

## VALUING ALTERNATIVE BUNDLES OF LANDSCAPE ATTRIBUTES:

### Cost-benefit analysis for the selection of optimal landscapes

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**Abstract** – The role of landscape valuation and extended cost-benefit analysis in landscape conservation decision-making is first addressed. It is stressed that substantial research on how to value alternative conservation schemes is required, in order to cope with emerging policy needs. An analytical frame for the valuation of alternative landscape conservation schemes is then discussed. This frame enables the analyst to sequentially disaggregate values for whole landscape changes over attributes. This is an essential operation if the optimal bundle of landscape attributes is to be selected by cost-benefit analysis. The concept of substitution between landscape attributes plays an essential role within the whole analytical frame. The circumstances that lead to anticipate substitution between landscape attributes are explored. A brief review of the alternative empirical strategies for landscape valuation is then carried out, to check whether they permit sequential desegregation of landscape value over attributes. Next, an empirical application to the valuation of landscape attribute changes in the Pennine Dales Environmentally Sensitive Area is presented. The empirical results confirm the idea of the prevalence of substitution in valuation in most practical contexts. To illustrate the potential of the proposed approach, a sequential cost-benefit analysis of attribute changes along consistent paths of aggregation is then carried out – which eventually leads to the selection of optimal bundles of landscape attributes. Some problems and limitations of the approach are also discussed. Among them, the question of non-uniqueness, or path dependency of the optimum is given particular consideration.

**Resumo** – VALORAÇÃO DE MUDANÇAS PAISAGÍSTICAS MULTI-ATRIBUTOS: UMA ANÁLISE CUSTO-BENEFÍCIO PARA A SELECÇÃO DE PAISAGENS ÓPTIMAS. Abordam-se primeiramente os papéis da valoração económica da paisagem e da análise custo-benefício no quadro da concepção e avaliação de programas de conservação da paisagem, colo-

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cando particular ênfase na necessidade de desenvolver métodos para a avaliação de programas de conservação alternativos. Neste contexto, propõe-se um quadro analítico para a valoração de programas alternativos, o qual permite ao analista desagregar sequencialmente os valores das diversas mudanças nos atributos paisagísticos elementares, que compõem uma determinada mudança multi-atributos. Esta é uma operação necessária para a selecção, através da análise custo-benefício, de combinações óptimas de atributos paisagísticos. O conceito de substituição entre os atributos da paisagem tem um papel essencial no quadro analítico proposto, pelo que se exploram as circunstâncias que podem levar a antecipar, em determinados casos, a existência de relações de substituição entre atributos paisagísticos. Procede-se também a uma revisão crítica das diversas estratégias empíricas alternativas para a valoração económica da paisagem, a fim de verificar se elas permitem desagregar sequencialmente o valor de uma determinada mudança multi-atributos nas diversas mudanças de atributos elementares que a compõem. Seguidamente, apresenta-se uma aplicação empírica relativa à valoração de mudanças nos atributos paisagísticos da Área Ambientalmente Sensível (*Environmentally Sensitive Area, ESA*) dos *Pennine Dales*, no Reino Unido. Os resultados empíricos confirmam a tese da dominância da relação de substituição entre atributos na maior parte dos contextos práticos de valoração económica do ambiente. Para ilustrar o potencial da abordagem proposta na avaliação de programas de conservação alternativos para uma mesma área, procede-se a uma análise custo-benefício de mudanças multi-atributos, a qual tem carácter sequencial, na medida em que percorre os diversos caminhos consistentes de agregação no espaço dos multi-atributos. Esta análise conduz-nos à selecção de combinações óptimas de atributos paisagísticos. Alguns problemas e limitações da abordagem são também discutidos, como é o caso da questão da não-singularidade do óptimo, ou seja, da sua dependência face ao caminho seguido para agregar as diversas mudanças nos atributos elementares que integram a mudança multi-atributos.

## I. A CASE FOR LANDSCAPE VALUATION AND EXTENDED COST-BENEFIT ANALYSIS

What can environmental economists say about landscape conservation policies? To anticipate the answer, it will be argued that explicit monetary valuation of landscape benefits of these policies is an important task for environmental economists. It will also be suggested that if environmental economists want their views to be more fully considered in environmental decision-making, then they must drop any claims to the use of cost-benefit analysis at the exclusion of other criteria. Moreover, it will be maintained that substantial research on how to value alternative conservation schemes is required, in order to cope with emerging policy needs. The discussion is divided into five points.

1. Decisions involving landscape changes necessarily assign (at least implicitly) a monetary value to the implied landscape benefits. PRICE (1978) illustrates this point with a simple example of a decision-maker facing two projects achieving the same purpose: one cheaper, the other providing

additional landscape benefits. Hence, the question is not *whether* monetary values should be assigned to landscape, but to clarify *whose* values are being assigned by particular decisions.

2. The analytical tools of environmental economists enable them to play a *specific role* in this task of value elucidation. Indeed, the theoretical foundations of cost-benefit analysis (CBA; see e.g. JOHANSSON, 1993) give this technique the possibility of checking the consistency of particular decisions with the trade-offs ordinary people would have made in the same circumstances. Hence, the application of CBA is particularly suitable to incorporate people's values for landscapes into decisions involving landscape changes.

3. The very nature of costs and benefits of landscape conservation policies leads to a systematic under-valuation of conservation benefits. In fact, there is a fundamental asymmetry between the cost and the benefit sides of these policies. Conservation costs are not only visible (for example, on farming balance sheets), but they are also «socially» conspicuous, because «commercial interests make sure that they are well documented.» (BRADEN & KOLSTAD, 1991: 4). On the other hand, benefits are mainly «intangible», and usually accrue to a vast number of landscape users who are not effectively organised for the defence of their interests as users. However, the fact that benefits are «intangible» does not mean they are unreal in welfare terms. Many studies using valuation techniques suggest that people are prepared to trade-off other commodities for landscape quality to an extent that is sufficient to justify the conservation option (see e.g. WILLIS et al. 1993).

All the previous arguments stress the advantages of explicit benefit estimation, if landscape benefits are to be given a fair place in public decision-making. Specially, *if a cost-benefit analysis is to be carried out* (because of a legal requirement, or within the usual practice of an agency), *all relevant impacts on individuals welfare should be valued and incorporated in the analysis*. Despite the fact this is a basic rule, recommended by all CBA handbooks (cf. e.g. HANLEY and SPASH, 1993), it is not very often followed with respect to landscape changes. BARDE and PEARCE (1991) illustrated this point by using an actual case of the approval of a final stretch of a motorway that threatened an area of great natural beauty in the UK. Though valuations of time savings and changes in risk to life (both favourable to road building) are current practice in CBA's carried out by the UK Department of Transport, landscape changes (in this case unfavourable to the cheapest option) were not valued nor incorporated in the formal CBA. The discussion by BARDE and PEARCE (1991) clearly shows how ambiguous can be the results of an *incomplete* CBA and the decisions based on these results<sup>2</sup>. The idea that the only good CBA is *extended* CBA emphasises the need

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<sup>2</sup> The incomplete nature of CBA is here related to the problem of institutional capture of CBA by the agencies that carry out the analysis (see also BOWERS, 1988).

to allocate a bigger share of the analysts' effort to estimate benefits of landscape changes.

4. Decisions involving landscape changes typically take place in a multi-dimensional environment, in which impacts on wildlife, water quality, and other resources are also present. CBA reduces all these dimensions to a single metric (money), based on the trade-offs people are willing to make among the diverse dimensions.

If it was assumed that only peoples' voluntary trade-offs are to be taken into account, CBA would achieve the purpose of compressing into a single figure (usually the net present value of a project) all the information that counts for the decision. Only technical difficulties (for example in valuing environmental features) could hinder this occurring. This idea can be called the *strong case* for using CBA in environmental decision-making. It will be argued below why this *strong case* is equivocal and hinders the full consideration of landscape valuations by policy-makers. Accepting the *strong case* means accepting that cost-benefit analysis alone (provided that it is extended to incorporate people's values for all relevant impacts) *does automatically dictate* a preferred course of action. As PRICE (1991) claims:

By offering, in a single net present value figure, a judgement on *overall* performance, cost-benefit analysis *supplants* the ostensible function of the political processes – to assign a balance to the competing interests.» (PRICE, 1991; emphasis added).

Therefore, the *strong case* has a political implication: CBA would provide a substitute for the instituted role of politicians. Here lies perhaps the explanation for why so many policy-makers are hostile to the use of CBA in public, decision-making. As an illustration, look at the arguments against economic valuation by a Senior Policy Officer at the Countryside Commission:

- 1) «valuation, through the very process of condensing complex issues into a single index, actually hides potential environmental conflicts» (MINTER, 1994: 4).
- 2) hence «it may be *politically* desirable to preserve separate, relevant dimensions for each issue.» (MINTER, 1994: 4, emphasis added).
- 3) «The Countryside Commission will do this [referring to point 2], because it believes that protecting the landscape for its cultural significance is not the same thing as protecting it to satisfy consumer preferences.» (MINTER, 1994: 4).
- 4) «... whilst monetary values will give a *reasonable* indication of the public's preferences, these may not accord with the *importance* ascribed to an environmental feature or species by science or other disciplines.» (MINTER, 1994: 3, emphasis added).

- 5) therefore [referring to the Commission] we «have not considered expressing examples of our projects or concerns in monetary terms, unless obliged to by government processes to which we are subject ...» (MINTER, 1994: 3).

Indeed, the strong case for CBA can genuinely be charged as a technocratic way of concealing important conflicts at issue (some of them non-reducible to the basic assumptions of CBA) under the mantle of a single-figure «objective» result. However, the only point that is being made in this paper – which can be referred to as the *weak case* for CBA in environmental decision-making – is that *extended CBA* conveys information structured in a way that is meaningful for the decision-making process.

