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Managing agriculture's impacts on the environment : basic economics and international evidence

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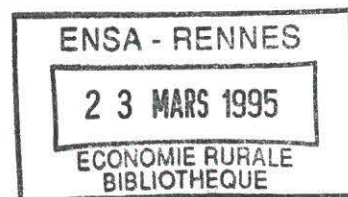
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Managing Agriculture's Impacts on the Environment:

Basic Economics and International Evidence

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**Managing Agriculture's Impacts on the Environment:
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March 1995

Introduction

ESAs represent a policy mechanism aimed at inducing changes in economic activity to increase social welfare. The value of the ESA mechanism results from the net social benefits it induces. It follows that estimation of the benefits of ESA applications must be grounded on an understanding of the change in economic activity that the application hopes to induce, the changes in economic goods that result, and the sources of social value of those changes in available goods. Most generally, ESAs can be viewed as a mechanism for inducing changes in current and future environmental performance. Authors have presented varied definitions of the goals of ESAs, however, the primitive concept that can be extracted from each of these definitions is that of change in environmental performance. More specifically, the environment of interest is the nonurban, land related environment. In the U.K. this environment has been labeled *countryside*, while in the U.S. literature the term *landscape* has taken on a generalized connotation.

A rather limited literature has considered the public economics of the ESA mechanism in general, or of estimation of the benefits associated with particular applications of ESAs. Colman et al. (1992) and Colman (1994) present a useful description of alternative conservation instruments and an assessment of their relative effectiveness based on a menu of criteria (e.g. capacity to protect and enhance, timeliness, targetability, monitorability, cost efficiency, political acceptability and transparency, promotion of conservation-mindedness). Bateman (1994) and Willis and Garrod (1994) present brief considerations of the microeconomics of benefit measurement and more thorough overviews of alternative approaches. However, in each case, these presentations are limited to considerations of the

traditional case of benefits attributable to provision of a public good. In the case of ESAs, the linkage between a program and public good provision is obscured by both the complexity of programs as well as numerous types of uncertainty of program effects. Further, the intended economic effects of ESAs often go beyond provision of public goods to include management of externalities and private provision of public bads.

The objective of this paper will be to reconsider the valuation of ESAs at both a conceptual level and to reassess available evidence of benefits generated by ESAs within this context. The next section reconsiders the origins of social value induced by ESA mechanisms. In so doing, the section considers the character of the economic goods that might result from use of the ESA mechanism, the mechanism through which the supply of these goods is induced, and the bases of social value of these goods might generate. Next, the microeconomics of valuation of these goods is reconsidered graphically and conclusions are drawn concerning what frameworks are appropriate for general application. The fourth section presents an assessment of available evidence concerning the value of particular ESA applications. Finally, conclusions are drawn.

ESAs as a Mechanism that Induces Increased Social Value

The objectives of the ESA approach has been popularly understood to be the safeguarding of particular characteristics (landscape, wildlife, and historical features) of rural land through the management of agriculture's role in the landscape, Willis et al. (1993). The concept emerged in the 1980s in response to public concern over two types of agriculture outputs: 1) externalities associated with intensive agriculture and their effects on the rural landscape and 2) perceived beneficial effects of less intensive, traditional agriculture and its role as a distinctive and valued element in rural landscapes. Traditional agriculture was perceived to contribute to preservation of valued landscapes (e.g. meadows, grasslands, heathers, etc.); landscape functions such as wildlife habitat provision; to reduced off-site effects through hydrologic stabilization, windbreaks that reduce soil displacement, etc.; and to preservation of archaeological and architectural features. In this sense, these beneficial effects were generically classified as public goods, although economists would classify them more precisely.

The notion of an ESA was introduced in European Community law within the context of agri-environment schemes which could be adopted on a voluntary basis by member states,

Beaufoy (1994), Whitby and Lowe (1994). Importantly, it emerged as a result of public recognition and valuation of the extensive variety of nonmarket economic goods or services produced by agricultural land use. In the U.K., ESA legislation emerged under Section 18 of the Agricultural Act of 1986 and led to five ESAs being designated in 1987, six in 1988, six in 1993, and six in 1994, Willis et al. (1993, p.5), Whitby and Lowe (1994). During the first rounds, farmers were offered fixed period contracts providing annual fixed payments in return for adoption of a set of farm production practices. These practices involved requirements as well as prohibitions on daily and seasonal activities with a focus on chemical application, stocking densities, infrastructure installation such as drainage or fencing, and management of landscape features. Often, these practices were adopted as part of an overall plan for the farm. Payments were based on area treated by the practice or prescription. Adoption rates have been high and shown steady growth in the U.K.

In France, ESA applications were initiated between 1991 and 1993. In 1993, ESA applications were brought under European Community initiative (CEE 2078/92). As in the U.K., the areas of application were spatially limited, however, the French applications involved more extensive negotiation with farmers of the terms of the contracts to be offered in each area. French applications focused on two goals: 1) *ex situ* preservation of biodiversity or specific biotopes (30 ESAs by 1993 encompassing about 220,700 ha with about 83,700 ha enrolled) and 2) preservation of extensive agriculture and reduction of land abandonment in areas of traditional, yet highly dispersed agriculture (24 ESAs by 1993 encompassing about 509,000 ha of which only about 115,800 ha was eligible for financing). The second goal was intended to protect natural areas from abandonment or encroachment of more intensive uses. In total these two goals involved only about 700,000 ha of which only 205,000 ha were eligible for participation (i.e. farmed land). The budgetary commitment was 92.5 million FF. A small number of ESAs were initiated, though later abandoned, which were focused on water pollution and fire protection in forested pasture. Implementation led to a substantially smaller number of ESAs due to administrative delay, or under enrollment and closure. Despite farm representation in the contract specifications, participation was low, CNASEA (1992).

Most recently, the agri-environmental regulation (2078/92) established a package of measures which are mandatory for all states to implement through zonal programs in their territories. Rather than limiting these schemes to particular areas, their zonal focus would allow them to be available to all areas within the territory with such zonal characteristics.

