 2005 (EU Regulation 1698). Unfortunately, this short-term increase of fiscal efficiency will likely prove very costly in the long run. From an economic point of view, the most fundamental cause of the lack of conservation in rural areas, i.e. of catastrophic losses of biodiversity, is the lack of adequate incentives. For decades, the economic setup has incited farmers not to conserve. On the other hand, if conservation paid, it would be carried out just like any other business. It is thus extremely unfortunate that the last, small financial incentive has been eliminated from ACMs.

Another very contentious problem is windfall profits. From the treasurys' point of view, payments are a waste of money if they reward farmers for ecological services that they would have provided anyway, for free. If, for instance, upland farmers have no intention of increasing fertilizer and pesticide input, why should the government pay them? The problem is aggravated by the fact that the conservation of existing items of ecological value takes priority over restoration. Yet, the very existence of valuable farmland biotopes results from their past and continuous relatively conservation-friendly management, and sometimes there is indeed little reason to believe that this will change. Nevertheless, I think that farmers who have voluntary undertaken such measures should be rewarded, for efficiency and fairness reasons.

6.3 Efficiency in a Welfare Economics Framework

The problems of incentives and windfall profits, and indeed of ACM efficiency as a whole, are best addressed in the framework of welfare economics and benefit-cost analysis. Unfortunately, the literature on PES generally argues in terms of public expenditures, relying on principal-agent or labour economics, or other approaches (e.g. ENGEL et al., 2008; ZABEL and ROE, 2009). The solutions offered are thus invariably second-best ones that fail to maximize overall social welfare. The relevant task, however, is not to please the treasurer but optimally to allocate society's resources.

Leaving aside ethical aspects (see section 9), a social optimum implies that all goods and services are supplied according to individual demands and that prices everywhere equal marginal costs (KOOPMANS, 1957). A perfect market for private goods automatically approximates this equilibrium. In contrast, the optimal allocation of public goods requires that the sum of individual bids (Lindahl prices) equals marginal costs (Lindahl equilibrium, CORNES and SANDLER, 1986). Unlike in the private-good market, there is no 'gravity' pulling the economic system as a whole towards a Lindahl equilibrium. To the contrary, information constraints, the possibility of free riding, and other circumstances provoke an under supply of public goods, so that collective institutions are necessary to correct market dynamics.

Let us start with the simplifying assumption that rural conservation is a perfect public good. In this model world, a perfectly informed and benevolent utilitarian planner (the state) ascertains how each and every citizen values the good in question. All individual demand curves are vertically aggregated into a societal demand curve, which is passed on to the suppliers of the public good. Every potential supplier, depending on their individual situation, then decides whether or not to supply a share of the good, and how much. For farms, commodities and conservation are typically rival goods; more commodities mean less conservation, and vice versa. Each farm allocates its factors so that their marginal product is the same in both commodity and conservation production (HAMPICKE, 2006), achieving an optimal mix of commodities and conservation. Still in the model world, consumers 'buy' conservation at Lindahl prices, receiving consumer surpluses just as they do on the commodity market. Producers, too, supply at marginal costs and receive producer surpluses (Ricardian rents) in both sectors. In other words, according to theory, a so-called 'windfall profit' from ACMs is nothing but a producer rent. Farmers in an ACM who produce conservation at little or no cost receive exactly the same kind of rent as farmers who happen to own good land that enables them to grow grain at low cost, but who sell their grain for the same market price as less lucky farmers (cf. RICARDO, 1817).

This is the optimum ideally achieved by the 'government-assisted invisible hand' (WELLISZ, 1964). Though based on abstract reasoning, it is a relevant result. In the realm of private goods, the market equilibrium, despite its abstractness, is the unchallenged prototype of societal organization, and many concrete economic policies are based on market metaphors. Likewise, the pure theory of public goods should serve as prototype of societal organization where public goods are concerned.

Theory needs to make certain important concessions to the empirical world. For instance, the assets involved in conservation are rarely pure public goods in SAMUELSON'S (1954) sense. They are often mutually exclusive, or complements rather than substitutes. Conservation is not the production of one homogeneous good. Therefore, competition between producers is not perfect; it is easy to find 'monopolists', e.g. the owner of the only meadow containing a rare plant. Conservation efforts are directed at entire ecosystems rather than at single organisms or populations, and are strongly restricted by irreversibilities and spatial considerations (RANDALL, 2007).

