NATURE AND LANDSCAPE CONSERVATION BENEFITS

The case of agri-environmental public intervention in the EU

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1. Nature and landscape conservation and the EU agri-environmental policy

- *A rich and diversified biological and landscape heritage*

The European countryside includes a rich and diversified biological and landscape heritage, which is the result of many centuries of human occupation and the historical interaction of agricultural and forestry practices with the underlying ecological processes.

Even profoundly modified landscapes exhibit much more than simple artefacts or mere results of man's presence. For instance, in England, the number of species of trees and shrubs per 30 meters of hedgerow is an approximate estimator for the age of that hedgerow in centuries[1]. Thus, even hedges enclosing fields are not a mere result of plantation by man for utilitarian purposes. On the contrary, the structure and composition of hedges can testify long-term processes of colonisation by new species; in

[1] This is the so called Hooper's rule (Rackham 1986).

some cases, they may even suggest that the hedge directly originated from the natural wood grubbed out to create farmland many centuries ago. In the same way, the floristic composition of a meadow can indicate centuries of use as permanent grassland. Ploughing the meadow for conversion into arable or nitrogen fertilisation to increase meadow productivity, on one hand, or, on the other, colonisation by shrubs if the grass is no more cut for hay can lead to disappearance of characteristic species of herbs and invertebrates for very long periods. So, these and many other components of the biological and landscape heritage of rural Europe depend for their conservation on the maintenance of the management practices that currently are in balance with the underlying ecological processes.

The diversity of the biological and landscape heritage of rural Europe is based on two factors: (1) the variability of ecological base-conditions from the Mediterranean areas to boreal tundra; and (2) the marked differentiation in levels of agricultural modernisation among countries and regions.

Hence, for example, in the lowlands of the Parisian Basin or the East of England, as well as in some irrigated valleys in Southern Europe, an intensive, specialised and large-scale style of farming prevails today. Here, most the territory is used as arable land, and only tiny patches of woodland, wetland and other 'islands' of semi-natural vegetation have been left behind, usually confined to the poorest soils. Intensification of farming within arable land and its marginal expansion into the small residual patches of semi-natural vegetation have strongly negative impacts on water, soil or air quality, as well as on the conservation value of these residual patches. In these agriculturally intensified regions, there are, in general, few natural or historic-cultural values left to be protected within arable land; these values are here usually associated with the mentioned residual patches of semi-natural vegetation.

On the other hand, in the non-irrigated Mediterranean dry lands as well as in European mountains and uplands, the pace of agricultural modernisation was slower. Land is here mainly occupied by farming systems making a non-intensive use of modern agricultural inputs. There are still high natural and historic-cultural values both within farmland and in woods, heaths, moors or wetlands, which typically occupy an important share of the land. Bignal and McCraken (1996) mentioned that the Iberian Peninsula represents, by itself, approximately half of the area occupied by these non-intensive farming systems with high natural value in the nine European countries they have studied[2]. The main risk of biological and landscape heritage degradation in the case of these systems is not associated with intensification or the expansion of agriculture to all land, but with the likely abandonment of economically marginal farmland, or its massive conversion to low-natural-value forest. In fact, a recent study about the abandonment of farmland in Europe (Baldock et al. 1996) shows that the regions where the risk of abandonment is higher include the dry lands, uplands and mountains around the Mediterranean Sea, as well as the western fringe of the British Islands. The coincidence with the geographic distribution of the extensive farming systems with high conservation value just referred to could not be more perfect.

- *Conservation policy in the EU*

National and European public decision-makers have been giving a growing weight to nature and landscape conservation goals in rural areas. This is aimed at meeting public demands for rural spaces of leisure with high ecological and historic-cultural quality, as well as meeting environmental organisations' appeals for more effective preservation of our natural and landscape heritage. A number of multilateral international agreements, signed by national and European authorities, such as the Convention on Biological Diversity, also explain such a policy trend to bestow higher priority to conservation goals in rural areas.

Some years ago, most policy objectives and measures related to nature and landscape conservation in Europe had a national or regional level; they were particularly focused on the designation and management of national or regional protected areas. In some countries, these measures were also complemented by policy initiatives for the countryside as a whole, i.e.: not only aimed at particular sites of special natural or landscape value.

In 1979, the EU Birds Directive gave, for the first time, a truly European dimension to conservation policy – "since migratory birds know no frontiers, Europe must ensure their conservation" (European Commission 1999a).

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2 France, Greece, Hungary, Italy, Poland, Portugal, Ireland, Spain and the UK.

Acknowledging the fact that its agricultural, forestry, rural and regional policies have had significant impacts on the state of nature conservation throughout the Union, the EU gave a very significant new step towards the creation of a truly European policy for nature conservation: the Habitats Directive of 1992. This initiated a process that will lead to Natura 2000, an European network of protected areas designed to meet European-level conservation goals. This network will comprise areas already designated under the Birds Directive of 1979, i.e.: the Special Protection Areas (SPA), as well as new areas to be proposed by Member States (MS) in accordance to the 1992 Habitats Directive's provisions, i.e.: the Sites of Community Importance (SCI). The SCI designation process should be ended by 2004, after which MS will be required to gradually introduce protection rules and management plans for the designated sites.

- *Integrating the environment into the CAP and the emergence of an agri-environmental policy*

Accompanying all this process of the EU conservation policy, it became increasingly clear for the European public opinion that a strong connection exists between the way the land is farmed and the level of environmental quality, including the state of nature conservation, in rural areas. The fact of farmers managing more than half of EU's land and the visible negative impacts of post-war agricultural modernisation on the environment were decisive factors leading to the emergence of this new perception of a significant environmental dimension in agricultural production.

According to European Commissioner Franz Fischler, "Taxpayers can no more accept that money is spent to support an agricultural sector that does not pay due attention to environmental factors. They are, however, increasingly prepared to pay for sound environmental protection." (European Commission 1997).

The need to integrate the environment in the Common Agricultural Policy (CAP) is often justified based on market failure, i.e.: market prices fail to reflect the environmental effects of farmers' decisions. These effects are, therefore, not taken into account as private benefits or costs for the farmer. This increases the likelihood of what is a private optimum for farmers not coinciding with the social optimum. Hence, “policies should provide the more adequate tools and take into account society’s willingness to pay for those [non-market] benefits.” (European Commission 1998: 126).

There have been different factors pressing for a better integration of environmental concerns in all sectors of economic activity, namely agriculture and forestry. These factors comprise rising public concern about the environmental effects of farming and international commitments of the EU and different MS, in particular those related to agreements signed during the Earth Summit at Rio de Janeiro in 1992. At the European level, this need to integrate the environment in policies for the different sectors of economic activity had already been given legal status in the Single Act of 1986. The Fifth European Environmental Programme of 1992 included agriculture among the five sectors considered top priority with respect to the integration of environmental concerns into community policies. It actually established a specific target as regards the agricultural sector: that of 15% of all agricultural land in the Union covered by agri-environmental management agreements with farmers by the year 2000.