Provided that users know how to interpret this meaning, the technique can be helpful in assisting policy-makers and policy analysts to explore the consequences of alternative policies with respect to the choices ordinary people would have done if faced with the same trade-offs. Of course, people's values are only one dimension in policy making, but CBA does not need to be used alone to dictate the best course of action (cf. BARDE & PEARCE, 1991). Together with other information relevant for dimensions that cannot be contained within the cost-benefit frame, CBA represents an important exploratory tool in clarifying the decisions to be made, and so improves the accountability of the political process (cf. LOWE *et al.*, 1993, and HANLEY & SPASH, 1993). Moreover the dimensions that *can* be contained within the cost-benefit frame are surely crucial for decisions on landscape conservation. Otherwise, how could we interpret some recent statements of the Countryside Commission on landscape conservation policies, like these:

«The unique importance of payment schemes is in paying land managers to provide public benefits, both environmental and recreational, which they cannot provide commercially...

«*Public money* should be used to pay for the benefits that *we* all desire...

«But public funds for countryside schemes are *limited* and must be carefully used to achieve *best value* for money.

«In effect, agencies buying countryside products *on behalf of the public* should be able to act *as discerning purchasers...*» (COUNTRYSIDE COMMISSION, 1993: 9 and 13; emphasis added).

However, if these statements are compared to MINTER'S (1994) critique of economic valuation quoted above, some inconsistencies seem inevitable – at least if we have in mind the *weak case* for landscape valuation and CBA. So: why could someone who acts on behalf of the public ignore the choices ordinary people would have made if confronted with the same trade-offs? (Moreover, notice that Minter acknowledges that monetary values do reasonably depict public's preferences.) However, if we have in mind the *strong case*, for valuation

and CBA, the inconsistency is *partly* removed: the refusal of valuation would be required to maintain the option of protecting landscape for the sake of its 'cultural' or scientific significance (cf. points 3 and 4 of Minter's quotation). The idea that 'cultural' significance cannot be accommodated within the concept of public's preferences can be accepted as a working hypothesis, even if it is charged with elitism. The same holds for the importance ascribed to environmental features by science and other disciplines (provided the term *importance* means more than the particular preferences of scientists as a group). But, if we return again to the *weak case* scenario, why not to maintain the two criteria, i.e.: – the 'benefits that *we all* desire' (evaluated through the trade-offs *we all* are willing to make, specially because *we all* are paying for them) as appraised by CBA, – the 'cultural significance' as appraised by some kind of expert opinion, and to ask policy-makers to state the trade-offs they are prepared to make between the two criteria'?

Environmental economists are maybe responsible for some of these ambiguous attitudes of policy-makers towards CBA, because they have not made clear enough their adhesion to a *weak case* for the role of landscape valuation and CBA in environmental policy-making (even though many of them are by no means firm believers of the strong case; see e.g. BARDE and PEARCE 1991, or HANLEY & SPASH, 1993), or because they depict trade-offs between ordinary people's values and other criteria as trade-offs between 'allocative efficiency' (a term with a much broader meaning for policy makers) and 'whatever else' (see e.g. BROMLEY, 1990). Otherwise, most of policy-makers' suspicions towards landscape valuation would probably vanish.

If this is to be the future trend, a vast field of empirical work can be opened up to landscape change valuation.

5. Provided that the political hurdle raised in the last point can be peacefully solved, the scope for using valuation techniques in decisions involving landscape changes can be considerably enlarged.

For example, one important new trend in countryside conservation payment schemes in the UK can be described as a shift from paying farmers for compliance with specified practice to paying farmers for well specified countryside products (cf. COUNTRYSIDE COMMISSION, 1993). The approach has the advantages of being more cost-effective, maintaining the managerial autonomy of farmers, and making farmers accountable for the achievement of the conservation targets to which they are to be bound by agreement (COUNTRYSIDE COMMISSION, 1993). Within this new frame, the targeting of conservation goals (landscape attributes to maintain, conservation standards for each attribute, and areas to be covered by schemes) is an essential part of the design of a particular scheme. Valuation techniques and extended CBA can be used as *exploratory* tools in clarifying the implied decisions. A whole range of new applications can be envisaged here that clearly matches current policy needs. For environmental economists to

cope with these new policy needs, substantial research and development on new valuation strategies is required. This is clearly one of the most urgent tasks in the field of landscape economics <sup>3</sup>.

In this paper, an analytical frame is first built to accommodate the main questions raised by the valuation of alternative landscape conservation schemes. An empirical application to the valuation of alternative conservation schemes for the same area is carried out. Alternative conservation schemes for the same area can be viewed both as alternative product specifications – using the Countryside Commission’s language – or as alternative bundles of landscape attributes – to use one term that is familiar to the economist.

## II. VALUING BUNDLES OF LANDSCAPE ATTRIBUTES (AN ANALYTICAL FRAME)

1. Landscape conservation schemes aim at conserving or enhancing particular landscape attributes at one or more areas of the countryside. Hence, the benefits of a scheme refer to the difference between the *state of landscape* that would evolve without the scheme and the *state of the landscape* that the scheme is assumed to deliver. Differences of this kind between two states of landscape will be thereafter called *landscape changes*.

2. A *state of landscape* can be described as a matrix  $\mathbf{Z}$ , with the generic element  $z_{ij}$  representing the level of attribute  $i$  (for example, percentage of heather cover) in area (for example, the Simonside Hills, Northumberland, UK). Thus, rows of  $\mathbf{Z}$  represent attributes, and columns represent areas of the countryside. To simplify the notation of expressions to be presented below, matrix  $\mathbf{Z}$  is rearranged into a column vector  $\mathbf{z}$  in which the columns of  $\mathbf{Z}$  are aligned one after another. The notation  $z_{ij}$  is maintained for the generic element of vector  $\mathbf{z}$  (in a not very orthodox use of matrix notation).

3. Within welfare economics, the benefits of a landscape conservation scheme are conceived as trade-offs people are prepared to make between

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<sup>3</sup> To time knowledge of the author, the term landscape economics was first given a systematic content by Colin Price, in 1978, in an historic book called precisely *Landscape Economics*. The book starts with the discussion of the first ‘hurdle’ to landscape economics: the objections to explicit valuation of landscape; a considerable length of the book is concerned with the valuation techniques, and the whole book is a persuasive case for valuation and extended cost-benefit analysis. Price must also be credited for being time first to explore the connections between environmental economics and the literature on landscape evaluation, developed by planners and landscape architects. Based on this connections, Price has developed a method for landscape valuation described later in this paper.

consumption goods (or, more simply, income) and the landscape change delivered by the scheme. The definition and measurement of these benefits is based on two functions: the *indirect utility* and the *expenditure* functions. To derive these functions, let us now examine the landscape user's consumption choices.

So, assume that landscape users' preferences can be described by a continuous utility function  $U=U(\mathbf{x}, \mathbf{z}, \mathbf{c})$ , where:  $\mathbf{x} = (x_p, \dots, x_M)'$  is a vector of marketed consumption goods;  $\mathbf{z}$  is the vector just described, representing the state of landscape; and  $\mathbf{c}=(c_p, \dots, c_k)'$  is a vector describing the user's landscape tastes and other socio-economic variable<sup>4</sup>.  $U(\cdot)$  is also assumed to be strictly increasing and strictly quasi-concave in  $\mathbf{x}$  and  $\mathbf{z}$ <sup>5</sup>.

Once produced, as a side-effect of the economic activities that take place in the countryside,  $\mathbf{z}$  (that is the levels of landscape attributes at several areas) is beyond the influence of the landscape user. This public good nature of  $\mathbf{z}$  implies that  $\mathbf{z}$  enters the choices of landscape users as a parameter. A landscape user faced with income  $y$  and prices  $\mathbf{p}$  for the  $M$  marketed goods is, therefore, assumed to choose the bundle of private goods  $\mathbf{x}$  that maximises  $U(\cdot)$  subject to  $\mathbf{x}' \mathbf{p} \leq y$ . The solution generates a set of market demand functions for the  $M$  consumption goods,  $\mathbf{x} = \mathbf{x}(\mathbf{p}, \mathbf{z}, y, \mathbf{c})$ .

Notice that the demands for marketed goods depend not only on prices, income, tastes and socio-economic variables, but also on the state of landscape  $\mathbf{z}$ . So the model allows for shifts in demand functions determined by exogenous landscape changes. For example, walkers can reduce their purchases of petrol and other private goods related to recreational trips to the Simonside Hills during August and September (the heather flowering period) if the percentage of heather cover in the area is reduced. The indirect valuation methods, like the travel cost models, explore precisely these traces of landscape changes in people's behaviour, to uncover people's values for these changes.

Replacing, the market demand functions back into the utility function yields the indirect utility function,  $V(\mathbf{p}, \mathbf{z}, y, \mathbf{c}) = U(\mathbf{x}(\mathbf{p}, \mathbf{z}, y, \mathbf{c}), \mathbf{z}, \mathbf{c})$ , that is the maximum utility achievable by an individual under the circumstances  $\mathbf{p}$ ,  $\mathbf{z}$ ,  $y$ , and  $\mathbf{c}$ .

The problem dual to constrained utility maximisation is the choice by the individual – facing prices  $\mathbf{p}$  and state of landscape  $\mathbf{z}$  – of the bundle of private goods  $\mathbf{x}$  that minimises the expenditure,  $\mathbf{x}' \mathbf{p}$ , of obtaining at least utility level  $U_0$  (that is  $U(\mathbf{x}, \mathbf{z}, \mathbf{c}) \geq U_0$ ). The solution generates a series of Hicksian compensated demand functions for the  $M$  marketed goods,  $\mathbf{x}^c = \mathbf{x}^c(\mathbf{p}, \mathbf{z}, U_0, \mathbf{c})$ . Substituting these back into the objective function yields the expenditure func-

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<sup>4</sup> Though all individuals are assumed to have 'similarly shaped' utility functions, the vector  $\mathbf{c}$  provides for shifts in preferences according to socio-economically determined differences across individuals.