7 Theoretically Correct ACMs

In a welfare-economic framework, farmers are not 'indemnified' for losses they incur by caring for biodiversity. Rather, farmers sell ecological services just as they would sell any other good and are paid market prices.

7.1 Supply

From a welfare-economic point of view, suppliers of ecological services should behave as closely as possible to how they would in a competitive market. A perfect market is an example of unintentional and domination-free self-organization. All participants are free to choose whatever alternative they prefer, but they are unable to deliberately influence data, especially prices. All they can do is optimally to adapt to circumstances. The result is a pattern of consumer and producer surpluses that emerges spontaneously and thus reflects no single actor's intentions or power. This 'objectivity' of the market is the reason that prices are accepted by all participants and that rent distribution goes undisputed.

But in the realm of public goods, the dynamics of the free market may lead to inefficient non-cooperative outcomes like the Nash equilibrium. Cooperation is thus called for; individual demands are bundled into collective demands; and collectives exert power over the market. Accordingly, suppliers of ecological services, including farmers in ACMs, do not face an anonymous mass of individually powerless buyers, but rather a set of powerful institutions.

On the other hand, the nature of the ecological 'goods' also gives unusual power to the suppliers. Instead of a multitude of suppliers offering the same good, or very similar goods, (a situation where an auction would make sense) there is typically a small number of suppliers who control non-substitutable goods. Interestingly, this type of monopoly is quite distinct from the textbook monopoly based on market power or the 'natural' monopoly based on technical effectiveness, and it would have been inconceivable in the world of ecological plenty that still existed 100 years ago. The main reason why today's land users hold a monopoly on ecological services is the extreme scarcity of biodiversity in vast regions of Europe.

Thus, suppliers on the market for ecological services are in a similar position as suppliers on the art market, where public agents buy pieces of art for museums. Fortunately, the quality of ecological goods can be determined more objectively than the quality of art, e.g. using Red Lists.

Given that both sellers and buyers have unusual power, it is futile to hope that a perfectly competitive market for ecological goods could be established. In particular, any market that could be established lacks an automatism to determine the distribution of rents, which will therefore remain contentious. But within these restrictions, there is ample room to bring the current ACMs in the EU closer to the socially optimal situation described by welfare economics.

7.2 Demand

In a very important paper, RANDALL (2007) outlines a consistent valuation and pricing framework for non-commodity outputs. He argues that the value of ecological services should be determined by the economic sovereign, the people. That implies that the public at large should decide what share of their resources they want to spend on various forms of conservation. State authorities only serve the purpose of creating suitable institutions to bundle individual bids. (An important qualification to this principle is explained in section 9, below.) This view stands in stark contrast to the present practice in the EU.

How can we know how much people want to spend on conservation? Methods to assess people's willingness to pay (WTP) for collective goods have been applied to conservation and landscaping for several decades. The Contingent Valuation Method, the Travel Cost Method, and others have been used to capture valuations of individual species, ecosystems, and landscapes (for a recent overview see MADUREIRA et al., 2007; for early examples see NAVRUD, 1992). Meta-analysis and benefit transfer studies have been added to the field more recently (ELSASSER, 2001; NAVRUD and READY, 2007). They aim to transfer results between similar cases in a methodologically correct way, because it is impossible to conduct valuation studies for every single ecological asset. In general, methodological progress, particularly in statistics and econometrics, has improved the validity and reliability of valuation studies (for overviews see BATEMAN and WILLIS, 1999; BATEMAN et al., 2002; CHAMP et al., 2003). The perfect measurement of valuations to allow for the construction of a Lindahl equilibrium will remain an utopian goal. But for the purposes of this contribution, I assume that the accumulated knowledge on what people are prepared to pay for ecological services, especially for conservation, provides a useful first approximation.