The integration of the environment in the CAP started in a moment at which this policy was suffering the most important crisis in its whole history. This was, in fact, an internal crisis caused by the very success of the CAP in promoting agricultural production and productivity growth during the previous three decades. Policy-based supply surpluses in different commodity markets covered by CAP regimes, which increased during the 1980s, required effective and strong measures to control supply and CAP-related public expenditure. Another source of pressure for CAP reform came from EU trade partners, especially in the context of GATT negotiations, who suffered the depressing effects of the CAP on world market prices of agricultural commodities. This internal crisis of the CAP made it easier for environmental interests to get inside the, until then very closed, debate forums on the future of agricultural policy (cfr. Winter 1996; and Baldock and Lowe 1996).

Besides some technical measures with an environmental content and incentive programmes for promoting agricultural extensification, with a strong supply-control content, an essential initial step in the integration of the environment in the CAP was article 19 of Regulation (EEC) n.º 797/1985. This authorised MSs to designate Environmentally Sensitive

Areas (ESAs) in which they could offer management agreements to farmers, so as to improve agricultural practices from an environmental standpoint or keep environmentally beneficial practices. In exchange for signing such agreements, farmers would receive an annual payment from the MS's budget – initially without any financial contribution from the EC[3].

With the 1992 CAP reform appeared the so-called accompanying measures, which comprised the agri-environment regulation – Reg. (EEC) n.º 2078/1992. This regulation required each MS to develop agri-environmental measures applicable throughout its territory. Thus, although these measures were kept as voluntary for farmers, as in the past, they became statutory for MSs. Community financial support was made available at 75% of the overall payment expenditure in objective 1 regions and 50% in other regions. These were measures accompanying the CAP reform. So, they were aimed at three goals which were largely seen as complementing this reform: (1) getting environmental benefits from CAP reform; (2) compensating farmers for income losses that could result from reduced support through prices; and (3) controlling surplus supply through environmentally benign extensification of farming. Regulation 2078 significantly broadened the type of environmental problems that could be targeted by agri-environmental payments. For the first time, these included some of the typical problems of Southern Europe, such as the abandonment of farmland and forests, the associated increased risk of fire, and the degradation of traditional terraced landscapes caused by the decline in upkeep practices. The fact the application of agri-environmental measures was now statutory for MS and this broadening of scope[4] led to a significant application success in many MSs, with an approximate 20% of the EU agricultural land covered by management agreements by 1998. This significantly exceeded the 15% proposed as a target for the year 2000 by the Fifth European Environmental Programme 1992. However, in Southern European countries, except Portugal, the uptake rates with respect to agricultural land were below that target: 1% for Greece, 3% for Spain, 14% for Italy and 17% for Portugal (European Commission 1998)[5].

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3 Meanwhile EU legislation has appeared that grants European funding for payments made within ESAs.

4 The introduction of article 19 about ESAs in the regulation 797/1985 had been the result of a demand by Northern European countries, namely the UK, Denmark and the Netherlands, and was mainly designed to respond to environmental problems typical of these countries: landscape and nature conservation; and preventing water pollution in the context of a modernized and intensive agricultural sector. Thus, when the CAP was reformed in 1992, only these 3 countries, followed later by France and Germany, had designed and applied ESA policy.

5 France (23%), which also includes an important share of Southern Europe, had an uptake rate slightly larger than the European average (20%).

Overall, EU public expenditure related to the agri-environment regulation (excluding the financial contribution of MSs) was circa ECU 1.7 billion, which is approximately 4% of all expenditure from the European Agricultural Guidance and Guarantee Fund (EAGGF) Guarantee section (European Commission 1998). Hence, though small in relation to CAP as a whole, agri-environmental policy represents today one of the most significant EU funds for environmental purposes in rural areas, possibly the most significant for nature and landscape conservation purposes alone.

Accordingly, one would expect some countries to use part of their agri-environmental budget to achieve European-level conservation objectives (though the regulation considers national and regional-level objectives as equally legitimate for the programmes applying the regulation at the national or regional levels). That use of the agri-environmental budget appears as particularly attractive when we look at the large territorial extent the Natura 2000 network will predictably acquire in some European countries, hence at the significant overall costs of managing Natura 2000 sites (European Commission 1999a). Reinforcing this argument is the fact that, in many Natura 2000 sites, agriculture is the main land user and the one with the greatest impact on conservation objectives. For example, in the case of Portugal, 65 sites have been proposed to the European Commission for designation as SCIs, which occupy 1,215,000 hectares, that is more than 12% of the national territory (European Commission 1999a). In a very significant part of this area, probably more than half, which comprises mountains, uplands and dry lands in the interior of the country, the most important biological and landscape values depend on successful maintenance of existing, extensive farming systems. Some countries are already making use of agri-environmental funds to manage conservation sites to be designated as Natura 2000 sites. One example is that of the 'opérations locales' of the French agri-environmental programme, which apply to such sites, especially those located in the uplands where conservation goals are mainly threatened by the abandonment of farmland and pastures (European Commission 1998).

The most characteristic design and application aspects of current EU agri-environmental policy are: (1) the type of policy tool that is used; (2) the scope of problems to which agri-environmental payments apply; and (3) the current, flexible and decentralised, application model, which operates through national, regional or local programmes submitted by MSs to the European Commission.

As regards the type of policy tool, agri-environmental measures are applied through management agreements between a public authority and a farmer. These agreements specify a number of management prescriptions, either positive or negative (e.g. making at least a cut of hay per year; do not cutting before July; or do not using more than 50 Kg of nitrogen per hectare), which the farmer voluntarily subscribes for a given period in exchange for a given annual payment during this period. These management prescriptions can mean a, small or large, departure from the usual practices of the farmer if not bound by the agreement. The agreement may refer to the all farm or only part of it; it may apply to all or part of a specific land use (e.g. permanent grassland, olive groves…) in the farm. Agreements have a minimum time length of five years. Usually a standard contract accompanied by a standard per-hectare payment rate is offered to all eligible farmers in a country, region or small area. The annual payment is computed based on the income loss and additional costs for the farmer of subscribing and complying with the management prescriptions; it may include a limited incentive premium as well.

As regards the scope for agri-environmental measures, it is important to realise that these measures are considered within a model where public authorities, acting on behalf of society, sign contracts with farmers so that these will supply specified environmental services. This contracting model only makes sense to the extent that supplying these services implies management constraints with respect to what the farmer would otherwise be allowed to do. The farmer is, in fact, temporarily selling a set of rights, and so it would require as compensation, at least, the (productive) value these rights have for him. To the extent that environmentally sensitive management is comprised in a set (or code) of good practices, which society has the right to expect from farmers (at their own expenses; i.e. if the polluter pays principle applies), there is no role for service-purchasing agri-environmental payments: farmers cannot sell rights they do not have. Likewise, prohibitions that already apply to farmers, e.g. because they are located inside a nationally designated Park or Reserve, or inside a Nitrate Vulnerable Area (EU Nitrates Directive), are outside the policy scope for agri-environmental payments[6].