<sup>5</sup> Other assumptions characterising 'well behaved' utility functions, and allowing the required mathematical operations to be carried out, are also assumed to hold as usual (for a list of these assumptions see e.g. JOHANSSON, 1987).



tion,  $e(\mathbf{p}, \mathbf{z}, U_0, \mathbf{c}) = \mathbf{x}^c(\mathbf{p}, \mathbf{z}, U_0, \mathbf{c})' \mathbf{p}$ , that is the minimum expenditure required to achieve  $U_0$ , under the circumstances  $\mathbf{p}$ ,  $\mathbf{z}$ , and  $\mathbf{c}$ . The expenditure function is strictly decreasing and convex in  $\mathbf{z}$  (and increasing and concave in  $\mathbf{p}$ )<sup>6</sup>.

Thereafter, price vector  $\mathbf{p}$  is ignored in the arguments of both the indirect utility function and the expenditure function. Prices are treated as constant parameters because the focus of the present discussion is in on welfare effects of landscape changes, not price changes<sup>7</sup>.

4. The definition of the benefits of a conservation scheme can now be carried out. Though these benefits are here defined based on a single example (to simplify the discussion) the definition is much more general than the example. It could be adapted with minor changes to any of the annual payment schemes currently operating in the UK (and other EU countries).

So, suppose that current trends in farming practice in various areas are changing some landscape attributes in a way that reduces the utility individuals derive from landscape. The landscape change can be represented as a change from state of landscape  $\mathbf{z}^0$  (current landscape) to state of landscape  $\mathbf{z}^1$  (impoverished landscape occurring in the near future without conservation). Consider now that some alternative conservation schemes aiming at avoiding the undesirable landscape change are proposed. The state of landscape resulting from each one of these conservation schemes can be specified as a vector  $\mathbf{z}$ . If a scheme is only about conservation, then:  $\mathbf{z}^1 \leq \mathbf{z} \leq \mathbf{z}^0$ ; if it also introduces some landscape enhancement, then some elements of  $\mathbf{z}$  exceed the corresponding elements of  $\mathbf{z}^0$ . How can the benefits of each of such schemes be defined? Within welfare economics, the question can be translated into: 'How much consumption goods (or simply, income) would be given up by landscape users to ensure that the state of landscape is  $\mathbf{z}$  and not  $\mathbf{z}^1$ ?'<sup>8</sup>. This maximum willingness-to-pay (WTP) for the scheme can be defined by using the indirect utility function:

$$V(\mathbf{z}, y - WTP, \mathbf{c}) = V(\mathbf{z}^1, y, \mathbf{c}). \quad (1)$$

Hence *WTP* is the amount that subtracted from the individual's income,

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<sup>6</sup> This follows from the assumptions on the utility function (cf. HOEHN 1991 and KOLSTAD and BRADEN 1991).

<sup>7</sup> Implicitly, it is assumed that the individuals are adjusting their demands for private goods so that utility maximisation (or expenditure minimization) is maintained as landscape changes. It is also assumed that these demand adjustments are small enough to allow the analysis to be carried out in a partial equilibrium frame (constant prices).

<sup>8</sup> Current conservation schemes pay farmers to obtain certain states of landscape. This implies farmers are entitled the properly rights over time state of landscape. Within this frame the proper benefit measure is landscape users' maximum willingness-to-pay to avoid the detrimental landscape change. Were landscape users entitled these property rights, and the proper benefit measure would be users' minimum compensation required to tolerate landscape degradation.

with the scheme on (state of landscape  $\mathbf{z}$ ), causes the individual to be as well-off as without the scheme (state of landscape  $\mathbf{z}^1$ ). This is equivalent to the maximum of marketed consumption goods  $\mathbf{x}$  the individual would trade-off by the scheme. If more was required from him he would prefer having the impoverished state of landscape  $\mathbf{z}^1$  rather than paying.

The definition of  $WTP$  can be made explicit by using the expenditure function,

$$WTP = e(\mathbf{z}^1, U^1, \mathbf{c}) - e(\mathbf{z}, U^1, \mathbf{c}); \quad (2)$$

in which  $U^1$  is the maximum achievable utility without conservation.

Because the expenditure function is strictly decreasing in  $\mathbf{z}$ , the minimum expenditure for obtaining  $U^1$  when the state of landscape is  $\mathbf{z}$  [second term of right hand side of (2)] is smaller than the minimum expenditure for obtaining the same level of utility when the state of landscape is  $\mathbf{z}^1$  [first term of RHS of (2)]. Hence, the conservation scheme is equivalent to a monetary saving. Because the landscape user is only entitled to  $\mathbf{z}^1$  (that is to  $U^1$ )<sup>9</sup>, then he would be prepared to give up an amount up to the monetary saving produced by the scheme. If he could pay less for the scheme to occur, then he would be better-off with the scheme. If he was required to pay more he would be better-off without the scheme. Hence  $WTP'$  measures the full monetary value of the scheme for the landscape user.

Note that, because  $U^1 = (z^1, y, \mathbf{c})$ , and because  $e(\mathbf{z}^1, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) = y$ , (2) can be simplified into:

$$WTP(\mathbf{z}, \mathbf{z}^1, y, \mathbf{c}) = y - e(\mathbf{z}, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}). \quad (3)$$

$WTP$  is a *valuation function*: for each scheme delivering a state of landscape  $\mathbf{z}$ , the function yields the maximum willingness-to-pay for this scheme. Within this frame, it is possible to value all of the alternative conservation schemes, provided that schemes' outcomes, in terms of states of landscape  $\mathbf{z}$ , are known.

5. Let us now derive some simple properties and implications of the valuation function. *First*, as  $\mathbf{z}$  increases from  $\mathbf{z}^1$ , the expenditure function declines, and so  $WTP$  rises; as  $\mathbf{z}$  declines and becomes closer to  $\mathbf{z}^1$ , expenditure rises, and so  $WTP$  declines (eventually, when  $\mathbf{z}=\mathbf{z}^1$ , expenditure is  $y$  and  $WTP$  is zero). *Second*, the partial derivatives with respect to the landscape attributes at various locations, that is  $\partial WTP(\mathbf{z}, \mathbf{z}^1, y, \mathbf{c}) / \partial \mathbf{z} = \partial e(\mathbf{z}, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) / \partial \mathbf{z}$ , represent the *inverse compensated demand functions* (or marginal  $WTP$ ) for landscape attributes at various locations. These inverse demands are strictly positive because the

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<sup>9</sup> Cf. the previous footnote.

expenditure function is strictly decreasing. *Third*, the second order derivatives of the valuation function with respect to attributes, that is  $\partial^2 WTP(\mathbf{z}, \mathbf{z}^1, y, \mathbf{c}) / \partial \mathbf{z} \partial \mathbf{z}' = -\partial^2 e(\mathbf{z}, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) / \partial \mathbf{z} \partial \mathbf{z}'$ , define a *substitution matrix* (the term was drawn from HOEHN and LOOMIS 1993). *Fourth*, the diagonal elements of the substitution matrix, that is  $-\partial^2 e(\mathbf{z}, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) / \partial \mathbf{z}_{ij}^2$  represent the slope of the inverse demands for attributes. So inverse demands are non-increasing because of the convexity of the expenditure function – which implies that marginal willingness-to-pay for an attribute does not increase with the level of that attribute. *Fifth*, the non diagonal elements of the substitution matrix, i.e. the cross derivatives  $-\partial^2 e(\mathbf{z}, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) / \partial \mathbf{z}_{ij} \partial \mathbf{z}_{kp}$ , for  $i \neq k$  (different attributes) or  $j \neq p$  (different areas of the countryside) define the *substitution effects* between landscape attributes. When these elements are negative, the landscape attributes are said *substitutes in valuation*; when they are positive, the attributes are *complements in valuation*; and when they are nil, the attributes are *independent in valuation* (cf. HOEHN and LOOMIS, 1993). There are three different types of substitution effects between landscape attributes: (i) substitution effects between different attributes in the same area ( $i \neq k$  and  $j = p$ ); (ii) between the same attribute in different areas ( $i = k$  and  $j \neq p$ ); and (iii) between different attributes in different areas ( $i \neq k$  and  $j \neq p$ ).

6. The analytical frame to deal with the valuation of alternative conservation schemes is now completed. However, suppose the conservation agency wants not only to chose between alternative schemes, but also to pick the scheme that is *optimal* with respect to the trade-offs people are prepared to make – i.e.: the optimal state of landscape, optimal attribute bundle or optimal product specification. This can be done through a cost-benefit analysis of sequential changes in each one of the individual attributes. Hence, this approach implies valuing separate attribute changes<sup>10</sup>.

Carrying out the tasks just described requires the analytical frame for landscape change valuation to be extended, so that we can apportion the value of the whole landscape change to its component attribute changes. In so doing, this paper relies mainly on the works of HOEHN (1991), and HOEHN and LOOMIS (1993). These authors addressed the problem of valuing multi-programme environmental policies. They have clearly shown the crucial role of substitution effects between environmental programmes when aggregating benefits over programmes. Because substitution in valuation between programmes prevails (as it seems to be the case), they argue that aggregating programmes' benefits, with each programme valued with other programmes at policy-off levels, will lead to an over-valuation of the multi-programme policy's benefits. Hence, with

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<sup>10</sup> The proposed approach implies, of course, that attributes are defined in a way that they are separable in production. Otherwise, a separate change in a single attribute would not have any practical implications.

their two applications, HOEHN and LOOMIS have empirically established the initial theoretical result of HOEHN and RANDALL (1989): ‘too many proposals pass the benefit-cost test’. In this paper, HOEHN and LOOMIS’s approach to multi-programme environmental programmes is adapted to the much smaller scale problem of sequentially valuing multi-attribute landscape conservation schemes.