In contrast, researchers have clearly neglected the question how the results of valuation studies can be put to good use in economic and environmental policy (HAMPICKE, 2003). Assume that a contingent valuation study provides reasonably reliable information on how much a sample of the population is willing to pay for different amounts of some ecological item. Assume also that correct means, variances, and other moments have been calculated and that we are able to make confident inferences from the sample to the parent population. Now, what do we do with these data? Asking people to donate the amount they claim to be willing to pay would invite free riding. Imposing a tax at the height of the mean WTP would be tantamount to imposing state-controlled commodity prices. Both options also suffer from the flaw that they aim to skim off the entire WTP, while in a competitive market, a consumer surplus remains. But establishing a market for ecological services with individual (non-institutional) buyers and sellers is close to impossible. As a consequence, the original motive of letting the people determine the prices is lost along the way. One way out of this problem is to use uniform price procurement auctions (see ROMSTAD, 2008).

8 Economic Recommendations

Not surprisingly, the previous section has shown that designing ACMs in strict accordance with the principles of welfare economics is impractical. Still, practice can be substantially improved by approximating theoretical standards. It is, therefore, very unfortunate that the EU Commission does not endorse output-orientied ACMs, and recent proposals for future agricultural policy suggest that this view is not likely to change soon (EUROPEAN COMMISSION, 2011). Nevertheless, I submit three contrary policy recommendations:

(1) Priority should be given to output-oriented ACMs.
(2) Suppliers of ecological services should enjoy producer surpluses.
(3) People's preferences should determine the values of ecological services.

The last recommendation is subject to an important qualification as discussed in chapter 9. I now add some details to each of my recommendations.

(ad 1) Table 3 summarizes the advantages of output-oriented ACMs, based partly on theory and partly on experience. The most important advantages of output-orientation are that land users take a genuine interest in the success of their conservation efforts, they can choose the best and cheapest 'production' method, and their spirit of innovation is incited. As noted in section 2, several authors warn of the risks inherent in output-oriented PES. For instance, it may take many years for a rare flower species to reappear on a given site following a switch of management methods, and there is a risk that it may not appear at all. Can farmers be expected to incur substantial costs year after year in the hope of a distant and uncertain reward? Future research should be directed at using output-oriented ACMs to restore valuable ecological structures, and in particular at the question how the risks can be shared fairly between suppliers and buyers. Information asymmetries to the disadvantage of buyers, though an important theoretical topic, are not a matter of practical concern in ACMs. The authorities are usually no less informed about ecologically valuable biotopes than land users are, if only because biotopes have to be registered under EU Directive 92/43/EEC (Habitat Directive).

(ad 2) Producer surpluses, or Ricardian rents, are accepted as a common and legitimate feature of any market. In fact, they are the most powerful incentive for the supply of commodities and services, for innovation, and thus for the reduction of scarcity. If conservation pays, it will be supplied. If orchid meadows generate sufficient income, they will not be converted into high-input production areas, but rather the reverse. And as more biotopes are restored, their scarcity, the owners' monopoly power, and producer rents will decrease. While these rents must always remain a matter of debate, there are comparable imperfect markets, e.g. the art market, where the same problem appears to have been settled more or less satisfactorily.

Table 3. Comparison of input and output oriented Agricultural Conservation Measures

	Input-oriented ACMs	Output-oriented ACMs
Incentive to contribute to success of measure	not extant, payment is subject to compliance EP	perfect, payment is subject to success P
Incentive to become informed on conservation matters	not extant or weak EP	high, at least target species must be known EP
Physical effectivity	dependent on design of programme, often weak E	perfect, payment is subject to success P
Versatility	not extant P	high, farm is able to choose best suited measure to achieve success P
Incentive to cooperate	not extant P	extant when success is achieved only upon combined effort of several farms P
Control of compliance	easy to difficult E	easy if easily identifiable target species are chosen E
Nearness to market principles	poor, payment is regarded as indemnification, bureaucratic way of thinking prevails E	high, payment is regarded as remuneration for valuable service just as any other EP
Compatibility to free trade	measures are prone to degenerate into ordinary income subsidies E	high in theory but not yet agreed by authorities, especially WTO

E: judgement based on empirical experience, see OPPERMAN and GUJER (2003)
P: judgement based on plausibility and theoretical prediction
Source: own presentation

