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STUDY ON THE VALUATION AND RESTORATION OF BIODIVERSITY DAMAGE FOR THE PURPOSE OF ENVIRONMENTAL LIABILITY

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FINAL REPORT (ANNEXES)

BY

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ANNEX A: ECONOMIC VALUATION: CONCEPTS AND TECHNIQUES

This Annex outlines the basic concepts relevant for the economic valuation of damages to natural resources and the selection of the most suitable restoration option (Section A.1), and techniques for estimating this economic value (Section A.2). There is a wide range of techniques that can be used for the purposes of expressing damage to natural resources including monetary valuation techniques (Section A2.1), benefits transfer (Section A2.2) and non-monetary scoring and weighting techniques (Section A2.3). Under each technique, the following issues are discussed: what the technique consists of, its pros and cons and applicability in the current context. Section A.3 concludes this annex by providing a set of criteria for choosing between different valuation techniques.

A.1 ECONOMIC VALUATION: CONCEPTS

The economic approach to valuing an environmental change (an improvement or a degradation) is based on individuals' (or households') preferences for that change. Preferences are assumed to be reflected in people's willingness to pay to secure the improvement, or avoid the degradation. Willingness to pay (WTP), in turn, is defined as the amount of goods, services or money individuals are willing to give up within a given time period to secure or avoid the change.

Preferences are also reflected in people's willingness to accept compensation to forgo the improvement or endure the degradation. Willingness to accept compensation (WTA), in turn, is defined as the amount of goods, services or money individuals are willing to accept to forgo or endure the change.

In actual markets, people's preferences are reflected in their consumption and production behaviour so that market price reflects the WTP of buyers and WTA of sellers. Generally an individual will only consume a good or a service when its price is less than or equal to the individual's WTP. When the price is less than WTP, the difference between the price and WTP is the surplus value that accrues to the individual beyond the amount he has to pay in the market. This surplus is known as *consumer surplus* in the economic literature. This implies that in most cases, the market price is only a lower-bound estimate of the individual's WTP.

For environmental goods and services which are not traded in actual market (typically because of their open access and/or non-excludable property right characteristics), the task of finding out about people's preferences and estimating the economic value in question is even more complex since when there is no price information, there is not even a lower-bound estimate of WTP. Note that the concept of consumer surplus is valid even for those goods and services that are not traded in markets.

The lack of markets and prices for many environmental goods and services means that the challenge for economists is twofold. The first task is to *identify* the reasons why individuals may have preferences for or against an environmental change, or in other words, the ways in which an environmental change affects individual's well-being. The second task is to *estimate the value of the environmental change* or *consumer surplus* through a variety of economic valuation techniques. This section outlines the first task. The economic valuation techniques discussed in Section A.2 can be used to estimate the consumer surplus both for marketed and non-marketed environmental goods and services.

There are a number of motivations behind people's preferences for environmental resources and the services these resources provide. They can be grouped as those that are related to the actual or future use of resources and their services (known as *use values*) and those that are not related to any use (*passive use* or *non-use value*).

Use values include:

- direct use values, where individuals make actual use of a resource either in a consumptive way (e.g. harvesting forest timber and abstraction of water for drinking or commercial use) or a non-consumptive way (e.g. bird watching and trekking);
- indirect use values, where society benefits from ecosystem functions (e.g. watershed protection or carbon sequestration by forests); and
- option values, where individuals are willing to pay for the option of using a resource in the future (e.g. future visits to a wilderness area, or possible future pharmaceutical uses of biological resources).

A convenient classification of *non-use values* is in terms of:

- existence values, which reflect WTP to keep a resource in existence in a context where the individual expressing the value has no actual or planned use for his/herself *or for anyone else*;
- altruistic values, which might arise when the individual is concerned that the resource in question should be available to others in the current generation; and
- bequest values, which reflect concerns that the next and future generations should have the option to make use of the resource.

These categories of motivations or preferences or value are collectively known as *total economic value* (TEV). TEV of a change, then, is the sum of both use and non-use values:

$$\text{TEV} = \text{use values} + \text{non-use values}$$
$$= \text{direct use} + \text{indirect use} + \text{option} + \text{existence} + \text{bequest values}$$

To arrive at the measure of society's preferences for an environmental change (or change in societal well-being due to the environmental change), TEV expressed by individuals need to be aggregated across the population affected by the change in question. Note that some may be affected positively, while others may be affected negatively and both parties should be accounted for in estimating the effect of the change to the society.

Determining the 'affected' population is a crucial step of the valuation of environmental change and choosing the restoration option and is discussed in more detail in Annex 2. Evidence from valuation exercises to date show that individual use values tend to be larger than individual non-use values while the aggregate non-use values tend to be larger than aggregate use values. This is mainly due to the fact that non-use values are likely to be held by a larger population than those who may be making actual use (or planning future use) of the natural resource in question.

Figure A-1 below shows the characterisation of TEV by types of value, while Table A-1 presents a taxonomy of TEV in the context of natural resource valuation.

Figure A-1: Total Economic Value

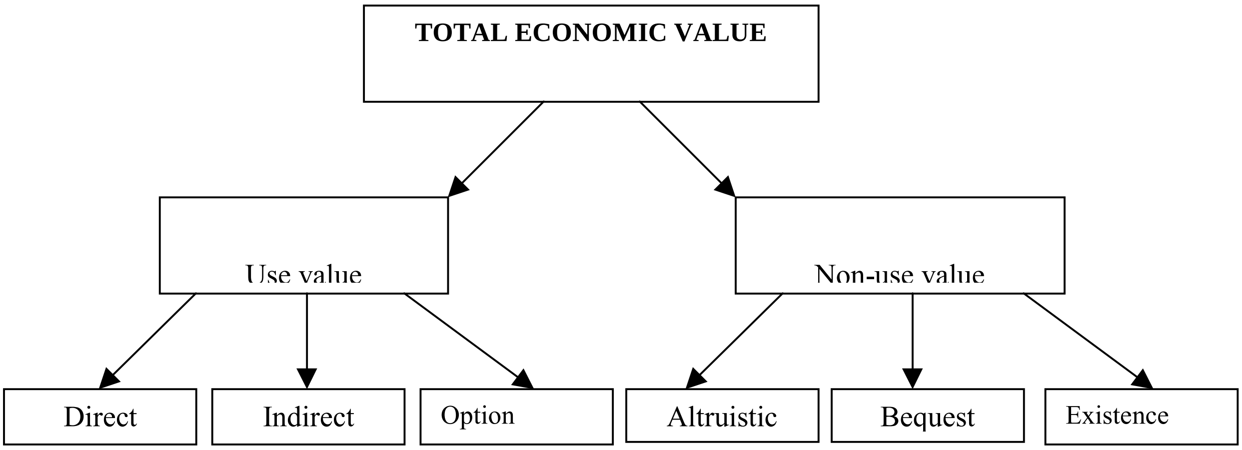

Table A-1: Total Economic Value of a Resource

Use values			Non-use values
Direct Use Values	**INDIRECT USE VALUES**	***Option Values***	***Existence, Bequest, Altruistic Values***
Products for direct use or consumption, including commercial uses, e.g.: ▪ recreation ▪ fish / meat ▪ paper / wood	Ecosystem services, e.g.: ▪ flood control ▪ storm protection ▪ carbon fixation	Potential future uses, e.g.: ▪ future visitation ▪ future use in pharmaceutical products	Value of continued existence of the resource, separate from any intent to use it (preservation for future generations, for others, or for itself)

A frequent question concerns how TEV is related to the notion of *intrinsic value.* Intrinsic value is often regarded as being a value that resides 'in' the asset in question, and especially environmental assets, but which is independent of human preferences. By definition, TEV relates to the preferences of individual human beings, so that if intrinsic value is defined to be independent of those preferences, TEV cannot encompass intrinsic values. However, notions of intrinsic value may well *influence* WTP/WTA, and some valuation techniques are particularly well-suited to revealing the *motives* for stated WTP/WTA. These motives vary and may well include notions such as 'a right to exist' for the asset in question. This is a fairly common motive when the asset is, for example, a living creature. Hence, although TEV cannot embrace a *measure* of intrinsic value, some valuation techniques may help to make the motivations for WTP explicit, and those motives may well involve a concern 'on behalf' of the object being valued.

TEV expressed by an individual for an environmental change is determined by a number of factors including but not limited to:

- price paid for marketed resources if available, (e.g. fees and other costs incurred to facilitate recreational use of resources);
- characteristics of the individual or household (e.g. income, age, gender, employment status, education, recreational choices, location, membership of an environmental group etc.);
- characteristics of the resource (e.g. whether use is made of the resource, availability and quality of the resource prior to the change in question, accessibility to the resource, the availability and characteristics of substitutes for the resources etc.);
- characteristics of the environmental change (e.g. reversible / irreversible, improvement / degradation, a voluntary change chosen by or an involuntary change imposed on those affected etc.); and
- other characteristics (e.g. weather, cultural characteristics etc.).

Economic valuation techniques discussed in Section A2 are designed to identify and collect information on these factors in order to estimate the TEV of an environmental change. In doing this, the techniques not only generate an estimate for TEV but other information which can be valuable in designing restoration options.

A.2 VALUATION TECHNIQUES

This section presents three approaches to valuing damage to natural resources: (1) economic valuation techniques, (2) benefits transfer and (3) scoring and weighting techniques. The first two approaches can generate both monetary and non-monetary expressions of people's preferences for or against damage to natural resources. The third approach is usually based on expert opinion and can generate non-monetary estimates of damage to natural resources.

A.2.1 Economic Valuation Techniques

There are three main types of economic valuation techniques:

Conventional market techniques: these techniques rely on readily observable market prices as measures of value. They are appropriate for valuing damage where there are observable impacts to commercial operations that depend on the natural resource (e.g. tourism, forest products, fisheries, etc.). These are currently used in determining compensation (e.g. fishery damage due to the Sea Empress oil spill). This technique is limited as it may only be applied to marketed goods, i.e. a subset of 'use values'. Because damage to commercial operations is usually site and resource-specific and straightforward to value (basically, the price – net of taxes and subsidies – times the quantity of resource lost or damaged), they are dealt with only briefly in this study[1].

Revealed preference techniques: these techniques attempt to find 'surrogate' or hidden markets for natural resources, where, through the price or costs of other goods and services, individuals implicitly express preferences for environmental resources. Methods of most relevance for the current context are *travel cost models* (mainly to estimate recreational value of a site) and *random utility models* (to estimate the value of different individual features of a site, which may be of interest where damage affects only certain aspects of a site). Other techniques include *hedonic property pricing* (to estimate the effect of environmental characteristics on property values) and *averting behaviour* (using the expenditure on aversion as a measure of damage cost). Revealed preference techniques are preferred in some contexts because of their explicit link with actual, observed market prices. However, these techniques are useful only in the context of estimating use values.

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[1] This assumes that prices do not change as a result of the damage, If prices of goods or services do change, then we need to look at impacts on consumers and producers surplus instead.

Stated preference techniques: these are survey methods in which hypothetical markets are created by way of carefully structured questionnaires for respondents to express their preferences. While these techniques may be used to estimate use and/or non-use values for a resource, they are the *only* techniques available for estimating non-use values. Non-use values have been shown to be a significant portion of total economic value in the context of many natural resources, especially where the resource concerned is unique or the impact is irreversible. Two variants of stated preference techniques exist: *contingent valuation* (which focuses on the natural resource as a whole) and *choice modelling* (which focuses on the individual attributes or characteristics of a given natural resource).

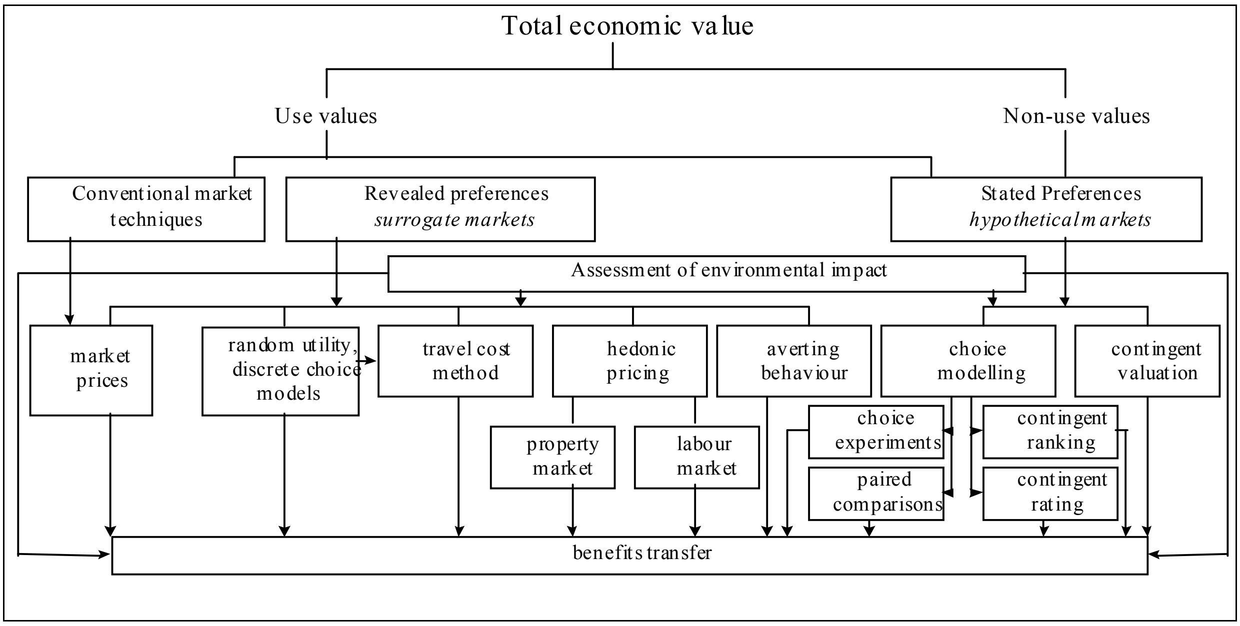
Figure A-2 overleaf provides a summary of the monetary valuation techniques, and their suitability to measuring different components of total economic value. The first feature of this figure concerns the assessment of the natural resource damage without which monetary valuation could not proceed. This can be undertaken by a number of different methods which are discussed in Chapter 3 of the main report. Whichever method is used, it is crucial to express the environmental impact in terms that are easily perceived by individuals. For example, take the case of a damage to a wetland. Impact assessment may express this as the size of wetland damaged, number of birds affected etc. For economic analysis of this damage, which estimates the change in TEV of the wetland due to the damage, we need to assess whether the wetland was used for recreation, its ecological functions etc. (different uses along the lines of those mentioned in Table A-1), and if so, how many people benefit from such uses, whether or not there are any substitutes etc. (see the factors affecting people's WTP/WTA for a resource listed at the end of Section A-1). We also need an indication whether the wetland is likely to attract non-use value[2].

The second feature of concern is *benefits transfer*. This is the process of taking information about benefits from one context (the 'study site') and applying it to another context (the 'policy site') and is the subject of a rapidly growing literature. The reason is obvious: if benefits transfer is a valid procedure, then the need for original (or 'primary') studies could be vastly reduced. In an ideal world, values would be taken 'off the shelf' and applied to new contexts. Less than ideally, this is how many actual cost-benefit studies have proceeded. It is 'less than ideally' since various requirements for a valid benefits transfer exercise are rarely met in practice. Further discussion on these requirements is presented in Section A.2.2.

Finally, note that restoration costs are omitted as a valuation technique in the figure. In some circumstances it may be legitimate to estimate benefits by the cost of replacing an asset (e.g. a particular habitat). Strictly, however, restoration cost is not a proper valuation technique: for (i) it relates to costs, not preferences; (ii) full restoration may not be possible, (iii) restoration only occurs after a time lag and (iv) restoration costs ignore psychic costs as a result of an incident. But where it is clear that the asset in question is unique and that benefits greatly exceed costs even on a limited inspection of the information available, then restoration cost becomes a minimum estimate of benefits. As a general rule, however, restoration costs as the measure of the 'value' of a resource should only be used in exceptional circumstances.

[2] The different ways in which 'damage' may be defined by damage assessment techniques and economic approach are sometimes referred to as 'correspondence' or 'translation' problem.

Figure A-2: Techniques for economic valuation



A.2.1.1 Revealed Preference Techniques

Revealed preference techniques (RP) are appropriate whenever the relevant WTP information can be inferred from individuals' decisions in actual markets. They assume that purchases in markets are reliable indicators of preference: if a person actually pays £x to buy a good, it is inferred that her WTP for that good is at least £x. Inferring current actions is a relatively simple process. However, predicting actions through RP techniques under a new set of conditions not yet experienced is not so straightforward.

The type of data and the absence of direct enquiry into individual preferences for environmental goods restrict RP techniques to the estimation of *use values* only. This suggests RP techniques are appropriate for valuing biological resources when they provide current and potential future use values, such as habitats for recreation, fisheries for commercial or recreational use and so on. However, considering that the use values represent only a part of TEV, RP are likely to underestimate the TEV of a particular resource. Estimation of non-use values is of particular importance in the current context so RP techniques are covered only briefly in this report.

The variants of RP are the travel cost method, random utility models, hedonic property pricing, and averting behaviour. The most relevant of these in the current context are travel cost method and random utility models. Restoration cost techniques are also covered here, although it should be noted that these are not true valuation techniques (see above).

A.2.1.2 Travel Cost Method

The travel cost method seeks to value *recreational use* of woodlands, wetlands, coastal zones and so on. The method assumes that travel costs (fares, fuel cost, wear and tear, out of pocket expenses such as entry fees and value of time spent travelling and on site) are a proxy for the recreational value of visiting a given site. More recent applications of the TCM are referred to as count data models. The demand for the site is estimated by observing variation in the number of site visits according to variation in these costs. The methodology assumes there is an inverse relationship between the visit costs and the number of visits observed. For example, people living further away from the site incur higher transport costs and hence visit less often than those who live nearer the site. Because travel cost models are concerned with active participation they measure only the *use* value associated with any recreation site. Table A-2 summarises the main advantages, disadvantages and the applicability of travel cost method in the current context.

Table A-2: Evaluation of travel cost method

Advantages	• use of real market data
Disadvantages	• can estimate use values alone • may have substantial data requirements, if data are not readily available • requires estimates of value of travel / leisure time • problems arise with multi-purpose trips • cannot predict the changes in use values due to environmental changes without precedence
Application	• recreational value (total) associated with national parks, reservoirs, woodlands, forests, wetlands, etc.

A.2.1.3 Random Utility Model

The random utility model is an extension of the travel cost method. It seeks to estimate recreational use values for individual features of a site. The random utility approach is concerned with explaining the choice between two or more goods with varying environmental attributes as a function of their characteristics. This can be useful where, for example, polluting activity causes damage to only some features of a recreational site. Travel costs and site attribute data are collected for a number of substitute sites in an area. The probability that an individual will visit site A rather than site B is then estimated depending on the costs of visiting each site and their physical characteristics relative to the characteristics of all sites in the individual's choice set. For example, for a forest, these could include species diversity and recreational facilities. Table A-3 summarises the main advantages, disadvantages and the applicability of random utility model in the current context.

Table A-3: Evaluation of random utility model

Advantages	• estimates recreational use value of (i) changing environmental quality of site attributes and (ii) recreational use value of site in total • use of real market data
Disadvantages	• can estimate use values alone • may have substantial data requirements, if data are not readily available • requires estimates of value of travel / leisure time • problems arise with multi-purpose trips • cannot predict the changes in use values due to environmental changes without precedence • can be hard to handle participation decisions (i.e. whether to make the visit or not)
Application	• open access resources such as national parks, public woodlands, forests, wetlands, fisheries

A.2.1.4 Hedonic Pricing Method

The hedonic pricing method seeks to estimate an implicit price for environmental attributes by observing actual markets in which those attributes are effectively traded. The market most frequently used is the housing market where property prices are determined not only by the structural characteristics (e.g. number of rooms, etc) and the access to public services (e.g. schools, etc) but also the environmental characteristics (e.g. clean air, peace and quiet, pleasant views, etc). By controlling for the non-environmental features and observing variations in house prices over the environmental attributes, the hedonic approach seeks to (i) estimate the implicit price that individuals are willing to pay for environmental characteristics associated with the house and (ii) infer how much people are willing to pay for an improvement in environmental quality, and hence estimate households' demand for the environmental characteristics of concern.

Current evidence suggests that hedonic pricing is mostly used for estimating the implicit price of an environmental characteristics ((i) above). Since the implicit price is unique to the property market for which it is estimated, there is no justification for transferring them across property markets. Only studies that estimate a demand function ((ii) above), can be transferred to other contexts (Day, 2000). Table A-4 summarises the main advantages, disadvantages and the applicability of hedonic pricing method in the current context.

Table A-4: Evaluation of hedonic pricing method

Advantages	• use of real market data
Disadvantages	• can estimate use values alone • requires extensive house market data • cannot predict the changes in use values due to environmental changes without precedence • current evidence suggests it is not suitable for use in benefits transfer
Application	• amenities (such as proximity of woodlands), visibility (air pollution), noise, etc.

A.2.1.5 Avertive Expenditure / Avoided Cost Approach

The averting expenditure approach infers a monetary value for an environmental problem by observing the costs individuals are prepared to pay in order to avoid it. The increase in averting expenditure such as purchase of water filters to ensure safe drinking water, or double glazing to reduce road / air traffic noise, is assumed to be a minimum measure of the welfare loss to the household of the decline in environmental quality. However problems of interpreting WTP estimates arise where the good used to trade-off against some environmental problem may have other benefits (e.g. double glazing provides multiple services such as energy conservation and anti-burglary), when averting expenditure is an imperfect substitute for the environmental loss, or when individuals engage in more than one form of averting behaviour. Table A-5 summarises the main advantages, disadvantages and the applicability of avertive expenditure method in the current context.

Table A-5: Evaluation of the avertive expenditure method

Advantages	• modest data requirements • use of real market data
Disadvantages	• can estimate use values alone • problems arise when: i) individuals make multiple averting expenditures, ii) there are secondary benefits of an averting expenditure and iii) averting behaviour is not a continuous decision but a discrete one, i.e. double glazing is either purchased or not • cannot predict the changes in use values due to environmental changes without precedence
Application	• water quality, noise nuisance, air pollution and radon contamination

A.2.1.6 Restoration Cost Method

The restoration cost approach values an environmental good according to the cost incurred in restoring it to its original state after it has been damaged. This approach is used extensively because it is relatively easy to find estimates of such costs. The approach also forms the basis of the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) 1980 in the USA. However, this method is not a proper valuation technique because there is no formal relationship between costs of restoration and TEV. The only link between restoration cost and TEV of a resource exist when there is a prior and unanimous social decision that the restoration must take place. Table A-6 summarises the main advantages, disadvantages and the applicability of restoration cost method in the current context.

Table A-6: Evaluation of restoration cost method

Advantages	• costs easily obtained from direct observation of actual spending on restoring damage or from professional estimates
Disadvantages	• not considered a proper valuation technique • if restoration must take place, then costs become a minimum estimate of TEV • potential to underestimate TEV because some damage may not be fully perceived, or may arise in the long term, or may not be fully restorable • potential to underestimate TEV because there may be secondary benefits from restoration
Application	Use as an indication of the TEV of a resource only in exceptional circumstances such as: • when restoration must take place due to quality standards • as first approximations when there is an overall constraint not to let environmental quality decline • useful for flood protection, water regulatory services supplied by forested watersheds, replacement of traditional medicines and the costs of crop insurance to replace natural insurance afforded by genetically diverse traditional cropping systems (Pearce and Moran 1994)

A.2.1.7 Stated Preference Techniques

Stated preference techniques (SP) use carefully structured questionnaires to elicit respondents' preferences for a given natural resource or environmental change. In principle, SP can be applied to any context. Thus, there exist numerous SP studies covering a wide range of non-marketed commodities. In addition SP are the *only* techniques that can estimate values for non-use benefits of environmental resources. Non-use values have been shown to be a significant portion of TEV in the context of many natural resources, especially where the resource concerned is unique or the impact is irreversible. SP are also the most appropriate techniques to estimate indirect use values of ecological stability, biodiversity, watershed protection and so on.

In the context of biodiversity, where respondents may not be entirely familiar with the resource in question (especially with the concept of 'diversity' as opposed to an individual biological resource), it is particularly important that surveys are designed in accordance with respected guidelines (Spash and Hanley, 1995). Such a questionnaire will provide adequate information to permit respondents to give reliable estimates of WTP even though they are not directly familiar with the resource described. Currently available guidelines are those of NOAA (Arrow *et al*, 1993) and EFTEC (forthcoming, 2001).

There are two variants of SP: contingent valuation and choice modelling. Both variants use similarly structured questionnaires but differ in the way they define the environmental resource of concern. Contingent valuation is concerned with the resource as the bundle of its attributes or characteristics, while choice modelling is concerned with the individual attributes of the resource.

A.2.1.8 Contingent Valuation

The aim of a CV study is to elicit individuals' preferences, in monetary terms, for changes in quantity or quality of a non-market good or service. In CV studies, a contingent market defines the good itself, the institutional context in which it would be provided and the way it would be financed. The WTP / WTA question can be asked in a number of different ways, known as elicitation formats. These formats include open-ended (What are you willing to pay?), dichotomous choice (Are you willing to pay £x?), bidding game (repetition of the dichotomous choice question with lower or higher amounts depending on the response to the initial question) and payment card (where the respondent chooses the WTP amount from a list of different amounts shown to him). More information about the elicitation formats and how they affect the resulting WTP or WTA estimate can be found in Annex 3.

Econometric techniques are then applied to the survey results to derive the average WTP or WTA and to explain the variance in these results depending on the differences in the characteristics of the environmental resource and/or change of concern and the socio-economic characteristics of the respondent population, i.e. to define what is known as the 'bid function'.

CV is likely to be most reliable for valuing environmental gains, particularly when familiar resources are considered. Using the standard CV approach to value complex and multidimensional scenarios raises some concerns, as complex and burdensome questionnaire designs as well as very large samples are required. In such situations, choice modelling may be preferred. There is also the issue of scope, which refers to the possibility that people may not be able to differentiate between different scales of the environmental impact of concern (especially very small changes). The academic debate on this suggests that estimated values may relate to some invariant 'moral satisfaction' or 'warm-glow' measure rather than the described impact. This issue affects both variants of stated preference techniques and is discussed further in Annex 3. Table A-7 summarises the main advantages, disadvantages and the applicability of contingent valuation in the current context.

Table A-7: Evaluation of contingent valuation

Advantages	• can estimate both use and non-use values • suitable for valuing environmental changes irrespective of whether or not they have precedence • completed surveys give full profile of target population
Disadvantages	• relatively expensive • complex and multidimensional scenarios may be too much of a cognitive burden for respondents • the concept of 'diversity' may similarly be difficult to put across to the respondents
Application	• potentially can be applied to all environmental resources and changes, though the application is limited by the complexity of the questionnaire • most suitable when all attributes of a natural resource are affected rather than individual attributes.

A.2.1.9 Choice Modelling

The term choice modelling (CM) represents a range of SP techniques which take a similar approach to valuing non-market goods including:

- choice experiments,
- contingent ranking,
- contingent rating, and
- paired comparisons.

CM were originally designed by marketing practitioners to isolate the value of individual product characteristics typically supplied in combination with one another. The techniques are based around the idea that any good can be described in terms of its attributes, and the levels that these take. For example, a forest can be described in terms of its species diversity, age structure and recreational facilities and a river can be described in terms of its chemical water quality, ecological quality and appearance. Changing attribute levels will essentially result in a different 'good' being produced and it is on the value of such changes in attributes that CM techniques focus (EFTEC, forthcoming 2001).

Thus, CM approaches provide a direct route to the valuation of the attributes of a good and of marginal changes in these characteristics, rather than the value of the good *in toto*. This is of interest for those instances where damage affects only certain aspects of the natural resource of concern. Contingent valuation could also be used to value such changes, but the number of scenarios that can be considered in any one study is limited. Thus, it is generally assumed that choice modelling approaches are preferred over contingent valuation in contexts where it is important to value individual attributes. However, not all of the CM techniques are founded in the theory of welfare economics. In fact, only the choice experiment approach definitely fits the theory, while contingent ranking may do so (EFTEC, forthcoming 2001). For this reason, this chapter considers choice experiments and contingent ranking only.

In a ***choice experiment*** respondents are presented with a series of alternatives and asked to choose their *most preferred* option. Each alternative is characterised by a number of attributes, of which one will be monetary (that is a price ticket), offered at different levels across options. Analysts can then see how respondents' choices change as the attributes and monetary amounts are varied and from this information infer the value placed upon each attribute. A baseline status quo alternative is usually included in each choice set in order to ensure estimates are consistent with the theory of welfare economics.

Choice experiments give welfare consistent estimates for four reasons. First, they force the respondents to trade off changes in attribute levels against the costs of making these changes. Second, the respondents can opt for the status quo, that is no increase in environmental quality at no extra cost to them. Third, because we can represent the econometric technique used in a way which is exactly parallel to the theory of rational, probabilistic choice. Fourth, because we can derive estimates of compensating and equivalent surplus from the "output" of the technique.

In a ***contingent ranking experiment*** respondents are asked to *rank* a number of alternative options according to their preferences. Each alternative is made up of a number of attributes and prices, which are set at varying levels across options. From the ordinal rankings (or choices), the WTP associated with each attribute can be indirectly calculated. As such, CR provides not only a monetary measure for each scenario but also uncovers the monetary value of each individual attribute within the scenario. In addition, CR avoids the need for an *explicit* elicitation of respondent willingness to pay by relying instead on the ranking of (or choices between) a series of alternative scenarios or packages of attributes. It is important that one of the options must always be *status quo* for estimates to be consistent with standard welfare economics.

Table A-8 summarises the main advantages, disadvantages and the applicability of choice modelling in the current context.

Table A-8: Evaluation of choice modelling

Advantages	• can estimate both use and non-use values • suitable for valuing environmental changes irrespective of whether or not they have precedence • completed surveys give full profile of target population
Disadvantages	• not yet as widely tested as CV • some techniques are not based on economic theory • the concept of 'diversity' may be difficult to put across to the respondents
Application	• potentially can be applied for all environmental resources and changes, though the application is limited by the complexity of the questionnaire • appropriate method when damage affects only certain attributes of a natural resource

Choice modelling can also be used to elicit respondents' opinion about the incident, damage and the characteristics of the restoration option without a monetary attribute. The choices respondents make provide information about their preferences about the different aspects of damage and restoration option and hence can be an input to designing the restoration options.

Note that choice experiments can be used to generate non-monetary estimates of damage to biodiversity or benefit of restoration and can also inform the choice of restoration options. This requires the design of the questionnaire according to the general rules but not including a cost or price attribute. This way, respondents can express their preferences for or against the damage and restoration without expressing their WTP/WTA in monetary terms.

Whether a SP study is successful or not depends on the existence of potential biases and how these are dealt with. The term 'bias' refers to the difference between the 'true' WTP/WTA of a respondent and his/her 'stated' WTP/WTA, where such difference causes validity problems for the results of the study. Note that differences between value types do not always constitute a validity problem. Information bias (WTP/WTA sensitive to the amount and quality of information provided), for example, is defined as a reliability problem by some experts but as an expected and normal outcome by others. A typology of biases and ways to avoid them are presented in Annex 3.