How is it possible to value separate attribute changes and to relate these values to the value of the whole landscape change defined in (3)?

For simplicity, consider that there are only two attribute changes involved. Hence the landscape change from  $\mathbf{z}^1$  (without scheme) to  $\mathbf{z}$  (with scheme) could be decomposed into say a change in  $\mathbf{z}_{ij}$  ( $\mathbf{z}_{ij}^1$  to  $\mathbf{z}_{ij}$ ) and a change in  $\mathbf{z}_{kp}$  ( $\mathbf{z}_{kp}^1$  to  $\mathbf{z}_{kp}$ ). These attribute components of the change can refer to: different attributes in the same area ( $i \neq k$  and  $j = p$ ); the same attribute in different areas ( $i = k$  and  $j \neq p$ ); or different attributes in different areas ( $i \neq k$  and  $j \neq p$ ). The empirical application to be presented in this paper refers to the first case. So, think, for example, of changes in drystone walls and meadows’ flower-diversity in one single dale (valley).

(a) One answer to the question above is to value each attribute change separately, that is: holding constant, at their policy-off level, the other attribute. This is what can be expected from survey respondents faced with separate attribute changes. The *separate valuation* would yield:

$$WTP(\mathbf{z}_{ij}, \mathbf{z}_{ij}^1, y, \mathbf{c}) = y - e(\mathbf{z}_{ij}, \mathbf{z}_{kp}^1, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}), \text{ and} \quad (4)$$

$$WTP(\mathbf{z}_{kp}, \mathbf{z}_{kp}^1, y, \mathbf{c}) = y - e(\mathbf{z}_{ij}^1, \mathbf{z}_{kp}, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}), \quad (5)$$

which added up produce the independent valuation and summation result (*IVS*, from HOEHN 1991):

$$IVS(\mathbf{z}, \mathbf{z}^1, y, \mathbf{c}) = 2y - e(\mathbf{z}_{ij}, \mathbf{z}_{kp}^1, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) - e(\mathbf{z}_{ij}^1, \mathbf{z}_{kp}, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) \quad (6)$$

(b) Another possibility is to disaggregate the value for the whole landscape change from  $\mathbf{z}^1$  to  $\mathbf{z}$  – see (3) – along a sequential path, *say* from  $(\mathbf{z}_{ij}^1, \mathbf{z}_{kp}^1)$  to  $(\mathbf{z}_{ij}, \mathbf{z}_{kp}^1)$ , and then from  $(\mathbf{z}_{ij}, \mathbf{z}_{kp}^1)$  to  $(\mathbf{z}_{ij}, \mathbf{z}_{kp})$ . This yields (cf. HOEHN, 1991):

$$WTP(\mathbf{z}, \mathbf{z}^1, y, \mathbf{c}) = y - e(\mathbf{z}_{ij}, \mathbf{z}_{kp}^1, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) \quad (7)$$

$$+ e(\mathbf{z}_{ij}, \mathbf{z}_{kp}^1, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) - e(\mathbf{z}_{ij}, \mathbf{z}_{kp}, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c})$$

where the first line corresponds to the change in  $\mathbf{z}_{ij}$  with  $\mathbf{z}_{kp}$  constant at its policy-off level, and the second line represents the change in  $\mathbf{z}_{kp}$  with  $\mathbf{z}_{ij}$  constant at its policy-on level. Note that the path of disaggregation is not unique

(for example, there are two possible paths when two attributes are changing). Thus the value of the same attribute change *is not unique*: it *depends on the integration path* (differently to the value of the whole landscape change, which is unique).

This sequential disaggregation procedure is, by definition, consistent with the value of the whole landscape change given by (3) (cf. HOEHN, 1991). Hence the result of the *IVS* approach (6) can be appraised by comparison with (7). Thus, it is easy to show that the *IVS* approach: over-estimates the value of the whole landscape change if the landscape attributes are substitutes in valuation; under-estimates this value if they are complements in consumption; and correctly estimates this value if the landscape attributes are independent in valuation (for a proof see HOEHN, 1991).

(c) A third option is to disaggregate the value for the whole landscape change along a simultaneous path from  $(\mathbf{z}_{ij}^1, \mathbf{z}_{kp}^1)$  to  $(\mathbf{z}_{ij}, \mathbf{z}_{kp})$  by integrating the inverse demands for the attributes along this path and summing up the results. This yields (cf. HOEHN, 1991):

$$WTP(\mathbf{z}, \mathbf{z}^1, y, \mathbf{c}) = \int_{\mathbf{z}^1}^{\mathbf{z}} [\partial e(\mathbf{z}_{ij}, \mathbf{z}_{kp}, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) / \partial \mathbf{z}_{ij}] d\mathbf{z}_{ij} + \int_{\mathbf{z}^1}^{\mathbf{z}} [\partial e(\mathbf{z}_{ij}, \mathbf{z}_{kp}, V(\mathbf{z}^1, y, \mathbf{c}), \mathbf{c}) / \partial \mathbf{z}_{kp}] d\mathbf{z}_{kp} \quad (8)$$

where the first integral represents the value of the change on  $\mathbf{z}_{ij}$  with  $\mathbf{z}_{kp}$  changing simultaneously, and the second the value of the change on  $\mathbf{z}_{kp}$  with  $\mathbf{z}_{ij}$  changing simultaneously. Note that now the values of attribute changes are *unique* or *path-independent* as the value of the whole landscape change. Values of attribute changes along simultaneous paths can be obtained in empirical applications with methods that ask respondents to apportion the value for the whole landscape change by component attributes (for example, the token method described below). However the attributes' values obtained by integration along simultaneous paths are not usable in the approach proposed in this paper. In fact, they do not enable the analyst to pick the optimal state of landscape (or optimal scheme) by carrying out a cost-benefit analysis of sequential changes in each one of the attributes. Only attributes' values obtained by sequential disaggregation (that is following approach b) are, in general, fitted to this task. Attributes' values obtained by separate valuation (approach a) or simultaneous path disaggregation (approach c) are fitted to this task only when attributes are independent in valuation.

7. The previous discussion raises an important question: under which

conditions can we anticipate that landscape attributes will behave as substitutes, complements or independent in valuation? The question is relevant in substantive terms, because it refers to the characteristics of people's preferences for landscapes. It is also important in methodological terms, because it sets the limits to be imposed on the applicability of the 'independent valuation and summation' (IVS) and the simultaneous path disaggregation approaches. Here it is again HOEHN's (1991) work that provides the most interesting insights.

HOEHN suggests that some environmental services are additively separable in utility and therefore independent *in consumption*, that is  $\partial^2 U(\cdot) / \partial z_{ij} \partial z_{kp} = 0$ , for  $i \neq k$  or  $j \neq p$ . One clear example is the one of spatially separated recreational activities. In this case, the household production technology (see SMITH, 1991) suggests that environmental services used in different activities are additively separable in utility. Alternatively, environmental services can be substitutes *in consumption*, if  $\partial^2 U(\cdot) / \partial z_{ij} \partial z_{kp} < 0$ , or complements *in consumption*, if  $\partial^2 U(\cdot) / \partial z_{ij} \partial z_{kp} > 0$ . (Notice that these definitions differ from the definitions of substitutes, complements and independent *in valuation*, which are based on the expenditure function).

HOEHN argues that, under very general assumptions about the household production technology: '(a) environmental services that are additively separable *in utility* are substitutes *in valuation* and (b) environmental services that are not additively separable *in utility* may be substitutes, complements, or independent *in valuation*.' (HOEHN, 1991: 293; emphasis added). This dominance of the substitution relationship *in valuation* is said to be related to the constrained nature of the consumer optimisation problem. Besides HOEHN has recovered substitution in valuation between improvements of air quality in Chicago and the Grand Canyon, based on a valuation function estimated from contingent valuation data. Because of spatial separability, these two environmental services were supposed to be independent in consumption. Hence, HOEHN's theorem was empirically confirmed. However the possibility remains that, in some contexts complementarity in consumption could be strong enough to outweigh the prevalence of substitution in valuation. Hence, HOEHN and LOOMIS (1993) have studied interactions in valuation between several environmental programmes for the same geographical area (the San Joaquin Valley in California) in search for complementarity in valuation. Complementarity has been here plausible because of both positive cross-program productivity effects and jointness in the household production technology (HOEHN and LOOMIS, 1993). However, only substitution in valuation between programmes was found.

Then, what about landscape valuation? Let us consider three cases.

(i) Relationships between the same landscape attribute in separate areas of the countryside (or what is the anticipated sign for  $\partial^2 U(\cdot) / \partial z_{ij} \partial z_{kp}$  with  $i = k$  and  $j \neq p$ ?). This seems to be the clearest case. Probably the relationship here is of substitution in consumption (negative sign). For example, the presence of the same attribute at high levels in several areas within the user's choice set (say

green corn fields in areas of eastern England) reduces the value of the attribute in any one of the particular areas. On the other hand, some green corn fields can be highly valued where they are scarce within the user's choice set (for example, in deserts or dry countries). Here the abundance *vs.* scarcity is the probable leading factor. Abundance of substitutes in consumption leads to substitution in valuation.

(ii) Relationships between different landscape attributes in separate areas of the countryside (or what is the anticipated sign for  $\partial^2 U(\cdot) / \partial z_{ij} \partial z_{kp}$  with  $i \neq k$  and  $j \neq p$ ?). HOEHN's (1991) discussion suggests independence in consumption to be expected in this context. However, with landscape attributes the case is not so clear. For example, it is plausible that areas of the Pennine Dales, with their enclosed structure of walls, meadows and small woodlands, are more highly valued because of their occurrence amid open moorland areas (see LAND USE CONSULTANTS, 1991). In this case, two sets of different attributes in two separate areas are possibly complements in consumption. Hence, in general, it can be expected that different attributes in separate areas behave as independent or complements in consumption. Is complementarity in consumption strong enough to determine complementarity in valuation? The answer is not clear and requires further empirical research.