Each of these programs offers standard payments to farmers to take actions which produce public goods and reduce externalities. These actions are intended to simultaneously maintain agricultural land use while shifting agricultural practices toward those that benefit the environment. Under the regulation, member states presented proposals for such zonal programs. The details of these new applications have been reviewed by other papers presented in the volume.

Throughout the historical evolution of ESAs, political and social debate has obscured the underlying public economics of ESAs. We next review these foundations to clarify the origins and characteristics of changes in social value they might induce. Accepting the focus of ESAs on altering the performance of the rural environment or countryside, we begin by reconsidering the nature of performance with which are concerned. Most typically cited as the target of ESAs are changes in externalities and environmental outputs produced by agriculture, Crabtree (1992), Whitby and Lowe (1994). This definition can be extended to include changes in the production of negative public goods, i.e. public bads, such as generation of CO₂ and ozone destruction or habitat destruction and displacement. Within a temporal sense, this list of performance goals appears to focus on contemporaneous effects. However, it is clear from most discussion of ESAs that their intent is to alter the intertemporal effects of agriculture on the environment. In this sense, the rural environment is viewed as dynamic system and the objective of public management is to divert that system from its current or future paths associated with purely private decisions to a path which is viewed as having greater social value. It is within this sense that ESAs encompass goals such as improvement of habitat, stimulation of wild species, or elimination of invasive species.

The focus of ESAs is on agriculture. Within the rural setting, man's impacts occur through agriculture, forestry, recreation, resource extraction, and infrastructure installation and use (e.g. highways, dams, etc.). Agriculture is the predominant occupation of rural land and, therefore, dominant as an economic activity through which man affects nature. Therefore, it is natural that agriculture was targeted as a basis for affecting the value of nature. Further, agriculture involves active management of the rural landscape. Numerous analyses of ESAs present commentary on the role that post-war period intensification (chemical use, varietal homogenization, more frequent cropping) and expansion (drainage, grassland conversion) of agricultural activities have played in altering the nonmarket goods supplied by such activity, e.g. Froud (1994), O'Carroll (1994), Saunders (1994), Coleman et al. (1994).

We next consider the mechanism through which ESAs operate. Here, we depart from the literature to focus on the economically operative elements of the mechanisms. We define the ESA mechanism as involving standardized contracts to purchase changes in agricultural production practices (by requirement or prohibition) affecting land through payment systems. They are offered to farmers within a targeted subset of the population. Until recently, targeting was accomplished on a geographical basis imposing spatial limits based on specific environmentally sensitive areas allowing targeting of the contracted group of providers, the ecological and other physical characteristics of the environment affected, and the beneficiaries of the public good or reduction of the externality. In the PAC reform, the basis for targeting was redefined to include land or ecological characteristics, or zone types.

Most generally conceived, an ESA represents a social contracting scheme which purchases changes in production practices which are hoped to induce private provision of public goods, reduction of public bads and reduction in externalities from private owners of rural land. Weaver (1994) presents an economic theory of *environmental effort* by farmers which clarifies the need for such social contracting. In brief, Weaver notes that where farmers hold nonhedonistic preferences, a contracting approach can encourage private voluntary contribution in a way which is not possible using traditional Pigouvian instruments such as regulatory standards or incentives such as penalties or taxes. Despite the fact that the ESA mechanism is a contractual purchase arrangement it is centrally directed and typical applications have offered homogenous standards and incentives to heterogeneous populations of farmers. In all past applications, the mechanism has operated indirectly in the sense that contracts are for private good production practices which are predicted to result in production of desired public goods or reduction in externalities or public bads. Often the contracted production practices may produce joint multiple outputs or effects. The contracts have involved restriction of production practices either by prohibiting use of particular activities which are perceived to result in public bads or externalities or by requiring use of activities that are perceived to result in public goods production.

From this perspective, Crabtree (1993) and others have questioned whether the ESA mechanism can be interpreted as a property rights transfer mechanism. For this paper, it is of interest to ask whether such an interpretation provides a useful basis for valuation of ESA applications. Crabtree (1993) properly noted that the ESA mechanism involves a restriction of the production possibilities set available to farmers. Whitby (1989) adds the additional

presumption that the farmers hold a property right to freely choose production practices. Based on this presumption, ESA mechanisms which impose restrictions on production practices can be viewed as involving a transfer of property rights from the farmer to society in return for a cash payment. The cash payment can then be interpreted as equal to or greater than the farmer's valuation of that property right. This perspective would seem appropriate for contracts which prevent the farmer from using practices or taking specific actions which produce public bads, or externalities. For the case of contracts for production of public goods, Crabtree (1992) concluded the property right transfer interpretation is less useful. In this case, the contract represents a purchase of private provision of a public good. Where contracting is for direct provision of a public good, the contract can be viewed as providing direct incentives for a particular output and no property rights transfer is implicit. In contrast, where public goods are only indirectly contracted, e.g. by contracting only for the use particular production practices which are perceived to result in production of the desired public good, a property right transfer interpretation would appear to remain useful. As a basis for valuing ESA applications, the property right transfer interpretation provides a basis by which the ESA value could be viewed as based on social valuation of ownership of the right. That is, social value would indicate society's valuation of the farmer not having the right to pursue particular production activities. However, from a practical standpoint, because this right is at best only indirectly linked to the reduced production of an externality or public bad, or increased production of a public good, this type of property right provides a very limited basis for establishing value of an ESA application.

The ESA mechanism applications have also allowed for more direct contracting for the provision of public goods. For example, the European Community legislation clearly recognizes farmers as having the potential to produce environmental goods through their direct management of the countryside. For example, farmers can privately invest to produce various landscape elements such as woodlands or wetlands, or landscape characteristics such as wildlife through introduction of species (see Crabtree and Appleton, 1992.). In these cases, property rights transfers are not involved, and instead fairly traditional production economics suffice to analysis the supply of these goods through acceptance of contracts. From the perspective of valuation, provision of these public goods will typically involve marginal changes in the aggregate supply allowing marginal valuation. In many cases, however,

valuation will be complicated by the fact that the goods are not pure public goods, but rather may involve some aspects of limited access, at least in a spatial sense.