A.2.2 Benefits Transfer

Benefits transfer (BT), i.e. the process of taking information about benefits from one context (the 'study site') and applying it to another context (the 'policy site'), is the subject of a rapidly growing literature. The reason is obvious: if BT is a valid procedure, then the need for original (or 'primary') studies could be vastly reduced. In an ideal world, values would be taken 'off the shelf' and applied to new contexts. Less than ideally, this is how many cost-benefit studies or other contexts in which economic valuation is used proceed, and have proceeded for some considerable time. 'Less than ideally' since various requirements for a valid benefits transfer exercise (as listed below) are rarely met in practice.

This section first outlines the general requirements for BT, and the different procedures, which may be used. This is followed by a discussion of the validity of benefits transfer and methodological considerations. Finally, the section is concluded with a review of the currently available electronic databases of valuation studies.

A.2.2.1 Requirements and Procedures for Benefits Transfer

It should be noted that certain conditions have to be met for a valid transfer of value to take place (Boyle and Bergstrom, 1992; Desvousges *et al*, 1992; EFTEC, forthcoming 2001). These are widely recognised to be the following:

- the studies included in the analysis must themselves be sound;
- the studies should contain WTP bid functions, i.e. regressions showing how WTP varies with explanatory variables such as those factors listed in Section A.1;
- the study and policy sites must be similar in terms of population and site characteristics, or differences in characteristics must be adjusted for;
- the change in the provision of the good being valued at the two sites should be similar, and WTP measures cannot be changed into WTA measures and vice versa; and
- property rights should be the same across the sites.

The process of benefits transfer is clearly less than ideal. Even if all the above criteria are met, benefits transfer are only as accurate as the original valuation study or studies. Thus, the quality of the data, and the methodology, of the original study or studies need to be examined.

In implementing benefits transfer, the three most common procedures are to (i) transfer an average WTP estimate from one primary study, (ii) transfer WTP estimates from meta-analyses, and (iii) transfer a WTP function. These are discussed in turn overleaf. The analysis applies equally to WTA but WTP is used throughout this section as a short hand.

1. Transferring average WTP from a single study to another site which has no study

One elementary procedure is to 'borrow' an estimate of WTP in context i (the study site) and apply it to context j (the policy site). The estimate may be left unadjusted, or it may be adjusted in some way. Transferring unadjusted estimates is clearly hazardous, although it is widely practised. Reasons for differences in average WTP include differences in the:

- socio-economic characteristics of the relevant populations;
- physical characteristics of the study and policy site;
- proposed change in provision between the sites of the good to be valued; and
- market conditions applying to the sites (for example variation in the availability of substitutes) (Bateman *et al*, 1999a).

As a general rule, there is little evidence that the conditions for accepting unadjusted value transfer hold in practice. Effectively, those conditions amount to saying that the various conditions listed above all do not hold, i.e. sites are effectively 'identical' in all these characteristics. An alternative is therefore to adjust the WTP estimates in some way.

A widely used formula for adjusted transfer is:

$WTP_j = WTP_i\ (Yj/Yi)^e$

where Y is income per capita, WTP is willingness to pay, and 'e' is the “income elasticity of WTP”, i.e. an estimate of how the WTP for the environmental attribute in question varies with changes in income. The typical practice in benefits transfers between countries has been to use the ratio of income in the two countries, as income is known to be one of the most important (if not the most important) factors resulting in changes in WTP. However, it is also possible to make a similar adjustment for, say, differences in age structure between the two sites, differences in population density, and so on. Making multiple changes of this kind amounts to transferring benefit functions (see overleaf). Another approach to adjusted transfer involves the selection of a sub-sample from the original study site and then transferring the WTP estimate for that sub-sample to the policy site on the grounds that the policy site is more like the sub-sample than the complete sample. This may work in some contexts but sub-dividing samples may render the transferred values less reliable due to small sample size.

2. Transferring benefit functions

A more sophisticated approach is to transfer the *benefit function* (or bid function) from i and apply it to j. Thus if it is known that $WTP_i = f(A,B,C,Y)$ where A,B,C and Y are factors affecting WTP at site i, then WTP_j can be estimated using the coefficients from this equation but using the values of A,B,C, Y at site j. Given that the characteristics of the population to which the estimate will be transferred is likely to differ from those of the study population, it is hoped that benefits estimates can be improved by using the transfer equation to modify the estimate of average WTP to account for these differences.

This approach relies on the availability of an appropriate valuation function in the original study, which may be used for BT purposes. When benefits transfer is the objective, the analysis of valuation results according to the 'best fit' model, which is usually the only model reported, will usually be inadequate for BT purposes. A BT model will contain only a limited number of covariates: those that can easily be gathered for the transfer population. Usually this amounts to basic socio-economic details, such as the respondent's income, age and sex, since details of these characteristics in the transfer population can be easily collected from census returns. For resources that have a spatial dimension, however, it is crucial to include a variable that measures the household's distance from the site of provision.

Although valuation practitioners are becoming more aware of potential uses of their results for BT, it is not yet standard practice to present the results of such a model in a report on the analysis of data from any valuation study.

3. Transferring benefit functions: meta analysis

An alternative procedure is to use *meta-analysis* to take the results from a number of studies and analyse them in such a way that the variations in WTP found in those studies can be explained. This should enable better transfer of values since we can find out what WTP depends on. In the meta-analysis case, whole functions are transferred rather than average values, and the functions do not come from a single study, but from a collection of studies.

Interest in the application of meta-analysis to the field of economic valuation has expanded rapidly in recent years, and many are of relevance to the current context of natural resource valuation. Studies have taken place in respect of, among others, outdoor recreation, the ecological functions of wetlands, and the local income generation effects of tourism (see, for example, Rosenberger and Loomis, 2000 and Smith *et al*, 1996).

Table A-9 summarises some relevant studies for valuation of natural resources.

Table A-9: A selection of meta-analyses from the literature

Study	Method	Resource
Walsh *et al* (1988, 1990)	TC and CV	recreational activities
Smith and Kauro (1990)	TC	recreational activities
Sturtevant *et al* (1995)	TC	freshwater fishing
Smith and Osborne (1996)	CV	visibility at national parks
Smith (1993)	CV	Visibility
Magnussen (1993)	CV	water quality
Boyle *et al* (1994)	CV	groundwater protection
Bateman *et al* (1995)	CV	woodland recreation
Loomis and White (1996)	CV	rare / endangered species
Brouwer *et al* (1999)	CV	Environmental functions of wetlands

TC: travel cost method; CV: contingent valuation.

Some interesting results of relevance to the valuation of biodiversity in Europe have emerged from meta-analysis. For example, in a meta-analysis of wetland values comprising over 30 studies (Brouwer *et al*, 1999), the basic model indicates that study location has a significant impact on average WTP. Average WTP appears to be substantially higher in North America than in Europe. Also interesting is the role played by the wetland functions themselves since they have a statistically significant role in explaining variance in average WTP. The average WTP is highest for flood control, followed by the function of supplying or supporting biodiversity, then water generation and lowest for water quality.

Brouwer *et al* (1999) indicate some study design factors with significant influences on WTP. C*eteris paribus*, an almost twice as high average WTP was found for an increase in income tax than for any other payment vehicle. Response rate is also found to influence the results. In low response rate studies only those who are really interested in the good, i.e. they also have a high WTP, make the effort to respond. Therefore mean WTP is high in such cases. Conversely, in high response rate studies, people with low WTP are also responding so that mean WTP for the whole sample falls. With respect to elicitation formats, the open-ended format is seen to produce a significantly lower WTP, by about 40%, than other formats. The dichotomous choice format yields the highest average WTP, followed by the iterative bidding procedure (See Annex 3 for details).

The suitability of the meta analysis for benefits transfer is again the subject of some cautionary remarks by Brouwer *et al* (1999). But the authors suggest that if low variance reflects the quality of the estimate for purposes of benefits transfer, then studies using income taxation as a payment vehicle are better suited than other payment vehicles, and studies valuing wetland biodiversity are more reliably transferred than estimates of the value of wetlands in their capacity of generating water or maintaining water quality.

The ELF (Environmental Landscape Features) model under development for the UK Ministry of Agriculture, Fisheries and Food (MAFF) uses a meta analysis of benefit function coefficients to develop a predictive BT model (Hanley, 1999).

A.2.2.2 Validity of Benefits Transfer and Methodological Considerations

The previous section illustrated the various ways in which BT might be practised. A further consideration is whether BT is affected in any particular way by the nature of the valuation studies that are included. For example, is there any reason to suppose that stated preference studies will perform better or worse than revealed preference studies? This is of concern in the current context since stated preference techniques are the only methods capable of estimating non-use values, which are likely to be important in the valuation of damage to natural resources.

On the basis of the requirements for BT outlined above, there is no particular reason to suppose that stated preference studies would fare any worse than any other form of study, although care will need to be taken to ensure that views about who has effective property rights are fully accounted for. Otherwise, the tests are very much a matter of carefully scrutinising the accuracy and professionalism of the original studies.

More formal tests might therefore involve seeing how far the transferred values are accurate. If transferred value from stated preference studies have more error than transferred values from revealed preference studies, then there may be a legitimate concern about the validity of stated preference transfers. However, it is only in the last decade that the question has been properly raised as to whether BT is valid, and formal testing of this type has taken place. Brouwer (1998) observes that there are comparatively few studies that test for the validity of BT, on whatever basis the original estimates were obtained.

There are three broad procedures for the validation of BT results, which have been undertaken to date (although these procedures cannot be required from each BT exercise due to time and budget limitations):

(i) transfer a value and then carry out a primary study at the policy site as well. Ideally, the transferred value and the primary estimate should be similar. If this exercise is repeated until a significant sample of studies exists in which primary and transferred values are calculated for policy sites, then there would be a justification for assuming that transferred values could be used in the future without the need to validate them with primary studies;

(ii) conduct a *meta-analysis* on existing studies to explain why studies result in different mean (or median) estimates of WTP. At its simplest, a meta analysis might simply take an average of existing estimates of WTP, provided the dispersion about the average is not found to be substantial, and use that average in policy site studies. Or, average values might be weighted by the dispersion about the mean, the wider the dispersion the lower the weight the estimate would receive. A more sophisticated approach takes a set of n primary studies and uses n-1 of the studies to estimate the value at the n^{th} site. That 'transferred' value can then be compared with the original primary value at that site; and

(iii) benefits transfer may be tested by estimating WTP before an actual change occurs and then revisiting the area later when the change is complete to see if people behaved according to their stated WTP.

Table A-10 summarises some recent work for validation of benefits transfer. Some general findings (EFTEC, forthcoming 2001) are:

- that transferring benefit functions is more accurate than transferring average values;
- stated preference studies appear to perform no worse than revealed preference studies in terms of transfer error;
- but transfer error can be quite large, 1 to 75% if 'outliers', i.e. responses which are unrealistically low or high, are ignored, but up to 450% if they are included;
- individuals' attitudes are often important determinants of WTP in SP studies, yet most BT makes little effort to test for variability in attitudes across sites. This suggests that BT would have to be supplemented by social surveys at the policy site;
- meta-analysis of contingent valuation studies can explain a reasonable proportion of the variation in the original studies, but the original studies do not include sufficient information to test whether more information would have increased the explanatory power of the meta-analysis; and
- the missing information may well be of the motivational type, i.e. why people adopt the value stances they do. This conclusion fits well with the current focus in economics on the analysis of motives for preferences.

The new focus on motives suggests that more attention should be paid to motives in SP studies so that there is a better chance of explaining WTP variation between studies. Study context may also be important (e.g. the historical factors affecting a particular site, or the 'causes' of the problem that is being valued). Efforts to conduct SP studies in a context-free environment should help to resolve this issue, although care must be taken not to render the questions meaningless to respondents.

Table A-10: Studies for valuation of benefits transfer

Study	Valuation technique	Environmental good	Transfer error
Loomis (1992)	TC	sport fishing	5-40%
Parsons and Kealy (1994)	random utility model	water quality improvements	1-75%
Loomis *et al* (1995)	TC	reservoir recreation	1-475%
Bergland *et al* (1995)	iterative bidding CV	water quality improvements	25-45%
O'Doherty (1996)	open-ended CV	green space	unknown
Downing and Ozuna (1996)	dichotomous choice CV	saltwater fishing	1-34%
Kirchhoff *et al* (1997)	payment card CV	white water rafting	6-228%
Brouwer and Spaninks (1998)	CV	agricultural wildlife	20%
Barton (1999)	dichotomous choice CV	water quality improvements	11-26%
Barton (1999)	payment card CV	water quality related illness episodes	5-120%
EC (1999)	CV	air pollution related illness episodes	4-45%
Desvouges *et al* (1998)	Meta-analysis of CV studies, plus hedonic property pricing	air pollution related illness episodes	50%+

Source: adapted from IERM (1999) which draws heavily on Brouwer et al (1998); supplemented with more recent studies.

It may be that values are transferable but that much more information is required before meta-analyses can explain the variation in WTP across studies. On this view, more research will improve the reliability of BT at some stage in the future. Navrud (1999) summarises the main challenges for future applications of BT as:

- increasing the number of high quality studies, constructed and reported with BT in mind;
- targeting of original studies at existing gaps in the literature – changes which are outside the range of previous experience, discrete versus marginal changes, increases versus decreases in environmental quality;
- increasing the availability of valuation databases;
- improving BT procedures, including research into differences between study and policy sites which are not accounted for in the specification of the valuation function or adjustment procedures (e.g. cultural and institutional differences);
- increasing the number of original studies and validity tests of BT for complex environmental goods with large non-use values (e.g. ecosystems) since the studies of use values currently account for the largest number of studies and most meta-analyses; and
- better determining the extent of the market, i.e. the relevant population holding values for a resource.

This last point, i.e. determination of the market, is of particular relevance for non-use values. Accurately determining the extent of the market is at least as important as accurate estimation of WTP values, as this will have a strong influence on the estimate of total economic value of a change. While there is an increasing literature which addresses this issue (see for example Bateman *et al*, 2000; Moran, 1999) the process has a source of dispute in UK policy-making in the recent years. It is likely that some survey work would be needed to give some indication of the boundaries of the relevant population.

Another view is that values are inherently not transferable because what is valued is site-specific and because the characteristics of those engaging in valuation are site specific too. At the moment, there is no consensus on these issues. This points the way towards (a) continued reliance on primary studies, and (b) conducting those primary studies in a manner that is consistent with future BT tests.

Given that benefits transfer is not yet accepted as a valid procedure in the academic community (IERM, 1999; Oglethorpe *et al*, 2000, EFTEC, forthcoming 2001) it may well be that original valuation studies are required where damage to environmental resources is significant, or irreversible, or where the resource concerned is unique. The results of an original valuation study are certain to be less controversial than any estimates obtained through benefits transfer. While the results of individual stated preference studies may still be open to debate[3], in the past decade these techniques have gained much credence in academic and policy-making communities, when properly conducted. Detailed guidelines for the structure of such studies do exist (Arrow *et al*, 1993; EFTEC, forthcoming 2001). If the responsibility for commissioning original studies valuing damage lies with the EC, then independent consultants can be commissioned, and the proper conduct of studies can be overseen.

As discussed above, results tend to differ by up to 75% if outliers are excluded, and by up to 450% if they are included. Whether this margin of error is considered 'large' or 'too large' depends on the use of the results.[4] For some project and policy applications it is probably acceptable, and uncertainty of the final results can be dealt with through sensitivity analysis. Indeed, it is not uncommon to find demand studies for market goods and services where, depending on the method of estimation, the functional form, and the selection of observations, the results can vary by a factor of five or more.

It may be the case that there is a role for benefits transfer in cases where environmental damage affects non-unique resources, is reversible, is less significant compared to other costs of an incident, or where similar damage has previously occurred and been the subject of a valuation study. A more important potential role, however, is likely to be its use in the evaluation of restoration options. It is probable that a range of restoration options will be identified for consideration, and it would certainly be infeasible to use original studies to value the likely effects of each. Different options for restoration are likely to affect similar environmental variables to different degrees. In these circumstances, benefits transfer may be used to value some or all of the changes affected by each option. This process will reduce the number of environmental impacts expressed in non-monetary terms, thereby facilitating cost-effectiveness analysis or any other methods used to aid in project selection. Sensitivity analysis may be used in conjunction with benefits transfer, for example to test the effects of using different unit values from the existing literature.

A.2.3 Review of Valuation Databases for Benefits Transfer

There are a number of web-based databases of valuation studies, designed for the application to benefits transfer. This section gives an overview of the databases currently available, with an assessment of their coverage in terms of subject, resource type, geographical area, completeness of information, study type (e.g. coverage of 'grey literature'), and on-going maintenance of the information (e.g. whether it is up-to-date).

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[3] For examples of high profile cases in the USA where stated preference studies have been used to value damages to natural resources, and the results have been challenged, see Breedlove (1999).

[4] The trade-off is essentially one between accuracy and cost. This relationship is modelled formally by Desvouges *et al* (1992).

Five currently available databases are reviewed here. These are:

- the Environmental Valuation Reference Inventory (EVRI) of Environment Canada;
- the Valuation Source List for the UK, compiled by the UK Department of the Environment, Transport and the Regions;
- the Australian New South Wales Environmental Protection Agency's database, Envalue;
- the Economy and Environment Program for South East Asia (EEPSEA) commissioned database, ValuAsia; and
- the New Zealand Non-Market Valuation Database, based at Lincoln University, Canterbury, New Zealand.

All of these databases are reviewed here to highlight features of design and construction. Most have some European content which may be relevant in the current context, although valuation studies conducted outside the EU may also be of interest due to cross-country effects (e.g. impacts to migratory birds). Of the existing databases, EVRI is widely acknowledged to be the most complete to date.

There are other databases currently under construction, but not yet publicly available. For this reason these are not reviewed here. However, they may be of interest in the future. Such databases include:

- the UK Environment Agency's National Centre for Risk Analysis and Options Appraisal database;
- the UK Countryside Agency's database of non-timber values; and
- the Environmental and Landscape Features Model (ELF) under development for the UK Ministry of Agriculture, Fisheries and Food (MAFF).

In addition, there are several existing guidelines for conducting benefits transfer, mainly for the purposes of project assessment, for various different media. While these manuals are not available electronically, they can often provide a good summary of existing valuation studies relating to particular resources. These include:

- Guidelines for Project Appraisal in Fresh Water Supply, Waste Water Treatment and Solid Waste Management Sectors, European Investment Bank, 1998;
- Framework to Assess Environmental Costs and Benefits for a Range of Total Water Management Options, UK Environment Agency, 1998; and,
- Project appraisal guidance documents published by international development banks such as the World Bank and the Asian Development Bank.

A brief review is made below of the databases currently available for assisting benefits transfer.

A.2.3.1 EVRI

The EVRI database was developed for Environment Canada, in conjunction with several other institutions, notably the United States Environment Protection Agency. The database is available to registered subscribers at http://www.evri.ec.gc.ca/evri/. This is without question the most complete database of non-market valuation studies internationally. The summary presented here is based on the information given in the 'Tour EVRI' pages on the site.

EVRI is designed for use in the application of benefits transfer and all of the information is presented with this in mind. Information is divided into six main categories, as follows:

1. **Study Reference** – basic bibliographic information;
2. **Study Area and Population Characteristics** – information about the location of the study along with population and site data;
3. **Environmental Focus of the Study** – fields that describe the environmental asset being valued, the stressors on the environment, and the specific purpose of the study;
4. **Study Methods** – technical information on the actual study, along with the specific techniques that were used to arrive at the results;
5. **Estimated Values** – the monetary values that are presented in the study as well as the specific units of measure; and
6. **Alternative Language Summary** – an abstract of the study available in English, French and Spanish.

Currently, the database contains information from over 700 studies, although further expansion of the database is underway, and improved coverage of European studies is one focus of this work. At the moment, entries are concentrated in the area of water valuation studies and the database is most complete in this area. This is a consequence of the initial focus during the testing and development of the database.

Tables A-11 to A-14 present a summary of the coverage of the studies available on the EVRI database at the time of writing. The geographical breakdown of the studies included in the database (Table A-11) shows strong USA coverage, with 591 studies covered. In contrast, the number of European studies, at 85, is relatively low at the moment.

Table A-11: Geographical distribution of valuation studies

Geographical location	USA	Europe	Asia	Africa	S. America	total
No. of studies	591	85	53	18	8	755
% of studies	78%	11%	7%	2%	1%	100%

Tables A-12 and A-13 show the distribution of studies according to environmental asset considered, and good or service valued respectively. As acknowledged by EVRI, this does show a bias towards water-based studies at the present time, with these studies accounting for a third of all papers considered. Valuation studies of animals and land resources are also fairly numerous, accounting for 26% and 17% of studies respectively. All of these categories may be relevant to the valuation of biodiversity, as defined in the White Paper on Environmental Liability (2000).

Table A-12: Distribution of valuation studies according to environmental asset

Asset	air	Animals	human	land	man	micro	plants	water	total
No. of studies	49	305	115	193	54	3	62	383	1164
% of studies	4%	26%	10%	17%	5%	0%	5%	33%	100%

The distribution of studies according to good/service valued (Table A-13) is concentrated on extractive uses (35%), non-extractive uses (28%) and passive uses (14%). This may also be a result of the current focus on water-based studies, and the related goods and services they produce. Other goods and services valued include human health (11% of studies), ecological functions (8%) and the built environment (3%).

Table A-13: Distribution of valuation studies according to good/service valued

Good / service	built env't	ecological functions	extractive uses	human health	non-extractive uses	passive uses	total
No. of studies	39	102	420	133	343	168	1205
% of studies	3%	8%	35%	11%	28%	14%	100%

Finally, Table A-14 shows the distribution of studies according to valuation technique. The bulk of studies (58%) use revealed preference techniques, while 28% are based on stated preference techniques. The remainder (14%) are classified as 'actual' techniques by EVRI, though the definition is not clear; presumably it refers to using actual market data.

Table A-14: Distribution of valuation studies according to valuation technique

Technique	stated preferences	revealed preferences	other ('actual'?)	total
No. of studies	262	549	130	941
% of studies	28%	58%	14%	100%

The search facilities of EVRI allow the user to identify relevant studies according to:

- Geographic characteristics;
- Economic measure and market characteristics; and
- Similarity of environmental issues.

The initial search may then be refined using a Screening Module, which helps assess the suitability of the candidate studies for benefits transfer, and evaluate the quality of the studies.

Suitability is determined based on similarities between the policy site and the study sites in the following areas:

- Geographic location
- Population
- Environment
- Timeliness of Data
- Economic Measure
- Estimated Values
- Abstract

A.2.3.2 DETR Valuation Source List for the UK

The Valuation Source List of the UK Department of the Environment, Transport and the Regions is a relatively new database, initially released in May, 2000. It is freely available on the internet at http://www.environment.detr.gov.uk/evslist/index.htm. This database is focused exclusively on valuation studies of environmental impacts and assets in the UK, and has been created as a step towards expanding the UK element of databases such as EVRI. Currently, the list is already very large, covering over 500 studies. However, it is not yet complete, though the DETR plan to expand it and keep it up to date.

In contrast to the EVRI database, the DETR Valuation Source List does not provide detailed information about the studies included. The list reads simply like a reference list, containing only basic bibliographic information for each study. While it may be possible to identify a range of studies that may be applicable to any given question, for example by using the search facility on Internet Explorer, this can only be the beginning point of any benefits transfer application. Relying on this source list runs the risk that relevant studies may be missed, *including those within the database*, as it is not always clear from a title what the study is about. For example, the type of resource or environmental change may be difficult to identify from the title alone. Evidently, copies of any studies identified as potentially relevant will have to be obtained and screened in the usual way, thus requiring a larger investment of time and effort to obtain the relevant information compared to, say, EVRI which provides much of the information online.

A very brief search of the database for references which might relate to biodiversity issues of relevance to this study was conducted by counting the number of times a given word appeared. The results were as follows: 'biodiversity' – 14; 'wetlands' – 17; endangered species – 2; river – 19; ecosystem – 9; forest – 256; habitat – 1; bird – 1; flora – 0; fauna – 0; coast – 27. Note that these numbers are almost certainly an overestimate of the number of studies relevant to each category, as words in journal titles, commissioning bodies, etc will also be picked up.

A.2.3.3 Envalue: database of valuation studies from the New South Wales Environmental Protection Authority

This database was developed by the NSW EPA and first released in 1995. It is freely available on the internet at http://www.epa.nsw.gov.au/envalue/. Similar to EVRI, the database was designed to facilitate benefits transfer for use in cost-benefit analyses, environmental impact statements, project appraisals and overall valuation of changes in environmental quality. The presentation of the information regarding benefits transfer is geared towards use in the Australian context.

The database covers approximately 425 studies. Table A-15 shows the geographical distribution of the studies, which are concentrated in Australia and the USA although the database also contains a number of studies from Europe.

Table A-15: Geographical distribution of valuation studies

Region	Number of studies
Australia, New Zealand, Pacific Islands	145
North America	202
Europe	64
South and Central America	5
Africa	4
Global	2
Asia	1
Total	423

The distribution of studies according to environmental asset demonstrates that the database contains a number of studies which may be relevant to the valuation of biodiversity: 125 studies relate to natural areas, while a further 74 cover water quality (see Table A-16).

Table A-16: Distribution of valuation studies according to environmental asset

Air quality	Conceptual studies	Land quality	Natural areas	Noise	Non-urban amenity	Risk of mortality	Radiation	Urban amenity	Water quality
84	32	47	125	35	4	1	6	16	74

The search facilities for the database are organised by medium, country and author, and the user may search within each of these categories individually, or using combinations of them. The site also provides quite detailed discussion of benefits transfer, both in terms of general considerations (e.g. rationale, approaches, etc) and specific considerations for different media (e.g. air versus water environments).

The information contained within the study summaries is quite detailed, similar to EVRI, and is designed for use in a benefits transfer context. For each entry, the following information is available:

- **study reference**: bibliographic information;
- **environment and locational information**: country, site, resource type, etc
- **methods and key results**: valuation methodology, estimated values (including units), functions (including both does-response functions where relevant and econometric results);
- **considerations for benefits transfer to the Australian context** for each study individually; and
- **evaluation criteria, comments and related studies**.

Overall, the database is well designed and organised, and is a source with much potential, particularly for benefits transfer in the Australian context. The coverage for the Australian context, and for the application of benefits transfer to this context, is particularly strong. However, the database is unfortunately not well maintained – the last update at the time of writing was in 1998.

A.2.3.4 ValuAsia (EEPSEA-commissioned database)

The organisation of this database is similar to that of EVRI or Envalue, designed with the purpose of use in a benefits transfer context. The author based the design on that of EVRI. The database was created in June 1999, and the last update (at time of writing) was in September 2000. ValuAsia was designed as part of an undergraduate project, with funding from the Economy and Environment Programme for South East Asia (EEPSEA). It is freely available over the internet at http://www.geocities.com/valuasia/.

The database is focused on the use of benefits transfer in developing country applications, with particular reference to South East Asia. While potentially useful in this context, the database currently covers only 20 studies, 16 of which are from South East Asian countries, and the reminder from the UK and the USA. Table A-17 presents the coverage by environmental asset.

Table A-17: ValuAsia database coverage by environmental asset

Air	Animals/plants	Health/human	Land (inc. wetlands, coastal resources)	Water
1	2	2	10	5

The information presented for each study follows a similar format to EVRI or Envalue, covering:

- **study reference**: bibliographic information;
- **environment and locational information**: country, site, resource type etc
- **methods and key results**: valuation methodology, estimated values (inc. units), functions (including both does-response functions where relevant and econometric results);
- **considerations for benefits transfer to the South East Asian context** for each study individually; and
- **comments**.

In addition, the site provides quite detailed information on how to conduct benefits transfer, in particular adjusting estimates to account for differences between countries. While this database is potentially useful for the South East Asian context in particular, it will probably need expansion in order to be of real benefit. Its usefulness in the context of valuation of European biodiversity or natural resources is limited.

A.2.3.5 New Zealand Non-Market Valuation Database

This is a database of valuation studies based at Lincoln University, Canterbury, New Zealand. It is focused entirely on studies in New Zealand, and aims to be comprehensive in this respect. It is freely available on the internet at http://learn.lincoln.ac.nz/markval/.

Currently, the database covers approximately 85 studies, searchable by date, valuation technique, environmental focus, and author. Only brief details of each study are presented in a summary table, comprising of:

- bibliographic information;
- resource valued, and valuation method;
- mean value; and
- reviews of the study.

Obvious omissions, which are necessary for effective benefits transfer, include characteristics of the population, details on the aggregation procedure, detail about the study site, etc. The database is potentially useful as a resource to uncover what valuation studies exist for given resources in New Zealand and what kind of values are produced from them. However, use of these estimates for benefits transfer will require that the original studies are obtained.

A.2.3.6 Overall Assessment

The current state of play is that none of the databases in their existing states of development provide sufficient coverage of relevant valuation studies to be used in the context of BT for the purposes of valuing damages to biodiversity in the EU. Current issues include:

- insufficient coverage of European studies;
- lack of existing studies to value the range of possible impacts to biodiversity;
- a general lack of studies valuing environmental damages rather than improvements.

Expansion of EVRI to cover more resources and more European studies should improve the potential for applications in the current context. However, in order for BT to form the basis of most valuations of damage to biodiversity, it is clear that gaps in the existing literature will need to be targeted as a priority.

A.3 SELECTION CRITERIA FOR VALUATION TECHNIQUES

There are four options for expressing biodiversity damage, which are discussed in Section A.2. Below these options are summarised in the order from the option with the least resource requirement to the option with the most resource requirement. The first option of 'scoring' is possibly the easiest option requiring the least time and resources. However, it is sufficient only for cost-effectiveness analysis (CEA), even though due to difficulties with estimating the monetary expression of damage they can also be used for cost-benefit analysis (CBA) (for further discussion on this see Annex B). The time and resource requirements of the options increase as we go down the list but so does the coverage of impacts and suitability of the option to the current context. Therefore, the choice between these options depends largely on balancing the time and resource requirements of an option with its usefulness in expressing biodiversity damage and choosing the restoration options. Some criteria that can help with this choice are presented after the options.

A.3.1 Available Valuation Techniques

1. Scoring and weighting generally uses expert judgement to identify the scale of the different aspects of biodiversity damage and the effectiveness of different restoration options in achieving the predetermined restoration targets. The scales can also be informed by the stakeholders to identify the relative importance of the aspects of use and non-use values. The stakeholder participation can be through the implementation of a choice modelling survey without the use of money as the unit of measure (see stated preference techniques below). Clearly, the scales are specific to the environmental issue, thus direct comparisons of environmental significance between different issues cannot be made. Such comparison would require weights to be assigned, based on local characteristics.

This is the most practical option since it generally only requires expert opinion to be collated based on their experience and the results of the damage assessment process. Thus, it can potentially be used for assessing all aspects of biodiversity damage.

Its applicability is limited to being an input to CEA (e.g. score per Euro of restoration cost) of primary and compensatory restoration options and the service-to-service approach for determining the scale of the compensatory restoration options. Since it cannot express biodiversity damage in monetary units, it should not be used in CBA, even though limitations in monetary damage estimates may necessitate its use in CBA.

2. Benefits transfer is the process of borrowing the WTP / WTA estimates from one study (study site) and applying it to the context in hand (policy site). The transfer usually involves some adjustment, which is generally based on the income differences between the study site and policy site.