Points (i) and (ii) are in accordance with the idea of *diversity* as a constituent of landscape values: diversity is 'the constituent whose value depends on the difference or contrast in type of a specific landscape from types recently experienced.' (PRICE, 1978: 161).

(iii) Relationships between different landscape attributes in the same area (or what is the anticipated sign for  $\partial^2 U(\cdot) / \partial z_{ij} \partial z_{kp}$   $i \neq k$  and  $j = p$ ?). This is the more difficult case.

Planners and landscape architects usually assign to fine landscapes a value that is much higher than the sum of the contributions of the various landscape attributes. APPLETON (1994) and PRICE (1991) stress the role of the *composition* of attributes in the whole scene in explaining this phenomenon. DEARDEN and ROSENBLOOD (1980) accept that one component can be differently valued in the presence of others. APPLETON (1994) makes exactly the same point when he stresses that some attributes that act as positive preference predictors in some types of landscapes are negative predictors for other types of landscape. This is supposed to occur with trees in open marsh landscapes that are unique because of the uninterrupted field of vision they afford. The same phenomenon is illustrated when preference prediction models estimated for some areas, containing only a limited number of landscape types, are not valid outside these areas (see e.g. LANDSCAPE RESEARCH GROUP, 1988).

The economic rationale for these examples lies on the fact that different attributes of the same landscape are jointly consumed whenever the landscape is

perceived. Hence, they can behave either as complements or substitutes in consumption. In APPLETON's example, trees and marshland could be substitutes in consumption; trees do not 'belong' to the open large-scale landscape where marshland occurs. However the opposite could be expected for the set of attributes that are supposed to 'belong' to some cherished landscape such as the Pennine Dales. Here, drystone walls and flowerrich meadows could be expected to behave as complements in consumption. This means most of us would prefer (*ceteris paribus*) the same amount of meadows conservation to occur in an enclosed setting of drystone walls than among extensive unenclosed corn fields. Indeed, this latter case possibly provides one of the instances in which complementarity *in consumption* could be expected to be strong enough to be transposed into complementarity *in valuation*. Hence this case should provide the 'difficult case' for Hohen's hypothesis of the prevalence of substitution effects in practical valuation contexts. The test of scientific hypothesis under difficult circumstances is recommended by BLAUG (1992) as good scientific practice. Hence, the empirical application to be presented later in this paper also aimed at testing for HOEHN's hypothesis using the case-study of the Pennine Dales Environmentally Sensitive Area.

### III. EMPIRICAL VALUATION STRATEGIES

The multi-attribute nature of states of landscape required the development, in the previous section, of a theoretical frame to landscape change valuation that provides for disaggregation of the value of the whole change over individual attribute changes. As it is argued in that section, such a frame is a precondition to select an optimal bundle of landscape attributes through sequential cost-benefit analysis. In the present section, alternative empirical strategies to value landscape changes are reviewed, focusing on their potential to provide for sequential disaggregation of benefits over attributes. Hence, valuation techniques are not appraised with respect to their general properties (for this, see: MITCHELL and CARSON, 1989; BRADEN and KOLSTAD, 1991 or JOHANSEN, 1987) but *only with respect to the object of this study*.

1. The first strategy consists of *valuing one discrete change in the state of landscape* that is completely specified regarding all the component attribute changes, and that corresponds exactly to the policy to be evaluated (see WILLIS *et al.*, 1993, using the contingent valuation method, CVM). Within this strategy, results are only usable for the evaluation of this particular policy. The disaggregation of benefits over attributes is not possible. One variant within this strategy is the valuation of as many landscape changes as the number of alternative management options under consideration (see WILLIS and GARROD, 1991, using the CVM). If this variant is extended to the modelling of effects of attributes on values of alternative management options, it will be equivalent (regarding its



potential for disaggregation) to the fifth strategy to be presented below. Another variant to the strategy holds constant the attribute mix, but models the effect on valuation of extending the area of the conservation scheme (see BERGSTROM *et al.*, 1985). This enables us to undertake a sequential cost-benefit optimisation as regards area covered by the scheme but not as regards attribute mix.

2. *Hedonic modelling of houses' prices can be used to uncover the implicit prices for diverse landscape attributes* such as woodland occurrence and woodland types near residential areas (e.g. GARROD and WILLIS, 1991). Hedonics cannot address the issue of aggregating and disaggregating the value of the whole landscape change over attributes simply because the value of the whole landscape change is not known (cf. WILLIS and GARROD, 1991; and GARROD, 1994). However, it could be used in sequential cost-benefit optimisation of the attribute bundle, provided that all relevant attributes and attribute interactions were included in the hedonic price function. This is often hindered because of multi-collinearity among attributes in data sets, and because separability among attributes is generally assumed in hedonic models (see PRICE, 1991).

3. The third strategy is the *direct valuation of individual landscape attributes*. One possible example is HANLEY and RUFFELL's (1993) use of CVM to value pairs of photographs depicting single attribute changes. If applied to the problem of the choice of all optimal landscape bundle, this valuation strategy is prone to the problems of the independent-valuation-and-summation (IVS) approach presented in the last section. Having the values of individual attribute changes is not sufficient to predict how do they combine to produce the composite value of the whole landscape change, because complementarity and substitution in valuation are expected to occur.

4. The fourth strategy starts by *valuing the whole change and then proceeds by subdividing this total value over attributes*. This subdivision is based on survey information about the way participants apportion an amount of points, or *tokens*, corresponding to total value over the diverse attributes (e.g. DRAKE, 1992, using the CVM to value the whole change; BENSON, 1992, and WILLIS and BENSON, 1989, using the travel cost method to value the whole change). This strategy disaggregates the whole value along an integration path in which all attributes are changing simultaneously. Hence, as it is argued in the last section, it does not secure the sequential disaggregation that is needed to select optimum bundles of landscape attributes.

5. The fifth strategy is the *modelling of landscape values using landscape attributes as the predictors* (see e.g. HANLEY and RUFFELL, 1993, using the CVM). The inclusion of interaction terms between predictors enables the analyst to take into account complementarity and substitution in valuation.

This paper proposes to develop this strategy as follows: first, the regression

equation, or *valuation function*, is used to predict values for all relevant bundles of attributes (within the data range); sequential disaggregation is carried out by simply taking differences between the point predictions generated by the valuation function, along a consistent path of change. This is precisely the strategy used in the empirical application presented below in this paper.

6. Another strategy is the *modelling of landscape values using a technical index of landscape quality* (resulting from expert evaluation of the landscapes) as the predictor. This strategy was proposed by PRICE (1978), has received further amendments in PRICE (1991), and was applied together with the travel cost model by BERGIN (1993). The approach is suited to value the impacts of changes that differently affect multiple views (for example possible views impacted by a new motorway along all its extension). However, this approach compresses all the information relative to the attribute mix into a single figure ('landscape quality') through expert evaluation. Hence, it does not allow for the sequential disaggregation of the whole landscape change benefit over landscape attributes. Moreover, landscape changes associated with agricultural changes seem to occur *more or less* homogeneously over large areas of the countryside. This enables an adequate description of the change to be made to respondents in a CVM survey (the same does not hold for the multiple-impact x multiple-view setting addressed by PRICE).

\*

Two other important elements of the valuation strategy are the type of used valuation techniques, and whether data are multi-site data, valuing access to actual areas, or single-area data, valuing hypothetical landscape changes in a particular area. The two issues are related as it is shown below. The discussion is divided in four points.

(i) *Types of valuation techniques and attribute multicollinearity.*

Concerning valuation techniques, hedonics, multi-site travel cost, and generally all '*revealed preference*' techniques explore differences across sites. They are, therefore, frequently prone to multicollinearity among landscape attributes (as already shown for hedonics). Some multicollinearity between attributes across sites is unavoidable. For example: stone walls and coniferous forests tend to prevail in uplands, whereas hedgerows and broadleaved woodlands tend to be their counterparts in lowlands. Multicollinearity causes a statistical difficulty in identifying the attributes with an impact on landscape value. On the other hand, with the CVM, and '*stated preference*' techniques in general, a proper scenario design can circumvent the problem by creating the hypothetical alternative landscapes so that multicollinearity among attributes is avoided (ADAMOWICZ *et al.*, 1994). HANLEY and RUFFELL (1993), though using the CVM,

could not take advantage of this possibility because the analysis was based on multisite data about access to actual landscapes, rather than on hypothetical landscape changes.

(ii) *landscape types and value-attribute relationships.*

Multi-site data present another important flaw: the same landscape attribute can be related to landscape value in ways that vary across different landscape types. This was already noticed with APPLETON's (1994) example of trees in open marshland. Thus, focusing on one particular area and considering only hypothetical landscape changes small enough to avoid shifts of landscape type has the advantage of ensuring stability in the relationships of attributes to value. Another more general solution to the problem is differently defining the attributes for areas with different landscape types. (This specification is possible to accommodate within the frame of the state of landscape matrix  $\mathbf{Z}$  introduced in a previous section). The concept of substitution in valuation between, for example, trees and marshland could also be used to improve the model's explanatory power.

(iii) *multi-site data and landscape location.*

Focusing on one particular area also implicitly defines landscape's location and the availability of substitute landscapes for users. On the other hand, when landscapes whose locations are not well defined are to be valued (e.g. using sets of photographs), large ambiguities about landscapes' locations and substitutes available are created. These ambiguities allow contradictory assumptions to be made by respondents about these important factors in valuation. The issue can also be solved in a multi-site setting by clearly defining to respondents the location of the areas under valuation.