Given that ESA mechanisms can be interpreted as contracting mechanisms, it is useful to clarify the extent to which they are interpretable as management agreements. Colman et al. (1992) define management agreements based on their historical application in the U.K. In this case, they are defined as individually negotiated contracts between a public agency and a land owner to restrict or initiate land management activities. They classify ESA mechanisms as standard payment systems in which contracted practices and payments are the same across participating farmers. Here, our intent is not to confuse the literature, but to distinguish among the operative elements of the ESA mechanisms. Clearly, they involve contracting for private management either restricting or requiring particular practices.

To summarize, several features of the ESA approach are important to note: 1) it may involve either indirect or direct provision of public goods, reduction in public bads and externalities, often these outputs are multiple and jointly produced (along with other public as well as with private outputs) by particular environmental efforts of the farmer, 2) it allows targeting in a variety of dimensions, 3) it relies on voluntary participation by farmers, 4) it involves contracts with standard payments as incentives and standard management agreements as regulatory constraints across participants.

The specific characteristics of particular ESA applications can now be placed within the context of this more general set of characteristics. Specific applications of ESAs have been reviewed by Willis (1994), Merlo and Puppa (1994), Riera (1994), Dubgaard (1994) (in Dubgaard, 1994), Moss (1994), Fraud(1994), O'Carroll (1994), Saunders (1994), Coleman et al. (1994), Hughes, (1994) all in Whitby(1994); Willis et al. (1993), Coleman et al. (1992), Todd (1993), and Bonnieux, et al. (1993). Each of these applications involved multiple standard management agreements and standard payments which farmers choose from a menu of available option. Thus, while the general mechanism is uniform across applications, the composition of the menu offered in terms of contracted or restricted production activities varies widely as do the perceived public goods, bads, or externalities that are the objective of the contract. In practice, the objectives of each ESA and the expected effects or outputs necessarily varies depending on the character of the demand for and potential supply of its public characteristics. Applications reported in Whitby (1994) illustrate this variation as well

as the variation of participation and choice of menu elements across the heterogeneous farm population.

A review of the possible nonmarket goods induced by ESAs highlights the variety of types of public outputs expected from these programs. Outputs include pure public goods such as preservation habitat diversity to encourage speciation to congestible, spatially limited quasi-public goods such as landscape use in hunting, viewing, or exercising. Importantly, the outputs are often multiple and jointly produced by the farmer's environmental effort. For example, actions taken by the farmer to enhance the visual amenities of the farm site will typically alter habitat inducing long term benefits related to changes in biodiversity and shorter term benefits related to wildlife use activities (e.g. hunting, viewing, etc.). Outputs also vary in the extent to which they generate utilitarian vs. nonutilitarian benefits, the length of the horizon over which these benefits will be generated, the uncertainty with which they will be generated, the dynamic path of supply associated with the goods or their characteristics, as well as their jointness in supply and/or demand or benefit characteristics. Finally, the interaction of these goods or outputs with private goods or services also varies substantially.

The Origins of Social Value Created by ESA Applications

The origin of social value that ESA mechanisms hope to create lies ultimately on the value that humankind places on nature. ESA mechanisms hope to create a change in the environmental performance of the rural landscape. In this sense, they attempt to affect marginal changes in the current and future state of nature. Operatively, ESA mechanisms affect nature through standard management agreements. While the ESA designation of an area may have some impact independently of any effect produced through changes in management, the basis of such changes is unclear. Here, we consider only changes induced through standard management agreements. At the most general level, the notion that nature has value is complex, Kim and Weaver(1994). It is clear, however, that across cultures and time humans have placed value on nature. Within this context, nature has been understood to mean all things not created by man. The value of nature has been extensively considered within a number of disciplines. Economists have focused on utilitarian value, Weaver (1994). While other disciplines have recognized nonutilitarian values associated with nature, Norton (1994), Rolston (1994), Katz (1994). Utilitarian value can be decomposed into direct consumption

value, option value, and existence value. Nonutilitarian value systems necessarily go beyond homocentric perspectives that view nature as an entity that serves humankind, Hargrove (1994), Weaver and Kim (1994). Within this taxonomy, it is clear from the list of environmental effects or outputs expected from past ESA applications that the social value of these effects or outputs may involve all elements of this taxonomy.

Based on the characteristics of the ESA mechanism as well as the diversity of economic goods its application is hoped to induce, the challenge of valuation of a particular ESA application is striking. While the challenges of valuing landscapes have often been noted (e.g. Weaver and Kim(1994), Price (1994)), their consideration has often been limited to the problem of valuing a particular known change. In the context of ESA applications, numerous types of environmental efforts are implemented, which are hoped to induce a diversity of public goods, bads, and changes in externalities. Thus, the valuation problem must necessarily consider a vector of changes. While the jointness in production of these outputs has been noted, their consumption may also be joint. Focusing on single output effects, Bateman (1994), Merlo and Puppa (1994) among others have presented reviews of valuation approaches that may be useful. However, these reviews consider only valuation of known changes in a particular public good. As was argued in the previous section, the problem of valuation of a particular ESA application is considerably more complicated. To simplify, while highlighting this complexity, suppose an ESA application produces a single public good, however, suppose we allow it to have characteristics the supply of which is dependent on 1) uncertain participation by farmers, 2) uncertain efficiency of farmers production, and 3) uncertain technological feasibility; and where they may be said to be characteristics 1) with utilitarian value, 2) with nonutilitarian value, 3) with uncertain current value, 4) with uncertain future value, 5) with pure public good aspects, 6) with spatially limited access aspects, 7) which substitute with some private goods, and 8) which complement other private or public goods. The problem of valuation must necessarily involve 1) valuation of different types of goods, 2) valuation over different horizon, 3) valuation under different extents of uncertainty, 4) valuation based on different approaches, and 5) aggregation of these heterogeneous sources of value, possibly recognizing complementarity in demand. Given this complexity, the necessity of treating each output of an ESA application is clear. Coleman et al. (1993), Willis et al (1993) provide illustrations of such individualized treatment.