The attraction of benefits transfer is clear: it can generate monetary expressions of biodiversity damage at a fraction of time and resource requirements of undertaking an original valuation study. All that is required for benefits transfer is a thorough literature review, which is becoming easier with the provision of on-line databases and an understanding of the adjustment processes. On the other hand, there can be significant limitations to benefits transfer. First of all, it is unlikely that there is currently sufficient literature covering all aspects of biodiversity damage. Secondly, some evidence in the literature show that using estimates from other studies can lead to results that are significantly different from those achieved if the estimates were obtained from site-specific valuation studies. This difference is larger the more site-specific the damage is and the more unique the resource is.

Since it can generate monetary expressions of biodiversity damage, benefits transfer can be an input to CBA. However, it is likely that not all impacts can be covered by benefits transfer and that the remaining has to be expressed in monetary terms using revealed or stated preference techniques or in non-monetary terms using scoring and weighting techniques. Moreover, due to the limitation mentioned above, its use is recommended to be limited to small impacts on non-unique resources. It is also useful in that it generates ballpark figures for the magnitude of the different aspects of biodiversity damage. For example, in the case of a damage to a wetland, if the valuation literature can tell us which functions of wetlands are valued more, we can make an informed judgement about the severity of the damage based on which functions are affected most. This can inform both the design of the restoration options (especially of the compensatory restoration) and the design of the monetary valuation exercise, if it is to take place. Brouwer *et al* (1999) presents just such a summary of the valuation literature about wetlands (see Section B.2.1).

3. Revealed preference techniques analyse consumers' behaviour in actual markets for goods and services, which have environmental attributes, in order to derive estimates of WTP for the environmental attribute. For hedonic property pricing, the marketed good is private property, the price of which (and hence the demand for it) is affected by environmental attributes such as landscape surrounding the property, peace and quiet etc. For travel cost and random utility models, the marketed good is the recreational site. The cost of travelling to and from a site and the money spent while at the site are indications of people's preferences for visiting that site based on its attributes, some of which are environmental. It is likely that for most biodiversity damage cases, travel cost method and random utility models will be more relevant than hedonic property pricing.

If the necessary data are available for these techniques to be implemented, the analysis takes a relatively short time. However, if such data need to be collected, it could take up to a year, which is as long as a full-scale stated preference study. In fact, in the case of hedonic pricing technique, which requires panel data (time series and cross-sectional), such data would not be possible to collect within the time frame of a damage assessment exercise. There are two other limitations to revealed preference techniques. Firstly, only those resources or habitats that are used for recreation, or possibly have an effect on property prices, can be covered. Secondly, only use value attached to a resource can be measured since the data analysed are collected from users' behaviour alone.

Since revealed preference techniques can generate monetary expressions of biodiversity damage, they can be an input to CBA. However, they are relevant for impacts on landscape and biodiversity especially when such impacts are experienced in areas used for recreation. Since a revealed preference study can only assess use value, if non-use values are thought to be affected, then revealed preference results need to be supplemented by a stated preference study. Consideration of non-use values is likely to be relevant where any of the following apply: (i) the magnitude of the damage is high, or irreversible; (ii) the damaged resource is considered to be of critical importance; or (iii) compensatory restoration with a resource of the same type, same quality and comparable value is infeasible. Where any of these criteria apply, especially in combination, it will be difficult to be reasonably sure that restoration really does provide an appropriate level of compensation without a thorough investigation of preferences, including non-use values. . Stated preference techniques such as contingent valuation or choice modelling would provide the most accurate information for this purpose.

Once a stated preference study is needed, however, it would be more cost-effective to use stated preference technique alone.

4. Stated preference techniques make use of carefully structured questionnaire to elicit people's preferences for or against an environmental change like damage to biodiversity. Surveys generate data on people's opinions and attitudes as well as their WTP/WTA. The WTP/WTA questions provide respondents with information about the change of concern (biodiversity damage), restoration options (if available), institutional context in which the restoration will take place and mechanisms through which people can make or demand a payment. Contingent valuation version of stated preference techniques concentrates on people's preferences for the whole of an environmental resources and asks WTP/WTA questions directly. Choice modelling version, on the other hand, concentrates on individual attributes of a resource and asks respondents to choose between or rank the scenarios they are presented with, where the scenarios show the different levels of different attributes of the same environmental resource. One of the attributes is usually the cost of providing the scenario, such as the cost of the restoration option.

If the whole of the environmental resource is damaged, contingent valuation is the more appropriate method, while if only some of the attributes of a resource are damaged (e.g. only some of the functions of a wetland), then choice modelling would be more appropriate. Moreover, choice modelling can be used to derive scores or weights if the choice question is asked without a monetary attribute.

Finally, both versions of the stated preference techniques have the advantage of being able to estimate non-use values associated with environmental resources, which can be significant especially if the damaged resource is unique (either locally, nationally or globally). The main drawback of stated preference techniques is the time and resource required for their implementation. Other drawbacks such as potential biases can largely be dealt by careful design (see Annex 3). While valuation estimates obtained through stated preference techniques may depend on whether the exercise is conducted *ex ante* or *ex post*, this should not be an issue of concern. In some cases, the latter is known to produce lower estimates than the former, once the damage is observed to be not as bad as expected. However, this is largely due to differences in information and knowledge at the time the surveys take place. There is no rule that this will always be the case. In the current context, since valuation studies for estimating the relevant compensation amount are likely to take place after the damage occurs. It is possible that, with poor design, survey respondents may perceive that their responses could influence the amount of compensation provided. Elimination of possible bias in the results of an *ex post* study is an issue of design. With suitable survey design and appropriate piloting, it should be possible to obtain accurate results. For example, use of a WTP question (rather than WTA) may be found to be more suitable, or indeed the use of choice modelling techniques which elicit preferences for trade-offs between resources directly without the use of monetary measures.

Since stated preference techniques can generate monetary expressions of biodiversity damage, they can be an input to CBA. In fact, given the extensive coverage of the techniques both in terms of different aspects of damage and in terms of both use and non-use values, they are the only option that can lead (at least close) to a full monetary implementation of CBA.

Before going into the discussion about the criteria for selecting between these options, it is crucial to note that any one of the above options can be only as good as the information provided by damage assessment. If damage assessment cannot provide information which are in an easily understandable format, it will be difficult for the experts to make the links between damage and the loss of economic value regardless of which of the above options is chosen.

A.3.2 Choice of Valuation Technique

The following are the criteria recommended for choosing between the above options that can generate monetary expressions of biodiversity damage. Since scores and weights are much more straightforward in their application and do not generate monetary estimates, they are not included in these criteria.

The skills and resources of the analysts undertaking the damage assessment are not included as a criteria since it is assumed that the assessment should be undertaken by an interdisciplinary team consisting at least of experts in ecology and related technical issues, (environmental) economists, lawyers and where necessary social consultation experts. Although there is a growing guidance literature about undertaking economic valuation and appraisal ranging from text books to official guidance documents, to an non-expert, they can only provide guidance in assessing the quality of the process and the results not in actual implementation.

- Likely magnitude of the damage: The more severe the magnitude of the damage to a natural resource, the more important it is that the valuation of damage is carried out thoroughly to ensure full compensation. Moreover, in the case of severe damage the required assessment is likely to be more complex, and it may well be the case that non-use values are affected. Benefits transfer techniques are unlikely to be appropriate in this case, as they are unlikely to be able to provide estimates which capture values for the full range of impacts, nor provide sufficient information regarding the relevant population who might hold non-use values for the damaged resource. Original studies, in particular stated preference techniques, are therefore likely to be the most appropriate.
- Critical importance of the environmental resource impacted, the significance of the impact and the type of value to be measured: the more important the resource and the more significant the impact, the greater the need for as comprehensive an analysis as possible. Benefits transfer is limited by the limited coverage of all aspects of biodiversity damage in the literature and its insensitivity to site-specific characteristics. Revealed preference techniques are limited to the use values associated with those resources that are reflected in actual markets. The scope of stated preference techniques is the largest in this context given that potentially all impacts can be covered and both use and non-use values can be estimated.
- Feasibility of compensatory restoration with resources of the same type, same quality and of comparable value: The less similar and the more distant the resources identified for compensatory restoration, the harder it will be to be reasonably sure that restoration really does provide an appropriate level of compensation without conducting valuation. This criterion should be considered in conjunction with the two criteria above, namely the magnitude of damage and the critical importance of the damaged resource. Where the magnitude of damage is relatively minor, and the resource is not of critical importance, benefits transfer techniques may provide acceptable estimates of the relative values of the damaged and replacement resources. On the other hand, where damage is relatively severe and the resource concerned is unique or of critical importance, there may be a strong case for a thorough investigation of preferences to provide some assurance that the scale of restoration is appropriate to provide full compensation. Stated preference techniques such as contingent valuation or choice modelling are likely to provide the most accurate information for this purpose.
- Applicability: the purposes for which the above options are implemented for determine which option should be chosen. If service-to-service approach is sufficient for the design of the compensatory restoration option and CEA is sufficient for the choice of the restoration option, then scoring / weighting systems are sufficient. As discussed above, choice modelling techniques can be used to elicit the public's view of the relative importance of the different aspects of damage and restoration. However, these do not need to include monetary

expressions of damage. If the value of damage needs to be measured for monetary compensation or value-to-value scaling of compensatory measures, or CBA is needed to choose the restoration option, then monetary expressions of biodiversity damage (or benefit of restoration) would be necessary. If there is sufficient literature available, benefits transfer could be used for small damages and interim losses. If not, original valuation exercise would be required.

- Time and data available for analysis: availability of data about the physical measure of environmental impacts is a concern for all valuation options. The availability of economic valuation data is typically not a concern for stated preference studies since the necessary data are generated by the study itself. However, for this reason, stated preference studies can take between six months to a year depending on the complexity of the resource and damage to it. Therefore, the decision to implement a stated preference study should be put in practice at the beginning of the assessment process to inform the damage and impact assessment. In the USA, litigation often is settled before stated preference applications are completed. Typically, damage assessment and compensation negotiations proceed on parallel tracks, so that information generated during the damage assessment may play a role in negotiations, and settlements may be reached before the damage assessment is completed. In the Exxon Valdez and Blackbird Mine cases, for example, settlements were reached after completion of stated preference pilot studies.

  Provided that the necessary data are available, revealed preference studies can take about six months to implement. But if such data do not exist, a revealed preference study can take as long as a stated preference one. Data for travel cost method and random utility models can be collected using surveys. However, since such surveys are likely to take similar time and effort as stated preference surveys, it would be more efficient to design a questionnaire that would enable the application of stated preference and travel cost and/or random utility models simultaneously. On the other hand, in the absence of data, hedonic pricing is unlikely to be appropriate since the necessary (especially time series) data cannot be collected within the time scale of a damage assessment. Benefits transfer can be applied in a matter of weeks provided that the appropriate literature exists.

- The cost of the valuation exercise depends on the complexity of the damage and restoration options which affect the complexity of the questionnaire design, the size of the sample and the complexity of the data analysis. Valuation of damage is relatively inexpensive for small injuries. Various versions of benefits transfer can successfully be used for small incidents. Benefits transfer is relatively inexpensive given that there is little need for data collection and significant less time input requirement. Stated preference studies can cost between £50,000 and £200,000 (although some studies have cost millions of dollars in the USA[5]), while a comprehensive revealed preference study could also cost towards the higher end of this range if necessary data are not readily available and towards the lower end of this range (or may be even lower) if such data are ready. However, the crucial criterion is not the absolute cost of a valuation exercise but its incremental cost in terms of additional information it provides and the increased accuracy and reliability of the results produced at the end of the assessment process. The cost of the valuation exercise should also be seen in the light of the cost of the total restoration activity. Even the most expensive option of a stated preference study is a small amount relative to the total cost of restoration option that can cost millions of pounds. It is usually more cost-effective to spend more at the beginning of an assessment process than to spend to mitigate the outcome of a wrong decision afterwards.

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[5] The 'Montrose' (Southern California Bight) contingent valuation studies for damage assessment cost about US$8 million. However, the case was recently settled out of court for US$160 million.

- Whether the results of a valuation exercise are legally defensible: this depends on how strongly a valuation approach is grounded in theory and how well it is implemented in the particular study of concern. For example, benefits transfer is largely based on assumptions about the suitability of WTP/WTA estimate borrowed. Further, assumptions need to be made regarding the boundaries of the population affected. Such assumptions can be easy to contest and difficult to defend.[6] Again, scoring and weighting techniques are generally based on expert opinion, which can be contested. For stated preference studies, approaches such as pairwise comparisons etc. that are not founded in economic theory, should be avoided. Moreover, the design specifications and assumptions behind econometric analysis of the responses to the stated preference surveys should be clearly stated and defensible.
- Timing of the valuation exercise: it can be difficult for a stated preference study to elicit people's preferences for the environmental impacts of concern and distinguish these from strategic choices once the incident causing damage becomes a political issue. However, this is an empirical issue and one which can be dealt with adjusting the questionnaire design. For example, the contingent valuation study undertaken for the Exxon Valdez oil spill asks the respondents how much they are willing to pay to avoid a similar incident in the future rather than asking for WTP for restoration or WTA for damage (Carson *et al*, 1992). This puts the valuation question in a more neutral setting. What is most important is to ensure that the respondents are given incentives to answer truthfully. This can be achieved if people care about the damage and believe that their responses will be taken care of (for more discussion on this, see Annex C, Section C.2).

Although whether to use WTP or WTA is also an important decision, this is not strictly relevant for the choice between different valuation options. The only guidance available about this choice is that the choice depends on the property rights: if the people affected have a right to an action, then they should be given the chance to state their WTA to forego this action. However, there are also issues of credibility such as respondents may not believe such a compensation will ever be paid out. The distinction between WTP and WTA estimates is discussed in detail in Annex 3, Section C2.1.

Table A-18 sets out the practical considerations in terms of time and cost associated with the implementation of different valuation techniques. However, it should be emphasised that such criteria should not be the determining factors in selection of an appropriate valuation technique. The trade-offs from lower time and cost components associated with the use of benefits transfer techniques are those of accuracy, reliability and defensibility of the results. The most important considerations are those outlined in the first three bullet points above, namely: the likely magnitude of the damage, the critical importance of the environmental resource impacted, and the feasibility of compensatory restoration with resources of the same type, same quality and of comparable value. Where damage is high, the resource is critical, and/or compensatory options provide different type and quality of resources, there is a very strong case for conducting an appropriate investigation of preferences through an original stated preference study.

[6] The Axford case in the UK, for example, was successfully challenged on the basis of the definition of the relevant population assumed to hold non-use values for the resource. It should be noted that the existence and magnitude of per-person non-use values were not an issue in this case, and this case alone by no means suggests that non-use values estimated through and original stated preference study are less able to withstand legal challenge than estimates of use values.

Table A-18: Time and cost considerations associated with different valuation techniques

Methodology	Time taken for study	Possible costs
Stated preference techniques	▪ a thorough original study is likely to take between 6 months to a year, perhaps longer ▪ pilot results, giving a preliminary estimate of value would be completed earlier	▪ stated preference studies can cost between £50,000 and £200,000, depending on the complexity required ▪ studies in the USA concerned with major damages have averaged 3% of total known incident costs, although a select few have been extremely costly
Revealed preference techniques	▪ if necessary data are available, a study may take about 6 months to implement (in the absence of necessary data, implementation of this technique is unlikely to be feasible)	▪ a comprehensive study could cost towards the upper end of the range provided for stated preference studies if necessary data are not readily available, and towards the lower end of the range (or lower) if such data are available
Benefits transfer	▪ can be applied in a matter of weeks provided that the appropriate literature exists	▪ relatively inexpensive, given that there is no need for data collection and significantly less time requirement

The following criteria can be taken into account for the choice between different stated preference techniques:

- The main point is to choose CV when we need WTP for the environmental good or service in total and CM when we need WTP for individual attributes. CM is also useful if we want to know about relative values for different attributes of an environmental good.
- CM is still young in the environmental theory and tests of validity need to be practised far more before we can be confident about their implementation.
- Some CM techniques are not consistent with underlying welfare theory. If welfare-consistent estimates are needed, then choice experiments (or, to a degree, contingent ranking) are the preferred option, relative to say contingent rating.
- Questions such as ‘what are you willing to pay?’ are thought by some critics of CV to present cognitive problems. CM does not explicitly ask about pounds so it is argued that CM is easier for people to understand.
- Finally, CM offers a more "efficient" means of sampling than CV, since we typically obtain more responses from each individual with CM than with CV.

ANNEX B: CHOOSING RESTORATION OPTIONS: COST-BENEFIT ANALYSIS, COST-EFFECTIVENESS ANALYSIS AND MULTI-CRITERIA ANALYSIS

B.1 OVERVIEW AND ISSUES THAT APPLY TO ALL ANALYSES

Chapters 4 and 5 of the main report outline cost-effectiveness analysis and cost-benefit analysis as the types of analysis that can be used for choosing (both primary and compensatory) restoration options.

Cost-effectiveness analysis (CEA), also known as the least-cost analysis, can be used for two purposes: (1) minimise the cost of restoration while reaching the restoration target and (2) maximise the benefit of restoration for a given restoration budget. Given that the main objective in the current context is to ensure that the restoration target is met, the first purpose is more relevant. CEA can assist in identifying the least cost way of achieving a restoration target but cannot answer the question whether the restoration target is desirable from the society's point of view.

Cost-benefit analysis (CBA) is a more encompassing framework. By comparing both the costs and benefits of different restoration options, it can assist with determining whether a restoration target is desirable as well as choosing the best restoration option. The direct comparison of costs and benefits requires both identities to be expressed in the same unit, i.e. money. However, this may not always be possible. Thus, CBA framework is usually expanded to take both monetary and non-monetary expressions of costs and benefits into account. Multi-criteria analysis decision analysis (MCDA) is developed to compare costs and benefits when they are expressed in different units. This annex covers MCDA as well as CEA and CBA.

This section discusses a number of issues, which apply to all of the above analyses, namely, the definition of costs and benefits, definition of the 'affected' population (also relevant to the valuation techniques discussed in Annex 1) and discounting. Section B.2 presents guidance on how to implement CEA, while CBA framework is discussed in Section B.3. Some relevant aspects of MCDA are discussed in Section B.4. Risk and uncertainty associated with costs and benefits and how to deal with these are discussed in Section B.5. A set of criteria for selecting the relevant analyses and information requirements for each is given in Section B.6. The Annex concludes with an illustrative example in Section B.7.

B.1.1 Definition of benefits

A benefit is defined as any addition to human wellbeing. Wellbeing sounds vague but it has a strong linkage to what people want, i.e. to their preferences. Essentially, if an individual prefers to be in situation A rather than situation B, he or she can be said to have a higher wellbeing in A than in B. In the context of restoration of biodiversity damage, benefits of a restoration option is equal to the 'avoided' damage to biodiversity (both initial damage and interim losses). Just as damage is assessed in relation to the baseline condition of the damaged resource, benefits should be measured in relation to the same baseline. This is referred to as 'with/without principle' of cost-benefit analysis.

Economic approach is based on the assumption that individuals are the best judges of their preferences and uses the valuation techniques presented in Annex 1 to estimate the value placed on the (damage to) resources in monetary terms[7]. For this value to be estimated, however, the link between the damaged resource, the services it provides and the value placed on those services and hence the resource needs to be established (see Chapter 5 for further discussion on

[7] Note that this involves estimating the total economic value (both use and non-use values) of the damage.

this). Given this resource – service – value link, benefits can be expressed in either resource, service or value units. Note that the first two are generally based on expert judgement[8], while the latter is held by individuals but elicited by experts.

For example, suppose that 100 ha of a wetland is damaged due to an oil spill and a restoration option is devised to recreate the same area of wetland in an adjacent site. Benefits of this restoration measure (which is equal to the cost of damage) can be expressed in *resource* units such as '100 ha of wetland recreated'. Benefits can also be expressed in *service* units. Suppose that the wetland was used for recreation and prior to damage there were 120,000 visits per year. The unit for benefits can be '120,000 visits to the affected wetland'. Finally (and, strictly, what is preferred for the economic approach), benefits can be expressed in *value* units, i.e. value of the services provided by the damaged resource expressed in monetary units. Supposing that a visit to the damage wetland was worth Euro 10, the value of damage (and benefit of restoration) would be Euro 1,200,000 per year. The value of recreational uses of the resource can be estimated using benefits transfer if the relevant literature is available or by travel cost or stated preference techniques if an original valuation study is to be conducted (see Annex 1 for details). Note that the only unit in which non-use values (see Annex 1) can be expressed is money. Also note that, in CEA, where the restoration target is predetermined, baseline or damage is assessed in order to set the restoration target but benefits of restoration do not need to be measured.

B.1.2 Definition of costs

A decrease in human wellbeing is defined as a cost. In the current context, there are two aspects to cost: (1) cost of damage to biodiversity and (2) cost of restoring this damage. Cost of damage to biodiversity is dealt with in the section B.1.1 within the definition of benefits since cost of damage avoided by restoration is assumed to be equal to the benefit of that restoration. This section deals with the second aspect: the cost of (both primary and compensatory) restoration.

Costs of restoration are expressed in monetary units and include items such as cost of undertaking the damage assessment and preparing restoration and monitoring strategy; costs of species population restoration, habitat restoration and cleaning; cost of implementing the monitoring and surveillance strategy.

It is important to note the distinctions between how a conventional financial analysis and how CBA and CEA conducted from the point of view of the society as a whole differ in the way costs are defined and measured. The latter uses a concept called *opportunity cost*. Given the limited availability of financial (and indeed all other) resources, funds that are used for one purpose, biodiversity damage restoration in this case, cannot be used for other purposes. Therefore, decisions whether or not undertake restoration should include considerations of not only benefit of restoration but also what else could have been done with, or the opportunity cost of, the funds spent for restoration.

It is usually assumed that market prices used to measure the cost items reflect the opportunity cost. To ensure this is the case, sometimes it is necessary to undertake *shadow pricing* or to use 'real' prices. At its simplest, this involves taking account of taxes and subsidies that maybe included in the market prices of goods and services used for restoration. Taxes and subsidies are known as 'transfer payments' since they are transferred from one group to another (from the consumers to the government in the cases of taxes and from the government to the recipients in the case of subsidies) but have no net effect on the society's resources. Thus, shadow pricing involves using market prices net of any taxes and subsidies. This is valid both for CBA and CEA.

[8] Experts assume that if x amount of resource is damaged, restoring x amount of the same resource under the same conditions as the baseline, will generate the same service and hence the value and will therefore be beneficial.

B.1.3 Whose benefits and costs should count?

The answer is all those who experience gains or losses of wellbeing because of the initial damage to biodiversity, interim losses and the restoration option. Initial damage to biodiversity and interim losses generally affect the same population. Primary restoration options are usually designed so that the same population affected from the initial damage to biodiversity benefits from restoration. If compensatory restoration option takes place on the damaged site, then as for primary restoration, the same population that suffers from interim losses benefits from compensatory restoration. If, on the other hand, compensatory restoration takes place on another site, the beneficiaries would be the population affected by the compensatory restoration where it takes place.

For example, if a new wetland is to be created as a compensatory restoration option, the benefits of this option would be determined by not only the restoration of the damaged wetland but also the benefits of the new wetland to the population surrounding this new site.

Within the affected population, there is likely to be both ‘users’ and ‘non-users’ which would have different preferences and hence need to be identified separately. The ‘user population’ is relatively easy to identify. For example, the households living in and/or around the damaged area can be said to be direct and/or indirect users and can easily be identified. If the site is used for recreation, the visitors are also included in the direct user population. Such visitor data could exist or could be collected either by simply counting the population or sampling and extrapolation. Some resources may provide services to populations other than those who live in and/or around the damaged area those who visit the damaged area. For example, some sites support migratory bird populations. If damage to such sites affect the migratory birds, the ‘user’ of such sites should also include those who enjoy these birds, such as birdwatchers, at other sites visited by the migratory birds. In other words, user populations are defined not by geographical boundaries of the damaged area but the geographical distribution of the services provided by the damaged resource. Again these ‘distant users’ can be identified either by existing data on visitors or residents of these other sites or by collecting data on visitor and resident numbers.

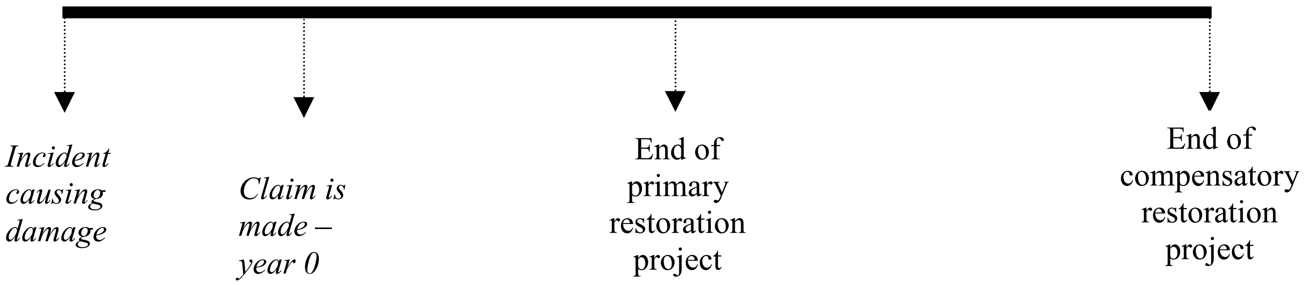
Determining the population holding ‘non-use’ values is not as straightforward. There is no clear-cut rule to predict the existence or absence of non-use values. The resource could be nationally unique, in which case, the relevant population is likely to be the whole nation, or locally unique or important, in which case the relevant population is the local one. In some cases, the resource could be of global importance such as a UNESCO recognised World Heritage Site, in which case whether the global population would be affected would be hard to predict in advance.

Many studies assume that if non-use value exists, it must exist for <u>all</u> non-users. This is the rationale for extrapolating mean non-use values to a whole population, usually the national population. It is more correct to sample the non-user population by geographical location. For example, for a site A, the presence of non-use value would be tested by sampling the population, say, 100 km from A, 200 km, and so on. One hypothesis is that non-use values will decline with distance – a ‘distance decay’ notion. This hypothesis is borne out in a few studies, but others have found no effect of distance on non-use value. Accordingly, there are no *a priori* rules for determining the aggregation procedure. Geographical sampling must take place. This requirement imposes a limitation on benefits transfer unless studies that report valuation results for populations in different distance bands (or other sub-samples) exist.

B.1.4 Treatment of costs and benefits occurring over time

In most circumstances, the various elements of a natural resource damage claim will be spread over different time periods. Choosing restoration options requires aggregation of all these various costs and benefits for comparison. However, it is generally not possible to simply add benefits and costs as they accrue over time; assessments must account for the timing of damage and restored benefits. Figure B.1 shows a time-line of a damage incident and recovery period.

Figure B-1: Damage and restoration over time



Therefore, we need a procedure which weights benefits and costs that occur in different years in this time-line so that they may be compared. In general, present gains or losses are weighted more heavily than future gains or losses for two reasons. The first reason is the productivity of capital (or opportunity cost of capital) argument: one unit of currency which is invested today will be worth more next year due to the interest on capital. Technically, if *r* is the interest rate, Euro 1 received this year has the same value as Euro 1+r received next year. This is referred to as *compounding*. The opposite is also true: the value of Euro 1 next year is Euro 1/(1+r) now, or in other words, the present value of Euro 1 next year is Euro 1/(1+r). This is referred to as *discounting*.

The second argument for estimating present values for costs and benefits is individual's time preference: we could simply observe the behaviour of individuals and conclude that, regardless of interest rates, people do prefer their benefits now rather than later. If what matters are individuals' preferences, their preference about the incidence of costs and benefits through time cannot logically be excluded. In turn, this means that future benefits and costs must be 'discounted' to be expressed in 'present value' terms.

An illustrative example in Section B.7 is provided to clarify how present values might be calculated in practice.

The main debate about treating costs and benefits occurring over time concerns the discount (and compound) rate. Conventionally, the same rate is used for costs and benefits. It could be argued that since we are measuring the costs and benefits as they accrue to the society as a whole, the time preference of society is what counts rather than the time preference of an individual. As the lifetime of an individual is much shorter than the lifetime of a society, it can be expected that individuals will be more impatient than society as a whole. Estimating the rate of time preference for the society as a whole is not straightforward. Pearce and Ulph (1999) show that the rate at which future costs and benefits are discounted is about 2.5-3% in the UK. Freeman (1993) derives a similar estimate of 2-3% for the USA. Several authors have found that individuals have lower discount rates for environmental goods than for market goods.

Guidance on appropriate social discount rates differs between government institutions, and is generally higher than these estimated social discount rates. Currently, the European Member States use a range of discount rates, ranging from 3% to 8%, while the European Commission employs a rate of 4%. While choice of the discount rate to be used in analysis is ultimately a political decision, for consistency in the implementation of the legislation across the EU it may be desirable to ensure the rate chosen is consistent across Member States. The effect of the chosen rate on the final results may be tested through sensitivity analysis.

Guidance on assessment of natural resource damage claims in the USA, however, recommends the use of different discount rates in the consideration of costs and benefits (DARP, 1999). In the assessment of benefits, they recommend use of a social discount rate of 2-3%, as above. However, the US Treasury rate (6%), which reflects the opportunity cost of capital, is recommended for use in the assessment of costs. The rationale for this is that restoration and assessment costs pose public sector capital budgeting problems, and the Treasury rate is perceived as most appropriate for these purposes.

B.2 COST EFFECTIVENESS ANALYSIS

The cost-effectiveness analysis (CEA) is a truncated form of cost-benefit analysis. It draws inspiration and guidance only from the cost side – or alternatively, only from the benefit side – of cost-benefit analysis. As mentioned above, the cost side is the appropriate use of CEA in the current context. Thus, to implement CEA, we need to undertake the following tasks:

- Damage assessment and significance analysis in order to identify the baseline and the restoration target (see Chapter 3);
- Determine primary restoration options, and identify and measure the costs of primary restoration options (see Chapter 4);
- Determine compensatory restoration options, and identify and measure the costs compensatory restoration options (see Chapter 5); and
- Calculate the present value of the costs of primary and/or compensatory restoration (see Section B.1.4).

CEA compares the primary or compensatory restoration options that are identified as technically capable of meeting the restoration target in terms of their costs. Costs should be expressed in present value terms as shown above in Section B.1.4. The option that achieves the restoration target at the lowest (present value of) cost is chosen as the optimum option according to CEA.

As noted above, CEA is sufficient when there is an agreement on the restoration target. It is also sufficient so long as the cost of the chosen (primary or compensatory) restoration option is not deemed to be 'excessive'. If the cost is deemed to be excessive, the only way to test whether this is the case is to compare the cost of the option with its benefit, in other words, cost-benefit analysis, which is presented in the next section.

B.3 COST BENEFIT ANALYSIS

Cost-benefit analysis (CBA) is a framework for comparing the present value of the costs and the present value of the benefits of an action. In the current context, CBA is needed only when the cost of the chosen primary and/or compensatory restoration option is deemed to be 'excessive'. CBA builds on the tasks of cost-effectiveness analysis. Thus, the following steps need to be taken to implement CBA:

- Damage assessment and significance analysis in order to identify the baseline and the restoration target (see Chapter 3);
- Determine primary restoration options, and identify and measure the costs of primary restoration options (see Chapter 4);
- Determine compensatory restoration options, and identify and measure the costs compensatory restoration options (see Chapter 5);
- Calculate the present value of the costs of (primary and/or compensatory) restoration options (see Section B.1.4); and
- Calculate the present value of the benefits of (primary and/or compensatory) restoration options (see Annex 1 for techniques to identify and estimate the benefits and Section B.1.4 to calculate the present value).