Unfortunately, both the WTO and the EU reject producer surpluses from ACMs. Agricultural 'subsidies' in return for conservation are admissible only if they do not provide extra income beyond the costs (WTO, 1994, EU Regulation 1698). In my view, it is curious that economic agents should have the right to earn money everywhere except in the business of nature conservation. Governments accept as a matter of course that suppliers earn fantastic surpluses even on low-competition markets, while even modest surpluses generated from public demand are prone to elimination for fiscal discipline and other reasons. Of course, public funds should not be wasted, but modest producer rents are no wastage if they serve as incentives for the provision of important services. The meticulous skimming of producer rents in ACMs is implausible especially when subsidies ten times their size are being meted out in the 'first pillar' of the CAP, without any justification, but with the blessing of the WTO. As long as financial incentives are stripped from ACMs, their success will be severely restricted, to the disadvantage of biodiversity conservation.

(ad 3) The value of ACM output should be determined, not by the 'whims' of governments, but through state-of-the-art assessment of people's valuations. We need to develop a framework of 'green prices' for 'green outputs', based on individual valuations and marginal production costs (RANDALL, 2007: 22). 'Individuals' preferences are to count' (SAMUELSON, 1975: 223).

RANDALL (2007) is open about the difficulties that inhere in such a scheme. But they might be greater than he supposes. While my two previous recommendations could easily be implemented if only policy makers acknowledged certain economic insights, the implementation of a green pricing framework still requires substantial theoretical and empirical research. How to value ecological services in a genuinely economic way and independently of their production costs is an unresolved problem. In the few existing output-oriented ACM programmes, especially in Switzerland, 'green prices' are fixed pragmatically and certainly in view of the total available funds.

9 A Side-Glance on Ethics and Economic Conclusions

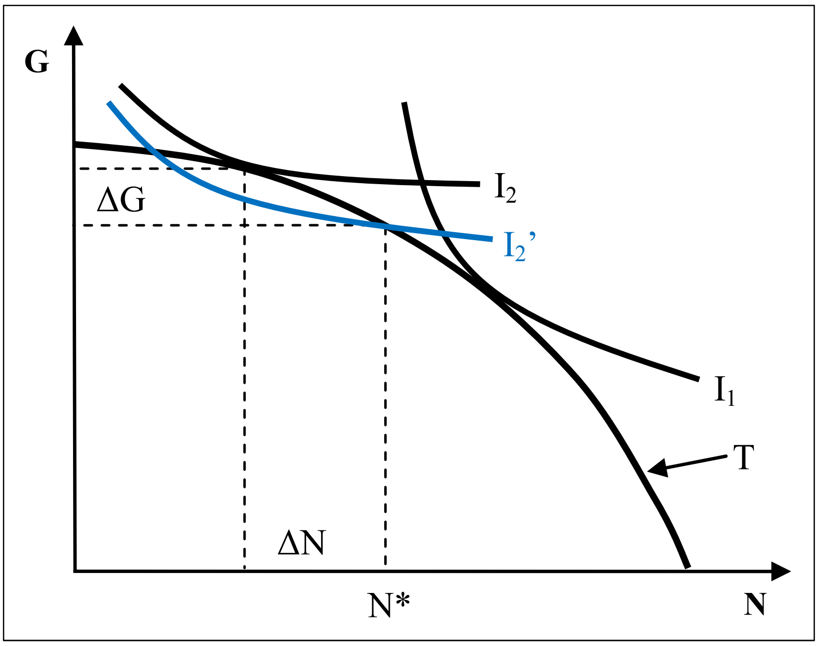
Economic thought is currently dominated by normative individualism which urges us to respect individual preferences, and this is indeed an important principle. Another important principle is intergenerational justice, a fundamental component of sustainability. Consequently, many agree that the world we hand over to future generations should not be impoverished. Thus, the preservation of biodiversity is not only an instrument for utility maximization in the sense of satisfying individuals' preferences, but also a matter of duty.