At this point, we are in a position to consider an obvious question: What are the implications of this complexity for comparative assessment of value across applications? To suppose comparative assessment is of interest requires that the same change is made across applications. The complexity of the ESA mechanism and its variation across applications implies that such a common change does not exist, in general. Even where the same practice is restricted in a set of applications, the output goals and effects may be different. Where the same goal or output effect can be found, comparative value assessment might be feasible. However, such feasibility would rely on independence of that output from others produced by the applications, unless those other outputs are also identical across applications. Where such independence does not exist, comparison would be difficult. Consider an example. Suppose in two applications the public good output of improved visual value of the landscape is sought and attempted by field perimeter plantings of poplar trees. Suppose in both applications, intensive field cropping occurs, however, assume in application A cold winters imply substantial wind induced soil erosion occurs while in application B, such erosion does not occur. To compare the visual value of the treatment across the two applications would clearly require identification of the value of the erosion protection and air quality services in application B. While this example is sufficiently simple to suggest that such evaluation would be possible, typical ESA applications involve a large number of outputs and interacting practices.

Valuation of When Public and Private Outputs are Joint

Before moving to consider and interpret available evidence of the value of ESA applications, we present an economic framework which clarifies many of the complications already discussed. In particular, the problem of valuation of the public good is extended to consider the case where it is jointly produced with private goods, as is the case in ESA applications. Consider the simple case where the ESA application results in a vector of public goods (e.g. ecological and amenity) and a vector of private goods (agricultural outputs). We would like to determine the value of public goods, i.e. the social rate of exchange or trade between the public and private benefits generated. Define social welfare U as a function of a vector of public goods Q and a vector of private agricultural commodities X :

- 1) $U = U(X, Q).$

Define the product transformation function that bounds the set of all technologically feasible combinations (X, Q) following Weaver and Harper (1993):

$$2) \quad G(X, Q; Z, \theta) = 0$$

where Z is a vector of environmental inputs and θ is a vector of quasi-fixed, private inputs

The social choice problem is to find a feasible combination (X^*, Q^*) which maximizes social welfare 1) subject to available technology described by 2). Graphically, the solution of the problem illustrated in Figure 1 by A^* which lies at the tangency of the isosocial welfare curve U^* and the production possibilities frontier G^* . At the tangency, the exchange rate defined in terms of social welfare (marginal rate of social welfare substitution MRSWS) between the goods is set equal to the exchange rate between the goods defined by technology (marginal rate of technical transformation MRTT) between the goods. Stated differently, the rate at which society is willing to trade Q for X is set equal to the rate at which society can actually trade Q for X as defined by technology.

The problem of valuation is clear from Figure 1 and the character of this tangency. For market goods, prices serve as signals to which consumption and production adjust to establish the conditions at point A^* . Where prices do not exist, we seek alternative means of evaluating the exchange rates of Q for X defined at point A^* . The first conclusion that is apparent from this simple graphic and the logic of economic optimization is that when prices are not available, it is more difficult to determine the position of A^* . What we need in terms of valuation is not simply a value at A^* , instead we need sufficient information to help us find A^* . In terms of the graphics of Figure 1, what we need are estimates of how the exchange rates MRSWS and MRTT change as Q and X are changed. Suppose point Q' were offered to consumers. The value of point Q' could be established by standard methods of valuation. One approach would be to measure the willingness of society to exchange Q for X (MRSWS) along U' , a lower level of welfare, finding $MRSWS' < MRTT'$ signaling that a reduction of Q and an increase in X would increase social welfare.

Figure 1.

From another perspective, consider a change in the environment from Q^* to Q' , i.e. such that $\Delta Q > 0$. In Figure 1, ΔX_1 measures the amount of X society would be willing to pay have Q' stay at the same utility level U^* as before the change in Q . Alternatively, ΔX_2 indicates how much X would have to be paid to allow Q' to be technologically feasible. Under standard curvature properties shown in Figure 1, $\Delta X_1 < \Delta X_2$ which implies society would not be willing to pay as much as technology would demand be paid for an increase in Q from Q^* to Q' . Conversely, the same logic could be applied for a decrease in Q from Q' to Q^* . In this case, society would be willing to give up Q at a rate that exceeded that which is demanded by technology and the adjustment to Q^* would be welfare increasing. Finally, in Figure 1, ΔR defines the amount of X that a market, if one were operative in both goods, would demand be accepted for a decrease from Q' to Q^* . Figure 2 presents the same logic for the case where an increase in Q is optimal. Initially Q equals Q^o , and $\Delta X_2 < \Delta R < \Delta X_1$ as $Q^o < Q^*$. Using the same logic as was applied to Figure 1, it is clear that welfare can be increased by increasing Q^o to Q^* .

Figure 2.

Finally, the same logic can be applied to when Q denotes a public bad as described in Figure 3. Here, free disposal is not assumed for the public bad.

Figure 3.

To consider hedonic price estimation approaches, Figure 1-3 are also useful. For example, in Figure 1, the slope of line P^* indicates the relative prices of X and Q , i.e. $P^* = P^*_Q / P^*_X$ where the asterisk indicate prices at the social welfare optimum A^* . Hedonic approaches attempt to estimate P_Q at the observed set of goods. However, where no markets are available to force P to P^* and goods to A^* , there is no basis for determining the relative prices and hedonic approaches are infeasible. In Figure 1, when the line P^* is not defined relative to the slopes of G and U^* at A^* ; no basis exists for fixing its position and its slope is indeterminate. Where an implicit market exists for Q , hedonic methods can be used to estimate the implicit price P_Q . For example, the use of a rural setting could be estimated from prices of leisure homes in the setting, Willis and Garrod (1993).