[6] For the same reasons, the formula for calculation of agri-environmental payments should only consider income losses and additional costs in the situation that represents the policy target as compared to that where the farmer is already complying with good agricultural practices – i.e.: it is impossible to pay to farmers for them to comply with their environmental obligations, either legal or derived from the good-agricultural-practice principle.

The new EU legislative frame for agri-environmental policy recently approved as part of the Commission's proposals (Agenda 2000) states this policy scope in a clear way: "Agri-environmental commitments should go beyond the mere application of current good agricultural practices." (Regulation (EC) n.º 1257/1999). It is up to MSs to define what is meant by good agricultural practices at a national or regional level.

It is also up to MSs to make eligibility to other CAP payments, such as direct payments under Common Market Organisations (CMOs) or the Hill Livestock Compensatory Allowances (HLCAs), dependent on environmental-performance criteria, whenever this seems appropriate. These environmental criteria required from farmers to be eligible for other CAP support measures cannot be covered by agri-environmental payments, since these payments "should deliver services that are not provided by other support measures" (Regulation (EC) n.º 1257/1999).

The agri-environmental needs, problems and potential of a country or region, which should be addressed by an agri-environmental programme, depend on particular characteristics of that geographic area. Hence, for consistency with the EU subsidiarity principle[7], agri-environmental policy is applied, in an extremely flexible way, through national or regional programmes to be designed and implemented by the competent public authority at the relevant territorial level. It is at this level that: (1) the specific environmental objectives of the programme (water quality, biodiversity, soil conservation…) are set; (2) a ranking of these objectives is decided upon; and (3) specific measures are designed to achieve these objectives. Among the environmental objectives and their priorities, it is considered that national- or regional-level objectives are as legitimate as community-level objectives (e.g. species and habitats lists in EU conservation directives). What matters is that the selected objectives are relevant with respect to the needs, problems and potential of the area to which the programme applies, and that the measures chosen to achieve these objectives are the most effective in that particular area's context.

The only, but important, role of the European Commission in this respect is the assessment of the programmes proposed by MSs, as regards e.g. its general architecture (internal coherence), relevance and consistency

[7] Principle of EU policy which states that each decision should be taken at the territorial level that is more adequate to get the highest possible effectiveness.

with the CAP and other EU policies, as well as the decision on whether to approve a programme. The Commission also assesses and approves proposals submitted by MSs to modify programmes currently in application.

According to the new rural development regulation (Regulation (EC) n.º 1257/1999), agri-environmental measures should be jointly programmed with the other so-called rural development measures. Thus, the main role of this regulation is providing a broad menu of policy tools, for which EU funds are available; these tools include e.g. investment grants for farmers and food processors, Hill Compensatory Allowances for marginal areas, and incentives for farmland afforestation. National or regional authorities responsible for the programming of rural development should set rural development objectives that address regional problems and potential, and pick the package of policy tools in the menu that would most effectively achieve these objectives in the regional context. This integrated character of the programming of all rural development measures (agri-environmental and others) enables public authorities to articulate all selected policy objectives and measures so as to eliminate conflicts and promote synergies between these objectives and measures. For example, investment grants are only awarded to investments not degrading the local landscape; the quality of this landscape is also promoted through specific agri-environmental measures and explored by grant-induced rural tourism, which should be designed so as to contribute to the region's income. Although not ignoring possible conflicts between these policy objectives, the OECD (1999a) also refers to the need of integrating environmental amenities in rural development as the best way to simultaneously achieve conservation and development goals for rural areas.

2. The evaluation of agri-environmental policy and programmes

- *The need for evaluation*

The flexible and decentralised way in which EU agri-environmental policy is designed and applied through national or regional programmes confers a crucial role to policy and programme evaluation. In fact, until 1998, 218 modifications to programmes in application have been approved versus only 137 new programmes. This gives an account of the experimental character of this policy and the high frequency at which evaluation exercises take place – an evaluation should support each MS's proposal of modification of a programme to be submitted to the Commission (European Commission 1998). Programme evaluation has had important roles at three steps of the programme cycle:

1 ex ante; roles for evaluation at this stage are that of improving the design of the programme and that of supporting the Commission's approval decision;

2 on going (i.e. when the programme is still in application); a role for evaluation at this stage is suggesting and supporting the need for modifications in the design or application of an on going programme, to be proposed to the Commission;

3 ex post; roles for evaluation at this stage are demonstrating, in a transparent way, the benefits delivered – i.e.: whether the programme was good value for taxpayers' money –, as well as learning from past experience with an eye on the design and application of future programmes.

Under the current legal frame, the public authority managing a programme is required to carry out an evaluation exercise at each of these stages (European Commission 1999c)[8].

Thus, the evaluation process meets the needs of three types of agents:

(1) the authority managing the programme, which gets help in designing new programmes, suggestions of improvements to programmes in application, or learning-by-doing knowledge to be used in future programmes;

(2) the European Commission, which gets support in approval decisions with respect to both new programmes and amendments to existing programmes, as well as learning-by-doing knowledge which can support proposals of agri-environmental policy reform; and, last but not the least,

[8] In the current legal frame, agri-environmental measures are programmed jointly with the other rural-development measures. Thus, in what follows, programme refers in rigor to the set of all rural-development measures and not only to the agri-environmental subset.

(3) taxpayers, NGOs and other organised interests in society, as well as the farmers themselves, to whom environmental benefits, resulting from public funds applied and management constraints accepted by farmers, should be clearly demonstrated.

Besides, agri-environmental policy evaluation could also be used to convince trade partners of the EU in the world market that this is not a disguised form of supporting non-competitive European farmers. The crucial demonstration for this purpose is that this is a policy aimed at real environmental benefits and based on the application of European taxpayers' money, with these being willing to pay for these benefits (cfr. Commissioner Franz Fischler's quotation above). Hence, this policy would simply correspond to the contracting of farmers for the delivery of a service for which there is an effective public demand. The effects of this service on world trade would not be trade-distorting provided that the service has certain public good characteristics, is paid on a cost basis and (if applicable) non-farmers are accepted as suppliers.

• *Evaluation criteria*

According to the stage of a programme's life cycle at which evaluation takes place, it will refer to a particular subset of the following set of criteria (European Commission 1999c).

˘ Relevance. Refers to appropriateness of a programme's objectives to the agri-environmental needs and potential of the context this programme is supposed to address.

˘ Coherence. Internal coherence refers to the appropriateness of planned resources and selected policy measures to the objectives of the programme. External coherence refers to, positive or negative, relationships between the programme at stake and other European or national programmes that will probably interact with it.

˘ Effectiveness is a measure of the extent to which the observed impacts of the programme (either intended or not, positive or negative) fulfil its objectives. For an effectiveness evaluation to be possible, specific objectives should be clearly defined and translated into (preferably quantitative) indicators, for which targets are set (e.g.: keep, at current levels, the frequency of plant species characteristic of meadow ecosystem to be conserved; include list of such species).