In this paper, I have focussed on the economic issues related to ACMs. As a result, a treatment of ethics in similar depth is ruled out, given limited space. Yet, circumventing ethics altogether is impossible in an epoch when intergenerational justice has become a commitment in Constitutions and International Conventions. Therefore, one aspect of conservation ethics must be presented in highly condensed form. I do not discuss *why* nature should be conserved. Rather, I take this for granted and show the economic consequences of such a value judgement. One is that the third recommendation in the previous section (individual valuation) which at first sight appears to pose the greatest difficulties loses some of its delicacy in practice, as shown below.

In Figure 1, G is the commodity output of the countryside and N is the conservation 'output', or the quality of biodiversity. All possible combinations of G/N that exhaust production resources are given by the transformation curve T. Experts may decide that, to the best of their knowledge, N^* is the minimal level of biodiversity quality that guarantees sustainability. Thus, on the principle of intergenerational justice, we have a duty to warrant N^* independently of individual preferences. This is the 'Safe Minimum Standard' SMS, introduced 60 years ago by resource economist CIRIACY-WANTRUP (1952). Not satisfying SMS is an option only in case when the costs of doing so are unbearable (BISHOP, 1980: 210).

Depending on the goods and on the characteristics of a given society, different policies are called for to respect the Safe Minimum Standard. A society characterized by the community indifference curve (CIC) I_1 fulfils its intergenerational duty while simultaneously maximizing its own utility under constraint T (for the CIC see MISHAN, 1981). The tangent point of T and I_1 being the optimal allocation, individual preferences need not be corrected by collective policies, either because preferences, although egotistic, unintentionally safeguard N^* or because they are partially altruistic. This is not a merely theoretical possibility. In fact, interviewees in contingent valuation studies often state that they are willing to pay for nature conservation both because they enjoy nature

Figure 1. Transformation curve between agricultural goods G and nature conservation C and different optimal allocations according to preferences



Source: own presentation

(egotistic preference) and because they feel a duty to contribute to its protection (altruistic preference).

In contrast, a society characterized by the community indifference curve I_2 will fail to steer clear of the Safe Minimum Standard if it relies only on individuals' preference-based behaviour. Duty requires it to adopt corrective policies that exchange ΔN for ΔG, lowering the utility level to I_2'. This is not at all unusual. Important environmental problems are routinely addressed by collective decisions, such as the policy to reduce greenhouse gas emissions to mitigate climate change.

Put in the language of mathematical optimization, society faces the problem of maximizing its welfare under the constraint to warrant N^*. The Lagrange multiplier λ is associated to this constraint and represents the shadow price of conservation. If willingness to pay (WTP) for conservation exceeds λ, the problem in question can be left to individual preferences. If WTP is smaller than λ, a collective policy is called for.

At the 2001 Gothenburg Summit, the EU pledged to stop biodiversity loss until 2010. The extension of the deadline to 2020 indicates that EU countries are failing to warrant N^*. In other words, they are failing to realize a central aspect of sustainability, which ostensibly guides their policies. Regardless of whether, in EU countries,

(a) individuals' WTP is really too low,
(b) individuals' genuine WTP is high enough but spoiled by free riding, or
(c) individuals' WTP is high enough but rests dormant due to inadequate institutions,

society *must* act immediately to stop the ongoing loss of biodiversity. We need not wait until RANDALL's (2007) framework of "green prices" for "green outputs" is completed although this completion is an important task for the future. In hypothetical societies of type (a) or (b), campaigning is required to convince people of the need for conservation and to dissuade them from free riding. Until this is done, conservation measures have to be implemented by state authority.

MEYERHOFF et al. (2012) estimate the aggregate WTP for conservation of the German population to be several times higher than HAMPICKE's (2013: 115) calculation of the costs of a substantial conservation programme in the rural countryside of at most € 2.10^9 per year. Thus, MEYERHOFF et al. (2012) reproduce with strongly refined methodology the results of an early estimate by HAMPICKE et al. (1991). There is reason to believe that in Germany, case (c) prevails, corresponding to CIC I in Figure 1. Provided that this result can be corroborated by more research, it follows that, next to missing incentives for farmers, lack of suitable institutions which elicit, bundle and activate dormant willingness to contribute to conservation, is the major reason for poor conservation in the countryside.

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