(iv) *Type of valuation techniques and ex ante valuation*

One further advantage of the CVM and *stated preference* techniques over *revealed preference* techniques is that the former are the only techniques available for *ex ante* valuation of future landscape changes, an important requirement for *ex ante* exploratory evaluation of proposed conservation schemes.

#### IV. AN EMPIRICAL SPECIFICATION

The valuation function presented in (3) is the centre of the analytical frame to landscape change valuation. This function can be estimated using either '*revealed preference*' or '*stated preference*' data. In the first case, the procedure is always indirect. It can be based on demand functions estimated

from behavioural data (using, for example, the travel cost model). Some assumptions about the individuals' preferences (for example, weak complementarity between the private goods whose demands have been estimated and  $\mathbf{z}$ ) are then required to permit the integration of the demand functions (cf. e.g. HAUSMAN 1981, or KOLSTAD and BRADEN, 1991).

The use of the CVM enables a much more straightforward approach, because CVM data can be directly used to estimate the relevant valuation function. In this context HOEHN and LOOMIS (1993) propose the use of a second-order Taylor series expansion for locally approximating the actual valuation function. Their approach is adopted in the empirical application presented in this paper. For the advantages of the Taylor series over alternative functional forms, in this context see HOEHN and LOOMIS (1993).

Hence, the second-order Taylor series approximation to (3) around the point  $(\mathbf{z}^1, y, \mathbf{c})$  yields:

$$\begin{aligned}
WTP(\mathbf{z}, \mathbf{z}^1, y, \mathbf{c}) &= WTP(\mathbf{z}^1, \mathbf{z}^1, \bar{y}, \bar{\mathbf{c}}) + (\mathbf{z} - \mathbf{z}^1)'(\partial WTP/\mathbf{jz}) \\
&+ (1/2) (\mathbf{z} - \mathbf{z}^1)'(\partial^2 WTP/\partial \mathbf{z}\partial \mathbf{z}') (\mathbf{z} - \mathbf{z}^1) + (\mathbf{z} - \mathbf{z}^1)'(\partial^2 WTP/\partial \mathbf{z}\partial y) (y - \bar{y}) \\
&+ (y - \bar{y}) (\partial WTP/\partial y) + (1/2) (\partial^2 WTP/\partial y^2) (y - \bar{y})^2 + (\mathbf{c} - \bar{\mathbf{c}})' (\partial WTP/\partial \mathbf{c}) \\
&+ (1/2) (\mathbf{c} - \bar{\mathbf{c}})'(\partial^2 WTP/\partial \mathbf{c}\partial \mathbf{c}') (\mathbf{c} - \bar{\mathbf{c}}) + (\mathbf{c} - \bar{\mathbf{c}})'(\partial^2 WTP/\partial \mathbf{c}\mathbf{jy}) (y - \bar{y}) \\
&+ (\mathbf{c} - \bar{\mathbf{c}})'(\partial^2 WTP/\partial \mathbf{c}\partial \mathbf{z}') (\mathbf{z} - \mathbf{z}^1) + \zeta
\end{aligned} \tag{9}$$

where:  $\bar{y}$  is the average income of the respondents,  $\bar{\mathbf{c}}$  is the vector comprising the respondents' average socio-economic and taste variables; and  $\zeta$  is the remainder of the polynomial approximation to the function. The valuation function and its first and second order derivatives are constants on the point of approximation. So they can be taken as the parameters of the empirical model, yielding:

$$\begin{aligned}
WTP(\mathbf{z}, \mathbf{z}^1, y, \mathbf{c}) &= \alpha + (\mathbf{z} - \mathbf{z}^1)' \beta + (1/2) (\partial WTP/\partial \mathbf{z}) (\mathbf{z} - \mathbf{z}^1)' \mathbf{A} (\mathbf{z} - \mathbf{z}^1) + (\mathbf{z} - \mathbf{z}^1)' \mu (y - \bar{y}) \\
&+ (y - \bar{y}) \beta_y + (1/2) \mu_y (y - \bar{y})^2 + (\mathbf{c} - \bar{\mathbf{c}})' \mathbf{f} + (1/2) (\mathbf{c} - \bar{\mathbf{c}})' \mathbf{B} (\mathbf{c} - \bar{\mathbf{c}}) \\
&+ (\mathbf{c} - \bar{\mathbf{c}})' \mathbf{g} (y - \bar{y}) + (\mathbf{c} - \bar{\mathbf{c}})' \mathbf{X} (\mathbf{z} - \mathbf{z}^1) + u,
\end{aligned} \tag{10}$$

where:  $\alpha, \beta, \mathbf{A}, \mu, \beta_y, \mu_y, \gamma, \mathbf{B}, \mathbf{g}, \mathbf{X}$  are the parameters to be estimated; and  $u$  is a random term assumed to be logistically distributed with 0 mean and  $k$  dispersion parameter.

The known theoretical properties of the valuation function constrain the empirical model. Because the valuation function is zero at  $\mathbf{z} = \mathbf{z}^1$ , then  $\alpha$  vanishes. Because  $\partial WTP/\partial y = 1 - (\partial e/\partial y) = 1 - 1 = 0$  (when  $z = z^1$ ), coefficients  $\beta_y$  and  $\mu_y$  also vanish (see HOEHN and LOOMIS 1993). This yields the simplified empirical model:

$$\begin{aligned}
WTP(\mathbf{z}, \mathbf{z}^1, y, \mathbf{c}) &= (\mathbf{z} - \mathbf{z}^1)' \boldsymbol{\beta} + (1/2) (\mathbf{z} - \mathbf{z}^1)' \mathbf{A} (\mathbf{z} - \mathbf{z}^1) + (\mathbf{z} - \mathbf{z}^1)' \boldsymbol{\mu} (y - \bar{y}) \\
&+ (\mathbf{c} - \bar{\mathbf{c}})' \mathbf{f} + (1/2) (\mathbf{c} - \bar{\mathbf{c}})' \mathbf{B} (\mathbf{c} - \bar{\mathbf{c}}) + (\mathbf{c} - \bar{\mathbf{c}})' \mathbf{g} (y - \bar{y}) \\
&+ (\mathbf{c} - \mathbf{c})' \mathbf{X} (\mathbf{z} - \mathbf{z}^1) + u = x' \mathbf{r} + u,
\end{aligned} \tag{11}$$

where:  $\mathbf{r}$  is a column vector comprising all parameters to be estimated (i.e.: the set  $\{\boldsymbol{\beta}, \mathbf{A}, \boldsymbol{\mu}, \mathbf{f}, \mathbf{B}, \mathbf{g}, \mathbf{X}\}$ ) and  $x$  represent the vector of all variables characterising the policy outcome and the individual respondent plus all interaction terms. The compact representation  $x' \mathbf{r} + u$  is only possible because the model is linear in all parameters. Among the parameters to be estimated, some are particularly interesting for this study:

- the  $\mathbf{b}$ , which are the first-order effects of attributes;
- $\mathbf{A}$ , which is the substitution matrix, non-diagonal elements of  $\mathbf{A}$  represent, as shown, the substitution effects between attributes;
- the  $\boldsymbol{\mu}$  which are the effects of income on marginal WTP for attributes;
- the  $\mathbf{X}$ , which are the effects of taste variables on marginal WTP for attributes.

Diagonal elements of  $\mathbf{A}$  were not estimated in the empirical application presented in this paper, as this is not possible with the dummy specification used for  $(\mathbf{z} - \mathbf{z}^1)$ . This implies the interpretation of the estimated parameters  $\mathbf{b}$  as describing the combination of first and second-order effects of attributes (HOEHN and LOOMIS, 1993). Given the theoretical properties of the valuation function, and the nature of the variables describing users' tastes (variables  $\mathbf{c}$ ), parameters  $\mathbf{f}$ ,  $\mathbf{B}$ ,  $\mathbf{g}$ , and the non-diagonal elements of  $\mathbf{X}$  have been constrained to be zero in the estimation procedure.

## V. DATA AND THE ESTIMATION TECHNIQUES

The empirical model was estimated with data from a CVM survey of 422 visitors to the Pennine Dales ESA. The main characteristics of the CVM scenario are summarised below (for a through account, see SANTOS, 1997 or SANTOS, 1998).

- (i) Each landscape attribute was presented in the survey scenario only at two levels: with and without one conservation programme.
- (ii) The relevant landscape attributes were encapsulated in three conservation programmes: programme 1 (P1), comprising the conservation of stone walls and field barns, programme 2 (P2), comprising the

conservation of hay meadows, flower diversity and *habitat* conditions for breeding birds in meadows; and programme 3 (P3), comprising the conservation of small broadleaved woods.

- (iii) These programmes grouped the landscape attributes to be valued so that the attributes included in different programmes are separable in production. As have already been noticed in this paper, this is an important criterion for the resulting valuations to have any policy implications.
- (iv) The 7 possible combinations of the 3 programmes (P1, P2, P3, P1-P2, P1-P3, P2-P3, and P1-P2-P3) were called the conservation 'schemes'. These conservation schemes were randomly assigned to and valued by respondents.
- (v) Each respondent answered to a series of different valuation questions, where each question was about a different scheme (as opposed to no-conservation at all) and was presented as a separate choice occasion.
- (vi) The discrete-choice format was used in the CVM survey. So, respondent *i* was asked whether or not he would be willing to pay £ *t* as an increase in his household income tax (the payment vehicle) to ensure that conservation scheme *j* could be carried out in the Dales.