˘ Utility is a measure of the extent to which the observed impacts meet the agri-environmental needs and potential the programme is supposed to address. Differs from effectiveness in that the evaluation does not refer to the objectives of the programme, but directly to the needs/potential it is supposed to address.

˘ Efficiency refers to an evaluation of the effects achieved with respect to the used (financial and administrative) resources, i.e.: how economically were these resources converted into results? Could the same results have been obtained with less resources? Or could more results be secured with the same resources? (European Commission 1999c).

The first two criteria are particularly relevant for ex ante evaluations, while the possibility of evaluating a programme with respect to the other three criteria depends on the observation of effects; hence it is circumscribed to the on going and ex post stages.

• *Specific aspects and difficulties in the evaluation of agri-environmental programmes*

This section identifies and discusses some specific aspects of the evaluation of agri-environmental programmes, exemplifying with cases of effectiveness evaluations.

For the effectiveness of a programme with nature and landscape conservation objectives to be subject to evaluation, these conservation objectives should be clearly and specifically defined, and converted into several indicators. For each of these indicators, a target level (or performance indicator) must be set. For example, see the case of objective 2 of the UK's Pennine Dales ESA scheme (adapted from ADAS 1996):

Objective 2 – Conserve the nature conservation value of pastureland.

Target 2.1. 80% of pasture area covered by management agreements;

Target 2.2. The occurrence of botanical species characteristic of non-improved pasture (list provided) should not decline with respect to initial levels, in pastureland covered by a management agreement.

Target 2.3. Breeding populations of waders, namely Lapwing (Vanellus vanellus), Curley (Numenius arquata) and Redshank (Tringa totanus), should not decline with respect to the national trend in the area covered by management agreements.

Target 2.1 is an uptake rate in terms of the coverage of the particular ecological resource (pastureland) that is targeted by the programme. The first evaluations of agri-environmental programmes were mainly based on assessments of the extent to which targeted uptake rates had been achieved, which implied implicitly assuming that the programme was effective in producing the aimed environmental effects inside the area covered by management agreements. This maintained assumption should, however, be submitted to the evaluation test. The goal of performance indicators such as targets 2.2 and 2.3 is precisely making this effectiveness evaluation possible[9] . As it is easy to see, many performance indicators of this type are specific to each programme. There has been a significant conceptual and analytical effort by OECD (forthcoming, 1999b and c) and the European Commission (2000, 1999a and b) to develop consistent and intelligible agri-environmental indicators, which have a sound scientific basis, are relevant for policy evaluation, and, if possible, are comparable between countries and computed based on available information.

Giving high priority to explicitly assessing performance indicators of this type does not mean that uptake indicators are less relevant. When uptake by farmers is voluntary, as it is the case with agri-environmental programmes, a significant uptake rate, specially when combined with some "in depth" effect (actual change with respect to the policy off scenario), is crucial to secure an effect. By definition, in the case of a voluntary scheme, without uptake there is no impact, simply because there is no application. In practice, when designing agri-environmental measures, one always faces a

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9 In this case, the quality indicators directly refer, as desirable, to aspects of the state of the environment. In other cases, it is only possible to indirectly measure these aspects, for example through the associated farming practices, such as fertilizer use or stocking rates.

trade off between more “in depth” effects (usually implying stricter management constraints) and more uptake (which tends to be higher the less restrictive management constraints are). These trade offs need to be assessed based on the particular objectives of the programme and the agri-environmental context in which this is to be applied.

The following step in an effectiveness evaluation is characterising the initial situation by assessing indicators such as those associated to targets 2.2 and 2.3 before the programme is implemented. Afterwards, monitorization of the evolution of these indicators, as the programme is applied, is required for at least two situations: fields (farms) with and without a management agreement. Biophysical effectiveness evaluation is aimed at using the results of this monitorization to demonstrate that: (1) targets were met; and that (2) this happened as a result of the programme itself and the used administrative and financial resources. For this latter demonstration, a comparison between areas covered and not covered by management agreements is generally required.

There are some particular problems when using biophysical indicators for an ex post assessment of the impact that a programme effectively had on its objectives. The first refers to the time that is required for the environmental impacts to become visible. For example, more rare and valuable plant species may take long time to appear in a meadow, even after conservation-friendly management practices have been introduced. Likewise, there can be a decades-long lag between adoption of adequate fertiliser-use practices and the aimed decline of nitrate concentration in the groundwater. The lime length of environmental processes is, therefore, not very compatible with the need to assess a programme’s impacts at the end of each 5-years period.

A second problem has to do with attributing the observed effects to the programme, which implies ruling out possible alternative causes. For example, a reduction in fertiliser use (and its positive effects on biodiversity and groundwater quality) could have been the result of declining output price or a lower level of agricultural-policy support, in which case it could not be imputed to the agri-environmental programme. This is why it is important to monitor the selected agri-environmental indicators both in areas covered and not covered by management agreements; these latter, while also subject to the same external effects (markets, general agricultural policy…), were not subject to the agri-environment programme, so providing a policy-off baseline for comparison.

However, this may be insufficient. For example, think of a case where older farmers without a successor did not subscribe an agri-environmental commitment implying the upkeep of their land, in the traditional way, for 5 years (as opposed to abandoning). This might happen precisely because these farmers thought they could not honour this commitment. On the other hand, younger farmers, who subscribed the commitment, would not anyway abandon their farmland during the 5-year period. In this case, we are going to observe a lower level of farmland abandonment among those who signed a management agreement, but this is not a result of the programme itself; it is the effect of demographic factors explaining farmland abandonment, which would operate in the same way without the programme. There are some solutions for this problem: to compare both groups (agreement and non-agreement farmers) to make sure they are comparable, i.e.: identical except with respect to uptake of the programme. Another solution is ex post surveying agreement farmers to try to find out what they would have done if not constrained by the commitments included in the management agreement.

A similar problem is to assess whether the results secured within the area covered by management agreements were offset by inverse-sign developments taking place outside this area. For example, the same farmer can reduce fertiliser use within eligible area and to intensify fertiliser use outside this area. This effect, known as a "halo effect" because it produces a halo of intensive agriculture around the area targeted by an extensification programme, was investigated for UK ESAs by Saunders (1994). This is a good case for requiring all the area of a farm to be included in a management agreement – there are other, equally convincing, arguments against this policy-design option (see European Commission 1998).

The existence of other factors (market prices, other policies, demographics…) that can produce the observed effects, as well as the type of negative offsetting developments exemplified with the halo effect, advise us to exercise great care when determining the net effects that are attributable to the particular programme at stake.

3. The economic valuation of nature and landscape conservation benefits

- *Economic valuation and cost-benefit analysis – potential uses in the context of EU agri-environmental policy*

The economic valuation of nature and landscape conservation benefits cannot be seen as an alternative to the biophysical evaluation method just described in the previous section[10] . Contrarily, economic valuation usually requires a previous evaluation (or prediction) of biophysical impacts. For example, in an ex post evaluation, the economic valuation will build on the results of a previous analysis of observed impacts of a programme. Likewise, in an ex ante evaluation, it will be necessary to assume or predict the relevant biophysical impacts for the economic valuation of these impacts to become a possible task. Thus, the validity and reliability of the economic valuation of a programme depend on the validity and reliability of the underlying evaluation/forecast of biophysical impacts.