Valuation of Rural Landscape Amenities

Given the conclusion that the value of ESA applications is best based on valuation of each of multiple and possibly joint public effects which are the goals of the application, we now provide a review of available evidence concerning the value of one of the most typically expected products of ESA applications: rural landscape amenities. Several studies have been conducted in which the value of agricultural landscape was estimated by means of contingent valuation. In comparison with CV studies in the field of recreation, application of CVM to rural amenities is more subject to misspecification of the product. In particular, geographical part-whole bias and policy-package part-whole bias are likely to result from inadequate specification of the amenity. Geographical part-whole bias occurs when a respondent values a landscape whose spatial attributes are larger or smaller than the spatial attributes of the researcher's intended landscape. There exists empirical evidence that people are able to discriminate a local landscape from a national landscape. Thus, specification of the geographical scope of the amenity is essential. Consider a respondent who is asked how much he is willing to pay for maintaining today's landscape in an ESA. If the respondent is unable to isolate that area in his mind from a larger area covering the specific ESA, the respondent may in fact value a larger area than intended by the researcher. Policy-package part-whole bias occurs where a respondent values a broader or a narrower policy package than the one intended by the researcher. This issue is of particular importance within the context of ESAs. Given that agricultural activities and particular ESA applications can produce a wide variety of public effects which may be associated with both current and future time periods, the precise nature of the public effect to be valued by respondents must be specified. Where this has not been accomplished, response can not be associated with particular public effects and in the extreme, may be based on a vague and poorly specified notion of the entire menu of public goods supplied by agriculture. In the context of the Common agricultural policy reform, people may treat the ESA scheme as symbolic of a larger policy goal (e.g. preservation of farming as a way of life) and assign to the scheme some of the values they have for this more general goal.

Table 1 gives a sample of studies whose objective is to value the preservation of today's landscape. Studies covering very limited portions of land such as specific habitats or sites of scientific interest (SSIs) are not included. First, the challenge of comparison of results from various ESA applications is clear. Not only does methodology differ across studies, but

studies rarely provide sufficient information to allow even the calculation willingness to pay in a common currency or measure.

Table 1.

As already noted, the particular policy package involved in each study differs and it is unclear how estimated values can be related to particular policy elements, or management practices. Drake (1986) considers WTP to maintain an open landscape in the whole Sweden whereas Le Goffe and Gerber (1994) and Dillman and Bregstrom (1991) address the landscape issue at a local level. Taking into account the average size of a Swedish household, Drake's figure is the highest and appears consistent with our expectations. However, no objective basis exists for validation of these estimates. Hackl and Pruckner (1994) estimated roughly a similar level of WTP for the provision of agricultural landscape cultivating services in Austria. Nonetheless, the context of their research is different since the countryside is considered an important input for the Austrian tourism service. The large difference in WTP estimates between the two other studies are attributable to the difference in local context. It is not easy to compare WTP across the other studies since some refer to the household and others to visits. Riera (1994) considers benefits of preserving a large area of high landscape quality in the Catalan Pyrénées and Rebolledo and Perez y Perez (1994). Consider a broader good which is the maintenance of a natural park close to the Aragon Pyrénées. While estimates from these studies appear consistent, a review of the literature shows a diversity of estimates. For example, Merlo and Della Puppa (1994) have reviewed Italian estimates of the public benefit values of forestry and farming areas and found a high degree of occurrence. This difference is due both to differences between sites and to differences in the way in which methods have been applied. The majority of benefits values range between 3 and 10 Ecu per person per visit. Bateman et al. (1992) in Willis, Garrod and Saunders (1993) consider ESA's. The first study refers to the Norfolk Broads which under threat from flooding whereas the second is based on the South Downs and the Somerset Levels and Moors. So differences in WTP could be attributable to differences in goods which are valued.

In our opinion a more important point has to be made: the value of a specific ESA is conditional on the continued protection and preservation of the other ESAs. So it can be assumed that the marginal benefits of an ESA decline as the number of ESAs increases. This

implies that aggregate total benefit curves increase at a decreasing rate with the protection of additional areas. This issue was made clear in the context of the preservation of wild and scenic rivers in Colorado (Walsh et al., 1985) and has been taken into account by Willis, Garrod and Saunders in their investigations.

Table 2 provides aggregate values for several areas. The range of WTP per hectare is very large. The dispersion cannot be attributed to differences in unitary values. It mainly comes from the definition of the population concerned by the good. For the resident component things are quite clear. It is more difficult to delineate the visitor population specially for a fuzzy good such as landscape. Moreover we are confused regarding what are the good criteria to define the non-user category.

Table 2.

There is a limited number of hedonic price studies whose objective is to value landscape and countryside amenities. Two specific studies referring to the UK (Garrod and Willis, 1993) and France (Anonymous, 1994) aim to be mentioned. The first study encompasses some 4 800 km² of central England and the Welsh borders and is concerned with the market of houses in rural areas (the data base includes 2 000 observations). It is constrained by empirical data since the environmental data are neighborhood-specific rather than house-specific, relating to 1 km² in which the house is located. It was not possible to estimate the demand for particular countryside characteristics. The second study encompasses all France and there are 3 300 observations, the dependent variable equals the rent for a vacation house ("gite rural"). It suffers the same limitations regarding environmental data as they are defined at the "Département" level. These variables include specific components of land use e.g. arable land, forest-land. So it is possible to infer the impact of various changes in land use on rents using the hedonic function. For example in a specific "Département" ("Manche") the increase in total rent would equal 45 F per hectare in response to the conversion of 1 000 hectares of arable land to permanent pasture.

Table 3.

Conclusions

The objective of this paper was to reconsider the public economics of the ESA mechanism and the valuation of the effects of ESA applications. At the core of the ESA mechanism lies a presumption that manipulation of private good production activities will affect environmental processes to generate a change in associated public good effects. This logic relies on the existence and nature of a relationship between private and public good production processes. Our first conclusion is drawn from our consideration of this relationship. As we noted, the technology of producing public effects through agricultural activities is one that involves three important characteristics: 1) multiple private and public effect outputs which are often joint, 2) uncertainty due to randomness of environmental inputs, and 3) site specificity. With respect to the latter characteristic, it is important to note that scale, scope, biophysical, and manager characteristics are likely to be important. The immediate implication of these characteristics is that even if the ESA mechanism involved only one instrument, multiple interacting affects would be expected, implying that valuation must span the set of generated effects. Unfortunately, the ESA mechanism involves multiple instruments rendering the problem even more complicated. Because multiple effects are likely to be interactive, and because of the jointness of the underlying technology, it is unlikely that the effects of an individual instrument will be identifiable. The conclusion can, therefore, be drawn that valuation of ESA mechanisms must simultaneously consider the application of the entire package of instruments involved in the ESA and value the package of effects generated, including both private and public effects. Going one step deeper into a potential mire, it is interesting to note that since application of the package of instruments induces changes in production practices, the cost of an application is not temporally separable from the benefit stream as in many cases of public investment. It follows that valuation of effects (i.e. ex post analysis) must be broadened beyond benefit analysis to include cost analysis as well.