CBA uses two decision-making criteria: net present value (NPV) and benefit-cost ratio (BCR). NPV is calculated as follows:

$$NPV = PV(B) - PV(C) = \sum_{t=t_I}^{T} (B_t - C_t) \times d_t$$

Where NPV is net present value, PV(B) is the present value of benefits, PV(C) is the present value of costs, B_t is the benefit which occurs in time period t, C_t is the cost which occurs in time period t, t_I is the period when the incident causing the damage occurs, T is the last period in the assessment, and d_t is the weight used to convert the past and future costs and benefits to present values.

NPV is useful for determining whether a given restoration project should go ahead:

- Negative NPV implies that the present value of the costs of a restoration option is *greater* than the present value of its benefit. On the basis of NPV alone, such options should be rejected.
- Zero NPV implies that the present value of the costs of a restoration option is *equal* to the present value of its benefits. Conventionally, if this is the result, CBA would be indifferent whether the option should be rejected or accepted since the main purpose of CBA is to identify options that generate net increase in social wellbeing (positive NPV, see below). However, in the current context, the main purpose of CBA is to demonstrate that meeting the restoration target does not entail 'excessive' costs. Thus, so long as the present value of costs is equal to the present value of benefits, it cannot be argued that the costs are excessive. This is why, the guidance in the USA (DARP, 1999) recommend that the options with zero NPV should be accepted.
- Positive NPV implies that the present value of the costs of a restoration option is *less* than the present value of its benefits. On the basis of NPV alone, such options should be accepted.

Note that if there is no option with zero NPV and that there are more than one options with positive NPV, the option with the highest positive NPV should be chosen. If there are more than one option with zero or positive NPV, either the option with the lowest cost or the option that generates the highest benefit (the highest positive NPV) should be chosen. The choice depends on whether the purpose is just to demonstrate that the restoration target is met and the cost is not excessive (zero NPV) or to generate net benefit (the highest positive NPV).

The benefit-cost ratio is another way of comparing the discounted (present) value of costs and benefits:

$$Benefit / Cost = \frac{PV(B_t)}{PV(C_t)}$$

This ratio enables a comparison of options on the basis of 'value per Euro spent'. Therefore, it should be used if the restoration budget is predetermined and fixed. The decision rules are the same as those for NPV:

- Benefit-cost ratio of less than one implies that the present value of the costs of a restoration option is *greater* than the present value of its benefits. On the basis of this ratio alone, such options should be rejected.
- Benefit-cost ratio of exactly one implies that the present value of the costs of a restoration option is *equal* to the present value of its benefits. Conventionally, if this is the result, CBA would be indifferent whether the option should be rejected or accepted since the main purpose of CBA is to identify options that generate net increase in social wellbeing (ratio greater than one, see below). However, as for NPV, in the current context, the main purpose of CBA is to demonstrate that meeting the restoration target does not entail 'excessive' costs.

Thus, so long as the present value of costs is equal to the present value of benefits, it cannot be argued that the costs are excessive.

- Benefit-cost ratio of greater than one implies that the present value of the costs of a restoration option is *less* than the present value of its benefits. On the basis of this ratio alone, such options should be accepted.

In the case of fixed restoration budgets, maximum benefits are obtained by first implementing the option with the highest benefit-cost ratio, then the second highest and so on. Once the restoration budget limit is reached the remaining projects cannot be implemented even if they have benefit-cost ratios greater than unity.

As discussed above, costs of restoration are expressed in monetary units. Therefore, the benefits should also be expressed in monetary units. This allows for the comparison of like with like and CBA can directly suggest options for implementation.

However, due to data, time and resource restrictions, this may not be possible and hence benefits may be partially expressed in monetary units and partially in non-monetary units. Although conventionally not incorporated in CBA, non-monetary expressions of benefits can still be incorporated in the CBA framework. Note that if this is the case the results of the monetary portion of CBA alone cannot directly suggest options for implementation. Professional judgement would be required to choose the option.

In this case, the procedure should be to compare all monetised costs and benefits as above, and list the non-monetised effects. Note that a non-monetised negative indicator constitutes a cost and a non-monetised positive indicator constitutes a benefit. There are then four ways of dealing with mixed outcomes:

1. If monetised benefits exceed monetised costs and the non-monetised indicators are judged mainly to be positive, then proceed since benefits more than outweigh the costs.
2. If monetised benefits exceed monetised costs and the non-monetised indicators are judged mainly to be negative, then compare net monetised benefits with the non-monetised costs. *Using professional judgement, ask if the non-monetised costs are likely to be greater than the net monetised benefits. If they are, the option is not worthwhile. If they are not, then the option is potentially worth pursuing*.
3. If monetised costs exceed monetised benefits and the non-monetised indicators are judged mainly to be positive, then compare net monetised costs with the non-monetised benefits. *Using professional judgement, ask if the non-monetised benefits are likely to be greater than the net monetised costs. If they are, the option is potentially worth pursuing. If they are not, then the option is not worthwhile*.
4. If monetised costs exceed monetised benefits and the non-monetised indicators are judged mainly to be negative, then the scheme is not worth pursuing.

Table B-1 summarises these four possible outcomes. Further discussion on how to treat non-monetary assessment of costs and benefits is presented in the next section.

Table B-1: The treatment of mixed (monetary and non-monetary) outcomes

	$B_m > C_m$	$B_m < C_m$
$B_{nm}>0$	1. Proceed since benefits more than outweigh costs	3. Judge if $B_{nm} > [C_m - B_m]$ *If so, proceed.* Judge if $B_{nm} < [C_m - B_m]$ *If so, reject.*
$C_{nm}>0$	2. Judge if $[B_m\text{-}C_m] > C_{nm}$ *If so, proceed.* Judge if $[B_m\text{-}C_m] < C_{nm}$ *If so, reject.*	4. Reject since costs more than outweigh benefits

Note: m denotes monetary estimates and nm denotes non-monetary indicators. All monetary measures should be considered in NPV terms.

Note that if the reason CBA is implemented is a suggestion that the restoration costs are 'excessive', using non-monetary expressions of benefits are unlikely to be sufficient to test whether this suggestion is justified. The only way to test this suggestion, as discussed above, is to express the benefits in the same units as costs, i.e. money.

B.4 MULTI-CRITERIA DECISION ANALYSIS

A form of multi-criteria analysis that has found many applications in both public and private sector is multi-criteria decision analysis (MCDA). MCDA is both an approach and a set of techniques, with the goal of providing an overall ordering of options, from the most preferred to the least preferred option. MCDA is a ways of looking at complex problems that are characterised by any mixture of monetary and non-monetary objectives, of breaking the problem into more manageable pieces to allow data and judgements to be brought to bear on the pieces, and then of reassembling the pieces to present a coherent overall picture to decision-makers. In doing this, the steps of MCDA are similar to those presented in this report for damage assessment framework, CEA and CBA:

- Establish the decision context: establish the aims of the decision analysis, and identify decision makers and other key players; design a framework for conducting MCDA and consider the context of the appraisal;
- Identify the options to be appraised;
- Identify objectives and criteria: identify criteria for assessing the consequences of each option and organise the criteria by clustering them under high-level and lower-level objectives in a hierarchy;
- Scoring: assess the expected performance of each option against the criteria. Then assess the value associated with the consequences of each option for each criterion. In other words, describe the consequences of the options; score the options on the criteria and check the consistency of the scores on each criterion. Note that scores can be based on WTP and/or WTA as well as expert opinion;
- Weighting: assign weights for each of the criterion to reflect their relative importance to the decision.

- Combine the weights and scores for each option to derive an overall value: calculate overall weighted scores at each level in the hierarchy of objectives and calculate overall weighted scores.
- Examine the results
- Sensitivity analysis: conduct a sensitivity analysis (see Section B.5); look at the advantages and disadvantages of selected options, and compare pairs of options; create possible new options that might be better than those originally considered; and repeat the above steps.

This section discusses the weighting element of MCDA as the other steps are covered elsewhere under CEA and CBA in this annex and scoring in Annex1. This discussion borrows from the multi-criteria analysis manual published by the UK DETR (available at http://www.environment.detr.gov.uk/multicriteria).

Most proponents of MCDA use the method of 'swing weighting' to elicit weights (say, from 0 to 100) for scores, where scores measure the performance of a given option against the objective of the option, i.e. performance of each restoration option against the restoration target, in the current context. Swing weighting is based on comparisons of differences between the options appraised. To make these comparisons, assessors are encouraged to take into account both the difference between the least and most preferred options, and how much they care about the difference. For example, in choosing a restoration option, cost might be considered to be important in some absolute sense. However, in making the choice of a particular option, there may already be a shortlist of, say, five options. If they only differ in cost by Euro 10,000, the cost may no longer be an important criterion for consideration. That criterion would receive a low weight because the difference between the highest and the lowest cost restoration option is so small. If the cost difference was Euro 1,000,000, the cost criterion may be given more weight - unless there is no budget constraint.

There is a crucial difference between measured performance (score) and the value of that performance (weight) in a specific context. Improvements in performance may be real but not necessarily useful or much valued: an increment of additional performance may not contribute a corresponding increment in added value.

Thus, the weight on a criterion for choosing one option over another reflects both the range of difference of the options, and how much that difference matters. So it may well happen that a criterion which is widely seen as 'very important', such as integrity of the ecosystem, will have a similar or lower weight than another relatively lower priority criterion, such as the cost. This would happen if all the options had much the same level of restoring the integrity of the ecosystem but varied widely in the cost of achieving this target. Any numbers can be used for the weights so long as their ratios consistently represent the ratios of the valuation of the differences in preferences between the top and bottom cores of the scales which are being weighted.

The process of deriving weights is thus fundamental to the effectiveness of an MCDA. Often they will be derived from the views of a group of people. They might reflect a face-to-face meeting of key stakeholders or people able to articulate those stakeholders' views, in which weights are derived individually, then compared, with an opportunity for reflection and change, followed by broad consensus. If there is not a consensus, then it might be best to take two or more sets of weights forward in parallel, for agreement on choice of options can sometimes be agreed even without agreement on weights. Even if this does not lead easily to agreement, explicitly awareness of the different weight sets and their consequences can facilitate the further search for acceptable compromise.

Note that in monetary expressions of costs and benefits, scores are the quantitative expressions of performance and weights are WTP and/or WTA which reflect the preferences of the affected population estimated through economic valuation techniques rather than discussions between stakeholders. Therefore, although the methods used for estimating weights are different between CBA and MCDA, the aim is the same.

Once weights are allocated to the scores, calculating the overall weighted scores is straightforward:

$$S_i = w_1 s_1 + w_2 s_2 + ... + w_n s_n = \sum_{i=1}^{n} w_i s_i$$

Where S_i is the overall weighted score for each restoration options, $w_{1...n}$ is the weight for each element of the restoration target and $s_{1...n}$ is the score of performance of each option against the restoration target. In other words, multiply an option's score against each element of the target by the importance weight of that element of the target, then sum the products to give the overall preference score for that option. Then repeat the process for the remaining options. If, for example, the restoration target is the restoration of damage to a wetland, elements of the target will be type and quantity of plant species, water quality, fish stocks etc.

There are two multi-criteria assessment models that are worth mentioning in more detail here: the Hessian Compensation Model (Hessian *Biotopwertverfahren)*, and the Andalusian compensation table. These models are used in two EU Member States for the purpose of determining the quantum of compensation to be paid for natural resource damage, and are specifically mentioned in the EU White Paper on environmental liability as potentially useful models for the valuing natural resource damage, and for determining at what point restoration cost become unreasonable.[9] The following paragraphs discuss the two above models.

B.4.1 The Hessian Compensation Model

In Hesse, Germany, a model has been developed to estimate the amount of money to be paid for interventions in nature and landscape that cannot be restored and for which replacement is not an option. The model, *Bewertungsverfahren nach Hessischen Ausgleichabgabenverordnung* or *Biotopwertverfahren*, is applied primarily in *ex ante* situations.[10] It is used to assess a compensatory fee if due to a planned intervention in nature, such as the building of railroads or houses in protected nature areas, there is a loss of nature and restoration measures cannot be taken or will not fully compensate the loss.[11] The aim of the model is to provide an instrument that makes it possible to assess damages on the basis of 'objective' criteria.

The Hessian compensation model is based on a classification of the Hessian territory into various different types of biotopes, thereby considering the environmental functions these biotopes provide for nature and people (*e.g.* aesthetic functions). Eleven main categories of biotopes are distinguished, including forests, grasslands, moors and heathlands, and poorly vegetated areas. These categories are further divided into about 180 different biotopes.[12] These biotopes are evaluated in the model on the basis of the eight different characteristics, which reflect the environmental value of the biotopes.

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[9] White Paper, para. 4.5.1.

[10] For a detailed description of the model, see: *Hintergründe zur Entwicklung des Hessischen Biotopwertverfahrens* (http://home.t-online.de/home/Klaus-Ulrich.Battefeld).

[11] Art. 6(b) Hessian Nature Conservation Act.

[12] For an overview, see: *Wertliste nach Nutzungs-/Biotypen*, Anlage 2 Ausgleichabgabeverortnung (http://www.mulf.hessen.de/umwelt/naturschultz/eingriffe.eingriffe.htm#ausgleich).

The following characteristics are distinguished:

1. Quality of the biotope [*Entwicklungsgrad des Biotoptyps*]
2. Naturalness of the biotope [*Natürlichkeit des Biotoptyps*]
3. Diversity of biotope structures [*Strukturvielfalt des Biotoptyps*]
4. Diversity of biotope species [*Artenvielfalt des Biotoptyps*]
5. Rareness of the biotope [*Seltenheit des Biotoptyps*]
6. Rareness of animal and plant species depending or occurring on the biotope [*Seltenheit der auf dem Biotoptyp üblicherweise vorkommenden Pflanzen- und Tierarten*]
7. Vulnerability of the biotope [*Empfindlichkeit des Biotoptyps*]
8. Developments/trends with regard to the quality and number of biotopes concerned [*Häufigkeit des Typs nimmt tendenziell ab, Beeinträchtigungen des Types nehmen tendenziell zu*)

Each of the above variables is awarded a point value (between one to six points). The environmental value (*Biotopwertes*) of the more than 180 biotopes covered by the model, is determined by adding up the first four characteristics and multiplying this figure by the sum of the remaining characteristics. This figure is divided by the maximum amount of points that can be attributed to a certain biotope (576) and multiplied by 100. So the following formula is used:

((1 + 2 + 3 + 4) * (5 + 6 + 7 + 8)) / 576] * 100 = number of points (3-100)

It is to be noted that in practice the point value of the listed biotopes ranges from 3 to 80 points (per square meter). To give a few examples:[13]

- Oak tree forest (*Eichen - Hainbuchenwald*)	56 points
- High-moor bog (*Hochmore*)	80 points
- Sand dunes (*Sanddünen natürlich*)	39 points
- Mixed forest (*Mittel wald*)	56 points
- Pine tree forest (*Natürlich Kiefern.*)	55 points

The amount of compensation payable for injuries to the more than 180 biotopes covered is finally reached by multiplying the final number of point attributed to a certain biotope (see above) with the amount of square meters affected and the average restoration cost (DM 0,62). The last mentioned figure is based on the average restoration costs that were made over the years to restore damage to nature and landscapes. The figure is thus based on the real costs of measures that were actually taken.

The money obtained is to be used for nature conservation and land management, and is preferably spent on projects having a close connection with the harm done. If this is impossible, the money will be used on projects a greater distance away from the impacted site.

As noted earlier, the Hessian compensation model is primarily applied in *ex ante* situations and used to calculate the amount of compensation to be paid if a certain project is expected to have negative consequences for the environment and cannot be fully compensated by taking restoration or replacement measures. The model may, however, also be applied in situations where a polluter causes a damage to nature but it appears impossible to (fully) re-establish the functions of the impacted nature area (*ex post* situations).[14] In that case the model is used to

[13] *Id.*

[14] Art. 8(2) Hessian Nature Conservation Act.

assess the level of compensation to be paid to compensate for the difference in quality of the impacted site before and after the incident. It is to be noted, however, that as far as known the model has never been used in such a context and there is no case law that affirms this application.

B.4.2 The Andalusian Compensation Table

Another abstract model that is used in one of the EU Member State is the 1986 Andalusian compensation table, which has been developed to assess damages for injuries to some particular natural resources. Since not many details are known on the model, it is noted that it is difficult to provide a complete overview of the model and to properly assess the model.

The Andalusian compensation table is a relative simple assessment model and has a limited scope of application as it is primarily used to assess damages for injuries to protected animal species. These injuries include the capture and killing of these protected animals, the disturbance of their breeding and nesting habitats, and the taking of eggs. The model exists of table with amounts that have to be paid if there has been caused an injury to the animal species covered. The animals that are covered are listed in the model. Apart from more general categories, such protected (marine) mammals and birds, some specific animal species are listed including seals, otters, wolfs, various species of eagles and the flamingo. The amounts vary from 1.500.000 ptas for the killing of a seal to 50.000 ptas for a weasel.[15] For a damage to non-specifically listed protected marine mammals the amount is 500.000 ptas per animal, for non-specifically listed mammals 25.000 ptas and for non-specifically listed protected birds 50.000 ptas per bird or egg. The quantum of damages is assessed by multiplying the number of animals killed or captured (or their eggs taken) with the listed monetary values.

The listed amounts reflect the cost of re-introduction of the animals concerned and is based on the average cost of maintaining and preserving the species covered (no further details were available).[16] The model is used primarily in *ex post* situations, but it is unclear under what conditions the model is being applied and what is being done with the monetary payments. It is also unclear how often the model is used, whether the model has been tested in court and whether or not it functions properly. It is to be noted that since the decree that establishes the compensation table allows to raise the listed amounts with about 20% if a person repeatedly causes harm to the wild life covered, that the model is more of a penal nature and may therefor be less suitable for liability situations.

B.5 RISK AND UNCERTAINTY

Discussion so far is based on the implicit assumption that the costs and benefits of restoration can be identified and measured with certainty. In reality, this may not always be the case. There are a number of different aspects to risk and uncertainty surrounding the restoration of biodiversity damage including, but not limited to:

- Pre-incident resource status may not be known with certainty if there are no data collected prior to the incident;
- The extent of the damage may not be possible to measure with certainty because: of (i) lack of data about the pre-incident resource, and (ii) lack of understanding of the process through which an incident causes damage;

[15] The here listed figures were published in 1992 and may now be higher. Part of the list is published in M.R. Will, H-U. Marticke, *Verantwortlichkeit für Ökologische Schäden. Rechtsvergleichende Untersuchung* (Vol. II), Saarbrücken 1992, p. 312.

[16] *Id.* at p. 311.

- Restoration options cannot be said to be certain to succeed because of: (i) lack of data about the time period required to meet the restoration target, and (ii) uncertainties about the future conditions;
- The costs of restoration options may not be known with certainty since more funds can be required if unforeseen factors threatening the success of restoration need to be compensated;
- The benefits of restoration options may not be known with certainty because (i) the extent of damage and the success with restoration option are not known with certainty, (ii) affected population may not be accurately identified, (iii) future uses of the damaged resource may not be predicted, (iv) measurement of the preferences may not be accurate especially if benefits transfer is not used appropriately, and (v) if non-monetary measure of benefits is used, expert judgements may be disputable; and
- Choice of restoration option can be significantly affected by the choice of discount (and compound) rate applied.

The different aspects listed above and others can be dealt with using different approaches. First of all, it is crucial to distinguish whether the above lead to *risk* or *uncertainty* since their treatment can be significantly different. Risk is defined as some known combination of the probability of an event occurring, and the scale of the event. Uncertainty arises when this probability distribution is not known and the scale of the event, if it occurs, may be known accurately or only imperfectly. The distinction between risk and uncertainty is important because the means of dealing with them are different. The rest of this section presents an outline of these different approaches to risk and uncertainty.

B.5.1 Incorporating Risk and Uncertainty into Costs and Benefits

For situations involving 'risk' there are three options depending on the level of available information about risk.

If the probability distributions of the costs and benefits are known, **stochastic simulation methods** (e.g. Monte Carlo method) can be useful for developing insights into these probability distributions. Although sophisticated and hence desirable, the information requirements of such simulation exercises can be too high. It can be applied only if the distribution of probabilities (across time and space) attached to costs and benefits are known, which is unlikely to be the case for most natural resource damage cases.

The second option involves estimating **expected values** for the outcomes of concern and can be chosen when we have only point estimates for the risk, i.e. we only have one absolute value for the outcome and one estimate of the probability of that happening.

For example, if an event with a cost valued at Euro100 occurs with a probability of 0.1, one approach might be to multiply the two numbers so that risk equals Euro10. This is an example of an expected value approach to representing risk. Equation below shows how the net present value estimate would change, when expected rather than absolute values for costs and benefits are used.

$$NPV = PV(B) - PV(C) = \sum_{t=t_I}^{T} [(p_b \times B_t) - (p_c \times C_t)] \times d_t$$

Where p_b is the probability of benefits occurring, p_c is the probability of costs occurring and the other variables are as above. The value these probabilities are likely to take depends on the characteristics of the resource in question, the restoration project alternative appraised and the background conditions at the site. Therefore, it is not possible to make a general statement about what these values should be other than to recommend that relevant experts in the field are consulted to identify these risks and their likely effect on the future costs and benefits of various restoration options.

Possibly the most sophisticated approach to incorporating risk and uncertainty into the values of costs and benefits is by estimating a **risk premium**. This involves measuring the certain benefits that affected individuals would accept in lieu of the uncertain benefits. The difference between the certain benefit and the uncertain benefits is referred to as the risk premium (Wilson, 1982). Risk premia may be estimated through the application of stated preference techniques to measure the certain benefit that is equivalent to an uncertain stream of benefits. For example, a stated preference study might ask respondents to choose between a certain stream of natural resource services and an uncertain stream of natural resource services. The choices thus elicited would yield information on individual risk premia (DARP, 1999). If a stated preference study is already implemented to estimate the benefits of restoration, the issue of risk premium can be incorporated into the design of the questionnaire. However, implementing a questionnaire with the sole purpose of estimating risk premium is not likely to be cost-effective. This is recognised by the USA guidance on the issue which states that the risk-adjusted measures of value (all three approaches mentioned here, namely, Monte Carlo simulation, expected values and risk premium) are rarely fully implemented due to their extensive requirements for generally unavailable information.

B.5.2 Incorporating risk and uncertainty into the discount rate

An alternative, less preferred, approach presented in the USA guidance (DARP, 1999) involves selecting discount rates that reflect the level of systematic, i.e. non-diversifiable, risk associated with the restoration option. A range of after-tax interest rates that reflect varying levels of risk are available for this purpose. For the USA, the lower end of the range of rates representing a risk-adjusted time preference is the riskless average real after-tax return on US Treasury bills. Again for the USA, the upper end of the range is the after-tax return on the market portfolio of stocks. Based on the historical data, the lower end is identified as about 1%, while the upper end is identified as about 7%. It is suggested that sensitivity analysis over the range of risk-adjusted rates to generate a range of outcomes (see Section B.5.4 for sensitivity analysis). We do not have historical data about Europe to recommend risk-adjusted values for Europe but the USA suggestions can be tested in the European context. However, the USA guidance notes that adjusting discount rates to accommodate risk and uncertainty is rarely fully implemented due to the extensive requirements for information.

B.5.3 Institutional mechanisms for accommodating risk and uncertainty

The USA guidance (DARP, 1999) states that the most common method for addressing uncertainty in restoration implementation is through institutional mechanisms. There are three approaches to institutional mechanisms: (i) performance standards, (ii) design standards, and (iii) contingency factors.

With a **performance standard**, the liable party agrees to undertake restoration and is held to a performance standard for the outcomes of primary and compensatory restoration projects. The performance standard shifts the uncertainty associated with restoration to the liable party rather than the public authority. Should the liable party fail to comply with the performance standard, such as restoring the damage resource to its pre-incident condition within a given time period, the liable party is subjected to a pre-determined penalty.

In some circumstances, there may be substantial uncertainty about project outcomes due to factors external to project implementation. In these cases, the liable party may not agree to any performance since performance is, to a large degree, out of its control. **Design standards** offer a mechanism for sharing uncertainty between the liable party and the public authority. With a design standard, the liable party is subject to standards on project actions and may incur penalties for non-compliance. The advice given by the English Nature in the UK (state nature conservation agency) can be likened to what is referred to as the design standard in the USA.

For example, if 100 ha of wetland is damaged as a result of an incident, the English Nature recommends that 200 ha of wetland is restored or recreated to account for the chance of failure that restoration of 100 ha may not be sufficient for the resource to return to its pre-incident level (personal communications, Jonathan Burney, economic adviser, English Nature, February 2001).

Blackbird Mine, which has been used as a case study in the main report, provides an example of performance standard as well. The responsible party is subject to stipulated penalties for each day they fail to complete a deliverable or fail to produce a deliverable of acceptable quality. They also must pay liquidated damages as compensation for interim loss damages in the event of the delays in the biological restoration plan. The responsible party must take additional action, with the approval of trustees and in consultation with the Environmental Protection Agency, to achieve water quality criteria if standards are not met any time after January 1, 2002. The consent decree includes dispute resolution and force majeure provisions, which are contingency plans for an event arising from causes beyond the control of the responsible party. Once the responsible party satisfies the relevant public authorities' action requirements, the uncertainty about the number of salmon produced for the specified actions falls on the trustees.

Another approach to incorporating risk and uncertainty, which is widely used in development projects around the world, and recommended by Army Corps of Engineers in the USA, is to establish **contingency factors**. Although this is the preferred approach in the USA guidance only when other approaches to risk and uncertainty cannot be implemented, due to its simplicity it is widely used. The factors are fixed percentages of the expected construction costs. For example, in the feasibility phase of project development, the Corps recommend adding 25% to the restoration cost for projects valued at less than US$10 million and 20% for projects valued at more than US$10 million (The Army Corps of Engineers, 1994).

The USA guidance for damage assessment (DARP, 1999) suggests that similar factors can be adopted in the current context and identified based on (i) the gap between predicted and realised service flows, and (ii) the gap between predicted and realised costs of restoration.

B.5.4 Adjusting the decision making rules to accommodate risk and uncertainty

Probably the simplest adjustment is **sensitivity analysis** in that it can be applied at any stage of damage assessment and choosing restoration option but it tends to be a rather superficial way of dealing with risk and uncertainty (though can be sufficient in some cases). It applies to both risk and uncertainty and involves reflecting them in 'what if…?' scenarios.

At its simplest, this will involve repeating the analysis (whether this is at the earlier stages of designing a restoration option or at the later stages of choosing a restoration option, say, the discount rate used) while using different assumptions about the value of the chosen factor. In addition to testing the effect of changing individual factors, combinations of assumptions may also be tested.

For example suppose that we are estimating the benefits of restoration in monetary units. Whether an original valuation study is undertaken or benefits transfer procedures are used, it is likely that we will have an interval within which WTP/WTA value lies. Sensitivity analysis allows the present value of the benefits to be estimated using lower bound, best and upper bound estimates based on this interval. Similarly, we may have different assumptions about the affected population, which again can be accounted for using sensitivity analysis.

Sensitivity analysis by itself resolves nothing: it simply shows the sensitivity of the cost-benefit calculation to changes in assumed values of parameters. However, this has the advantage of focusing attention on the values of the parameters in question. Note the usual values chosen for key factors are (i) the minimum possible value, (ii) best estimate and (iii) the maximum possible value.

Several situations might emerge:

- benefits exceed costs for the restoration option regardless of the value chosen for the key parameter. Then the result is robust;
- costs may exceed benefits for the restoration option regardless of the value chosen for the key parameter. Again the result is robust; and
- the project may pass (or fail) a test for some values of the chosen factor(s) but not for others. This forces the decision-maker to express a judgement as to which value of the parameter is 'most likely'. Effectively, an uncertainty problem is converted to something akin to a risk problem by the assignment of judgmental probabilities.

More sophisticated approaches to uncertainty can also be applied by employing decision analysis. This involves constructing a **payoff matrix**. It should be noted that the decision analysis discussed here does not involve any sophisticated modelling exercise but simply a structured framework for making professional judgements about uncertain situations.

An illustrative payoff matrix is constructed in Table B-2. If the objective is to maximise net benefits, the numbers in the payoff matrix record values of net benefits. These net benefits depend on both what decision (D) is taken (e.g. D1 involves undertaking a restoration measure and D2 involves no restoration) and what the 'state of the world' (S) is. The state of the world simply reflects the possibilities that may occur in the future. The pay-off matrix shows the net present values of decisions D1 and D2 in the states of the world S1 and S2. These can be estimated using sensitivity analysis, i.e. re-running the CBA with different assumptions. However, the probabilities attached to the states of nature and hence the outcome of the decisions D1 and D2 are not known.

Table B-2: Pay-off matrix

Decision	State of the world 1	State of the world 2
Decision 1	+ Euro 100	-Euro 15
Decision 2	+ Euro 90	+Euro 30

If S1 occurs, the best decision is D1. But choosing D1 is risky because S2 could occur and there could be a loss of 16. The following decision rules are possible:

Maximax**:** choose the option that maximises benefits (here D1 with +Euro 100). This criterion would be chosen by an optimist since there is a risk that S2 would occur and losses would be incurred.

Maximin**:** choose the option that minimises losses (here D2 with +Euro 30). The minimum payoffs are -15 and +30, so the decision-maker maximises these minima. The decision-maker using this criterion is cautious: he or she avoids the worst outcomes.

Other criteria focus on what would happen if the wrong decision is made. To determine this first construct a **regret matrix**. An illustrative regret matrix is constructed in Table B-3.The regret payoff is defined as the difference between what is actually secured and what could have been secured had the correct decision been made. For example, choosing D1 with S1 occurring involves no regret since D1 has the highest payoff. Choosing D1 with S2 occurring involves foregoing Euro 30 (had the correct decision, D2 been made) and losing Euro 15, a regret of Euro 46. Choosing D2 in S1 yields Euro90 but had D1 been chosen it could have been Euro100, so the regret is Euro 10. Choosing D2 in S2 involves getting Euro 30 but choosing D1 in S2 would have produced –Euro 15, so the regret is zero. The regret matrix is shown in Table 6-3 overleaf.

Table B-3: Regret matrix

Decision	State of the World 1	State of the World 2
Decision 1	Euro 0	- Euro 45
Decision 2	- Euro 10	Euro 0

A criterion for choice is now ***minimax regret***. This involves taking the maximum regrets from the regret matrix (Euro 10 and Euro 45) and minimising these (choosing Euro 10), i.e. D2.

B.6 SELECTION OF TYPE OF ANALYSIS

The necessity of implementing CBA in the selection of project alternatives is a factor that can be largely influenced by the choice of legal system. In the USA, the statutory goal of a restoration plan is to restore natural resources to baseline, and to compensate the public for interim losses from the time of injury until the return to baseline. Since restoration is a statutory requirement, possible restoration alternatives that could achieve the goal of compensation are selected on the basis of CEA. CBA comes into play in two instances:

(i) If the cost of restoration is thought to be 'excessive': If the cost is not deemed excessive, then CEA is sufficient. If the cost is deemed excessive, however, then CBA needs to be implemented.