Justifying the need for an economic valuation could not, therefore, rely on the claim that it would provide a substitute for the biophysical evaluation of the policy impacts, but on the fact that economic valuation provides to public decision-makers, programme managers and the general public with information that could not be extracted from observed or predicted biophysical impacts. This new information that is generated by valuation has to do with the public's preferences for the conservation benefits at stake.

Let us see a first example in which it is predicted that a particular programme will result in the conservation of meadow vegetation and dry stone walls, which are characteristic of the landscape of a particular countryside area. These attributes will be maintained by the programme at their current conservation status as opposed to a known level of degradation they will suffer without the programme in the near future. The cost of the programme is estimated at Euro 100,000 a year[11] . Should we go ahead with such a programme?

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[10] These alternative character of the two methods seem to be implied by statements such as "The two main methods to measure and monitoring change in environmental benefits of agriculture are the physical and the monetary methods." (European Commission 1998: 30).

[11] Besides the cost related to payments to farmers, we should also consider the administrative costs of the programme.

Given the conceptual and empirical difficulties of the economic valuation of the meadows and walls conservation benefits, we could simply (i.e.: without further information) leave this decision to the competent policy decision maker.

Note, however, that, once such a decision has been taken, this decision will necessarily have attributed a money value to those benefits. A decision of going ahead with the programme will mean that, for the particular decision maker, these benefits worth at least Euro 100,000. On the other hand, a not going ahead decision would imply that those benefits are below this amount.

The question of whether we should evaluate conservation benefits in money terms is, therefore, a false question: once a decision is taken, whatever its outcome, a money value will necessarily be imputed to these benefits by the decision outcome itself. Note, as well, that, if the decision proceeds as above, this implicit valuation will only rely on the particular decision-maker's preferences for those benefits. Is this desirable (appropriate)?

To avoid this difficult question, many decision-makers make resort to the opinion of ecological or landscape experts. Money values imputed by this decision will only rely on the decision-maker's and expert's preferences as well as on expert knowledge. Is this desirable? Shouldn't the decision take into account general public's preferences, namely those of taxpayers who pay for the programme?

The economic valuation of conservation benefits assumes that these preferences should count. Hence, we need to explicitly estimate the public's willingness-to-pay (WTP) for the conservation benefits, so that these are directly comparable to programme's cost. Even if the value of a programme so estimated results to be offset by programme's costs, the decision maker can, of course, decide to go ahead with the programme; or even if benefits are estimated as exceeding the cost, he can decide to reject the programme. However, in these cases, he should explain on which criteria, other than economic efficiency, has he based the decision. Economic valuation has, thus, an important role in enhancing the transparency of the decision.

Note that some quotations above in this paper, namely those from the European Commission, indicate that decision makers tend to explain the need for an agri-environmental policy based on the failure of the market in supplying environmental goods and services for which there is an actual (i.e.: accompanied by WTP) demand by the public. Thus, the task of demonstrating that such a WTP exists and offsets policy costs should be seen as crucial for any justification of public agri-environmental intervention at least in two fronts. First, with respect to taxpayers, whose money is spent in this policy. Second, with respect to trade partners of the EU in the world markets for agricultural commodities, to whom it is necessary to demonstrate that this is a policy aimed at contracting farmers for the provision of environmental services for which there is an actual demand (and WTP) by the European public, and not a form of hidden support to European farmers, which would cause significant cost for trade partners in the form of market distortion. The mere biophysical evaluation of policy impacts, though crucial in other respects (demonstrating effectiveness), could not be put to these uses. It would not demonstrate the intensity of the public's preferences for the relevant policy impacts, nor even that these impacts have a positive effect on social welfare. It also does not demonstrate that, even if existent, this positive welfare effect is sufficient to offset the negative effect of added taxes, which are required to pay policy costs.

Taking forward the previous example of the conservation of meadows vegetation and stone walls will allow us to make clear some other limitations of biophysical evaluation for some policy uses, which can be overcome when biophysical evaluation of impacts is complemented by an economic valuation of these impacts. So, consider now that there is not only the possibility of conserving both meadows and walls at their current conservation levels but also the possibility of improving the meadows provided one is prepared to accept some light degradation of the walls. This degradation would be implied by the budget constraint of the programme manager: allocating more funds to the meadows would require allocating less to the walls. Ex post evaluation of biophysical impacts will only reveal that, if programme objectives were met, meadows improved while walls suffered light degradation. But would this be preferable to keep both attributes, i.e.: meadows and walls, at their current conservation levels? A merely biophysical analysis cannot provide an answer to this question.

Information on public's relative preferences for meadows and walls, namely on the corresponding WTP for each attribute, will make possible to build an indicator of the value of each of the policy alternatives (i.e.: improving the meadows with some degradation of walls versus keeping both at their current conservation levels) for the public. This would allow us to select the policy alternative (i.e. programme design) that best satisfies public preferences while respecting the relevant budget constraint of the programme manager. This type of application of economic valuation methods has a wide range of uses in the design of efficient agri-environmental programmes.

Selecting the alternative that best satisfies the public while respecting the budget constraint, as exemplified above, requires what the European Commission (1999c; evaluation criteria quoted above) would call selecting the (ex ante) most relevant or (ex post) utility-maximising policy design. The mere evaluation of biophysical impacts does not support, by itself, a quantified evaluation of the compared relevance or utility of two policy alternatives (programme designs). Without economic valuation, we would be required to make a host of assumptions, mostly on an ad hoc basis, to assess the compared relevance or utility of a programme design.

A third and last example refers to the evaluation of programme efficiency. Recall the two questions used above to illustrate this criterion, as it is understood by the European Commission (1999c), i.e.: could the same results be secured with less resources? Or more results with the same resources?

This criterion is known by environmental economists as cost effectiveness. It is a criterion that, avoiding the need for an economic valuation of conservation benefits (which sounds attractive, as valuation has its own complexities and difficulties) leads to a partial cost-benefit analysis. This analysis is only valid if one of two conditions are met:

- the policy alternatives to be compared produce exactly the same results (relevant for the first question above); if this condition is not met, it is meaningless to compare costs;

- the policy alternatives to be compared have exactly the same cost, and all biophysical impacts of one of the alternatives are unequivocally superior to the corresponding impacts of all other alternatives (i.e.: more water quality, more biodiversity…; this is relevant for the second question above); if this condition is not met, it is impossible to select the best alternative without a system of weights for the different impacts under which these impacts become additive, so as to determine which alternative secures the best overall result (providing such a weighting system is precisely the role of economic valuation).

These two conditions are rarely met in practice.