Concerning the interpretation of the answers, each 'yes' or 'no' answer only revealed whether or not the maximum WTP of respondent *i* for conservation scheme *j* was above the offered bid amount. The individual WTP amounts (though non-observed) were modelled assuming that the (observed) binary answers were generated by the following model:

$$\begin{aligned}
 WTP_{ij} &= x_{ij}'\mathbf{r} + u_{ij} \\
 I_{ij} &= 1 \quad \text{if } WTP_{ij} > t_{ij} \\
 I_{ij} &= 0 \quad \text{if } WTP_{ij} \leq t_{ij}
 \end{aligned} \tag{12}$$

where:  $WTP_{ij}$  is the empirical specification of the valuation function (see 11);  $I_{ij}$  is the observed binary response variable (= 1 for a 'yes' answer and = 0 for a 'no' answer); and the  $t_{ij}$  are the offered £ amounts. So, the probability of a 'yes' answer is:

$$\begin{aligned}
 \Pr(I_{ij} = 1) &= \Pr(WTP_{ij} > t_{ij}) = \Pr(x_{ij}'\mathbf{r} + u_{ij} > t_{ij}) = \Pr(u_{ij} > t_{ij} - x_{ij}'\mathbf{r}) = \\
 &= \Pr\{u_{ij}/k > (t_{ij} - x_{ij}'\mathbf{r})/k\} = \Pr(\mathbf{y}_{ij} > (t_{ij} - x_{ij}'\mathbf{r})/k), \tag{13}
 \end{aligned}$$

where  $\mathbf{y}$  is the standard logistic distribution with mean 0 and dispersion parameter 1. This distribution has a known cumulative density function, that is  $F(a) = \Pr(\mathbf{y} < a) = 1 - \{1 + \exp(a)\}^{-1}$ . Hence, the log-likelihood function is:

$$\begin{aligned} \log L = & \sum_i \sum_j \{-I_{ij} \log \{1 + \exp [(t_{ij} - x_{ij} \mathbf{r}) / k]\} \\ & + (1 - I_{ij}) \log \{ \exp[(t_{ij} - x_{ij} \mathbf{r}) / k] / (1 + \exp[(t_{ij} - x_{ij} \mathbf{r}) / k]) \}\}. \end{aligned} \quad (14)$$

This estimation method was proposed by CAMERON (1988) as the ‘censored logistic regression’. The log-likelihood function of the censored model can be directly maximised to estimate the parameters  $\mathbf{r}$ . This requires some programming within a general optimisation computer programme. The other possibility to estimate parameters  $\mathbf{r}$ , and the one followed in this study, relies upon an interesting similarity between log-likelihood functions of censored models and the log-likelihood function of the ordinary logit model, first detected by CAMERON (1988). So, with the following reparameterization:

$$\begin{aligned} \mathbf{h} &= (-1/k, \mathbf{r}'/k)' \text{ (new parameter vector)} \\ \mathbf{w}_{ij} &= (t_{ij}, -x_{ij})' \text{ (augmented vector of predictors)} \end{aligned}$$

the log-likelihood function becomes:

$$\begin{aligned} \log L' = & \sum_i \sum_j \{-I_{ij} \log \{1 + \exp[-\mathbf{w}_{ij} \mathbf{h}]\} \\ & + (1 - I_{ij}) \log \{ \exp[-\mathbf{w}_{ij} \mathbf{h}] / (1 + \exp[-\mathbf{w}_{ij} \mathbf{h}]) \}\}, \end{aligned} \quad (15)$$

which is exactly the log-likelihood function that is maximised by any ordinary logit computer programme.

## VI. THE ESTIMATED VALUATION FUNCTION

The estimated model is presented table I. Let us consider now the question of whether the complementarity *in consumption* between landscape attributes is strong enough to counteract the generalised trend for the constrained nature of users’ choices to produce substitution in valuation. Remember that in the case of the Pennine Dales, all attributes somehow «belong» to the same cherished landscape type, which led to expect strong complementarity *in consumption*. This makes this case-study a difficult test ground for HOEHN’s hypothesis of generalised occurrence of substitution in valuation across all practical valuation contexts. So, let us look at the parameters for the interactions between the programmes; as shown before, these parameters represent the substitution in valuation between attributes. All these 3 parameters are negative, and statistically significant. Therefore, the results clearly suggest that, despite the plausibility of a strong *complementarity effect in consumption*, the *substitution effects*

prevail in landscape change *valuation*, as well as for the other classes of environmental resources studied by HOEHN (1991), and HOEHN and LOOMIS (1993).

TABLE I – Estimated valuation function

<b>Dependent variable:</b> WTP per household for the landscape conservation scheme in the Dales				
		<b>Prediction success (%):</b>		
		Actual		Total
		Would	Wouldn't	
<b>Number of observations:</b>	2293			
<b>Correct predictions:</b>	80.1 %			
<b>- 2 Log-likelihood ratio:</b>	1225.59	Predicted	<b>Would pay</b>	49.0
<b>D.F. :</b>	13		<b>Would not</b>	13.9
<b>Level of significance:</b>	P < 0.0001			62.9
		Total		37.1
				100.0
Variables	Parameter estimates	t-ratios s.l.	Label	
P1	22.75	2.978***	Programme 1 - stone walls and field barns (0-1)	
P2	20.50	2.800***	Programme 2 - flower diversity in meadows (0-1)	
P3	32.75	4.494***	Programme 3 -broadleaved woodland (0-1)	
PI * P2	-14.11	-2.068**	Interaction between programmes 1 and 2 (0-1)	
PI * P3	-18.05	-2.658***	Interaction between programmes 1 and 3 (0-1)	
P2 * P3	-26.81	-3.961***	Interaction between programmes 2 and 3 (0-1)	
INCOME * P1	0.86	3.668***	Interaction between income and programme 1	
INCOME * P2	0.84	3.547***	Interaction between income and programme 2	
INCOME * P3	0.52	2.220**	Interaction between income and programme 3	
PREFE_P1 * P1	9.88	1.718**	Programme 1 present and first in programmes' ranking (0-1)	
PREFE_P2 * P2	22.66	3.631***	Programme 2 present and first in programmes' ranking (0-1)	
PREFE_P3 * P3	20.42	2.953***	Programme 3 present and first in programmes' ranking (0-1)	
<i>k</i>	33.67	20.704***	Dispersion parameter of the logistic distribution	

Note: Significance level (s.l.) are: \* -0.10; \*\*-0.05; and \*\*\* -0.01

Regarding the other parameter estimates, some more conclusions can be drawn.

- (i) The parameters for the interactions INCOME\*PROGRAMMES are positive (and statistically significant) as expected, because landscape attributes are supposed to be normal goods. The WTP-flexibility of income for the 3-attribute bundle that is implied by the model is 0.63. According to HANEMANN (1991), the interpretation of this variable is not straightforward. In fact, the price-flexibility of income is a ratio between: (1) the income elasticity of demand for the public good and (2) the aggregate Alien-Uzawa elasticity of substitution between the



public good and a Hicksian composite commodity comprising all marketed consumption goods. If the latter is not known, then nothing can be deduced about the former. Because of the intuitive uniqueness of the Dales landscape, the latter would be expected to be small, in which case the income elasticity of demand for the Dales landscape would be small.

- (ii) The parameters for the interactions between respondents' preferences for the programmes and the programmes are positive (and statistically significant), as expected. In fact, it is reasonable to anticipate that someone who has strong preferences for a particular attribute would pay more than others for a conservation scheme including this attribute.
- (iii) Note that all parameters are statistically (as well as practically) significant, and that they are revealed to have the signs that could be predicted from economic theory. Moreover the model reveals a reasonable goodness-of-fit, evaluated both through the log-likelihood ratio and the rates of prediction success.

The estimated valuation function was used for *sequentially* disaggregating the value of the whole landscape change over programmes. This was carried out by predicting, with the valuation function, the expected values for all possible bundles of programmes (schemes). Differences between these point estimates for programme bundles were then calculated along all possible sequential paths. The result of these differences is the sequential values of each programme at different steps in different sequential paths, which are presented in table II. They measure the value of each programme when added to particular bundles of other programmes already in the scheme. These results point to some general trends which are clearly consistent with the presented analytical frame and the particular circumstances of the studied case.

- (i) The estimated model implies that, if attribute changes were valued separately and then summed up, then the whole landscape change benefit (with the three attribute changes on) would be overestimated by 65%. This is a measure of the potential for *aggregation bias* with the *independent-valuation-and-summation* (IVS) approach in this case.
- (ii) The value of each attribute change is dependent on the bundle of attributes already available. This illustrates the non-uniqueness, or path-dependency, of sequentially disaggregated values of landscape attribute changes. Because of the accumulation of substitution effects, attribute's values always decline with the number of attributes already present in the attribute bundle.
- (iii) Different landscape attributes are revealed to have different levels of substitutability. So, when added to an increasing number of other

attributes, 'stone walls and field barns' suffer a decline in value that is smaller than the decline suffered by 'meadows' and 'broadleaved woodland' under the same circumstances. 'Meadows' and 'woodland' seem to be more inter-substitutable: these attributes have a smaller value when added to each other than when added to 'walls and barns'. 'Woodland' adds a particularly small value when the other two attributes are already present. The substitution relationships revealed by these phenomena are somehow understandable, given the nature of the several landscape attributes. Thus, stone walls and field barns were frequently described by respondents as the 'character of the Dales', and this is compatible with the higher degree of non-substitutability of these attributes. On the other hand, 'meadows' and 'woodland' have been (correctly) perceived not only as aesthetic attributes, but also as *habitats* for wildlife. This was particularly stressed by some respondents with strong preferences for '*habitat* attributes', and can explain why these two attributes are more functionally inter-substitutable.

TABLE II – Expected value of willingness-to-pay for each landscape attribute change with different bundles of landscape attributes previously available

Attribute	Bundles previously available						
	None	P1	P2	P3	P1 P2	P1 P3	P2 P3
Changes							
walls (P1)	49.58	-	35.47	31.53	-	-	17.42
Meadows (P2)	49.89	35.78	-	23.07	-	8.96	
woods (P3)	50.40	32.35	23.58	-	5.53	-	-

Note: Values in £ per year per household.