A second conclusion can be drawn from the theoretical consideration of the ESA mechanism. In many cases, ESA applications will generate both local public goods which verge toward club goods for which exclusions are feasible, and, more classic public goods. Consider the case of actions directed toward biodiversity. Maintenance or redevelopment of hedges can be expected to generate local public benefits in the form of aesthetics or use (e.g. hunting) while at the same time generating a classic public good of biodiversity preservation. While the local/national dichotomy has often motivated concern for geographical part-whole

bias, we note that the local vs. classic public good nature of effects will also involve what we label conceptual part-whole bias. That is, if respondents hold a conceptual basis for valuation that is not identified in the survey process, their response may follow from either local geographical or a local conceptual base. While geographical part-whole bias may be addressed through off-site surveys, the conceptual part-whole bias must also be addressed through specification of the effects of the ESA. Clearly, where the conceptual basis is unknown to the researcher, aggregation of individual benefit estimates will be problematic.

With respect to methods, it is clear from our analysis that contingent valuation approaches, though limited by the realm of complications we have discussed, remain preferable to hedonic or travel cost approaches. From the perspective of part-whole biases, it is clear that these latter approaches offer very limited opportunities for consideration of such biases. Nonetheless, our review provides a solid basis for the conclusion that the utility of CVM studies could be expanded by the adoption of common bases for their implementation bases for their implementation. Through such harmonization in approach and method the development of an expanding data base would be ensured as well as would an increased level of comparability across studies. For this reason, we recommend that effort be placed toward such harmonization in both the valuation and in the design of ESA applications. Such harmonization, in our view, should be targeted at ensuring that an adequate base is developed to provide scientific comment on the following policy questions: 1) what is the level of net benefits generated by ESA applications? 2) to what extent do ESA applications? 2) to what extent are ESA instruments neutral in their affects on production decisions? and 3) what are implications of past ESA applications for future applications?

To provide at basis for comment on such questions, we see it as essential that guidelines be established to harmonize data collection and valuation approaches. In both cases, we do not support rigid standards, instead we propose the adoption of guidelines that ensure room for innovation while retaining a sufficient basis for comparative analysis. We propose that the bases for such harmonization be developed through a multinational effort that ensures the eventual convergence of approaches and confidence in results.

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Figure 1 - Valuation of a decrease in environmental goods