An interesting example of the non-meeting of the first condition is that of a comparison between two alternative policy measures to conserve meadow vegetation: (1) using incentive payments to farmers to continue with traditional meadow cultivation and grazing practices; (2) allowing farmers to discontinue their traditional practices (i.e.: meadows abandonment from a productive point of view) and contracting private firms providing landscape conservation services such as annual cut of the grass, but without keeping grazing livestock. It is very likely that measure (2), without grazing, does not succeed in keeping the vegetation exactly as it is now, which is ensure by measure (1). Thus, knowing that (2) is cheaper than (1) is not sufficient to select (2) as the preferable alternative if the results of (1) are perceived as superior to those of (2). An economic valuation of how much (1) is better than (2), in terms of public's WTP for having (1) rather than (2), would be required to decide according to public preferences for the two alternatives, while also taking account of public preferences for the added money income that would spent by (1) with respect to cheaper alternative (2).

As regards the second condition above, it often happens that each of the several alternatives that are compatible with the budget constraint is superior to the others in some respects, and inferior in other respects, as it happens with the example discussed above of better meadows at the cost of worse walls. In these cases, there is not a policy alternative that is unequivocally superior to all others, and knowing WTP for more of each of the relevant impacts (water quality, biodiversity…) is required to know the total value of each alternative for the public, so that we can select that which best satisfies the public while being consistent with available budget.

In this way, and excepting the (few) cases in which one of the two conditions above is met, the evaluation of the compared efficiency of two programme designs will require the economic evaluation of biophysical impacts of conservation (including negative impacts) and a full cost-benefit analysis. A cost-effectiveness analysis (partial cost-benefit analysis), though simpler and not requiring economic valuation of impacts, is not in general valid.

Summarising, there is a wide range of potential uses for the economic valuation of nature and landscape conservation benefits in the context of the design and evaluation of agri-environmental programmes. Economic valuation is often the only way of validly carrying out a number of analytical tasks, such as for example:

- justifying, for taxpayers, the relevance, utility and efficiency of public expenditure in agri-environmental policy, as a whole, or in a particular agri-environmental programme;
- helping in ex ante selecting the most efficient programme alternative at the agri-environmental-programme design step;
- supporting the evaluation of the efficiency or utility of a particular programme, the suggestion of amendments to on going programmes, aimed at increasing their efficiency or utility, or the design of new, more efficient programmes (roles in ex ante, on going and ex post evaluation stages);
- demonstrating to EU trade partners in the world market that agri-environmental policy truly is the contracting of services for which there is an effective demand (and WTP) by the European public.

For all these policy uses, the mere quantification of biophysical impacts, although necessary to assess effectiveness and as a starting point for economic valuation, is not, per se, an alternative to economic valuation.

- *Types of methods for the economic valuation of conservation benefits*

A general problem to be dealt with by all methods for the economic valuation of conservation benefits is the absence of markets for most of these benefits. Hence, there is also no information on prices and quantities, which is required to reveal WTP by measuring areas under demand curves, as it is usual in most cost-benefit analyses of public policies.

The fact that there are no markets does not necessarily mean that individuals have not well defined preferences for conservation benefits, or that they are not prepared to trade off money income (i.e.: purchasing power for market commodities) for more of these conservation benefits. The estimate of the economic value of a (positive) conservation impact is based on the concept of maximum willingness-to-pay (WTP) for this impact. WTP represents the amount that we need to charge for this impact so that the individual is exactly as well off as before benefiting from that impact. Thus, WTP reveals us the intensity of the individual's preferences for the impact at stake when compared to his (her) preferences for other, also useful, goods and services he (she) can buy with available income.

We can distinguish different types of methods used to reveal WTP for non-market environmental benefits, according to two main criteria:

(1) type of information used, i.e.: the type of data; and

(2) analytical method used to reveal WTP from data.

There are two types of data that can be used to estimate WTP for non-market conservation benefits: hypothetical and behavioural data.

Hypothetical data refer to choices or values that have been stated by individuals when replying to questions involving hypothetical scenarios, for example:

(i) how much would you be prepared to pay at the maximum, as an entrance fee, if this was required to get access to this area of upland farming and to enjoy its landscape and natural value?

As its name indicates, behavioural data refer to observed, actual choices and behaviours of individuals, when reacting to actual settings, for example:

(ii) how many visits did you make to this area of upland farming to enjoy its landscape and natural value during last year?

The main problem with hypothetical data is that people have a limited capacity to absorb all the information that they need in order to be able to correctly value an environmental benefit (e.g.: scarcity, quality, and existence of substitutes) and arrive at a considerate decision during the short time available for an interview. Besides, people can engage in strategic answering, i.e.: answers that aim at influencing the policy decision they perceive underlies the survey without necessarily revealing their true WTP (for example, they may declare a high WTP amount if they like the conservation programme, as they know the survey is anonymous, and thus nobody is going to ask them to pay the stated amount if there is a decision for the programme to go ahead)[12].

For these reasons and because they, in general, mistrust people's answers to surveys, many economists (who ignore that statistical figures they work with most of the time come from people's answers to surveys) prefer to work only based on observed behaviour. Actual choices in actual settings would lead people to more seriously considering the costs of a wrong decision, and so behavioural data would provide a more reliable window into people's preferences.

There are, however, problems typically associated to the analysis of observed behaviour. Behaviour does not speak by itself. It needs to be interpreted to reveal people's preferences and WTP. For this purpose, economists make resort to the modelling of decisions such as that implied by question (ii). In this modelling, we need to take account of all aspects of the actual context that determined the decision of visiting area X times during the last year (i.e.: quality of the area's landscape; travel cost from the individual's home; leisure time he has available for visiting the area; and existence of substitute landscapes closer to home). These aspects need to be described in the model in the same way they are perceived by the individual, as people do not react to objective reality but to reality as subjectively perceived.

For example, the travel cost (TC) method[13] is based on the estimation of the relationship between the number of visits to a site and the travel cost (as a measure of the implicit price of visiting that site). This is in fact a demand curve for the recreational services of the site. However, the price is not directly observed. It needs to be constructed by the analyst, based on distance travelled, time spent in the journey and the cost the individual

12 On the other hand, giving a strategic answer does not always implies lying about own preferences. All depends on the type of economic incentive created by the way the question is framed. It is, thus, important to analyse, in each particular case, the type of incentive induced by the valuation question, rather than having to choose in general between the two following extreme positions: "economic valuation techniques are never valid" or they are "always valid" (Carson et al. 1999).

13 Cfr. Ward and Beal (2000).

imputes to each travelled Km and hour spent of his (her) available leisure time. Will the cost per Km, as perceived by the individual, only comprise petrol costs or also include a share of the fixed costs of having a car (insurance, legal periodical tests…)? What is the value of each hour of leisure time for the individual? These questions raise difficult problems and the right answer will, probably, vary across individuals. If these aspects of the decision context are not interpreted in the same way as the individual perceives them, the final model will misinterpret behaviour and will lead to biased estimates of WTP.