(ii) If the best available compensatory restoration action concerns resources which are *not* of comparable quality or value compared to the lost resources: In this case, the service-to-service approach for design of compensatory restoration projects is not applicable. If there are significant impacts to the resource, such that the time and cost of valuation studies can be justified, then the value-to-value approach is implemented to scale compensatory restoration projects. This is equivalent to a form of cost-benefit analysis, as trade-offs between the lost and replacement resources are elicited and compared. The comparison here, however, is one of the value of services lost versus services gained, for the purposes of *designing* a restoration option. This is distinct from (i) which compares the benefits of restoration versus the costs of restoration.

As discussed above, CEA does not require the measurement of the benefits of restoration so long as restoration target is identified and agreed. If required, scoring techniques can be used for expressing benefits so that the restoration options can be compared on the basis of the scores they get per Euro spent – if such a comparison was desired. Therefore, CEA requires less information, time and effort than CBA.

Again as discussed above, strictly, CBA requires the benefits of restoration to be expressed in monetary units for direct comparison with the costs of restoration. Once the decision to implement CBA is taken, all care should be taken to ensure that monetary values of benefits are estimated. Thus, the difficulty with the CBA approach lies mainly in the difficulties with monetary estimation not in the implementation of CBA, which is usually little more than a spreadsheet exercise. Options for estimating monetary values for benefits are discussed in Annex A.3.

In the event, that monetary expression of benefits is not possible, CBA can include both monetary and non-monetary expressions as discussed above. However, this should *only* be undertaken if it is proved that monetary assessment is either not possible or feasible. As with most choices of this kind, the decision lies in the level of accuracy and robustness expected from the results and the time, effort and skills that are required for each approach. Notwithstanding the uncertainties and difficulties attached to the monetary assessment, it is likely to be easier to defend in a legal setting than non-monetary assessments entirely based on expert opinion and assumptions. Note that the reliability of non-monetary assessments increase if they are based on public opinion through, for example, the application of choice modelling techniques which use rankings of options and impacts rather than money as measure of WTP or WTA (see Annex 1, Section B.1.2).

The treatment of risk and uncertainty in choosing the restoration option is applicable regardless of whether CEA or CBA is chosen. Section B.5 discusses the options for dealing with risk and uncertainty. Some of these options such as estimating risk premia are complex and possibly not feasible given their information requirements. However, others such as sensitivity analysis and decision making rules have relatively less information requirements but can add significantly to explaining the uncertainties and hence improving the quality of the resulting decision. Again the choice is between the desired level of accuracy and robustness and information, time and resource requirements. Such choices are site and event specific and depends on factors such as the scale of the damage, importance of the damaged resource, the scale of the affected population and so on. It is not possible at this stage to make recommendations that would apply to every possible case in the future.

Finally, the level of difficulty with any analysis depends on the analysts undertaking the analysis. As with any other interdisciplinary work, assessment of damage, choice of restoration options and assessment of costs and benefits require experts from different disciplines to be involved in the process. A minimum requirement would be ecologists, economists and legal professionals. Given the time and financial requirements of involving such experts, it is crucial to include 'assessment costs' within the cost of restoration and for this cost item to include all aspects of assessment.

B.7 AN ILLUSTRATIVE EXAMPLE

The following example is based on DARP (1999) but has been adjusted to illustrate the main discussion points in this annex.

Suppose an oil spill occurred in 1997 and injured 50 acres of inter-tidal wetland. Further assume that 100% of the wetland services were lost in the initial period, recovery does not begin until 1999, and recovery is linear over a five-year period. If the interim loss calculation is conducted in 1998 (claim year), Table B-4 shows the calculation of the present value of damage to wetland, which is equal to the benefit of restoration. Note that benefit is expressed in non-monetary unit, i.e. acres of wetland damaged and then gradually restored.

Table B-4: Discounting Interim Losses (= benefits of restoration) in non-monetary units

Year	% of loss – start of period	% of loss – end of period	Loss in terms of acres of wetland	Discount factor (dt)	Discounted acres of interim losses *
1997	0	100	50	1.03	51.50
1998	100	100	50	1.00	50.00
1999	100	80	40	0.97	38.83
2000	80	60	30	0.94	28.28
2001	60	40	20	0.92	18.30
2002	40	20	10	0.89	8.88
2003	20	0	0	0.86	0
Present Value of interim losses (Benefits of restoration) in units of acres of wetland					195.80

*: Discounted acres of interim losses are calculated by multiplying the loss in terms of acres of wetland in each year with the discount factor for that year.

The discount (or compound) rate is chosen as 3%. Other rates can be tested as sensitivity analysis. The factor d_t in the fifth column serves the function of a compound factor for the year before the claim is made, i.e. 1997 and calculated as $(1+0.03)^{1998-1997}$. The value in 1998 is already in present value terms, or in other words, dt equal 1 ($((1+0.03)^{1998-1998})$. The years after 1998, the discount factor is estimated on the same basis for each year as $(1+0.03)^{1998-t}$ where t is each year until the end of project life-time.

The total costs of two technically feasible restoration options over the project life-time are presented in Table B-5. The costs consist of assessment costs, land acquisition (for the compensatory measure of converting an area adjacent to the damaged wetland into a wetland), restoration, monitoring and maintenance. The discount rate used to calculate the present value of costs is the same as that used for calculating the present value of benefits, i.e. 3%.

Table B-5: Costs of restoration options (Euro)

Year	Discount factor (dt)	Costs of restoration option A	Present value costs of option A*	Costs of restoration option B	Present value costs of option B*
1998	1.00	110,000	110,000	50,000	50,000
1999	0.97	95,000	92,150	250,000	242,500
2000	0.94	64,000	60,160	87,000	81,780
2001	0.92	66,000	60,720	95,000	87,400
2002	0.89	68,000	60,520	75,000	66,750
2003	0.86	72,000	61,920	45,000	38,700
Present Value			445,470		567,130

*: Present value of costs is calculated by multiplying the cost in each year with the discount factor for that year.

The first analysis to apply is the cost-effectiveness (or least-cost) analysis. Given that both restoration options generate the same benefit in terms of restoring the damaged 50 acres of wetland and assuming that each has the same likelihood of success, the choice is simple: the option with the lowest present value of cost should be implemented, i.e. option A with present value of costs of Euro 445,470.

It is also possible to express the result of the cost-effectiveness analysis in terms of present value of the interim loss restored per Euro spent (present value of benefits divided by present value of costs). In this case, option A scores about 0.0004 acres per Euro (195.80/445,470), while option B scores about 0.0003 acres per Euro (195.80/567,130). Again, the choice is clear: option A.

Now suppose that option B has a likelihood of success of 90% while option A has a likelihood of success of 10%. The scores of acres of wetland restored to Euro spent become about 0.00004 for option A and 0.00027 for option B. The introduction of risk element makes option B more cost-effective given that it is more likely to achieve the restoration target despite costing more in the process.

Now suppose that the costs of both restoration options are deemed to be excessive. Thus, the cost of restoration needs to be compared with the benefit of restoration. Note that although the acre per Euro makes such a comparison, it can only be used to choose the most cost-effective of the restoration options. It cannot answer the question whether any restoration should take place – which is what is effectively being suggested by arguing the cost of restoration to be excessive.

Therefore, the benefits of restoration should be expressed in terms of money to be directly comparable to the costs, or in other words, cost-benefit analysis should be implemented. Wetlands are valued by individuals because of their services that enhance the individual wellbeing. Such services include geo-hydrological, production/habitats, recreation, ecosystem integrity, cultural and heritage, health and scientific. Assume that the damaged wetland was used for angling and that there were 10,000 visits per year. Again suppose that a valuation study conducted estimates the value of a visit to be Euro 10. In other words, the recreational use value of the wetland is Euro100,000 per year or Euro 2000 per acre per year. It can be assumed that recreation will not be possible in the damaged wetland until full recovery. Thus, Table B-6 repeats the benefit assessment exercise but with recreational use of wetland in monetary terms. Note that recreation is only one component of the use values attached to the services of the damaged wetland and as such can only be a lower-bound estimate of all use values.

Let us assume that there is also a study that estimates the non-use value attached to the wetland as Euro 5 per person per year. Assuming that only the regional population of 12,000,000 people hold non-use values for the wetland, the total non-use value would be Euro 60,000,000 per year or €1,200,000 per acre per year. For a conservative estimate, it can be assumed that since the wetland recovers in a relatively short-time, non-use value is not lost in the interim. Thus, non-use values are not included in the damage assessment presented in Table B-6. Note that if the damage was irreversible, the effect of including non-use values is easy to see: a relatively small amount of non-use value per person adds up to a large sum across a large population.

Table B-6: Discounting Interim Losses (= benefits of restoration) in monetary units

Year	Acres of wetland	Recreational use (Euro) *	Discount factor (dt)	Discounted interim loss in Euro **
1997	50	100,000	1.03	103,000
1998	50	100,000	1.00	100,000
1999	50	100,000	0.97	97,000
2000	50	100,000	0.94	94,000
2001	50	100,000	0.92	92,000
2002	50	100,000	0.89	89,000
2003	0	0	0.86	0
Present Value of interim losses (Benefits of restoration) in Euros				575,000

*: Recreational use value is calculated by multiplying the lost acres in each year with Euro 2000 per acre per year. Note that it is assumed that recreational value per acre stays constant over the project life-time.

**: The discounted damage is calculated by multiplying the recreational use value for a year with the compound and discount factor for that year.

Based on the present value of benefits presented in Table B-6 and the present value of costs in Table B-5, the Net Present Value (NPV) of option A is Euro 129,530 (575,000 – 445,470) of option B is Euro 7,870 (575,000 – 567,130). The benefit-cost ratio for option A is 1.3 and for option B, 1.01. If both options are certain to succeed, option A should be chosen.

Let us this time assume that the uncertainty in the success of the restoration options is dealt with by creating a contingency fund in 1998 to the value of Euro 200,000 for option A and Euro 50,000 for option B increasing the present value of the costs of option A to Euro 645,470 and that for option B Euro 617.130 (note that the discount factor in 1998 is 1). This changes the NPV result for option A to minus Euro 70,470 and that for option B to minus Euro 42,130. Benefit cost ratios become 0.89 for option A and for 0.93 option B. Neither of the option passes the CBA test. However, other considerations may still lead to a restoration action.

ANNEX C: DIFFERENCES IN VALUES OBTAINED FROM DIFFERENT STUDIES

This annex[17] expands on the economic valuation techniques outlined in Annex 1 of this report. In particular, it addresses the question of whether it is legitimate to expect differences in estimated values from the application of different valuation techniques. There are two basic issues here:

1. First, does economic theory lead us to expect differences in values obtained from different valuation techniques? There are several reasons why well-designed and well-conducted, theoretically sound studies might result in different estimates of economic values. *These are the subject of Section C.1.*
1. Second, given the relative importance of stated preference techniques, it is relevant to explore the issues to do with the design of SP questionnaires that reduce the unexpected differences in valuation results and that improves the suitability of individual studies for future benefits transfer. *These are the subject of Section C.2.*

C.1 DIFFERENCES IN ESTIMATES OF ECONOMIC VALUES: THEORETICAL EXPLANATIONS

As noted in the terms of reference for this project, it is possible that different valuation techniques might result in different estimated values. The issue of whether such differences are expected and hence valid and reliable or they are signs of invalidity and unreliability has recently been the subject of an involved debate.

The central problem in assessing the validity of value measures obtained from any economic valuation technique is the absence of an unambiguously clear and definitive criterion against which to compare those measures. This is not a generic problem of all survey research (e.g. election opinion polls can be compared against the results from the subsequent elections they set out to predict). However, it is generally a problem for public goods in that, with very few exceptions, *actual values are unobservable*. The issue with all consumers surplus measures, be they for marketed or non-marketed goods, is that they are inherently 'unobservable' measures with respect to actual transactions because they represent the difference between what an agent is willing to pay or willing to accept and what they actually pay (or receive). It is possible with respect to market transactions to identify some (but not all) behaviour related to the consumer surplus measure. This implies that, at least, part of the consumer surplus estimate must always be driven by assumption. Analogous problems arise generally in psychological attitude-behaviour research (American Psychological Association, 1974) where validity is treated as a multidimensional issue.

One type of validity assessments (convergent validity) compares measures obtained from different valuation studies with:

(i) those obtained from other techniques;

(ii) multiple studies using the same technique (e.g. of the same or similar resources) in a process known as *meta-analysis*, or transferred across applications using *benefits transfer* techniques; and

(iii) those obtained via experimental simulated markets.

A separate but related issue concerns the 'reliability' of values, in other words:

(iv) consistency of estimated values obtained from the same technique at different points in time.

[17] This annex draws heavily on EFTEC (forthcoming 2001).

In convergent validity testing no measure can automatically claim superiority in terms of being a naturally closer approximation of the value of the underlying construct. This could be thought of as being 'validity by association'. However, strictly speaking, even this would be overstating the case. Just because two approaches deliver similar or logically related measures does not mean that those measures are valid; instead they may be equally invalid. Nevertheless, it is clear that a large and unexpected difference between estimates would show that at least one measure is invalid or two different questions are being addressed. The main results from these validity assessments are presented below.

The following factors may cause differences in WTP/WTA estimates and hence discussed further below:

1. type of economic value of concern;
2. coverage of relevant population;
3. characteristics of damage;
4. whether or not WTP or WTA is used as the measure of economic value;
5. use of different valuation techniques (comparison of SP and RP, comparison with simulated markets and differences due to different variants of SP);
6. changes in values over time, and
7. factors specific to stated preference studies (reviewed in Section C.2)

C.1.1 Type of economic value of concern

Annex 1 outlined the components of total economic value of an environmental change. In summary, total economic value may be disaggregated into *use values*, which relate to actual, planned or possible use of a natural resource, and *non-use values*, which reflect values of preservation of a resource in the absence of actual, planned or possible uses (e.g. for future generations). The main categories of value are outlined in Figure A1-1.

One important source of differences in valuation studies is the *category(ies) of values* being examined. As outlined in Section A1.2, revealed preference techniques (RP) depend on WTP information which can be inferred from individuals' actual decisions in the marketplace. As such, these methods are restricted to the estimation of *use values*, since these are the only categories of value which leave a behavioural trail.

This approach is perfectly adequate when the values of interest are use values only, for example, in the context of damage to a park which is temporary, reversible, and which affects users only. However, if, for example, damage is irreversible, this approach is liable to understate the total value of the park, by not capturing all of the ways in which people value it. In particular, *non-use values* are associated with individuals who do not currently visit the park, or plan to visit in the future. Such people may want the park to be protected, independent of any intention to visit it. Stated preference techniques (SP) are the only techniques capable of estimating these non-use values. In comparison of different studies of similar resources, therefore, it is essential to be clear of which values are being estimated. Stated preference studies of users only may also capture non-use values in addition to the use values of these respondents. Typically it is impossible to separately estimate the different components of value captured in respondents' stated values, although motivations for stated values are typically explored in survey methods, which may give an indication of whether stated values embrace non-use as well as use values. This is an important consideration when comparing stated preference and revealed preference studies of users of the same resource. RP and SP techniques typically address overlapping but not identical value sets.

C.1.2 Relevant population

A related issue is the *relevant population* considered in different valuation studies (see Annex 2). In both revealed and stated preference studies, often only a single group of users is considered. For example, in the context of a water resource the focus may be exclusively on WTP of anglers. However, it is reasonable to expect that different groups of users – boaters, anglers, and other waterside recreationists – will hold different values for the same natural resource just as values of non-users for protection of a resource would be expected to differ from those of users.

C.1.3 Characteristics of damage

Values of both users and non-users of any given damage would be expected to vary according the *scale of the damage* under consideration. It may be expected that the greater the scale of damage, the higher WTP/WTA should be. However, this increase may not be direct. In other words, a 10% loss of a habitat may not necessarily be twice as valuable as a 5% loss of the same habitat. This is often referred to as the scoping effect: the TEV estimates are insensitive to the scale or scope of the damage concerned. This is indeed a major challenge as a lack of scope sensitivity suggests that estimated values relate not to the specified impact but to some other measure most typically identified as being some invariant 'moral satisfaction' or 'warm glow' measure. In recent years, a heated empirical debate over scope has permeated the environmental economics literature: while some studies have demonstrated scope sensitivity, others have not and still others show that it is possible to observe scope and scope insensitivity within the same study. It is likely that the better the design of a valuation study, the more likely it is to demonstrate the existence of scope sensitivity.

The valuation estimate may also depend on whether the exercise is conducted *ex ante* or *ex post*. In some cases, the latter is known to produce lower estimates than the former, once the damage is observed to be not as bad as expected. However, there is no rule that this will always be the case. In fact, this difference may not be relevant in the current context, since valuation studies for estimating the relevant compensation amount are likely to take place after the damage occurs. Elimination of possible bias in the results of an *ex post* study is an issue of design. It is possible that, with poor design, survey respondents may perceive that their responses could influence the amount of compensation provided. However, with suitable survey design and appropriate piloting, it should be possible to obtain accurate results. For example, use of a WTP question (rather than WTA) may be found to be more suitable, or indeed the use of choice modelling techniques which elicit preferences for trade-offs between resources directly without the use of monetary measures.

Whether respondents have *direct experience* of the level of natural resource of concern and/or the damage to it. It could be argued that familiarity of this kind may lead to higher estimates of WTP or WTA but there is no rule about this.

Finally, in the context of damage to human health, it has been shown that *voluntary* risks are more acceptable than *involuntary* risks. This acceptance shows itself in higher WTP or WTA estimates for environmental changes that are involuntarily imposed on the affected population.

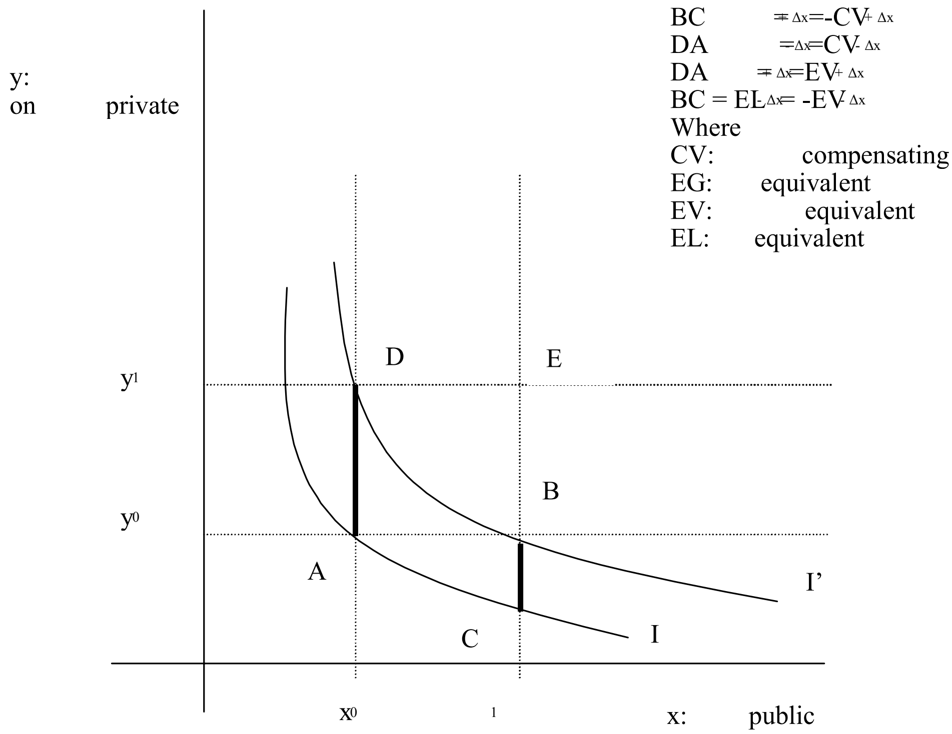
C.1.4 Willingness to pay and willingness to accept

The two basic concepts relevant for economic valuation are willingness to pay (WTP) and willingness to accept (WTA). While these two measures would be expected to be *similar* in magnitude in most circumstances, evidence leads us to expect WTA to exceed WTP. This section outlines the reasons for this disparity, and its consequences for valuation measures.

The choice of which valuation measure to use depends on whether the change in question is perceived as a benefit (in which case the relevant concept is WTP) or a cost compared to the status quo (in which case WTA should be used). The concepts of WTP and WTA, and the relationships between them, can best be explained by using indifference curves.[18] Figure B.1 represents the preferences of a given individual. The vertical axis measures the individual's expenditure on private goods (y). This is measured in money units, on the assumption that prices are given, and it can be thought of as the quantity of a single composite good. The horizontal axis measures the quantity (x) that exists of some public good. The indifference curves I and I′ link combinations of the two goods between which the individual is indifferent. Each curve can be thought of as corresponding to a level of welfare, utility, or well-being, with I′ corresponding to the higher level.[19]

There are four measures of the value of a change in the quantity of a public good. First, consider the value to the individual of an *increase* in the quantity of the public good from x_0 to x_1. Suppose that initially the individual has y_0 private consumption, and so is at A. Compare point C. At C the individual can enjoy x_1 of the public good but his private consumption is less by the amount BC. Since A and C are on the same indifference curve I, we can infer that his WTP for the increase in the public good is BC. In welfare economics, the negative of this amount is called the *compensating variation* for the *increase* in the public good, since the loss of BC in private consumption exactly compensates for that increase.

Figure C-1: Measure of change in human welfare



[18] Indifference curve analysis rests on certain fundamental assumptions about the nature of preferences, which are used almost everywhere in economics.

[19] The absence of any word which satisfactorily describes what these 'levels' are levels of is a consequence of the fact that the economic analysis of preference does not invoke any absolute standard of value. Terms such as 'welfare', 'utility' and 'well-being' are used in philosophical discussions to refer to particular aspects of an individual's life which can be asserted to have value. In contrast, an indifference curve simply describes an individual's willingness to accept some things in exchange for others. Strictly speaking, it is not a level *of* anything.

Second, consider the opposite case, in which the individual, again starting with y_0 private consumption, faces a *decrease* in the public good from x_1 to x_0. Now the initial position is B. Compare point D. At D, the individual enjoys only x_0 of the public good, but his private consumption is greater by DA. Since B and D are on the same indifference curve I′, we can infer that his WTA for the reduction in the public good is DA. This is the *compensating variation* for the *reduction* in the public good.

Third, it is useful to consider two other measures of the value to the individual of the increase in the public good from x_0 to x_1. Suppose that the individual starts off with y_0 private consumption and x_0 of the public good: he is at A. We may ask what additional amount of private consumption would be just as preferable as an increase in the public good to x_1. This is the *equivalent gain* measure of the value of the change in the public good. Since D is on the same indifference curve as B, equivalent gain is equal to DA. In the language of welfare economics, DA is the *equivalent variation* for the *increase* in the public good. Notice that equivalent gain and WTA are both equal to DA. The equality of these two measures is an implication of the standard economic theory of preference. In terms of stated preference questionnaires, however, equivalent gain and WTA are distinct concepts, elicited by different types of question. ('How much money would just compensate you for losing X?' is a different question from 'How much money would be just as good as gaining X?') The theory tells us to expect that these two types of question will yield the same answers.

To arrive at the fourth measure, suppose the individual starts off with y_0 private consumption and x_1 of the public good, i.e. at B. We may ask what loss of private consumption would be just as preferable as a decrease in the public good to x_0. This is the *equivalent loss* measure of the change in the public good; since C and A are on the same indifference curve, equivalent loss is equal to BC. In the language of welfare economics, the negative of BC is the *equivalent variation* for the *decrease* in the public good. Notice that equivalent loss and WTP are both equal to BC. As in the case of equivalent gain and WTA, this is a theoretical implication about the equivalence of what, in stated preference terms, are two different methods of eliciting valuations.

These fine distinctions are significant only to the extent that different measures yield different valuations. In the diagram, DA > BC. That is, WTA is greater than WTP (and likewise, equivalent gain is greater than equivalent loss). It can be shown theoretically that this inequality holds whenever the indifference curves are convex to the origin and the good is 'normal', i.e. if the good could be bought at constant prices, the amount consumed would increase with income. It should be clear from the diagram that the ratio between WTA and WTP will tend to be greater, the more convex the indifference curves are, i.e. the less substitutability there is between private consumption and the public good, and the greater the difference between x_0 and x_1.[20]

However, in most cases, the divergence between WTA and WTP, as predicted by the theory, should be very small. To see why, consider the case in which the individual starts with private consumption of y_1. In this case, WTP for an increase in the public good from x_0 to x_1 is EB. EB is greater than BC: if the individual is richer, he can afford to spend more in order to increase the public good. But notice that EB = DA. Thus, the difference between WTA and WTP (when both are evaluated in relation to an initial private consumption level of y_0) is exactly the same as the difference between the two measures of WTP – one evaluated in relation to y_0, the other in relation to y_1. Notice also that the difference between y_1 and y_0 is WTA, i.e. a measure of the individual's money valuation of the change in the public good. The size of the difference between WTP and WTA hinges on the magnitude of the income elasticity of WTP, i.e. the responsiveness of WTP to changes in income. Economic theory shows that this elasticity

[20] These qualitative conclusions are derived formally by Hanemann (1999).

depends on the elasticity of substitution between money and the good in question. But, whether this elasticity is large or small is an empirical question.

In empirical studies, it is common to find that *stated* WTA is greater than *stated* WTP. However, the WTP / WTA disparity issue is not specific to SP. For example, a recent review article (Horowitz and McConnell, 1999) shows that there are large differences between WTP in both SP and RP data. The size of the difference does not appear to be related to the SP versus RP distinction. For a recent theoretical paper that postulates why there should be differences between WTP and WTA, see Kolstad and Guzman (1999). One possible explanation is that these disparities are artefacts of the ways in which stated preference questions have been asked – the implication being that such disparities could be greatly reduced by improved survey design. A rather different possibility is that the disparity reflects fundamental limitations in the standard theoretical underpinnings.

In the rare cases in which a policy has such a large impact on individual welfare as to cause a large divergence between WTA and WTP, there are grounds for questioning whether money is an appropriate standard of value. For example, suppose we are trying to value the benefits of a costly medical treatment that, for a small number of identifiable individuals, would eliminate a 20% risk of immediate death. We might expect such an individual to be willing to pay a significant part of her expected lifetime income to gain this benefit; but WTP is inevitably constrained by income. Now suppose the same individual has entitlement to the treatment. What is her WTA for giving up this entitlement and accepting a 20 per cent risk of death? She might be unwilling to accept *any* amount of money as compensation for this risk. Thus WTA may be infinitely greater than WTP. Clearly, however, it would be wrong to conclude that a 20 per cent risk of death is infinitely bad. Some things (like a 40% risk of death) are a lot worse. A more appropriate conclusion is that in this case, money is not a satisfactory standard of value, since it is not seen as a substitute for the benefit in question.

Such a case is exceptional. In principle, the question of whether WTP or WTA valuations should be used in any instance should be addressed by taking one policy option (usually the ‘do nothing’ scenario) as the datum in relation to which costs and benefits are defined. Then benefits, i.e. changes that are more preferred than the datum, should be measured by WTP, while costs, i.e. changes that are less preferred than the datum, should be measured by WTA. However, the chosen measure should be tested in focus groups and pilot surveys before being implemented in the full-scale survey. It is important to test the credibility of the measure since evidence shows that people may not find either measure credible in certain circumstances (e.g. they may not believe that the compensation will ever be paid or the payments they make will be used for the purpose stated in the questionnaire).

C.1.5 Different valuation techniques

(i) Stated versus revealed preference techniques

Soon after the first applications of stated preference techniques (SP), and perhaps motivated by an underlying (although possibly misplaced) trust in revealed preference techniques (RP) (by both decision makers and many economists alike), analysts began comparing hypothetical market estimates (stated preference measures) with those of other non-market valuation methods, most commonly (RP) approaches such as the hedonic pricing or travel cost method[21]. SP measures are the product of hypothetical markets whereas RP measures are based upon observations of actual behaviour. However, this does not mean that RP measures should be treated as criterion values against which SP measures should be assessed.

[21] Early comparisons include Knetsch and Davis (1966), Thayer (1981) and Brookshire *et al.*, (1982). Note that for open-access recreation studies the CV and TC techniques require a heavily overlapping set of predictor variables and the design of a common questionnaire to facilitate the execution of both methods is relatively straightforward.

This arises because:

- The first obvious fact to note is that all studies (SP, and RP) may be subject to bias – this is not an issue restricted to survey techniques;
- The measures produced by RP methods, in particular travel cost methods, are essentially based upon economic interpretations of behaviour and therefore cannot unambiguously claim to be precise reflections of true values. Randall (1994) notes that in travel cost method studies it is the analyst who calculates the travel expenditure and travel time cost against which observed visitation behaviour is modelled. While such an approach may be reasonable for calculating relative value comparisons between recreation sites, i.e. where any errors between analyst-calculated costs and those perceived by visitors will be reasonably consistent, this method may be suspect when estimating the absolute value of visits to any given site for use within CBA. As Randall points out, at least in SP studies respondents are fully aware of the cost amount they are supposed to be reacting to;
- The hypothetical market underpinning SP estimates allows the possibility of disparities between formulated, stated and actual values whereas RP studies concern only actual values, and
- The RP and SP techniques typically address overlapping but not identical value sets. While RP measures are ex-post and exclude non-use values, SP measures are ex-ante, may or may not be based upon direct experience of the level of provision described in the scenario and often embrace both use and non-use values.

High quality SP and RP comparisons seem to have consistently found a satisfactory similarity between value estimates (e.g. Hoehn and Randall, 1985). Such single, within-study, comparisons of, typically, just two measures are relatively weak tests where these estimates have wide confidence intervals. An interesting variant on this approach is presented by Carson *et al*, (1996a) in a meta-analysis of 84 separate studies of quasi-public goods yielding 616 comparisons of CV with revealed preference methods. This analysis found a *high correlation between the CV and other measures, with the former slightly, although significantly, lower on average than the latter*. This result contrasts with the widely held prior belief that CV estimates would exceed revealed preference measures. Although, as indicated above, the relationship between such measures is complex, the fact that these measures were (in most cases) not wildly different does indicate that specific CV estimates cannot be dismissed by such cross method convergent validity tests.

The increasing accessibility of valuation databases such as EVRI (see Annex 1) is one development which may assist in the conduct of cross method convergent validity comparisons.

(ii) Comparisons with simulated markets

Simulated markets are frequently used by *experimental economists* to test hypotheses and a variety of techniques have been devised for designing such markets to be incentive compatible with truth telling. CV researchers have appealed to simulated markets as a method of validating their research findings. However, the major limitation to such attempts (indeed the major problem of all attempts to validate CV results) is that, in all but a few exceptional cases (discussed below), there exist no criterion values for public goods against which either simulated market or CV value estimates may be measured. Many environmental public goods are non-rival and non-exclusive and as such are not paid for in any direct manner. Even for those which are funded through central taxation the level of payment is not individually determined and therefore a valid criterion measure is unobservable.

Given this difficulty, the majority of CV/simulated market testing has concerned quasi-public goods (such as permits for hunting) or private goods with the implicit assumption being that results obtained in such circumstances may say something of relevance regarding the public goods case. There have been a number of surveys of this literature (see, for example, Mitchell and Carson, 1989; Carson *et al*, 1996b, 1999; Fisher, 1996; Hannemann, 1996; Schulze *et al*, 1996; Cummings, 1996). There is some considerable divergence in interpretation of this literature. Balistreri *et al* (forthcoming) argue that while survey dichotomous choice WTP responses typically exceed survey open-ended WTP amounts, the latter prove to be reasonable predictors of actual payments in simulated markets. In contrast Mitchell and Carson (1989), Carson *et al*, (1996b) and Hannemann (1996) argue that the available literature on simulated markets provides a strong endorsement of the validity of CV estimates of quasi public good values in general and dichotomous methods in particular.