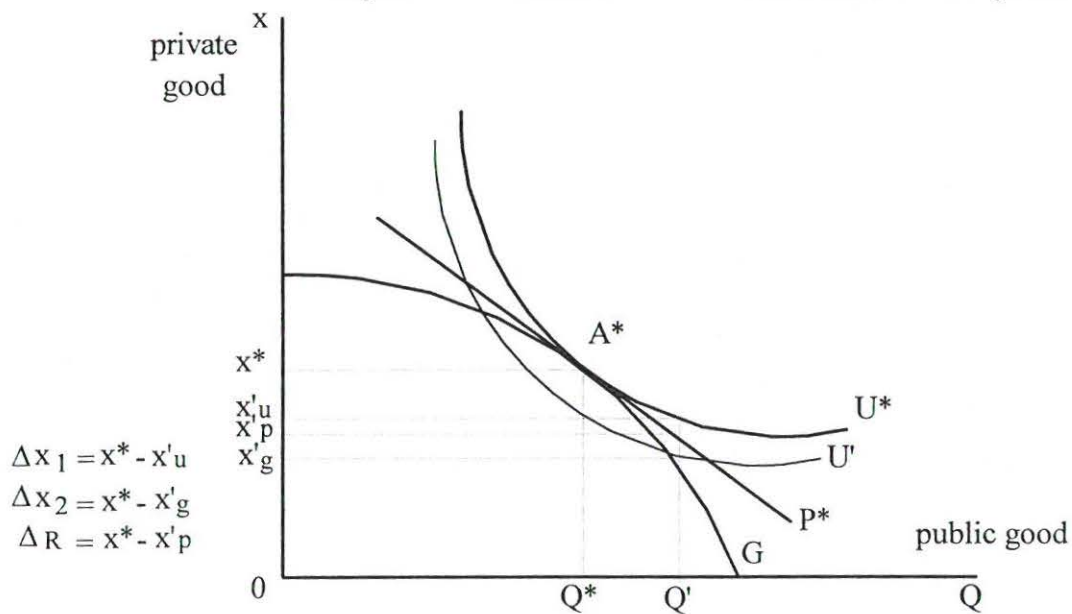


Figure 2 - Valuation of an increase in environmental good

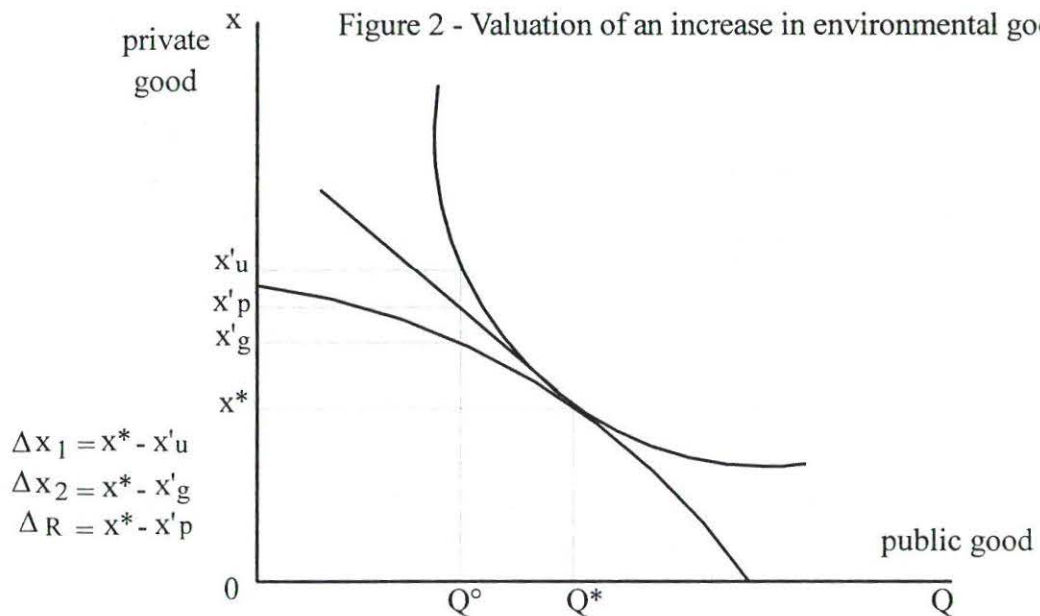


Figure 3 - Valuation of a decrease in environmental bads

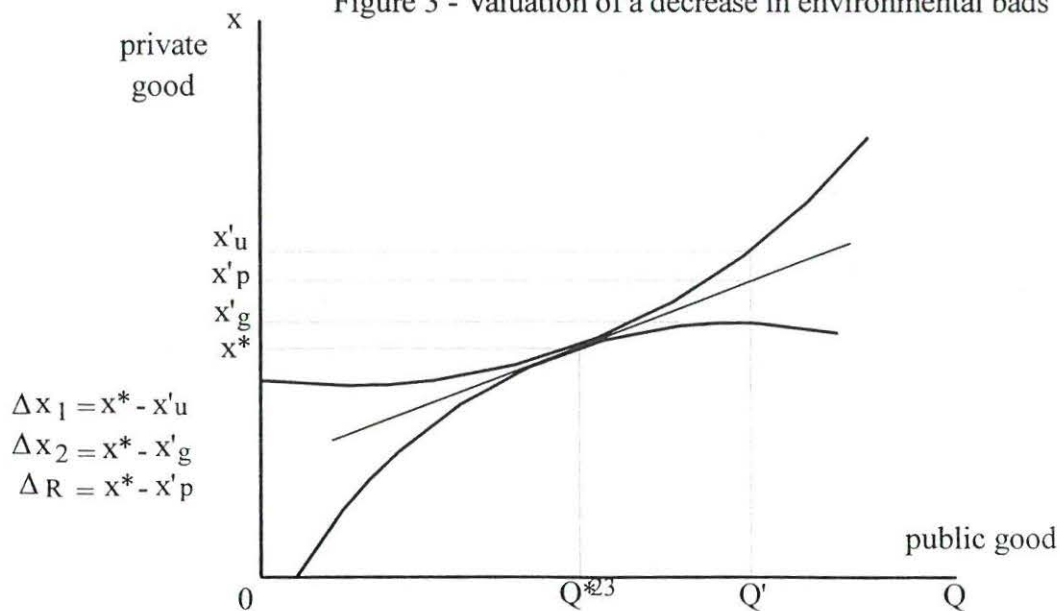


Figure 1 - Valuation of a decrease in environmental goods

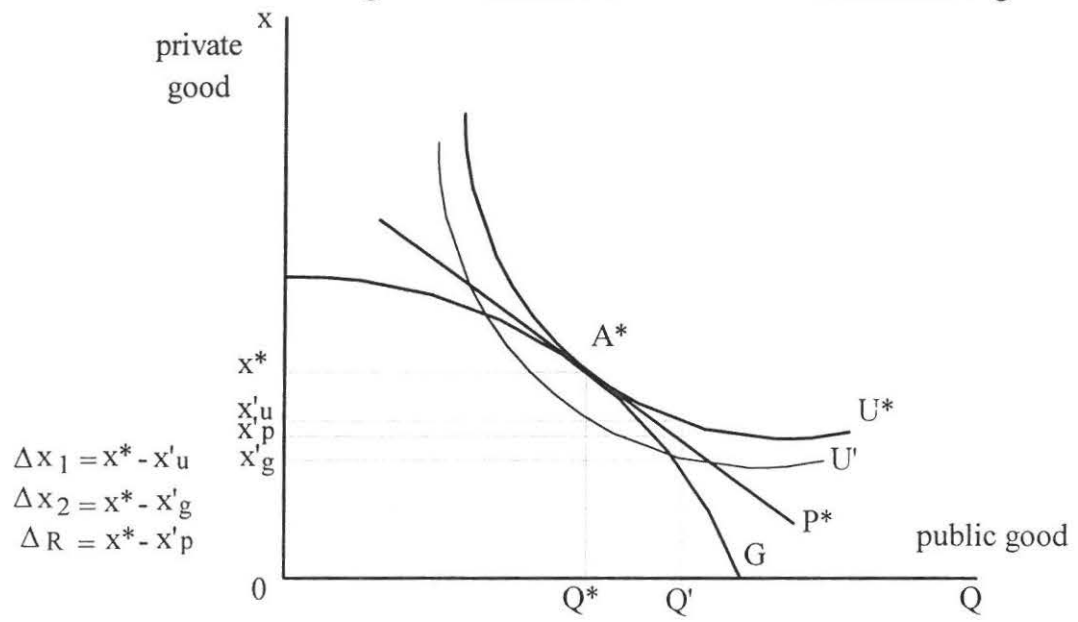


Figure 2 - Valuation of an increase in environmental good

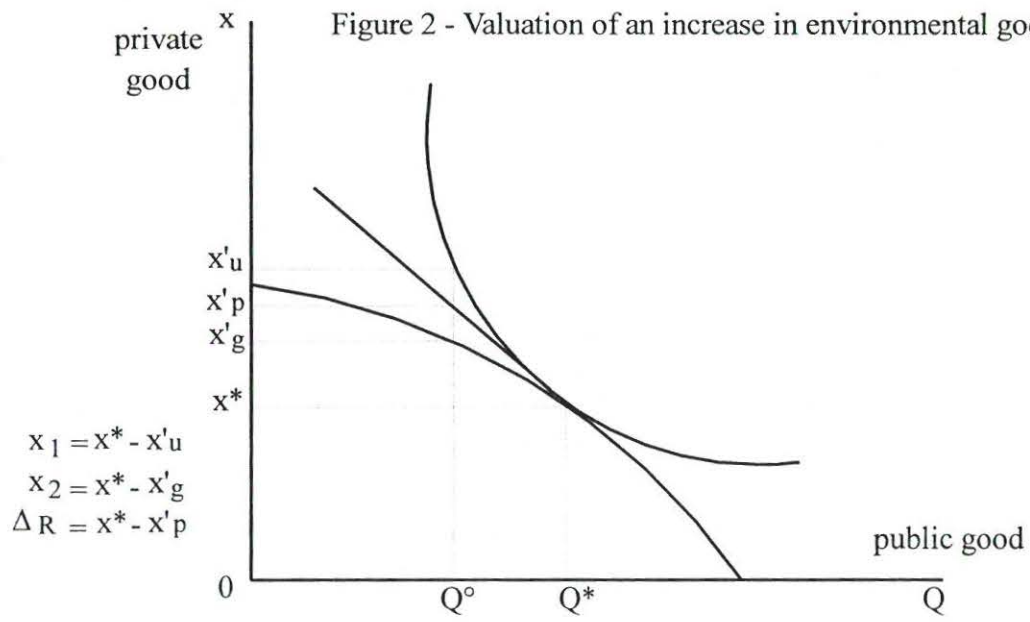


Figure 3 - Valuation of a decrease in environmental bads

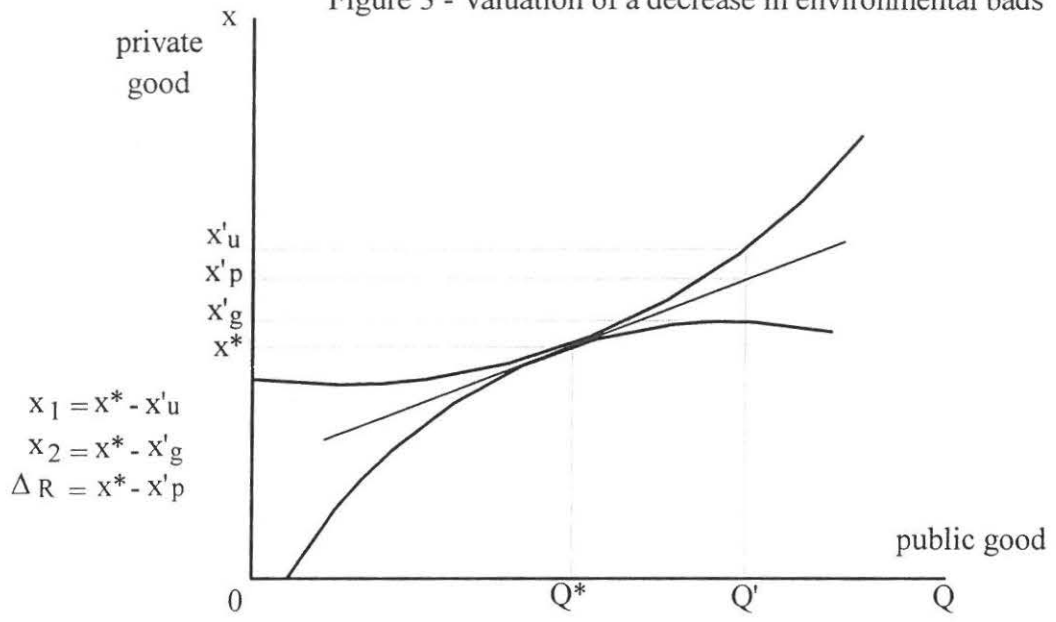


Table 1. WTP for landscape and countryside according to geographical level

| <i>Author</i> | <i>Geographical level</i> | <i>WTP (1992 Ecu)</i> | |
|---------------------------------------|---------------------------|---------------------------------|--|
| Drake 1986 | national | resident | 92/person/year (16-74 years old) |
| Bateman et al. (1992) | regional | visitor non-user | 95.3 - 280.6/household/year 5.1/household/year |
| Willis, Garrod and Saunders (1993) | regional | resident visitor non-user | 22.0 - 34.5/household/year 14.8 - 24.4/household/year 2.5 - 3.1/household/year |
| Riera (1994) | regional | user and existence value | 6.5/visit |
| Rebolledo and Perez y Perez (1994) | regional | user and existence value | 9.6/visit |
| Le Goffe and Gerber (1994) | local | resident | 60.6/household/year |
| Dillman and Bergstrom (1991) | local | resident | 6.7 - 10.5/household/year |