Different ways of questioning reality originate different data formats. For example, we may ask directly to people exactly what we need to know (WTP in this case), as in question (i) above. This defines the open-ended format of the contingent valuation (CV) method. The same information could in principle be secured by framing the question in an indirect way, such as:.

(iii) would you pay Euro 10, as an entrance fee, if this was required to get access to this area of upland farming and to enjoy its landscape and natural value or would prefer not to pay and giving up the visit? (each individual faces a different price).

This way of questioning defines the discrete-choice format of the CV method (Bishop and Heberlein 1979).

The second criterion to classify non-market valuation methods has to do with the analytical procedure used to reveal WTP from data. In this respect, there are direct and indirect methods.

Direct methods require that we ask people exactly what we need to know (i.e.: maximum WTP), as in question format (i), that is: the open-ended format of the CV method. The only analysis that is required in this case is computing the sample average of WTP and multiplying this by an estimate of the relevant population (total number of visits x people / year, in this case). In this way, we estimate the recreational benefit of keeping the site open to the public at the current state of landscape quality and natural value.

All the other data format types referred to above require a more complex analytical task to indirectly reveal WTP from data. For example, data with format implied by question (ii), i.e.: coming from the TC method, require that we regress the number of visits (demand) on travel cost (price) and other determinants of demand, such as landscape quality at the site. Then, the estimated demand-price relationship is interpreted as a conventional demand curve, and thus used, in the usual way, to estimate WTP (to be more precise consumer's surplus) per visit for the current state of landscape quality. Changing landscape quality will shift the demand curve, which will enable the analyst to also estimate WTP for landscape improvements.

- *Application of economic valuation methods in the context of the EU agri-environmental policy: some examples*

Several types of non-market valuation methods have been applied to estimate the public's WTP for nature and landscape conservation benefits. The applications in the particular context of the agri-environmental policy of the EU are less numerous. Willis et al. (1993b) analyse the potential and limitations of each available method (for a synthetic presentation see also Willis and Garrod 1994). Empirical applications comprise, namely, a valuation study of the English Environmentally Sensitive Areas (ESAs) by Willis at al. (1993a), a study of the Scottish ESAs by Hanley et al. (1996), a study of the Pennine Dales ESA, UK (Santos 1998) and more three studies of policy measures applied in the context of the Portuguese agri-environmental programme (Almeida 1999, Madureira 2000, and Santos 1998). The first two studies were commissioned by the UK's Ministry of Agriculture Fisheries and Food (MAFF) and submitted to the European Commission as part of an ex post evaluation report referring to the different agri-environmental schemes in operation in the UK. These results include estimates of WTP for different programmes and their comparison with the corresponding costs.

In what follows, we discuss some of the results of the evaluation of the Pennine Dales ESA (Santos 1998), as these allow us to illustrate most of the uses for the economic valuation of conservation benefits in the design and evaluation of agri-environmental programmes.

This study was based on the discrete-choice format of the CV method. In 1995, a survey of visitors to the area secured 422 complete and usable questionnaires. Visitors were asked to choose between (1) the continuation of a specified agri-environmental programme for the ESA, which would only be possible if funds were raised through a specified income tax rise; and (2) giving up the programme and its benefits without a financial cost. Each respondent was presented with a different level of price/tax rise and a different programme, which consisted of a particular combination of the following three basic measures:

P1, or the maintenance of currently existing dry stone walls and field barns in their current state of conservation, as opposed to some dereliction and decay in the future without the policy;

P2, or the conservation, at their current state, of meadow vegetation with a high botanical and scenic interest, as well as the habitat conditions for ground nesting birds in the meadows and pastures; these botanical and habitat benefits were both threatened by the intensification of meadows use and specially by increases in nitrogen fertiliser;

P3, or the conservation of small, remaining broadleaved woods threatened by excessive grazing and bad management.

The answers were interpreted based on a logistic regression model, which enabled us to predict the sample average of WTP for each of the 7 possible combinations of the three basic policy measures. Aggregating for the total population of households visiting the Dales per year (circa 167 220 households) yielded the nature and landscape conservation benefits for visitors presented in chart 1.

Chart 1
Aggregate conservation benefit for different programme alternatives

Programme alternatives (P1, P2, P3)	Aggregate benefit (thousands of £ per year)
(1, 0, 0)	7.192
(0, 1, 0)	7.127
(0, 0, 1)	7.174
(1, 1, 0)	10.229
(1, 0, 1)	10.067
(0, 1, 1)	9.466
(1, 1, 1)	12.048

Source: Santos (1998)

Taking differences between the relevant values in chart 1 yields the sequential benefits of each policy measure when offered with none, one or two other measures already available; these sequential benefit estimates are presented in chart 2.

Chart 2
Sequential benefits of each measure with none, one or two other measures already available

Measure	Bundle of measures already avaliable						
	(0, 0, 0)	(1, 0, 0)	(0, 1, 0)	(0, 0, 1)	(1, 1, 0)	(1, 0, 1)	(0, 1, 1)
P1	7.192	_	3.102	2.895	_	_	2.582
P2	7.127	3.037	_	2.293	_	1.982	_
P3	7.174	2.876	2.339	_	1.819	_	_

Source: Santos (1998)

Note: values in thousand £ per year.

Based on benefit information in chart 1, which assumes 100% uptake, and the actual programme's uptake rate, the ex post benefit of the Pennine Dales ESA for visitors is estimated as circa £6,585,332 a year. Expressed on a per hectare basis, it is possible to notice that this value significantly exceeds the value of agricultural output in farms with management agreement. Hence, from the point of view of society as a whole, the nature and landscape conservation functions of farming in the Pennine Dales is already more important than its food production function.

The total cost of the ESA programme was estimated as £2,599,129 of annual payments to farmers plus £1,389,000 of administrative costs, which gives a gross total of £3,988,129 a year. Yet, the programme originates some extensification of agricultural production, which leads to save £257,494 a year in reduced support expenditure related to the Common Agricultural Policy (CAP). So, the net cost of the programme for taxpayers is £3,730,635 a year.

As all of the estimated nature and landscape conservation benefit accrues to part of the taxpayers (those who are also visitors), the net benefit for all taxpayers (including visitors) is £2,854,691 a year, that is: 66% of the net social benefit generated by the programme.

Farmers, who receive £2,599,129 a year in the form of agri-environmental payments, incur a cost of complying with ESA requirements estimated as about £1,099,159 a year, which leaves a surplus of £1,499,970 for farmers, that is: 34% of the net social benefit generated by the programme.

Globally the programme generates £1.77 (i.e.: £6,585,332 /£3,730,635) of nature and landscape conservation benefits for visitors per each £ spent. This demonstrates that this ESA clearly is good value for taxpayers' money (i.e.: it is an efficient programme). So, it is also demonstrated that the ESA scheme really is contracting the supply of a conservation service for which there is an actual demand, accompanied by a WTP that significantly offsets the programme's costs. It is, therefore, not a mere hidden subsidy to farmers.