However, it is in relation to public goods that the CV debate is fiercest since in the vast majority of cases criterion values are not observable for such goods. In the US the use of real referenda to determine the provision of certain public goods at stated costs provides such a criterion. Furthermore, because the results of referenda are binding, i.e. the provision is made and posted prices are coercively enforced, surveys which emulate the referendum decision can be made incentive compatible. Two such assessments have been carried out to date. Carson *et al* (1987) used a survey referendum duplicating an actual referendum on water quality which was implemented several months after the CV survey. Findings indicated that survey results quite closely predicted the subsequent actual vote. A similar result is reported by Polasky *et al* (1996) concerning a referendum to purchase open-space in Oregon. Although this is only a very small empirical literature, the incentive compatibility of such studies and their public goods nature make their findings of considerable importance.

A larger set of studies has attempted convergent validity assessment of public goods under conditions which are not fully incentive compatible (e.g. Navrud, 1992). A typical example is provided by Foster *et al*, (1997) who compare survey open-ended WTP for wildlife habitat with actual donations to wildlife conservation bodies in respect of such goods. The problem here is that the latter donations are voluntary rather than coercive and are therefore not incentive compatible. In such situations a reasonably consistent pattern can be identified wherein, compared to actual markets, survey markets tend to overstate the willingness of respondents to participate in paying for public goods but provide reasonable predictors of the amounts that those who do participate actually pay. However, given the imperfect incentive properties of such comparisons the implications for the validity of CV estimates are somewhat speculative (Randall, 1996).

(iii) Elicitation method effects

One focus of research in the implementation of stated preference techniques is the choice of 'elicitation method' used in the valuation scenario. This is the key part of a stated preference questionnaire where, after the presentation of the scenario, the provision and payment mechanisms, respondents are asked questions to determine how much they would value the good if confronted with the opportunity to obtain it, under the specified terms and conditions.

The elicitation question can be asked in a number of different ways. Table C-1 (overleaf) summarises the principal formats of eliciting values. The examples in the table all relate to the elicitation of WTP but could easily be framed in terms of WTA.

Table C-1: Examples of common elicitation formats

Open ended	*What is the maximum amount that you would be prepared to pay every year, through a tax surcharge, to improve XXX in the ways I have just described?*
Bidding game	*Would you pay £5 every year, through a tax surcharge, to improve the XXX in the ways I have just described?* If Yes: Interviewer keeps increasing the bid until the respondent answers No. Then maximum WTP is elicited. If No: Interviewer keeps decreasing the bid until respondent answers Yes. Then maximum WTP is elicited.
Payment card	*Which of the amounts listed below best describes your maximum willingness to pay every year, through a tax surcharge, to improve XXX in the ways I have just described?* 0 £0.5 £1 £2 £3 £4 £5 £7.5 £10 £12.5 £15 £20 £30 £40 £50 £75 £100 £150 £200 >£200
Single-bounded dichotomous choice	*Would you pay £5 every year, through a tax surcharge, to improve XXX in the ways I have just described?* (the price is varied randomly across the sample)
Double-bounded dichotomous choice (or bidding game)	*Would you pay £5 every year, through a tax surcharge, to improve XXX in the ways I have just described?* (the price is varied randomly across the sample) If Yes: *And would you pay £10?* If No: *And would you pay £1?*

It should be noted that the different elicitation formats have different properties with respect to their incentives for strategic behaviour, how much information they convey to respondents, and how much information they collect from respondents. As a consequence, *theoretically one should not expect the elicitation formats to result in the same WTP estimates*. Possible biases from the various techniques are discussed overleaf.

Open-ended: The direct open-ended elicitation format is a straightforward way of uncovering values, does not provide respondents with cues about what the value of the change might be (i.e. no *anchoring bias*) is very informative as maximum WTP can be identified for each respondent and requires relatively straightforward statistical techniques. However, it has been progressively abandoned by CV practitioners due to a number of problems. Open-ended questioning leads to large non-response rates, protest answers, zero answers and outliers and generally to unreliable responses (Mitchell and Carson, 1989). This is because it may be very difficult for respondents to come up with their true maximum WTP 'out of the blue' for a change they are unfamiliar with or have never thought about valuing before. Moreover, most daily market transactions involve deciding whether or not to buy goods at fixed prices, rather than stating maximum WTP values.

The **bidding game** was one of the most widely used formats in the 1970s and 1980s. In this approach, as in an auction, respondents are faced with several rounds of discrete choice questions, with the final question being an open-ended WTP question. This iterative format was thought to facilitate respondents' thought processes and encourage them to consider their preferences carefully. A major disadvantage lies in the possibility of anchoring or *starting bias*, that is, respondents were found to be influenced by the starting values and succeeding bids used. It also leads to a large number of outliers (that is, unrealistically large bids) and to a phenomenon that has been labelled as 'yea-saying' (that is, respondents accepting to pay the specified amounts to avoid the socially embarrassing position of having to say no).

Payment card approaches were developed as improved alternatives to the open-ended and bidding game formats. Presenting respondents with a visual aid containing a large number of monetary amounts facilitates the valuation task by providing a context to their bids, while avoiding starting point bias at the same time. The number of outliers is also reduced in comparison to the previous formats. Some versions of the payment card show how the values in the card relate to actual household expenditures or taxes (benchmarks). The payment card is nevertheless vulnerable to biases relating to the range of the numbers used in the card and the location of the benchmarks.

Single-bounded dichotomous choice or referendum methods became increasingly popular in the 1990s. This elicitation format is thought to simplify the cognitive task faced by respondents (respondents only have to make a judgement about a given price, in the same way as they decide whether or not to buy a supermarket good at a certain price) while at the same time providing incentives for the truthful revelation of preferences under certain circumstances (that is, it is in the respondent's strategic interest to accept the bid if his WTP is greater than or equal to the price asked and to reject otherwise, see Carson *et al* (1999) for a detailed explanation of incentive compatibility). This procedure minimises non-response and avoids outliers. The presumed supremacy of the dichotomous choice approach reached its climax in 1993 when it received the endorsement of the NOAA panel (Arrow *et al*, 1993). However, enthusiasm for closed-ended formats gradually waned as an increasing number of empirical studies revealed that values obtained from dichotomous choice elicitation were *significantly and substantially larger than those resulting from comparable open-ended questions*. Such differences between elicitation formats are to be expected. Some degree of yea-saying is also possible, but the problem of nay-saying, typically from protesting an element of the scenario or disbelief that the government can actually provide the good, is likely to characterise a larger fraction of the respondents than is yea saying. In addition, dichotomous choice formats are relatively inefficient in that less information is available from each respondent (the researcher only knows whether WTP is above or below a certain amount), so that larger samples and stronger statistical assumptions are required. This makes surveys more expensive and their results more sensitive to the statistical assumptions made.

Double-bounded dichotomous choice formats are more efficient than their single-bounded counterpart as more information is elicited about each respondent's WTP. For example, we know that a person's true value lies between £5 and £10 if she accepted to pay £5 in the first question but rejected £10 in the second. But all the limitations of the single-bounded procedure still apply in this case. Another problem is the possible loss of incentive compatibility due to the fact that the second question may not be viewed by respondents as exogenous to the choice situation. Finally, anchoring and yea-saying biases can also occur.

Recently, a number of variants of the standard elicitation formats described above have also been proposed in the literature. One issue of concern in CV studies is respondent's preferences or imprecise preferences regarding the change of interest (Ready *et al*, 1995; Wang, 1997; Dubourg *et al*, 1996). Payment ladder approaches are designed to identify the range of values over which individual valuations are uncertain. This approach has been successfully used in a number of recent studies (Day *et al*, 1999; Maddison and Mourato, 1999; Mourato and Day, 1998; EFTEC, 1998b).

Hanemann (1999) proposed a **one and a half bound dichotomous choice** procedure whereby respondents are initially informed that costs of providing the good in question will be between £X and £Y (X<Y), with the amounts X and Y being varied across the sample. Respondents are then asked whether they are prepared to pay the lower amount £X. If the response is negative no further questions are asked; if the response is positive then respondents are asked if they would pay £Y Conversely respondents may be presented with the upper amount £Y initially and asked about amount £X if the former is refused.

Also promising is a **randomised card sorting procedure**, which is essentially a variant of the payment ladder approach described above. Here respondents are shown a pack of cards each depicting a monetary value. Cards are then shuffled in front of the respondent who is then asked to sort the pack into three piles: amounts which the respondent definitely would pay; amounts the respondent definitely would not pay; and amounts about which the respondent is uncertain.

As mentioned above, the choice of elicitation format is of dramatic importance as different elicitation formats typically produce different estimates. That is, *the elicitation format is a non-neutral element of the questionnaire.* Carson (2000) summarises a number of stylised facts regarding elicitation formats. These are depicted in Table C-2.

Table C-2: Elicitation formats: some stylised facts

Open-ended	Large number of zero responses, few small positive responses
Bidding game	Final estimate shows dependence on starting point used
Payment card	Weak dependence of estimate on amounts used in the card
Single-bounded dichotomous choice	Estimates typically higher than other formats
Double-bounded dichotomous choice	The two responses do not correspond to the same underlying WTP distribution

Overall, considering the pros and cons of each of the formats reviewed above, two procedures are currently in favour: payment cards and dichotomous choice formats. Payment cards are more informative and simpler to implement than dichotomous choice and are superior to both direct open-ended questions and bidding games. Dichotomous choice formats may be incentive compatible and facilitate respondents' valuation task. The new variants described (the one and a half bound approach and the randomised card sorting procedure) also show potential although further research is needed before they become established.

Whatever the elicitation format adopted, respondents must be reminded of substitute goods and of their budget constraints and the related need to make compensating adjustments in other types of expenditure to accommodate the additional financial transaction implied by the survey. The former reminds respondents that the good in question may not be unique and that this has implications upon its value. The latter reminds respondents of their limited incomes and of the need to trade-off money for environmental improvements. However, the wording of valuation questions is often not reported in stated preference studies, and it may therefore be difficult to check this aspect of the original study in a benefits transfer exercise.

Finally, since in the valuation of biodiversity damage WTA will usually be the appropriate measure to use, it is worth mentioning some adjustments that have to be made in the arguments presented above when WTA is used rather than WTP:

- first, contrary to what happens when WTP is used, under a WTA format, open-ended elicitation procedures will produce higher average values than dichotomous choice procedures. Open-ended elicitation may also yield very large outliers. In this case, dichotomous choice is the conservative approach; and
- given that WTA measures are not constrained by income, respondents may have a tendency to overbid. Some mechanisms must be found to counteract this tendency. Different approaches may be used to successfully elicit WTA amounts. WTA amounts should provide the same quality of life if the change occurs, not a better one.

C.1.6 Reliability of values over time

The time dimension raises a number of issues for valuation studies. Perhaps the most studied issue concerns the stability of stated values for the same good over time, i.e. the reliability of estimates. Reliability has been assessed both within and across samples. Comparisons across different samples collected using the same survey instrument administered at two points in time indicate that estimates are reasonably reliable. For example, Carson and Mitchell (1993) find that two surveys of national water quality improvement benefits conducted three years apart gave (inflation adjusted) values which were very similar to each other. Similarly the Exxon Valdez study (Carson, *et al*, 1992a, 1994a) was repeated two years after the initial survey yielding both per household values and regression equation coefficients which were almost identical to those originally estimated (Carson *et al*, 1997)[22]. Whitehead and Hoban (1999) administered the same WTP survey involving air and water quality improvements to two separate samples of the same population five years apart and found the estimated valuation function unchanged, even though WTP estimates were different because values of some of the main predictor variables had changed. However, while this suggests that in many cases attitudes towards a good may be reasonably stable, intervening events may shift these attitudes. In some cases these shifts may be merely transitory (e.g. attitudes to transport safety in the wake of an accident) while in other cases these changes may be more permanent (e.g. attitudes towards the gender/employment issue). Therefore, analysts should consider whether consistency or change is to be expected prior to conducting replicability exercises.

[22] The studies by Carson *et al*, (1987) and Polasky *et al*, (1996) discussed previously with respect to convergent validity comparisons with simulated markets also provide strong support for the reliability of CV estimates.

Comparisons taken within samples, i.e. classic test-retest experiments using the same sample of respondents, across different points in time have exhibited reasonable if variable degrees of reliability with correlations in the 0.5 to 0.9 range (see, e.g. Loomis, 1989; Reiling *et al*, 1990; Teisl *et al*, 1995). A number of valid reasons may explain differences in a given individual's answers at different times. As Carson *et al*, (1996b, p34) state; "*Respondents may not give the same answer for many reasons, such as changes in the respondent's financial situation, changes in expenditure opportunities, and perhaps most importantly, a retesting effect.*" In a more ambitious variant of this type of test, McConnell *et al* (1998) interviewed respondents at two different points in the fishing season, and found that the valuation function obtained was similar in both instances. After accounting for the differences in the nature of the fishing opportunities in the second time period, they were able to predict the results of the second interview based upon the first interview.

Another issue concerns the responsiveness of stated values to the specified payment period, i.e. the period over which payments might be made. Empirical studies suggest that significant sensitivity can be observed. However, simple relationships should not be expected. For example, budget constraints and time preference should mean that a lump-sum payment covering 10 years of benefits should be significantly lower than 10 times the annual WTP stated for the same benefits. Again empirical evidence supports such an expectation (see e.g. Bateman *et al*, 1992).

A further issue concerns the stability of stated values with respect to the amount of time respondents are given to consider their response. Empirical studies from the developing world have shown that increasing the amount of time which respondents have can substantially change stated values, typically by reducing them (Whittington *et al*, 1992 and Lauria *et al*, 1999). Such results do not appear to be inconsistent with theory as the extra time can presumably be used to gather extra information or consider other existing expenditure commitments further. If the difference in values is primarily due to such additional information then neither the immediate nor delayed response can be considered invalid. Two caveats should be noted. First, from a relative perspective the decisions that these studies consider are quite large and to be incurred over a long time frame relative to the CV surveys and goods in developed countries. Second, much of what appears to be taking place is an internal household discussion on household priorities for a very large purchase. Disagreement in such a case tends to move the numbers downward. In a developing country, if the commitment to a good offered in a CV survey is smaller, there is likely to be less disagreement over its desirability and there is typically random selection of respondents within the household with the notion that each responsible member can 'vote' independently. However, given that economic theory emphasises the importance of information in decision-making it would seem that the delayed, considered values are preferable for policy use. Where feasible, allowing respondents sufficient time to think seems a desirable feature in any CV study.

There are a number of open issues regarding reliability. Kahneman and Knetsch (1992) question the responsiveness of stated values to the temporal distribution of contingent benefits and costs[23]. Similarly there is a relative lack of studies examining how stated values may respond to scenarios concerning different goods distributed over time (e.g. tradeoffs of current road risks against future acute air pollution mortality effects). Further research into such issues is needed.

[23] Kahneman and Knetsch (1992) found that respondents' WTP answers were unresponsive to significant changes in the length of time to which valuation questions related. However, Carson *et al*, (1992a) do find significant responsiveness in this respect.

C.2 STUDY DESIGN ISSUES FOR STATED PREFERENCE TECHNIQUES

An area of debate in survey design examines what type of questions are likely to deliver useful information even when they are asked in respect of some survey scenario. Carson *et al* (1999) examine this issue by considering whether survey respondents will consider questions of some wider consequence or not. A question may be considered **consequential** if, and only if:

- the respondent feels that their response may influence the actions of relevant agencies; and
- the respondent cares about the outcome.

Only if both conditions hold will the question be consequential. For such questions respondents can be expected to answer on the basis of whatever underlying preferences they hold. Some assessment of validity is, in principle, feasible.

However, if either or both conditions do not hold then the respondent will consider the question to be of no consequence and any response has the same influence on the his/her utility. For consequential questions we can consider two response possibilities:

(i) respondents will answer so as to maximise their expected wellbeing, therefore they will respond to the incentives as set out in the survey design; or

(ii) respondents will answer truthfully irrespective of (i).

If respondents always answer truthfully then the analyst's problem is to verify whether respondents' concepts of the truth in a survey situation correspond with actual payments or compensation amounts demanded. This is no simple task given the absence of markets for public goods. However, the task becomes considerably more complex where (i) is in conflict with (ii), i.e. the respondent feels that the survey market provides some incentive to do other than truthfully reveal their preferences. There are a considerable number of possible situations in which truth telling might not seem the best way in which to maximise expected utility (wellbeing) and these are reviewed briefly here. However, the objective of the CV practitioner is to design a valuation mechanism in which truth telling and utility maximisation coincide. This is the issue of *incentive compatibility* and determining whether a given CV study design is incentive compatible is one of the major foci of validity analysis.

This drive for incentive compatibility can be complicated by what is known as the *face-value dilemma* (Carson *et al*., 1999). Taking answers at face value implies both that respondents always answer truthfully and they also answer the specific question asked. Both assumptions may be suspect (Sudman *et al*, 1996). Survey respondents are not automata and interpret questions in the light of their own prior knowledge and beliefs. For example, if respondents feel that the specified government agencies are incompetent or wasteful, that the specified scenario is implausible, or the good described is unlikely to be as specified or will cost some amount other than that stated, then their response is likely to be conditioned by these opinions. This should not be surprising as it has long been recognised in marketing that beliefs about the reliability of the seller of a good have a strong influence on durable goods purchases. In effect the respondent will be answering a different question to that understood by the analyst and stated values are not commensurate with the face value question.

There are three value types worth distinguishing which are pertinent to any SP experiment. These are defined in Table C-3.

Table C-3: Value types

Formulated Value	This is the WTP or WTA amount that a respondent *genuinely* believes they would be prepared to pay or accept in respect of the provision change scenario presented in a CV survey.
Stated Value	This is the WTP or WTA amount that the respondent tells the interviewer that they would be prepared to pay or accept in respect of the provision change scenario presented in a CV survey.
Actual Value	This is the WTP or WTA amount that the respondent actually does pay or accept in respect of the provision change when it occurs.

These definitions are necessary to allow for the possibility that the corresponding amounts are not all identical. Note first that formulated values are those which respondents hold as being true. However, as discussed above, in certain circumstances respondents may perceive some strategic advantage in misreporting their values within a CV study such that stated value may either exceed or be less than formulated value. Economists argue that this may occur if respondents feel that they can increase their utility through such a strategy. This explains the emphasis upon making CV questions compatible with truth telling such that it is in the respondents interest to ensure that stated value equals formulated value.

As an example, in a survey concerning possible biodiversity protection strategies, a respondent will only consider a valuation question consequential if they feel that their response may influence, say, the decisions made by the European Commission *and* they gain use value from the improved protection of natural areas and/or non-use value from the improved protection enjoyed by others. If they either do not care about the outcome or feel that their response will have no impact upon that outcome then they will consider the question inconsequential. With a coercive payment mechanism, the agent may have an incentive to say 'no' to avoid the loss of having to pay for something they do not care anything about.

If incentives are such that stated values differ from formulated values we would expect the former also to differ from the actual values which respondents would express if given the opportunity to conduct a real exchange. However, it may also be the case that formulated values differ from their actual counterparts. This can happen if time and information change between elicitation of these values. Another reason why this might occur is that respondents may expend greater cognitive effort upon determining actual as opposed to formulated values. Such a problem could, in theory, be overcome through improved study design. However a more fundamental problem may be that, while CV studies provide respondents with information regarding the good under evaluation, they can rarely offer respondents the *experience* of paying or receiving compensation for the good in question. If experience is a fundamental part of determining actual values then a difference between formulated and actual values cannot be ruled out. In many cases, a CV respondent spends more time considering information and the relevant decision than that spent for many on voting decisions or market goods decisions with similar monetary expenditures. As such the divergence may come from the agent spending less time in actual markets. The task of the CV instrument is to provide an unbiased and transparent vehicle which gives respondents the best possible chance to deliberate about their preferences and approach as closely as possible to the values that they would affirm in the light of experience. The instrument must also incorporate whatever validity tests may be useful for examining how far responses are the product of constructed or ‘true’ preferences and how confident we may be of the relationship between stated and actual values.

The principal biases that may occur in SP valuation estimates are outlined in Table C-4. The table adapts the typology of biases from Mitchel and Carson (1989).

Table C-4: Typology of potential biases in SP studies

1. Incentives to Misrepresent Responses:

Biases in this class occur when a respondent misrepresents his or her true willingness to pay (WTP).

A. *Strategic Bias*: where a respondent gives a WTP amount that differs from his or her true WTP amount (conditional on the perceived information) in an attempt to influence the provision of the good and/or the respondent's level of payment for the good.

B. *Compliance Bias*

1. *Sponsor Bias:* where a respondent gives a WTP amount that differs from his or her true WTP amount in an attempt to comply with the presumed expectations of the sponsor (or assumed sponsor).
2. *Interviewer Bias:* where a respondent gives a WTP amount that differs from his or her true WTP amount in an attempt to either please or gain status in the eyes of a particular interviewer.

2. Implied Value Cues:

These biases occur when elements of the contingent market are treated by respondents as providing information about the "correct" value for the good.

A. *Starting Point Bias*: where the elicitation format or payment vehicle directly or indirectly introduces a potential WTP amount that influences the WTP amount given by a respondent. This bias may be accentuated by a tendency to yea-saying.

B. *Range Bias*: where the elicitation method presents a range of potential WTP amounts that influences a respondent's WTP amount.

C. *Relational Bias*: where the description of the good presents information about its relationship to other public or private commodities that influences a respondent's WTP amount.

D. *Importance Bias*: where the act of being interviewed or some feature of the instrument suggests to the respondent that one or more levels of the amenity has value.

E. *Position Bias*: where *the* position or order or sequence in which valuation questions for different levels of a good (or different goods) suggests to respondents how those levels should be valued.

3. Scenario Mis-specification:

Biases in this category occur when a respondent does not respond to the correct valuation scenario. Except in A in the outline that follows, it is presumed that the intended scenario is correct and that the errors occur because the respondent does not understand the scenario as the researcher intends it to be understood.

A. *Theoretical Misspecification Bias*: where the scenario specified by the researcher is incorrect in terms of economic theory or the major policy elements.

B. *Amenity Misspecification Bias*: where the perceived good being valued differs from the intended good.

   1. *Symbolic:* where a respondent values a symbolic entity instead of the researcher's intended good.

   2. *Part-Whole*: where a respondent values a larger or a smaller entity than the researcher's intended good.

      a. *Geographical Part-Whole*: where a respondent values a good whose spatial attributes are larger or smaller than the spatial attributes of the researcher's intended good.

      b. *Benefit Part-Whole:* where a respondent includes a broader or a narrower range of benefits in valuing a good than intended by the researcher.

      c. *Policy-package Part-Whole*: where a respondent values a broader or a narrower policy package than the one intended by the researcher.

   3. *Metric*: where a respondent values the amenity on a different (and usually less precise) metric or scale than the one intended by the researcher.

   4. *Probability of Provision:* where a respondent values a good whose probability of provision differs from that intended by the researcher.

C. *Context Misspecification Bias:* where the perceived context of the market differs from the intended context.

1. *Payment Vehicle*: where the payment vehicle is either misperceived or is itself valued in a way not intended by the researcher.
2. *Property Right*: where the property right perceived for the good differs from that intended by the researcher.
3. *Method of Provision*: where the intended method of provision is either misperceived or is itself valued in a way not intended by the researcher.
4. *Budget Constraint*: where the perceived budget constraint differs from the budget constraint the researcher intended to invoke.
5. *Elicitation Question*: where the perceived elicitation question fails to convey a request for a firm commitment to pay the highest amount the respondent will realistically pay before preferring to do without the amenity. (In the discrete-choice framework, the commitment is to pay the specified amount.)
6. *Instrument Content*: where the intended context or reference frame conveyed by the preliminary nonscenario material differs from that perceived by the respondent.
7. *Question Order*: where a *sequence* of questions, which should not have an effect, does have an effect on a respondent's WTP amount.

Guidelines for the appropriate design of SP questionnaires, including incentive-compatibility considerations are given in NOAA (1993) and EFTEC (forthcoming 2001). However, while these guidelines are useful in the design of original SP studies, in practice it may be difficult to assess the extent of possible biases in existing studies, as this depends largely on the analysis and reporting of results by the authors. This complicates the task of study assessment for use in benefits transfer applications,

ANNEX D: OVERVIEW OF THE USA PROCESS FOR NATURAL RESOURCE DAMAGE ASSESSMENT

This annex aims to give an overview of Natural Resource Damage Assessment (NRDA) in the USA. This annex begins with an overview of the relevant legislation in the USA in Section D.1. This is followed by a summary of the relevant US guidelines for conducting NRDAs in Section D.2. Finally, Section D.3 outlines practical experience with the implementation of the liability regime and NRDAs, highlighting the lessons learned and the changes in implementation over time.

D.1 USA LEGISLATION

There are several pieces of legislation in the USA which endorse the estimation and recovery of damage to natural resources. Of these, two have received the lion's share of attention due to high profile cases: Superfund – the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) of 1980, and the Oil Pollution Act of 1990. Both of these provide for damage recovery, including the recovery of non-use values using stated preference techniques, among other methods. However, other federal laws also provide for damage recovery, and thus may implicitly authorise use of these techniques (Breedlove, 1999; DARP, 1999). These include the National Marine Sanctuaries Act, the Clean Water Act (which authorises the government to act as natural resources trustee to recover damages – originally equal to restoration costs – for hazardous discharges into navigable waters or near the coastline), the Deepwater Port Act, the Trans-Alaska Pipeline Act and the Outer Continental Shelf Lands Act (all of which provide for the recovery of damages). Some state laws also allow damage recovery and provide various types and levels of coverage. Extracts from the OPA, on which the NRDA guidelines are based, are given below.

990.10 Purpose

The goal of the Oil Pollution Act of 1990 (OPA), 33 U.S.C. 2701 et seq., is to make the environment and public whole for injuries to natural resources and services resulting from an incident involving a discharge or substantial threat of a discharge of oil (incident). This goal is achieved through the return of the injured natural resources and services to baseline and compensation for interim losses of such natural resources from the date of the incident until recovery. The purpose of this part is to promote expeditious and cost-effective restoration of natural resources and services injured as a result of an incident. To fulfil this purpose, this part provides a natural resource damage assessment process for developing a plan for restoration of the injured natural resources and services and pursuing implementation or funding of the plan by responsible parties. This part also provides an administrative process for involving interested parties in the assessment, a range of assessment procedures for identifying and evaluating injuries to natural resources and services, and a means for selecting restoration actions from a reasonable range of alternatives.

990.11 Scope

The Oil Pollution Act of 1990 (OPA), 33 U.S.C. 2701 et seq., provides for the designation of federal, state, and, if designated by the Governor of the state, local officials to act on behalf of the public as trustees for natural resources and for the designation of Indian tribe and foreign officials to act as trustees for natural resources on behalf of, respectively, the tribe or its members and the foreign government. This part may be used by these officials in conducting natural resource damage assessments when natural resources and/or services are injured as a result of an incident involving an actual or substantial threat of a discharge of oil. This part is not intended to affect the recoverability of natural resource damages when recoveries are sought other than in accordance with this part.

D.2 GUIDELINES FOR NATURAL RESOURCE DAMAGE ASSESSMENT

This section outlines the process for conducting natural resource damage assessments (NRDAs) in the USA. It draws heavily on NOAA (1997), which provides detailed guidance for each step in the damage assessment process. The NOAA guidelines for NRDA are available on the internet at http://www.darcnw.noaa.gov/opa.htm.

According to the Oil Pollution Act (OPA) the costs of implementing a restoration plan form the basis of a damage claim. The statutory goal of a restoration plan is to restore natural resources to baseline (primary restoration) and compensate the public for interim losses from the time of injury until they return to baseline (compensatory restoration).

The USA guidance for conducting natural resource damage assessments outlines five steps in the decision-making process for developing and scaling restoration projects. These are summarised in Box D.1.

While guidelines are provided for each step in the NRDA process, the focus of this section is on the classification, selection and scaling of restoration projects. In particular, emphasis is on the economic components, i.e. the role of cost-benefit analysis, cost-effectiveness analysis, and the scaling procedures.

D.2.1 Review preliminary restoration objectives

This section is based on text drawn from NOAA (1997). The purpose of this phase is to evaluate potential injuries to natural resources and services. The review aims to provide the following information:

- preliminary identification of natural resources and services that have been injured or lost; and
- preliminary identification of the degree, spatial and temporal extent of the injury, including a determination of the potential recovery period.

This process involves both the estimation of *baseline conditions*, and *injury assessment*. The baseline conditions are defined as

> *'... the condition of the natural resources and services that would have existed had the incident not occurred'.* (OPA regulations).

Although injury quantification requires a comparison to a baseline condition, site-specific baseline information that accounts for natural variability and confounding factors prior to the incident is often difficult to obtain and may not be required. In many cases, injuries can be quantified in terms of incremental changes resulting from the incident, rather than in terms of absolute changes relative to a known baseline. For example, counts of birds killed by an oil spill can be used to quantify incremental bird mortality resulting from an incident, thereby providing the planning base for restoration.

The OPA regulations do not distinguish between baseline, historical, reference or control data. Types of information that may be useful in evaluating baseline include:

- information collected regularly in the area of the incident both before and after the incident;
- information identifying historical patterns or trends in the area of the incident and injured natural resources;
- information from areas unaffected by the incident, that are judged sufficiently similar to the area of the incident with respect to the parameter being measured; or
- information from the area of the incident after particular natural resources or services have been judged to have recovered.

Injury assessment requires trustees to assess damages for 'injury to, destruction of, loss of, or loss of use of' natural resources. Injury can include adverse changes in chemical or physical quality, or viability of a natural resource (e.g. direct, indirect or delayed effects). Potential categories of injuries include adverse changes in:

- survival, growth and reproduction;
- health, physiology and biological condition;
- behaviour;
- community composition;
- ecological processes and functions;
- physical and chemical habitat quality or structure; and
- services to the public.

The definition of injury under OPA regulations is quite broad, including changes in biota but also injuries to non-living natural resources (e.g. oiled sand on a recreational beach) as well as injuries to services (e.g. lost use associated with a fisheries closure to prevent harvest of tainted fish, even though the fish themselves may not be injured).

D.2.2 Identifying Restoration Options

This section is based on text drawn from NOAA (1997). To identify restoration alternatives, the trustees may consult a variety of sources to ensure that they consider a comprehensive set of actions. Available sources include planning/management agencies and the general public. The first steps in the restoration selection process are to review preliminary restoration objectives from the injury process and to then identify possible restoration actions or projects. Each restoration option must be designed so that, as a package of one or more actions, the option would satisfy the OPA's objective to restore natural resources and services to baseline and compensate for the interim losses resulting from an incident. Incident-specific restoration objectives are developed by identifying the key characteristics and quality attributes of the natural resources and services lost due to the incident. This information is generated in the injury assessment process.

The OPA regulations identify criteria for selecting a preferred restoration option from the options under consideration. Factors to be considered include:

- the cost of the option;
- the extent to which each option is expected to meet the trustees' goals and objectives in returning the injured natural resources and services to baseline and/or compensating for interim losses;
- the likelihood of success of each option;
- the extent to which each option will prevent future injury as a result of the incident, and avoid collateral injury as a result of implementing the option;
- the extent to which each option benefits more than one natural resource and/or service; and
- the effect of each option on public health and safety.