## VII. SEQUENTIAL COST-BENEFIT ANALYSIS AND OPTIMAL BUNDLES OF LANDSCAPE ATTRIBUTES

Sequential benefit estimates are used, in this section, to illustrate the application of the proposed approach to the selection of the attribute bundle that is optimal with respect to the trade-offs people are prepared to make between income and the state of landscape.

Sequential benefit estimates were aggregated for the whole population of households visiting the Dales per year.

Social resource costs corresponding to the aggregated benefits were also estimated. Cost estimation was somewhat simpler than benefit estimation, in this case. In fact, landscape attributes were defined to be independent in

production, and hence cost estimation was carried out, separately, for each attribute change. Given their importance for policy evaluation, transaction costs were also estimated and included in costs. Details about benefit aggregation and cost estimation in this case-study are given elsewhere (SANTOS, 1997 and SANTOS, 1998).

Table III provides the benefit-cost ratios for sequential cost-benefit selection of the optimal bundle of landscape attributes. These ratios were estimated from lower-bound estimates for benefits (not only for per-household benefits but also for the visitor population) and upper-bound estimates for costs, so as to lead to only conservative conclusions as regards the merits of conservation. To carry out the optimisation of the bundle of landscape attributes, a sequential approach is required, in which landscape attribute changes are successively added to a previously existing bundle of attributes. In this sequential approach, additional costs and benefits resulting from adding up an attribute to an existing bundle of attributes are compared, to judge whether the additional conservation can be considered a potential Pareto improvement (PPI). In principle, including at each step only attribute changes that are PPIs will lead to the selection of a bundle of landscape attributes that is optimal with respect to people's trade-offs.

It could be argued that each conservation programme included in the CVM scenario covers several attributes, and hence that the approach does not allow enough flexibility for the precise selection of the optimal attribute bundle. However, to the extent that attributes comprised in each programme are jointly produced (which clearly happens, for example, with flower diversity and bird *habitat* in meadows, in P2), the approach encompasses all flexibility that is possible under the technical constraints on management.

Returning to table III, note that all attributes, when they are the first to be included in the attribute bundle, have positive benefit-cost ratios of at least 1.8 (P2). In this situation, P3 has a large benefit-cost ratio, explained by its very low costs. When attributes are added to another attribute already included in the bundle, a benefit-ratio lower than 1 appears when P2 is added to P3. This occurs partly because the two attributes are close substitutes in valuation, and hence the sequential value of P2 is much lower when added to P3 than when added to P1.

Another result of the analysis is that even if the adding of P2 to P3 causes the additional benefit to fall below the additional cost, the adding of P3 to P2 generates a marginal benefit much larger than marginal cost (approximately 40 times larger). So, bringing together the two attribute changes in the same attribute bundle can be justified as a PPI if the path of aggregation is one, but could not be justified as a PPI if the opposite path of aggregation is chosen. This can be explained by the much lower costs of P3 when compared to P2, together with the high substitutability between the two. However, the fact reveals one major weakness of cost-benefit analysis when applied to the sequential aggregation of policies: the fact that the optimal policy mix, in this case the optimal attribute bundle, depends on the path of aggregation. For example, if

the path of aggregation  $P3 \rightarrow P2 \rightarrow P1$  is considered, then the optimum bundle will comprise  $P3$  plus  $P1$ . If the path of aggregation  $P1 \rightarrow P2 \rightarrow P3$  is considered, then all the landscape attributes should be included in the optimal bundle (check in table III).

Since attribute changes should be sequentially evaluated to select the optimum bundle of attributes, and since the results of this evaluation depend on the path of aggregation, the role of the attribute bundles already available – as a result of past policies – is surely crucial when deciding about future conservation programmes. For programmes not already implemented, the selection of a unique optimum will generally depend on a choice about the correct path of aggregation – cost-benefit techniques left to themselves have no criterion to decide between the several optimum bundles that can, in principle, be selected by the analysis.

TABLE III – Benefit-cost ratios for each landscape attribute change with different bundles of landscape attributes previously available

Attribute changes	Bundles previously available						
	None	P1	P2	P3	P1 P2	P1 P3	P2 P3
walls (P1)	2.09	-	3.91	1.22	-	-	1.22
meadows (P2)	1.8	2.62	-	0.9	-	0.45	
woods (P3)	78.32	45.34	39.18	-	6.2	-	-

Note: lower bound estimates for benefits; upper bound estimates for costs.

## VIII. CONCLUSIONS

In the first section of this paper, it has been argued that: monetary values are necessarily assigned to landscape by decisions involving landscape changes; that environmental economists monetize landscape benefits in a way that is meaningful for conservation decisions; and that explicit valuation and extended cost-benefit analysis ensure, at least, that public's preferences for landscape are given a fair place. However, explicit valuation and extended cost-benefit analysis were only defended as *one among other* dimensions in the political process (weak case), and not as the only judgement on the *overall* performance of each course of action (strong case). This weak case gives room for some explicit political trade-offs to be made between different optimal solutions that emerge from different analytical perspectives. This multi-dimensional approach to landscape conservation decision-making was defended on grounds of reasonableness and political acceptability.

The discussion of a proper analytical frame to landscape change valuation provided two main insights: (1) that, whilst complementarity *in consumption* between attributes is sometimes plausible, economic theory predicts that substitution *in valuation* prevails over a wide range of practical contexts; and (2) that only sequential disaggregation can be validly used to select an optimal bundle of landscape attributes, by comparing the costs and benefits of small additional changes in the attribute bundle. With substitution in valuation, the two alternatives to sequential disaggregation (independent valuation, and simultaneous path disaggregation) lead to significant biases if used in sequential cost benefit analysis (over-valuation of the whole landscape change, and under-valuation of the independent attribute changes, respectively). The extent of the biases depends, of course, on the strength of the substitution effects. On the other hand, sequential disaggregation (the correct procedure) has a strong limitation: it is not unique, in the sense that it is path dependent.

Of all empirical valuation strategies reviewed only the modelling of landscape values using landscape attributes as the predictors permitted us to carry out sequential disaggregation of landscape change's values. The proposed procedure is as follows: first, the valuation function is estimated; second, it is used to generate point predictions of value for all relevant attribute bundles; third, individual attribute changes are sequentially valued by taking the differences between these point predictions along all possible sequential paths of disaggregation.

The use of contingent valuation and studies of spatially homogeneous and hypothetical changes in a single-area (or few precisely located areas) were identified as the context that maximises the potential of the proposed modelling approach.

The presented empirical results shown that, in a context where complementarity in consumption was anticipated, only substitution relationships in valuation were found. These relationships exhibit a clear pattern, which is understandable given the nature of the landscape attributes involved. Moreover, the prevalence of substitution would have caused the independent valuation and summation approach to yield, in this case, a strong over-valuation of the whole landscape change. Disaggregation along a simultaneous path would have caused an identical bias, but with the opposite sign, for the independent attribute changes. Though being the only correct procedure for the selection, sequential disaggregation produced non-unique, or path dependent, values of the individual attribute changes. Hence, sequential cost-benefit analysis selected two optimal bundles, namely (walls, woods) and (walls, meadows, woods), depending on the aggregation path. Besides, there is no internal criterion that can assist the analyst in selecting between the two optima. This non-uniqueness of the solution reveals an important limitation of sequential cost-benefit analysis <sup>11</sup> that makes

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<sup>11</sup> Notice that this limitation cannot be equated with technical imperfections, but only with the very substance of the approach.

clear the need for other criteria. For example, the high biodiversity of flower-rich meadows, their scarcity, and the irreversibility of their loss could have led an ecologist to propose that meadows should be conserved anyway, except if conservation resulted in clearly unacceptable costs. This is equivalent to the safe minimum standards criterion (SMS) proposed by some environmental economists. This proposal is not based on people's preferences for the flowers, insects, or whatever that is to be conserved in the meadows, but on expert opinion based on a range of information about ecological significance and ecological functions. If the ecologist's views had been accepted by the political process then the cost-benefit analyst would have been *constrained* to 'put the meadows' on the scheme. Notice, however, that this constraint does not produce a constrained optimum different from the unconstrained one. It only enables to pick one of two possible optima. This clearly illustrates that there is no need for the cost-benefit criterion to be always inconsistent with other decision criteria. Here, there are no trade-offs to be made between the two criteria. Moreover, the cost-benefit criterion, faced with its internal inability to pick a single optimum, definitely requires the ecological or other criterion.

Here there is a return back to the beginning of the paper: to the need for a multi-dimensional approach to landscape conservation. And, now, this need stems not only from reasonableness and political acceptability, but also from the very internal limitations of sequential cost-benefit analysis. However, *let us not confuse this with a refusal of the use of landscape valuation and extended cost-benefit analysis* in environmental decision-making. This can only be interpreted as a refusal by sympathisers of the *strong case* for CBA. Nothing in the analytical and empirical results presented so far denies the *weak case* for landscape valuation and CBA presented in the first section of this paper. Besides, the limitation just discussed is not congenital to all applications of cost-benefit analysis: non-optimising applications of the technique (for example, the appraisal of a single discrete landscape change) do not exhibit the same problem <sup>12</sup>.

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<sup>12</sup> Further arguments presented in Santos (1997) also suggest that *in the particular case* of the Pennine Dales ESA there is indeed a cost-benefit case for the inclusion of meadows in the conservation scheme. This happens because the lower bound cost estimate is the more realistic for meadows, in this context, and because this estimate leads to a unique optimum: the scheme comprising of walls, meadows and woodlands altogether. This does *not* reduce general concerns with the potential for multiple optima in sequential cost-benefit optimisation.

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