Besides, the results of the cost-benefit analysis of this programme enable us to characterise, as seen above, how the policy costs and benefits are distributed among the several groups of agents involved. This is a type of information which is often more relevant for policy decision-makers than global economic efficiency (i.e.: positive net benefits, or benefit/cost ratio > 1) by itself.

Chart 3
Sequential benefit/cost ratios

Measure	Bundle of measures already avaliable						
	(0, 0, 0)	(1, 0, 0)	(0, 1, 0)	(0, 0, 1)	(1, 1, 0)	(1, 0, 1)	(0, 1, 1)
P1	3,51	_	1,96	1,41	_	_	1,63
P2	4,98	3,14	_	1,60	_	2,05	_
P3	82,46	33,06	26,89	_	20,91	_	_

Source: Santos (1998)

Based on the analysis of the sequential benefits of each measure presented in chart 2 and the corresponding costs, it is possible to estimate the sequential benefit/cost ratios for each measure, which are presented in Chart 3. These ratios enable us to carry out a sequential cost-benefit analysis aimed at selecting an optimal bundle of measures, i.e.: the best programme design. In this particular case, we verify that, whatever the sequence in which the three measures are evaluated, and taking into account not only visitors' WTP for the different measures but also policy costs, we select as the optimal programme design that comprising the three measures altogether (notice all benefit/cost ratios are above 1). Note the relevance of this type of information if the design stage of a programme.

4. Conclusions

This paper started by stressing the importance of the evaluation of programmes in the context of the EU agri-environmental policy. Several evaluation criteria were defined and discussed, as well as the roles for evaluation all over the life cycle of a programme (design, implementation, ex post appraisal). The basic methodology used to evaluate, in biophysical terms, the impacts of a programme on several conservation objectives was also discussed. In this context, the importance of a clear and specific definition of the objectives, associated to indicator definition and target setting was underlined. Biophysical evaluation of impacts was said to be aimed to verifying the effectiveness of a programme. Besides, a good biophysical evaluation is a necessary (but not sufficient) condition for a good economic valuation.

The economic valuation cannot, therefore, replace the biophysical evaluation. However, it conveys a set of information on the intensity of public preferences with respect to the impacts of the programme, which supports some evaluation tasks that cannot be validly carried out only based on biophysical evaluation. Some examples of these tasks are: (1) the demonstration that a programme or policy constitutes an efficient application for taxpayers' money; (2) the verification that agri-environmental policy de facto consists of contracting farmers for the delivery of a service, for which there is an actual demand, accompanied by a WTP that offsets the policy costs; and (3) the selection of optimal programme design (bundle of measures) by taking into account both the intensity of public preferences for the different impacts of the programme and the respective costs. These diverse uses for methods of economic valuation of conservation benefits were illustrated based on the results of an empirical study in the context of the evaluation of agri-environmental programmes.

There are, however, some obstacles hindering a more systematic use of economic valuation methods in the context of EU agri-environmental policy. One of these obstacles has a very general nature and involves a series of methodological issues that still wait for completely satisfactory answers. The technical complexity of these issues and the extent of the theme do not allow us to undertake here a satisfactory report of these issues, even a synthetic one[14]. Another of these obstacles refers to the attitudes expressed by policy makers towards the different methods and their empirical results. We discuss next an example of this based on the reactions of the European Commission (1998) to the results of the valuation studies of ESA programmes in the UK.

In general, the Commission thinks that the monetary valuation provides an interesting view on the social value of programmes, but is still as very imprecise method. It is a fair assessment, but we could add that the same document of the European Commission (1998) provides many examples that show that biophysical evaluations of impacts, even those well conducted, are also rather imprecise. Lack of precision might not be a sufficient reason for not using a policy evaluation tool.

In the context of its work group on agri-environmental indicators, OECD (forthcoming) also underlines the interest of methods for the economic valuation of conservation benefits, which, in the absence of market prices, can be used to “establish the value of agricultural landscapes for society” and thus can “help decision makers to determine the benefits of conserving and restoring landscapes”.

When facing the results of the study of UK’s ESAs, the European Commission (1998) shows some perplexity as regards the disproportion between the value of the benefits and that of the costs when aggregated for all studies ESAs: “the claim the UK programmes secure ECU 275 million of

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[14] The interested reader is referred to a number of reference texts: Cummings et al. (1986), Mitchell and Carson (1989), Carson et al. (1999), and Hanemann and Kaninen (1998), as regards contingent valuation; Bockstael et al. (1991), and Ward and Beal (2000) for the travel cost method; and Palmquist (1991) as regards the hedonic prices method.

benefit for only ECU 11 million of expenditure seems difficult to sustain…”. Although there is something reasonable as regards this feeling of perplexity, we should add some comments:

- for non-rival public goods, as it is typically the case with many agricultural landscapes, the vertical sum of marginal benefits for the same unit of the good (e.g.: hectare of land or individual of a threatened species) can lead to extremely large (though correct) unit values of WTP for preservation (non-rival good) when compared to WTP for private uses (farming, fishing, hunting…) of a unit of the same good (Bishop and Welsh 1992);

- we are dealing with goods for which there is no markets that establish an equivalency between marginal benefit and marginal cost (through market equilibrium; see McConnell 1992); hence, there is no scientific basis on which to assess whether aggregate marginal benefit is disproportionate as compared to marginal cost;

- to secure an aggregate benefit estimate of ECU 275 million for the set of all studied ESAs, the Commission simply summed the estimated benefits for each ESA; this is a practice subject to a well known aggregation bias, known as the independent-valuation-and-summation (IVS) bias; Santos (2000) suggests that this bias (overvaluation in this case) can be rather large; thus the fact that the sum of benefits for all studied ESAs seems uncomfortably large is, at least in part, due to an invalid aggregation procedure.

Eventually some should be said in favour of the reasonableness of the feeling of perplexity of the Commission facing such seemingly uncomfortably large estimates. In fact, there are two serious problems in many studies of contingent valuation of public goods, which simply multiply a per-capita sample-average WTP estimate by the estimated population to secure an aggregate benefit. First the population estimate can be very imprecise. Second, the sample that delivered the average per-capita WTP estimate can be no-representative of the population as a whole. Bateman et al. (1999) show that, in an empirical example, the aggregate WTP estimate delivered by the usual process (average WTP per capita x estimated population) is reduced by 75% when one adequately controls for the following factors: (1) decline in WTP per capita when one moves away from the site; (2) spatial distribution of the population around the site; (3) spatial differentiation of the population's characteristics; and (4) increase in the non-response rate (mail survey) with the distance to the site.

Solving all these problems requires a significant advance in research on non-market valuation methods, so that it is possible to more effectively ensure the validity and reliability of particular empirical applications to the estimation of nature and landscape conservation benefits. However, in the absence of alternative methods for many of the uses that economic valuation can have in the context of EU agri-environmental policy, e taking account that also low levels of precision characterise biophysical methods (currently accepted and recommended by the Commission), the discussion in this paper suggests that a claim for a more systematic (though cautious) use of economic valuation methods in this context is warranted.

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