These criteria are used, along with incident-specific restoration objectives, to guide the development of restoration options. Trustees may decide to add to these criteria, depending on applicable laws, regulations, or other site-specific or case-specific requirements.

If the trustees conclude that two or more options are equally preferable based on these factors, the trustees must select the most cost-effective of the two or more equally preferable options.

This step applies equally to the identification of primary and compensatory restoration options. In practice, the most favourable scientific option for primary restoration may be 'no intervention'. In particular, for oil spills, it has often been the case that following the emergency response and clean-up, the preferred primary restoration option has been natural recovery. Identification and choice of compensatory measures, however, is often more complex.

D.2.3 Classifying and Selecting Compensatory Restoration Options

This section is based on text drawn from NOAA (1997). In the classification process, projects are classified by whether or not they provide services of the same type and quality and of comparable value to the services lost due to the injury. The classification criteria are useful in determining which restoration options should be considered for scaling. Selection criteria are then used to select a preferred option from among the scaled options.

There are four possible outcomes of this classification process. Starting with the most desirable class, they are:

Class I: Same type, same quality and comparable value;

Class II: Same type, same or different quality and *not* of comparable value;

Class III: Comparable type and quality; and

Class IV: *Not* of comparable type and quality.

The aim of the classification process is to evaluate how well the injured natural resources and services match the replacement natural resources and service on key characteristics and quality attributes. Even when a proposed action provides the same type of natural resources and services, a variety of substitutions (in time, space, species, etc) may be unavoidable. The result will be differences – in quality, economic value, and in populations who experience the service losses and those who experience the gains provided by the restoration options.

The key questions to be considered in this evaluation are discussed below. A decision-making tree summarising the evaluation process is presented in Figure D.1.

- *Question A. Does the option provide the same type of resources and services?*

This is the first question to be addressed by the trustees. It involves a determination of whether natural resources and services – both on and off-site – that are increased or enhanced by the option are of the same type as those lost.

Two judgements are required here. First, trustees must identify the key services provided by injured natural resources at baseline. These may include ecological services (e.g. hydrological, habitat, nutrient cycling, primary and secondary productivity) and human services (such as recreation, commercial opportunities, cultural/historic use and non-use services).

Second, trustees must determine whether the option may increase site *capacity* to provide the same type of services as those that were lost. However, an increase in the capacity to provide services does not necessarily result in an increase in the services provided. Trustees must therefore also evaluate whether the features of the landscape context at the restoration site suggest that the *opportunity* to provide the same type of services exists. For example, will the action increase public value by either increasing the quantity of uses (services) or enhancing the quality (or reducing the cost of access) or current uses?

If natural resources are judged to be of the same type, trustees move on the question B; otherwise, they turn to question AA.

- *Question AA: Does the option provide resources of comparable type and quality?*

When proposed restoration actions do not provide natural resources of the same type as those injured, natural resources and services that are similar or complementary to the injured resources may be considered and classified as 'comparable type and quality'.

For example, consider a pollution incident resulting in lost beach use. A compensatory option to create an off-shore reef for recreation-based snorkelling and fishing, where none was available before, would expand the range of water-based activities available at the site. While the option does not provide the same type of services as those lost, they might be considered of comparable type and quality.

The determination ultimately requires the trustee to exercise judgement. If the option is determined to provide natural resources and services of comparable type and quality as those lost, the action is classified as Class III.

- *Question B: Does the option (providing resources of the same type) also provide resources of the <u>same quality</u>?*

This question follows on from question A. For natural resources and services of the same type as those lost, the relative quality of these resources must then be assessed.

In order to compare type and quality of services, trustees must select a metric, or an index of metrics, to quantify services. Proxies, which may capture a range of services, are often employed in practice. For example, salmon populations may reflect the health of many other aspects of an ecosystem. However, the relationship between the proxy metric and service levels needs to be carefully considered, as it may not be a simple linear relationship. For example, stem (vegetation) density may serve as a resource-based proxy for primary productivity. However, in many cases a minimum threshold of stem density must be met before secondary productivity gains occur, and there may be another threshold of stem density at which productivity gains per stem decline at an increasing rate.

Consideration must be given to quality factors of both ecological services (e.g. natural resource density, genetic diversity, species diversity, and water, land or air pollution levels) and human services (e.g. access costs, diversity of activities, congestion, isolation, level of development).

If trustees determine that quality, as captured by the selected metric, is different between injury and restoration sites, it may be possible to *adjust the metric* or to choose a different metric to capture the quality differences. Consider the case where one of the proposed restoration sites has a diversity of on-shore or off-shore habitat (sand dunes, off-shore reefs, seagrass beds, etc.) providing a range of recreational activities (bird watching, snorkelling, fishing, etc.) and the other site has only sandy beaches. If all other quality attributes were equal between the two sites, the site with a wider range of habitat and recreational opportunities is likely to have a higher value for an additional beach trip. Using *valuation methods*, trustees may calculate an adjustment factor to capture the greater relative value of the higher quality/lower cost sites.

*If the chosen metric does capture quality differences, or may be adjusted to capture these differences, then options may be classified as the same quality. In this case, trustees turn to question C. If not, then the options are of the same type and comparable quality, in which case the proposed action is classified as **Class II**.*

- *Question C: Does the option provide services of comparable value?*

If projects provide services of the same type and quality as those lost, trustees answer Question C. This question guides the assessment of whether the simplifying assumption that the lost and restored services are of comparable value reasonable by directing attention to two potential causes for non-comparable values: differences in the aggregate supply or demand conditions. Evaluating the possible differences requires trustees' judgement, because the restored services and the future aggregate supply and demand conditions are not observable when compensatory restoration actions are being classified.

For example consider a recreational context, in which the closure of hiking trails in a wildlife refuge due to oiling results in a substantial loss of use. If demand for hiking was fully satisfied but the original quality of trails, a compensatory restoration action to add more hiking trails after the oiled trails are reopened will not add much additional value.

The smaller the injury and restoration action(s), the less likely it is that the change in aggregate supply of natural resources is significant, and consequently the less likely that the value of the last available unit of natural resources and services will change.

If trustees determine that the proposed restoration action will provide natural resources and services of the same type and quality and of comparable value as those injured, the proposed action is classified as ***Class I.***

If the lost and replacement services are of the same type but the values are not comparable because of differences in aggregate supply and/or demand, then the proposed action is classified as ***Class II.***

The OPA regulations place a priority on compensatory restoration actions that provide natural resources and services of the same type, quality, and of comparable value to those lost or impaired. Selection of options is done according to classification, with Class I options being the most preferred, and the rest following in numerical order.

D.2.4 Choosing a Scaling Approach

This section is based on text drawn from NOAA (1997). After identifying and classifying the restoration actions, the next stage of the restoration process is selecting approaches and methods for scaling the restoration alternatives. The process of 'scaling' a compensatory restoration option involves adjusting the size of the action such that the *present discounted value of the project gains* equals the *present discounted value of the interim losses*. The two major scaling approaches are the *service-to-service* approach and *valuation*.

Both approaches frame the scaling question in terms of what trade-offs exist between services lost due to the injury and services provided by potential compensatory restoration actions. However, the valuation approach is based on quantitative estimation about the trade-offs people make between services, whereas the service-to-service approach is based on simplifying assumptions about these trade-offs. Specifically, the implicit assumption of the service-to-service approach is that the public is willing to accept a one-to-one trade-off between a unit of lost services and a unit of services provided by a restoration project. This may be appropriate when the proposed restoration action provides services of the same type and quality, and of comparable value, as those lost due to the injury (i.e. options classified as Class I , as discussed in Section D.2.2).

According to the OPA regulations, trustees must consider using a *service-to-service approach.* However, where the assumption of a one-to-one trade-off does not apply, trustees are to consider the *valuation approach*. This approach relies on the assumption that last value can be determined using one of a variety of possible units of exchange, including natural resource units or dollars. Some stated preference methods, such as choice experiments, are flexible enough to elicit trade-offs between lost and replacement services in terms of dollars or in terms of natural resource services.

One final approach is the *value-to-cost approach*, where the restoration actions are scaled by equating the cost of restoration to the value (in dollar terms) of losses due to the injury. This approach may be used where valuation is possible, but would impose unreasonable time or cost requirements. This will generally occur where literature values from previous research are available to value lost services but are not available to value the gains from restoration actions.

Criteria for selecting a scaling approach are also given in the guidance documents. The OPA regulations require assessment procedures to comply with the following standards:

Figure D.2 (at the end of this Annex) presents a flow chart of the decision-making process for the selection of scaling options.

- *Applicability:* This pertains primarily to the choice of an approach. If an action provides services of the same type, quality, and comparable value as lost services, then the assumptions of the service-to service approach seem reasonable. Otherwise, the valuation approach should be considered.
- *Incremental Cost:* The criteria of reasonableness of incremental costs and of reliability and validity (discussed below) need to be considered together. The information gains of more complex approaches and methods are to be weighed against any expected increase in costs and the expected change in quality and quantity of information. Some methods may provide better information at a lower cost, e.g., construction of a unified model that values the losses due to the injuries as well as the benefits from all the restoration projects under consideration.
- *Validity and reliability:* This criteria requires that scaling methods used to implement an approach be consistent with the best technical practices appropriate for the level of precision required in the context. Validity pertains to the accuracy and completeness of the measurements of the specific concepts of interest. Reliability pertains to the precision and replicability of those measurements.

D.3 PRACTICAL IMPLEMENTATION OF NRDA IN THE USA

This section outlines the basic process of Natural Resource Damage Assessment (NRDA) in the USA, highlighting how the practical implementation of NRDA and the liability regime has altered over time.

Incidents covered by the liability regime include both one-time events (e.g., the *Exxon Valdez* oil spill), or the release of contaminants over a long period of time (e.g. the *Blackbird Mine* case discussed throughout the main text, where mining began in the 19th century with "new" contamination occurring each time precipitation releases contaminants from spoil piles in the basin).

The responsibility for undertaking NRDAs rests with public trustees in the USA. The Department of the Interior (DOI) and the National Oceanic and Atmospheric Administration (NOAA) are among the more prominent Federal trustees. Other examples include agencies of Federal, state, or local governments, and Native American Tribes. DOI and NOAA have co-ordinated their efforts to develop protocols for NRDA performed on behalf of trustees. The recommended procedures are outlined in more detail in Section D.2, and the guideline documents are available at http:// www.darcnw.noaa.gov/opa.htm.

Following an incident, baseline resources and services are catalogued and projected into the future. In practice, this step is done retroactively, after the incident occurs. After any incident, there are three types of response, and a typical injury involves all three:

- *Remediation*, which cleans-up the contamination, stops its spread, etc. Remediation is the emergency response necessary following an incident, and precedes NRDA. Costs of remediation are always included in liability for damages;
- *Primary restoration* (PR), which restores the injured resource and its flow of services. Primary restoration, in the default case, is complete. However, an exception may be granted if the costs of PR are grossly disproportionate to its benefits. Such a claim would bring cost benefit analysis (CBA) into play, but the responsible party has to demonstrate costs grossly disproportionate to benefits in order to get relief; and
- *Compensatory restoration* (CR), which compensates for interim lost use (ILU). Note that "interim" lost use will continue indefinitely if PR is incomplete.

The statutory goal of a restoration plan is to restore natural resources to baseline (primary restoration) and compensate the public for interim losses from the time of injury until they return to baseline (compensatory restoration). **The liability regime is aimed at restoring injured natural resources and economic assessment of natural resources damage is mostly about determining the amount of CR required.** Compensatory restoration may be on-site or off-site, and may enhance resources similar to or different from those injured. These possibilities complicate the task of determining the appropriate scale of CR.

D.3.1 Practical experience with the liability regime

The practical implementation of the regime has altered significantly over time. In particular, a shift in emphasis occurred in the mid-1990s, with respect to approaches to determining the scale of compensatory restoration.

Early 1990s:

In the early 1990s, economic assessments of natural resources damage were conducted with the *objective of determining a money value of damage* that, if paid as compensation, would make the public whole again. This process involved applying the theory and methods of welfare change measurement, and often made use of monetary valuation techniques. The money amount of liability included the costs incurred by the public trustees in assessing the damage, and the *value* of the appropriate scale of compensatory restoration.

Following the economic assessment of damages, restoration options would be evaluated and a restoration plan (RP) determined. Ideally, the RP would be selected as the least-cost suite of restoration activities that would make the public whole again. A good deal of practical politics was often inherent in this step, given that compensatory restoration could include off-site restoration and resources different in kind from those injured, and the constituencies for these (among the public and within the trustee agencies) may be quite different.

The dollar value of ILU (as determined by the courts or agreed in a settlement) would be paid by the responsible party to the trustees. These monies would be spent on implementing the restoration plan, until the money ran out. There was, in principle, no assurance that the RP will be completely implemented: the money could run out first, indicating that the costs of the RP in fact exceeded the benefits. Alternatively, the responsible party would always maintain the option of directly implementing the RP, and would do so if it were able to complete the RP at a cost less than the value of ILU.

In this process, CBA would be used in different contexts as follows:

- If it could be shown that the costs of primary restoration were grossly disproportionate to the benefits, incomplete primary restoration may be permitted. The responsibility for demonstrating this rested with the party responsible for the damage; and
- Compensatory restoration would be scaled by determining the *value* of ILU, and implementing a restoration plan that provides as much compensatory restoration as possible within a budget limited to the value of ILU.

Since the Mid-1990s:

The procedures for NRDA have altered somewhat since the mid-1990s. In the current guidelines for NRDA, and the applicable legislation, there is a pronounced tilt towards resource compensation and the resource-to-resource (or service-to-service) approach to determining the scale of compensatory restoration.

Economic assessments of natural resources damage are conducted with the *objective of determining the scale of compensatory restoration that would make the public whole* again. In general, the preferred approach for determining the scale of compensatory restoration is resources-for-resources (or service-to-service) compensation, where possible. However, it should be noted that this does not eliminate welfare-economic considerations. Rather, the welfare-economic task is to determine the welfare-restoring scale of compensatory restoration, where CR could be considered an in-kind payment (rather than a money payment) to compensate the public. Economic techniques, such as choice modelling, which may be used to determine trade-offs between different resources, or between resources and money, are therefore relevant for the assessment procedures.

Monetary valuation procedures are still used when there are no appropriate compensatory restoration options which meet the requirements of ‘Class I’ actions (see Section D.1). Current guidance documents from trustee agencies continue to include the full suite of non-market valuation methods among those acceptable for NRDA. In practice, the service-to-service and value-to-cost methods are used more often than value-to-value methods in this context.

Following this assessment, restoration options are then evaluated and a restoration plan (RP) is determined. Again, ideally, the RP is the least-cost suite of restoration activities that would provide the appropriate scale of CR. A good deal of practical politics is inherent in this step, given that compensatory restoration may include off-site restoration and resources different in kind from those injured, and the constituencies for these (among the public and within the trustee agencies) may be quite different.

Again, the responsible party may seek relief on the grounds that the costs of providing the appropriate scale of compensatory restoration are grossly disproportionate to the benefits. Such a claim would trigger an economic evaluation of the benefits of compensatory restoration, and may lead to modification of the restoration plan.

The restoration plan selected is then implemented to provide the appropriate scale of compensatory restoration (as determined by the courts or agreed in a settlement).

In this process, CBA may be used in different contexts as follows:

- If it can be shown that the costs of primary restoration are grossly disproportionate to the benefits, incomplete primary restoration may be permitted. The responsibility for demonstrating this rested with the party responsible for the damage;
- Welfare-economic principles are implemented in determining the appropriate scale of compensatory restoration (viewed as an in-kind compensating payment); and
- If it can be shown that the costs of appropriately-scaled compensatory restoration are grossly disproportionate to the benefits, a modified restoration plan is developed.

In simple cases, dealing with modest injuries to homogeneous resources, scaling is a relatively straightforward matter. Unsworth and Bishop (in Randall, 1997) dealing with a few acres of damaged wetlands, assume that restored wetlands will be homogeneous to injured wetlands and, from that point, scaling is largely a matter of determining the time-path of resource recovery and applying the appropriate discount rate.

For larger and more complicated injuries, methods such as choice experiments are appropriate. However, it is important to note that such methods, while promising, have yet to be validated in large-scale application under litigation conditions.

D.3.2 Cost implications

Economic assessment of natural resources damage is relatively inexpensive for small injuries. Various versions of benefits transfer are used, including the standard models developed for “Type A” damage assessment for small coastal and lake spills. At the other extreme, the “Montrose” (Southern California Bight) CV studies for damage assessment cost about $8 million. This case was recently settled out of court for $160 million.

Costs for natural resource damages and assessment have attracted much attention in the USA since the Exxon Valdez spill and the Oil Pollution Act of 1990, which extended liability to cover natural resource damages. A recent paper (Helton and Penn, 1999) explores the total private and social cost of oil spills in the USA, with a particular focus on costs of natural resource damages and their assessment. The authors emphasise that publicly available cost data for oil spills are often limited to public response costs and natural resource damage costs. By contrast, much of the private cost data, such as private response costs, third party claims and vessel repair costs, is not in the public domain. The authors suggest that it is failure to consider all elements of the total cost of spills which has created a false public perception that natural resource damage assessment (NRDA) costs comprise the major portion of the overall cost of spills.

The study examines 30 incidents across the USA between 1994 and 1997, which were selected on the availability of cost data. As such, the data are skewed towards larger incidents, where natural resource damage claims occurred.[24] For each incident examined, data were collected on costs of: response (public and private); assessment; natural resource damages; third party claims; penalties; and other costs. It appears that the data typically

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[24] The authors note that there are between 5,000 and 10,000 oil spills reported along the US coastline every year, and natural resource damage assessments are conducted in less than 1% of these cases.

exclude litigation costs, which would likely add significantly to the total costs.[25] Difficulties in obtaining data on costs meant that NRDA costs were the only category of costs where data were fully available[26], with the implication that this element is biased upwards as a percentage of total costs. Even given these biases, their data show that:

> *'...contrary to the perception, costs for natural resource damages and assessment comprise only a small portion of total liability from an oil spill.' (p.1)*

While for individual cases, NRDA costs can be a major component of total costs of a spill, in the authors' data set they averaged approximately 26% of the *known* costs of an incident. *Costs of assessment*, including the costs of an original valuation study, on average *account for 3% of total known costs*. With full cost information, this proportion would no doubt be lower. Viewed in this light, it may be that commissioning original studies can be justified, particularly where 'off the shelf' values are inadequate, the BT process is likely to be challenged, or where damages to natural resources are large.

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[25] An estimate in the *White Paper on Environmental Liability* puts transaction costs – mainly legal fees – at approximately 20% of total enforcement and compensation costs, although the source of this estimate is not clear.

[26] Data in each of the cost categories were available in only three of the incidents examined. Difficulties included: confidentiality of private costs and private settlements; unknown costs; government costs which are dropped from a case to facilitate settlement negotiations.

Box D-1: Outline of the USA NRDA Process

1. Review Preliminary Restoration Objectives

This is the starting point in an NRDA, which aims to assess injury to natural resources and the loss or impairment of the ecological and human services they support. The review aims to provide the following information:

i. preliminary identification of natural resources and services that have been injured or lost; and

ii. preliminary identification of the degree, spatial and temporal extent of the injury, including a determination of the potential recovery period.

With this information, trustees may define restoration objectives in terms of specific resources and services to be restored or replaced.

2. Identify Possible Restoration Actions

The aim of this step is to identify a range of possible primary and compensatory actions that address restoration objectives.

3. Classify Restoration Actions

Restoration actions are classified according to whether they provide services of the same type, quality and value as those that were lost due to injury. Services considered include geo-hydrological, habitat, recreation, commercial, cultural, health and passive uses. The classification of restoration actions serves two purposes:

i. Prioritising compensatory restoration actions: the OPA regulations place a priority on compensatory actions which provide resources and services of the same type and quality and of comparable value. If this is not possible, actions which provide services and resources of at least comparable type and quality may be considered; and

ii. Selecting a suitable approach for scaling: the type, quality and value of the services provided by restoration has implications for the choice of scaling method (see points 4 and 5 below).

4. Scaling of Primary Restoration Actions

For primary restoration, this addresses the question of what scale of primary restoration is necessary to return the stock of resource and service flows to baseline levels in a timely manner. Once primary restoration actions are selected, this allows quantification of the extent and duration of injury, i.e. estimation of interim losses, which informs the analysis of compensatory restoration actions (see step 5).

5. Scaling of Secondary Restoration Actions

The relevant question to be addressed here is: what scale of compensatory restoration action is necessary to compensate for the interim loss of natural resources from injury until full recovery? Scaling in this case involves adjusting the size of the action to ensure that (present discounted) gains from the action equal the (present discounted) losses from the injury. Scaling requires:

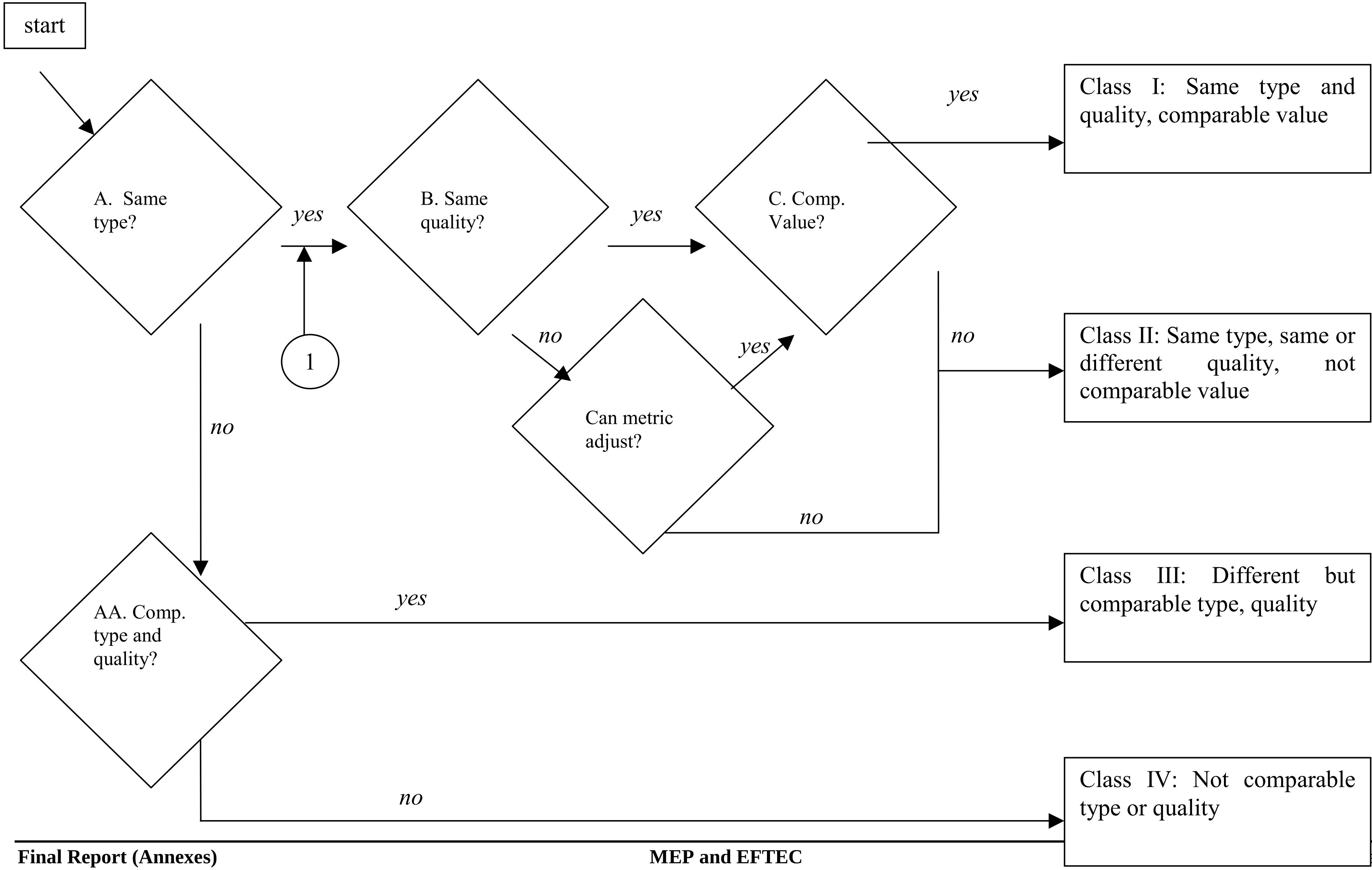
i. Quantifying the extent and duration of service losses;

ii. Quantifying the extent and duration of gains for different scales of compensatory action; and

iii. Determining trade-offs between services lost due to injury and gains from restoration actions.

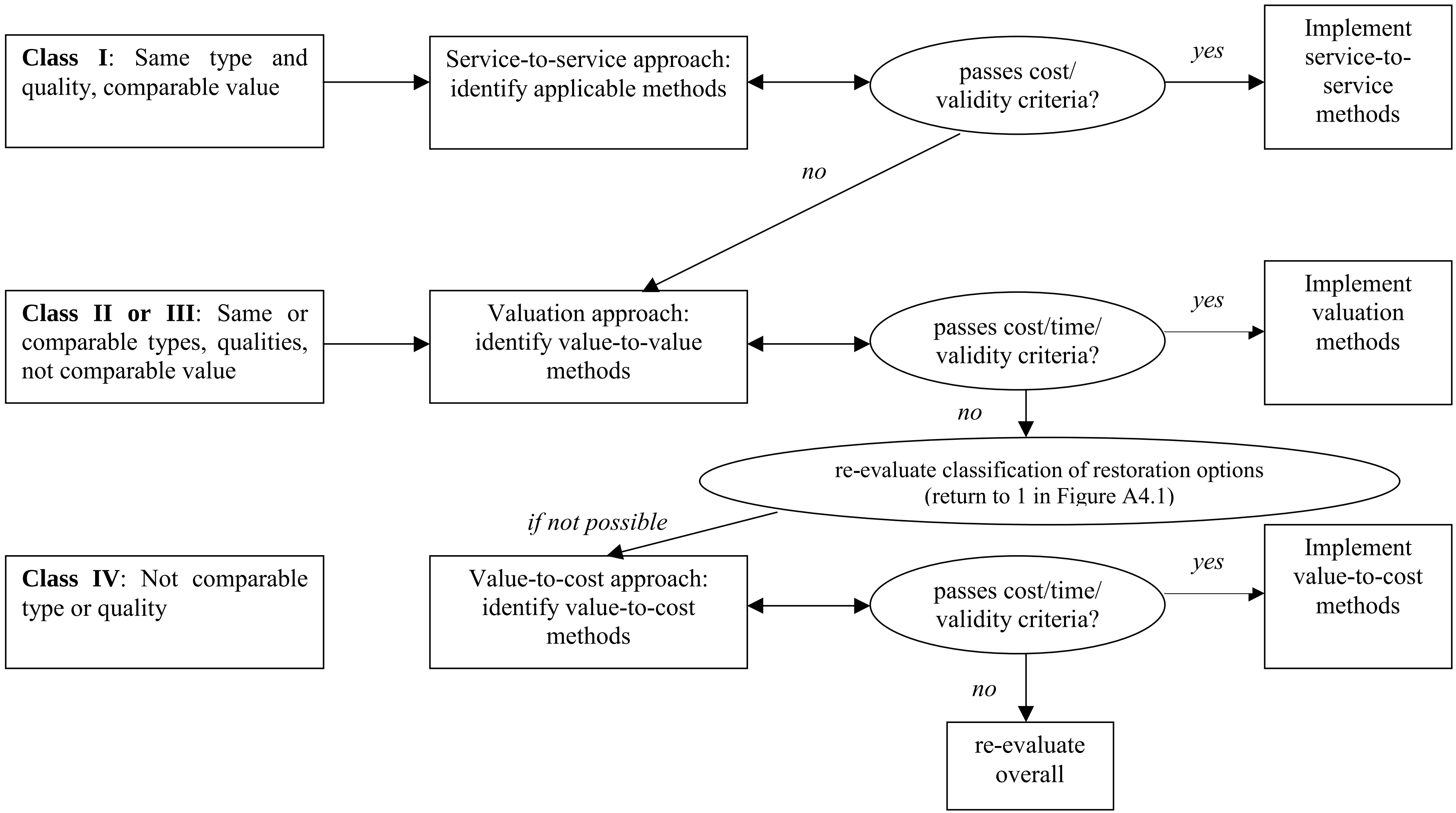
Source: NOAA (1997)

Figure D-1: Classifying Restoration Actions



1 starting point for re-evaluating Class II and Class III restoration actions

Figure D-2: Selecting Scaling Approaches and Methods



ANNEX E: NATURA 2000 STANDARD DATA FORM

Central to the success of NATURA 2000 is the level of information on habitats and species of Community interest which will be assembled during the coming years. Experience in data collection in Europe has been built up through the CORINE biotopes project, which at present describes over 6000 sites in the European Union. The base for the core data fields incorporates this experience, amended and expanded in the framework of the directives concerned.

As the sites classified under the "Birds" and the "Habitats" directives will together form NATURA 2000, a common baseline for both types is essential to achieve the objective of creating a coherent network. The **standard data-entry form** takes all aspects of both directives into account and there is only a need for one form. All data fields from the existing data sheet for the 'Birds' directive are fully compatible with the new entry form. So, where the data from the 1100 Special Protection Areas (SPAs) exist, they can be transferred automatically. This form will be used for all sites designated as SPAs under the Birds Directive. As regards the Habitats Directive it will initially be used to supply the necessary information for sites eligible for identification as Sites of Community Importance (SCIs) in application of Article 4.1 of the Directive (Stage 1).

The legal basis for providing the data to implement this phase of NATURA 2000 is outlined in article 4 of the Habitats Directive which defines that 'information shall include a map of the site, its name, location, extent and the data resulting from application of the criteria specified in Annex III (Stage 1) provided in a format established by the Commission in accordance with the procedure laid down in Article 21'. Under Article 4 paragraph 3 of the Birds Directive Member States are already required to 'send the Commission all relevant information so that it may take appropriate initiatives with a view to the coordination necessary to ensure that the areas provided for in paragraph 1 and 2 (of Article 4) form a coherent whole which meets the protection requirements of these species in the geographical sea and land area where this Directive applies.

The main objectives of the database are :

1. to provide the necessary information to enable the Commission, in partnership with the Member States, to co-ordinate measures to create a coherent NATURA 2000 network and to evaluate its effectiveness for the conservation of Annex I habitats and for the habitats of species listed in Annex II of Council Directive 92/43/EEC as well as the habitats of Annex I bird species and other migratory bird species covered by Council Directive 79/409/EEC;
2. to provide information which will assist the Commission in other decision making capacities to ensure that the NATURA 2000 network is fully considered in other policy areas and sectors of the Commission's activities in particular regional, agricultural, energy, transport and tourism policies;
3. to assist the Commission and the relevant committees in choosing actions for funding under LIFE and other financial instruments where data relevant to the conservation of sites, such as ownership and management practice, are likely to facilitate the decision making process; and
4. to provide a useful forum for the exchange and sharing of information on habitats and species of Community interest to the benefit of all Member States.