Table 2. Comparison of nature conservations estimates

| <i>Author</i> | <i>Area</i> | <i>Good being valued</i> | <i>Aggregate WTP (thousand Ecu 1992)</i> | <i>Average WTP (Ecu 1992/ha)</i> |
|---------------------------------------|--|--|---|--------------------------------------|
| Bateman et al. (1992) | The Norkolk Broads (UK) 29 980 ha | Preservation of the landscape from increased risk of flooding | user values 8 620 non-use benefits 137 613 | 288 4 590 |
| Willis, Garrod and Saunders (1993) | The South Downs (UK) 26 738 ha | Maintaining the ESA scheme in an environmentally sensitive area | user values 60 985 non-use benefits 3 951 | 2 281 148 |
| | The Somerset Levels and Moors (UK) 27 170 ha | sensitive area | user values 13 477 non-use benefits 52 462 | 496 1 931 |
| Le Goffe and Gerber (1994) | Pacé and Saint-Gilles (F) 5 580 ha | preservation of today's landscape | residents 803 | 144 |
| Rebolledo and Perez y Perez (1994) | Parque Natural de la dehesa del Moncayo (SP) 1 389 ha | Maintaining a natural park | user and existence values 1 334 | 960 |

Table 3. WTA for changing farming practices - Ecu/ha/year

| | | |
|--|--------------|-----|
| Substituting grassland for maize crops | | 515 |
| Reduction in stocking rate | - .25 GLU/ha | 144 |
| | - .50 GLU/ha | 306 |
| Joining an organic farming network | | 350 |

Source: Bonnieux and Rainelli, 1994.