The Standard Data Forms for the following three areas are attached overleaf:

i. the Parque Nacional de Doñana

ii. Carmarthen Bay and Estuaries

iii. Pembrokeshire Marine

E.1 PARQUE NACIONAL DE DOÑANA

NATURA 2000

STANDARD DATA FORM

FOR SPECIAL PROTECTION AREAS (SPA)

FOR SITES ELIGIBLE FOR IDENTIFICATION AS SITES OF COMMUNITY IMPORTANCE (SCI)

AND

FOR SPECIAL AREAS OF CONSERVATION (SAC)

1. SITE IDENTIFICATION

1.1. TYPE	*1.2. SITE CODE*	*1.3. COMPILATION DATE*	*1.4. UPDATE*
C	ES0000024	199712	199904

1.5. RELATION WITH OTHER NATURA 2000 SITES:

NATURA 2000 SITE CODES

ES6150009
ES6150008
ES6150019
ES6180005

1.6. RESPONDENT(S):

DIRECCION GENERAL DE PLANIFICACION
CONSEJERIA DE MEDIO AMBIENTE
JUNTA DE ANDALUCIA
AVD. DE ERITAÑA 1
41071 - SEVILLA
ESPAÑA
e-mail: sv.exn@cma.junta-andalucia.es

1.7. SITE NAME:

PARQUE NACIONAL DE DOÑANA

1.8. SITE INDICATION AND DESIGNATION/CLASSIFICATION DATES:

DATE SITE PROPOSED AS ELIGIBLE AS SCI:	*DATE CONFIRMED AS SCI:*
199712	
DATE SITE CLASSIFIED AS SPA:	*DATE SITE DESIGNATED AS SAC:*
198709	

Site code: ES0000024 NATURA 2000 Data Form

2. SITE LOCATION

2.1. SITE CENTRE LOCATION

LONGITUDE	LATITUDE
W 6 24 32	36 59 1
W/E (Greenwich)	

2.2. AREA (HA):

56393.22

2.3. SITE LENGTH (KM):

2.4. ALTITUDE (M):

MINIMUM	MAXIMUM	MEAN
0	47	9

2.5. ADMINISTRATIVE REGION:

NUTS CODE	REGION NAME	% COVER
ES615	Huelva	70
ES618	Sevilla	21
Marine area not covered by a NUTS-region		9

2.6. BIOGEOGRAPHIC REGION:

Alpine	Atlantic	Boreal	Continental	Macaronesian	Mediterranean
☐	☐	☐	☐	☐	☒

3. ECOLOGICAL INFORMATION

3.1. HABITAT types present on the site and assessment for them:

ANNEX I HABITAT TYPES:

CODE	%COVER	REPRESENTATIVITY	RELATIVE SURFACE	CONSERVATION STATUS	GLOBAL ASSESSMENT
2270	4	A B C D	A B C	A B C	A B C
2260	4	A B C D	A B C	A B C	A B C
2250	3	A B C D	A B C	A B C	A B C
2150	2	A B C D	A B C	A B C	A B C
1310	1	A B C D	A B C	A B C	A B C
1320	1	A B C D	A B C	A B C	A B C
1420	1	A B C D	A B C	A B C	A B C
1510	1	A B C D	A B C	A B C	A B C
2120	1	A B C D	A B C	A B C	A B C
2133	1	A B C D	A B C	A B C	A B C
2230	1	A B C D	A B C	A B C	A B C
3110	1	A B C D	A B C	A B C	A B C
92D0	1	A B C D	A B C	A B C	A B C
92A0	1	A B C D	A B C	A B C	A B C
9330	1	A B C D	A B C	A B C	A B C
3140	1	A B C D	A B C	A B C	A B C
3150	1	A B C D	A B C	A B C	A B C
3160	1	A B C D	A B C	A B C	A B C
3170	1	A B C D	A B C	A B C	A B C
4020	1	A B C D	A B C	A B C	A B C
5331	1	A B C D	A B C	A B C	A B C
6310	1	A B C D	A B C	A B C	A B C
6420	1	A B C D	A B C	A B C	A B C
7210	1	A B C D	A B C	A B C	A B C
91B0	1	A B C D	A B C	A B C	A B C
1150	1	A B C D	A B C	A B C	A B C

3.2.a. BIRDS listed on Annex I of Council directive 79/409/EEC

CODE	NAME	POPULATION Resident	Migratory Breed	Migratory Winter	Migratory Stage	SITE ASSESSMENT Population	Conservation	Isolation	Global
A157	Limosa lapponica			100i		A B C D	A B C	A B C	A B C
A015	Oceanodroma leucorhoa				P	A B C D	A B C	A B C	A B C
A135	Glareola pratincola		<4400p		P	A B C D	A B C	A B C	A B C
A131	Himantopus himantopus		<3500p	>1000i		A B C D	A B C	A B C	A B C
A180	Larus genei	<60p				A B C D	A B C	A B C	A B C
A176	Larus melanocephalus			P		A B C D	A B C	A B C	A B C
A170	Phalaropus lobatus			P		A B C D	A B C	A B C	A B C
A166	Tringa glareola			>3000i		A B C D	A B C	A B C	A B C
A189	Gelochelidon nilotica		>700p		P	A B C D	A B C	A B C	A B C
A140	Pluvialis apricaria			>150i		A B C D	A B C	A B C	A B C
A190	Sterna caspia				P	A B C D	A B C	A B C	A B C
A133	Burhinus oedicnemus	<1000p				A B C D	A B C	A B C	A B C
A132	Recurvirostra avosetta			>5000i		A B C D	A B C	A B C	A B C
A139	Charadrius morinellus				P	A B C D	A B C	A B C	A B C
A126	Fulica cristata		P	30i		A B C D	A B C	A B C	A B C
A124	Porphyrio porphyrio	>200p				A B C D	A B C	A B C	A B C
A119	Porzana porzana	P		P		A B C D	A B C	A B C	A B C
A151	Philomachus pugnax			>1000i		A B C D	A B C	A B C	A B C
A222	Asio flammeus			P		A B C D	A B C	A B C	A B C
A245	Galerida theklae	P				A B C D	A B C	A B C	A B C
A243	Calandrella brachydactyla		P		P	A B C D	A B C	A B C	A B C
A242	Melanocorypha calandra	P				A B C D	A B C	A B C	A B C
A231	Coracias garrulus				P	A B C D	A B C	A B C	A B C
A181	Larus audouinii				20i	A B C D	A B C	A B C	A B C
A127	Grus grus			P		A B C D	A B C	A B C	A B C
A391	Phalacrocorax carbo sinensis			<450i		A B C D	A B C	A B C	A B C
A205	Pterocles alchata		>100p		600i	A B C D	A B C	A B C	A B C
A420	Pterocles orientalis				P	A B C D	A B C	A B C	A B C
A197	Chlidonias niger		20p		P	A B C D	A B C	A B C	A B C
A196	Chlidonias hybridus				P	A B C D	A B C	A B C	A B C
A195	Sterna albifrons		<150p			A B C D	A B C	A B C	A B C
A193	Sterna hirundo		2p		P	A B C D	A B C	A B C	A B C
A191	Sterna sandvicensis			50i	100i	A B C D	A B C	A B C	A B C
A229	Alcedo atthis	P	>25p			A B C D	A B C	A B C	A B C
A030	Ciconia nigra			25i		A B C D	A B C	A B C	A B C
A121	Porzana pusilla	P		P		A B C D	A B C	A B C	A B C
A057	Marmaronetta angustirostris		P	400i		A B C D	A B C	A B C	A B C
A045	Branta leucopsis			P		A B C D	A B C	A B C	A B C

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Site code: ES0000024 NATURA 2000 Data Form

A035	Phoenicopterus ruber		>5000p	>10000i		A B C D	A B C	A B C	A B C
A034	Platalea leucorodia		1100p	>70i		A B C D	A B C	A B C	A B C
A094	Pandion haliaetus			P	P	A B C D	A B C	A B C	A B C
A031	Ciconia ciconia		>25p	P		A B C D	A B C	A B C	A B C
A399	Elanus caeruleus			<5i		A B C D	A B C	A B C	A B C
A029	Ardea purpurea		500p			A B C D	A B C	A B C	A B C
A026	Egretta garzetta		500p	>1000i		A B C D	A B C	A B C	A B C
A024	Ardeola ralloides		150p	10-15i		A B C D	A B C	A B C	A B C
A023	Nycticorax nycticorax		500p			A B C D	A B C	A B C	A B C
A022	Ixobrychus minutus		250p			A B C D	A B C	A B C	A B C
A032	Plegadis falcinellus		<40p		15i	A B C D	A B C	A B C	A B C
A084	Circus pygargus		>15p		P	A B C D	A B C	A B C	A B C
A103	Falco peregrinus	3p		P		A B C D	A B C	A B C	A B C
A098	Falco columbarius			P		A B C D	A B C	A B C	A B C
A095	Falco naumanni				P	A B C D	A B C	A B C	A B C
A093	Hieraaetus fasciatus				P	A B C D	A B C	A B C	A B C
A071	Oxyura leucocephala		10p	P		A B C D	A B C	A B C	A B C
A405	Aquila heliaca adalberti	13p				A B C D	A B C	A B C	A B C
A120	Porzana parva	P		P		A B C D	A B C	A B C	A B C
A090	Aquila clanga			P		A B C D	A B C	A B C	A B C
A060	Aythya nyroca		P	P		A B C D	A B C	A B C	A B C
A021	Botaurus stellaris	1p		P		A B C D	A B C	A B C	A B C
A089	Aquila pomarina			P		A B C D	A B C	A B C	A B C
A003	Gavia immer			P		A B C D	A B C	A B C	A B C
A192	Sterna dougallii				P	A B C D	A B C	A B C	A B C
A001	Gavia stellata			P		A B C D	A B C	A B C	A B C
A014	Hydrobates pelagicus			P		A B C D	A B C	A B C	A B C
A396	Branta ruficollis			P		A B C D	A B C	A B C	A B C
A128	Tetrax tetrax	P		P		A B C D	A B C	A B C	A B C
A091	Aquila chrysaetos				P	A B C D	A B C	A B C	A B C
A272	Luscinia svecica			P		A B C D	A B C	A B C	A B C
A224	Caprimulgus europaeus				P	A B C D	A B C	A B C	A B C
A042	Anser erythropus			P		A B C D	A B C	A B C	A B C
A397	Tadorna ferruginea		P			A B C D	A B C	A B C	A B C
A390	Oceanodroma castro			P		A B C D	A B C	A B C	A B C
A384	Puffinus puffinus mauretanicus			P		A B C D	A B C	A B C	A B C
A379	Emberiza hortulana				P	A B C D	A B C	A B C	A B C
A294	Acrocephalus paludicola				P	A B C D	A B C	A B C	A B C
A255	Anthus campestris				P	A B C D	A B C	A B C	A B C
A246	Lullula arborea	P				A B C D	A B C	A B C	A B C
A454	Cyanopica cyana	P				A B C D	A B C	A B C	A B C
A302	Sylvia undata	P	.			A B C D	A B C	A B C	A B C
A082	Circus cyaneus			>300i		A B C D	A B C	A B C	A B C
A081	Circus aeruginosus			>100i		A B C D	A B C	A B C	A B C
A117	Turnix sylvatica	P				A B C D	A B C	A B C	A B C
A080	Circaetus gallicus					A B C D	A B C	A B C	A B C
A074	Milvus milvus		25p	>200i		A B C D	A B C	A B C	A B C
A092	Hieraaetus pennatus			P		A B C D	A B C	A B C	A B C
A002	Gavia arctica			P		A B C D	A B C	A B C	A B C
A073	Milvus migrans		>50p			A B C D	A B C	A B C	A B C
A077	Neophron percnopterus			25i	P	A B C D	A B C	A B C	A B C
A078	Gyps fulvus			>200i	P	A B C D	A B C	A B C	A B C
A079	Aegypius monachus			1-5i	P	A B C D	A B C	A B C	A B C

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3.2.b. Regularly occuring Migratory Birds not listed on Annex I of Council Directive 79/409/EEC

3.2.c. MAMMALS listed on Annex II of Council directive 92/43/EEC

CODE	NAME	POPULATION Resident	Migratory Breed	Winter	Stage	SITE ASSESSMENT Population	Conservation	Isolation	Global
1355	Lutra lutra	P				A B C D	A B C	A B C	A B C
1362	Lynx pardinus	P				A B C D	A B C	A B C	A B C

3.2.d. AMPHIBIANS and REPTILES listed on Annex II of Council directive 92/43/EEC

CODE	NAME	POPULATION Resident	Migratory Breed	Winter	Stage	SITE ASSESSMENT Population	Conservation	Isolation	Global
1221	Mauremys leprosa	P				A B C D	A B C	A B C	A B C
1220	Emys orbicularis	P				A B C D	A B C	A B C	A B C
1219	Testudo graeca	P				A B C D	A B C	A B C	A B C

3.2.e. FISHES listed on Annex II of Council directive 92/43/EEC

CODE	NAME	POPULATION Resident	Migratory Breed	Winter	Stage	SITE ASSESSMENT Population	Conservation	Isolation	Global
1149	Cobitis taenia	P				A B C D	A B C	A B C	A B C
1142	Barbus comiza	P				A B C D	A B C	A B C	A B C
1116	Chondrostoma polylepis	P				A B C D	A B C	A B C	A B C
1101	Acipenser sturio					A B C D	A B C	A B C	A B C
1151	Aphanius iberus	P				A B C D	A B C	A B C	A B C

3.2.f. INVERTEBRATES listed on Annex II of Council directive 92/43/EEC

CODE	NAME	POPULATION Resident	Migratory Breed	Winter	Stage	SITE ASSESSMENT Population	Conservation	Isolation	Global

3.2 - 4

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Site code: ES0000024 NATURA 2000 Data Form

1044	Coenagrion mercuriale	P	A B C D	A B C	A B C	A B C

3.2.g. PLANTS listed on Annex II of Council directive 92/43/EEC

CODE	NAME	POPULATION	SITE ASSESSMENT Population	Conservation	Isolation	Global
1616	Thorella verticillatinundata		A B C D	A B C	A B C	A B C
1635	Armeria velutina		A B C D	A B C	A B C	A B C
1717	Linaria tursica		A B C D	A B C	A B C	A B C
1879	Micropyropsis tuberosa		A B C D	A B C	A B C	A B C

3.2 - 5

3.3. Other Important Species of Flora and Fauna

GROUP B M A R F I P	SCIENTIFIC NAME	POPULATION	MOTIVATION
M	Dama dama		A B C D
M	Felis silvestris		A B C D
M	Herpestes ichneumon		A B C D

(B = Birds, M = Mammals, A = Amphibians, R = Reptiles, F = Fish, I = Invertebrates, P = Plants)

4. SITE DESCRIPTION

4.1. GENERAL SITE CHARACTER:

Habitat classes	% cover
Marine areas, Sea inlets	8
Tidal rivers, Estuaries, Mud flats, Sand flats, Lagoons (including saltwork basins)	1
Salt marshes, Salt pastures, Salt steppes	34
Coastal sand dunes, Sand beaches, Machair	7
Inland water bodies (Standing water, Running water)	11
Heath, Scrub, Maquis and Garrigue, Phygrana	19
Dry grassland, Steppes	4
Broad-leaved deciduous woodland	3
Coniferous woodland	10
Artificial forest monoculture (e.g. Plantations of poplar or Exotic trees)	3
Total habitat cover	100 %

Other site characteristics

4.2. QUALITY AND IMPORTANCE:

Imprescindible para hábitats de la Directiva 92/43/CEE
Imprescindible para diversos taxones de la Directiva 92/43/CEE, incluido el Lince ibérico
Imprescindible para aves

4.3. VULNERABILITY

Vulnerabilidad del espacio según riesgo de amenaza de los hábitats naturales:
Distribución de la superficie en grados de amenaza
1% : Muy Alto
2% : Alto
3% : Moderado
7% : Bajo
87% : Muy Bajo

4.4. SITE DESIGNATION:

4.5. OWNERSHIP

4.6. DOCUMENTATION

5. SITE PROTECTION STATUS AND RELATION WITH CORINE BIOTOPES

5.1. DESIGNATION TYPES at National and Regional level:

CODE	% COVER
ES09	100

5.2. RELATION OF THE DESCRIBED SITE WITH OTHER SITES:

designated at National or Regional level:

designated at International level:

TYPE	SITE NAME	OVERLAP TYPE	% COVER
Ramsar Convention site	PARQUE NACIONAL DE DOÑANA	=	100
UNESCO Biosphere Reserve	PARQUE NACIONAL DE DOÑANA	=	100

5.3. RELATION OF THE DESCRIBED SITE WITH CORINE BIOTOPE SITES:

CORINE SITE CODE	OVERLAP TYPE	% COVER
B00000001	*	96
B00000505	*	2

6. IMPACTS AND ACTIVITIES IN AND AROUND THE SITE

6.1. GENERAL IMPACTS AND ACTIVITIES AND PROPORTION OF THE SURFACE OF THE SITE AFFECTED

IMPACTS AND ACTIVITIES WITHIN the site

IMPACTS AND ACTIVITIES AROUND the site

6.2. SITE MANAGEMENT AND PLANS

BODY RESPONSIBLE FOR THE SITE MANAGEMENT

SITE MANAGEMENT AND PLANS

Plan Rector de Uso y Gestión aprobado por RD 1772/1991, de 16 de diciembre

6. IMPACTS AND ACTIVITIES IN AND AROUND THE SITE

6.1. GENERAL IMPACTS AND ACTIVITIES AND PROPORTION OF THE SURFACE OF THE SITE AFFECTED

IMPACTS AND ACTIVITIES WITHIN the site

IMPACTS AND ACTIVITIES AROUND the site

6.2. SITE MANAGEMENT AND PLANS

BODY RESPONSIBLE FOR THE SITE MANAGEMENT

SITE MANAGEMENT AND PLANS

Plan Rector de Uso y Gestión aprobado por RD 1772/1991, de 16 de diciembre

7. MAPS OF THE SITE

Physical map

NATIONAL MAP NUMBER	*SCALE*	*PROJECTION*	*DIGITISED FORM AVAILABLE (*)*
5-21	100000	UTM (ES)	
5-22	100000	UTM (ES)	
6-21	100000	UTM (ES)	
6-22	100000	UTM (ES)	
1017	50000	UTM (ES)	
1018	50000	UTM (ES)	
1033	50000	UTM (ES)	
1047	50000	UTM (ES)	
1000	50000	UTM (ES)	

(*) Reference to availability of boundaries in digitised form

Aerial photograph(s) included:

8. SLIDES

E.2 CARMARTHEN BAY AND ESTUARIES

NATURA 2000

STANDARD DATA FORM

FOR SPECIAL PROTECTION AREAS (SPA)
FOR SITES ELIGIBLE FOR IDENTIFICATION AS SITES OF COMMUNITY IMPORTANCE (SCI)
AND
FOR SPECIAL AREAS OF CONSERVATION (SAC)

1. Site identification:

1.1 Type K

1.2 Site code UK0020020

1.3 Compilation date 199601

1.4 Update 200101

1.5 Relationship with other Natura 2000 sites

U	K	9	0	1	5	0	1	1

1.6 Respondent(s) International Designations, JNCC, Peterborough

1.7 Site name **Carmarthen Bay and Estuaries/ Bae Caerfyrddin ac Aberoedd**

1.8 Site indication and designation classification dates

date site proposed as eligible as SCI	199601
date confirmed as SCI	
date site classified as SPA	
date site designated as SAC	

2. Site location:

2.1 Site centre location

Longitude	*latitude*
04 22 21 W	51 40 44 N

2.2 Site area (ha) 65898.32

2.3 Site length (km)

2.5 Administrative region

NUTS code	*Region name*	*% cover*
UK912	Dyfed	12.0%
0	Marine	81.1%
UK924	West Glamorgan	6.9%

2.6 Biogeographic region

Alpine	Atlantic	Boreal	Continental	Macaronesia	Mediterranean
	√				

3. Ecological information:

3.1 Annex I habitats

Habitat types present on the site and the site assessment for them:

Annex I habitat	% cover	Representativity	Relative surface	Conservation status	Global assessment
Sandbanks which are slightly covered by sea water all the time	30	B	C	B	**B**
Estuaries	14	A	B	A	**A**
Mudflats and sandflats not covered by seawater at low tide	11	B	B	A	**B**
Coastal lagoons	0	D			
Large shallow inlets and bays	66	B	B	B	**B**
Salicornia and other annuals colonising mud and sand	0.1	A	B	A	**A**
Spartina swards (*Spartinion maritimae*)	0.5	D			
Atlantic salt meadows (*Glauco-Puccinellietalia maritimae*)	4	A	B	A	**A**
Dunes with *Hippophae rhamnoides*	0	D			
Submerged or partially submerged sea caves	0	D			

3.2 Annex II species

	Population				*Site assessment*			
	Resident	Migratory						
Species name		Breed	Winter	Stage	Population	Conservation	Isolation	Global
Petromyzon marinus	Common	-	-	-	C	C		**C**
Lampetra fluviatilis	Common	-	-	-	C	C		**C**
Alosa alosa	Rare	-	-	-	C	C		**C**
Alosa fallax	>10,000	-	-	-	A	B		**A**
Rhinolophus hipposideros	Present	-	-	-	D			
Rhinolophus ferrumequinum	Present	-	-	-	D			
Lutra lutra	Present	-	-	-		B		**C**
Halichoerus grypus	Present	-	-	-	D			

4. Site description:

4.1 General site character

Habitat classes	% cover
Marine areas. Sea inlets	82.00
Tidal rivers. Estuaries. Mud flats. Sand flats. Lagoons (including saltwork basins)	14.00
Salt marshes. Salt pastures. Salt steppes	4.00
Coastal sand dunes. Sand beaches. Machair	0
Shingle. Sea cliffs. Islets	0
Inland water bodies (standing water, running water)	0
Bogs. Marshes. Water fringed vegetation. Fens	0
Heath. Scrub. Maquis and garrigue. Phygrana	0
Dry grassland. Steppes	0
Humid grassland. Mesophile grassland	0
Alpine and sub-alpine grassland	0
Improved grassland	0
Other arable land	0
Broad-leaved deciduous woodland	0
Coniferous woodland	0
Mixed woodland	0
Non-forest areas cultivated with woody plants (including orchards, groves, vineyards, dehesas)	0
Inland rocks. Screes. Sands. Permanent snow and ice	0
Other land (including towns, villages, roads, waste places, mines, industrial sites)	0
Total habitat cover	**100%**

4.1 Other site characteristics

Soil & Geology:

Alluvium, Biogenic reef, Boulder, Clay, Cobble, Gravel, Limestone/chalk, Mud, Peat, Pebble, Sand, Sandstone/mudstone, Sedimentary, Shingle, Slate/shale

Geomorphology & Landscape:

Cave/tunnel, Cliffs, Enclosed coast (including embayment), Estuary, Intertidal rock, Intertidal sediments (including sandflat/mudflat), Islands, Lagoon, Open coast (including bay), Pools, Subtidal rock (including rocky reefs), Subtidal sediments (including sandbank/mudbank)

4.2 Quality and importance

Sandbanks which are slightly covered by sea water all the time

- for which this is considered to be one of the best areas in the United Kingdom.

Estuaries

- for which this is considered to be one of the best areas in the United Kingdom.

Mudflats and sandflats not covered by seawater at low tide

- for which this is considered to be one of the best areas in the United Kingdom.

Large shallow inlets and bays

- for which this is considered to be one of the best areas in the United Kingdom.

***Salicornia* and other annuals colonising mud and sand**

- for which this is considered to be one of the best areas in the United Kingdom.

Atlantic salt meadows (*Glauco-Puccinellietalia maritimae*)

- for which this is considered to be one of the best areas in the United Kingdom.

Petromyzon marinus

- for which the area is considered to support a significant presence.

Lampetra fluviatilis

- for which the area is considered to support a significant presence.

Alosa alosa

- for which the area is considered to support a significant presence.

Alosa fallax

- for which this is considered to be one of the best areas in the United Kingdom.

Lutra lutra

- for which the area is considered to support a significant presence.

4.3 Vulnerability

The Bay is both a fisheries resource and important nursery ground. Developments in fishing practices and target species could threaten the integrity of both the benthic communities and the sea-duck population (for which the Bay is also proposed as an SPA). Most of the potential threats come from fisheries and related activities such as shellfish management and access issues related to mussel and cockle gathering.

However two groups exist which discuss these issues: a group of statutory agencies and voluntary organisations, and the more recent cSAC relevant authorities group. CCW maintains close liaison in particular with the South Wales Sea Fisheries Committee.

CCW is liaising with the relevant local authority over the Millennium Coastal Park near Llanelli.

CCW is consulted over applications to dredge aggregate from Helwick Bank. These works may have an effect locally on the biology of the Bank, and in conjunction with other coastal defence works may also affect sediment budgets and characteristics over a wider area. CCW has encouraged extensive monitoring and further research.

5. Site protection status and relation with CORINE biotopes:

5.1 Designation types at national and regional level

Code	*% cover*
UK04 (SSSI/ASSI)	17.2
UK01 (NNR)	1.0

E.3 PEMBROKESHIRE MARINE

NATURA 2000

STANDARD DATA FORM

FOR SPECIAL PROTECTION AREAS (SPA)
FOR SITES ELIGIBLE FOR IDENTIFICATION AS SITES OF COMMUNITY IMPORTANCE (SCI)
AND
FOR SPECIAL AREAS OF CONSERVATION (SAC)

1. Site identification:

1.1 Type K

1.2 Site code UK0013116

1.3 Compilation date 199710

1.4 Update 200101

1.5 Relationship with other Natura 2000 sites

U	K	9	0	1	4	0	4	1
U	K	9	0	1	4	0	5	1
U	K	9	0	1	4	0	6	1
U	K	9	0	1	4	0	6	2

1.6 Respondent(s) International Designations, JNCC, Peterborough

1.7 Site name **Pembrokeshire Marine/ Sir Benfro Forol**

1.8 Site indication and designation classification dates

date site proposed as eligible as SCI	199710
date confirmed as SCI	
date site classified as SPA	
date site designated as SAC	

2. Site location:

2.1 Site centre location

longitude	*latitude*
05 36 56 W	51 43 33 N

2.2 Site area (ha)

136166.8

2.3 Site length (km)

2.5 Administrative region

NUTS code	*Region name*	*% cover*
UK912	Dyfed	1.5%
0	Marine	98.5%

2.6 Biogeographic region

Alpine	Atlantic	Boreal	Continental	Macaronesia	Mediterranean
	√				

3. Ecological information:

3.1 Annex I habitats

Habitat types present on the site and the site assessment for them:

Annex I habitat	% cover	Representativity	Relative surface	Conservation status	Global assessment
Sandbanks which are slightly covered by sea water all the time	30	C	B	B	**C**
Estuaries	1	A	C	B	**B**
Mudflats and sandflats not covered by seawater at low tide	1.2	B	C	B	**C**
Coastal lagoons	0	C	C	C	**C**
Large shallow inlets and bays	16.4	A	B	B	**B**
Reefs	30	B	B	A	**A**
Atlantic salt meadows (*Glauco-Puccinellietalia maritimae*)	0.3	C	C	C	**C**
Submerged or partially submerged sea caves	0	C	C	B	**C**

3.2 Annex II species

	Population				*Site assessment*			
	Resident	Migratory						
Species name		Breed	Winter	Stage	Population	Conservation	Isolation	Global
Petromyzon marinus	Present	-	-	-	C	C		**C**
Lampetra fluviatilis	Present	-	-	-	C	C		**C**
Alosa alosa	Present	-	-	-	C	C		**C**
Alosa fallax	Present	-	-	-	C	C		**C**
Tursiops truncatus	Very rare	-	-	-	D			
Phocoena phocoena	Common	-	-	-	D			
Lutra lutra	Present	-	-	-		B		**C**
Halichoerus grypus	1001-10,000	-	-	-	B	A	B	**B**

4. Site description:

4.1 General site character

Habitat classes	% cover
Marine areas. Sea inlets	98.00
Tidal rivers. Estuaries. Mud flats. Sand flats. Lagoons (including saltwork basins)	1.50
Salt marshes. Salt pastures. Salt steppes	0.50
Coastal sand dunes. Sand beaches. Machair	0
Shingle. Sea cliffs. Islets	0
Inland water bodies (standing water, running water)	0
Bogs. Marshes. Water fringed vegetation. Fens	0
Heath. Scrub. Maquis and garrigue. Phygrana	0
Dry grassland. Steppes	0
Humid grassland. Mesophile grassland	0
Alpine and sub-alpine grassland	0
Improved grassland	0
Other arable land	0
Broad-leaved deciduous woodland	0
Coniferous woodland	0
Mixed woodland	0
Non-forest areas cultivated with woody plants (including orchards, groves, vineyards, dehesas)	0
Inland rocks. Screes. Sands. Permanent snow and ice	0
Other land (including towns, villages, roads, waste places, mines, industrial sites)	0
Total habitat cover	**100%**

4.1 Other site characteristics

Soil & Geology:

Biogenic reef, Boulder, Chert/flint, Clay, Cobble, Gravel, Igneous, Limestone/chalk, Maerl, Metamorphic, Mud, Peat, Pebble, Sand, Sandstone/mudstone, Sedimentary, Shingle, Slate/shale

Geomorphology & Landscape:

Cave/tunnel, Cliffs, Coastal, Enclosed coast (including embayment), Estuary, Geos (rocky inlets), Intertidal rock, Intertidal sediments (including sandflat/mudflat), Islands, Lagoon, Open coast (including bay), Pools, Ria, Sound/strait, Subtidal rock (including rocky reefs), Subtidal sediments (including sandbank/mudbank), Surge gullies, Tidal rapids

4.2 Quality and importance

Sandbanks which are slightly covered by sea water all the time
- for which the area is considered to support a significant presence.

Estuaries
- for which this is considered to be one of the best areas in the United Kingdom.

Mudflats and sandflats not covered by seawater at low tide
- for which the area is considered to support a significant presence.

Coastal lagoons
- for which the area is considered to support a significant presence.

Large shallow inlets and bays
- for which this is considered to be one of the best areas in the United Kingdom.

Reefs
- for which this is considered to be one of the best areas in the United Kingdom.

Atlantic salt meadows (*Glauco-Puccinellietalia maritimae*)
- for which the area is considered to support a significant presence.

Submerged or partially submerged sea caves
- for which the area is considered to support a significant presence.

Petromyzon marinus
- for which the area is considered to support a significant presence.

Lampetra fluviatilis
- for which the area is considered to support a significant presence.

Alosa alosa
- for which the area is considered to support a significant presence.

Alosa fallax
- for which the area is considered to support a significant presence.

Lutra lutra
- for which the area is considered to support a significant presence.

Halichoerus grypus
- for which this is considered to be one of the best areas in the United Kingdom.

4.3 Vulnerability

Water quality issues such as those associated with dredge-spoil disposal are kept under review through liaison with the Environment Agency, Ministry of Agriculture, Fisheries and Food and Milford Haven Port Authority.

Pollution originating from the transport or exploration/production of oil and gas are of concern. Management of shipping using Milford Haven following the *Sea Empress* oil-spill in 1996 has improved and will be kept under review by the Port Authority. Improved contingency planning, which better reflects environmental priorities, involves many statutory agencies and is reflected in a revised national contingency plan published in January 2000.

Marine communities are vulnerable to damage by certain fishing methods. South Wales Sea Fisheries Committee bylaws control activities within Skomer Marine Nature Reserve. Other environmental requirements of management of fisheries are addressed through liaison with the SWSFC.

5. Site protection status and relation with CORINE biotopes:

5.1 Designation types at national and regional level

Code	*% cover*
UK04 (SSSI/ASSI)	1.4
UK01 (NNR)	0.0
UK02 (MNR)	1.0

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