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REPORT

MONETARY AND NON MONETARY METHODS FOR ECOSYSTEM SERVICES VALUATION SPECIFICATION SHEET AND SUPPORTING MATERIAL

Hadley, D., D'Hernoncourt, J., Franzén, F., Kinell, G., Söderqvist, T., Soutukorva, Å, and Brouwer, R.

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University of East Anglia CSERGE, Norwich
Université Libre de Bruxelles – CEESE, Brussels
Enveco Environmental Economics Consultancy Ltd, Stockholm
Vrije Universiteit – Institute for Environmental Studies, Amsterdam

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Note to the reader - How to use this material

This report has been designed as a hyperlinked pdf document.
The main text in the specification sheet synthesises the economic assessment method, its relation to systems approach and the appropriate use of the method. It also gives some hints on how to best present the results of your assessment to stakeholders, along with an example of the use of the method.
In the text and in the “further information” section, you will have access to links to the accompanying material available in the rest of the report (page numbers are also provided along the links in case you would like to print this report).
A back button on the bottom of each page of supporting material helps you go back to the main text.

SPECIFICATION SHEET

Monetary and non monetary methods for ecosystem services valuation

Method and assumptions

Ecosystems services provided by the environment are essential to our survival and our welfare. To incorporate this in policy making, it is sometimes interesting to establish how ecosystem services and their identified changes are valued by individuals. Economists use the term value to describe "a fair or proper equivalent in money, commodities, etc", where *equivalent in money* represents that sum of money that would have an equivalent effect (either increase or decrease) on the welfare/well-being or utilities of individuals (learn more about the theoretical grounds behind this here, p. 6). Valuing non market impacts in monetary terms makes it possible to compare, in a cost benefit analysis (CBA) frame, these costs and benefits to market impacts or financial revenues or costs.

A variety of approaches can be used to estimate values of ecosystem services. They fall in two main categories: techniques that estimate economic values –valuation approaches- and techniques that produce estimates equivalent to prices –pricing approaches. It is important to know that the price of a good or service and its economic value are distinct and can differ greatly: pricing approaches are not able to capture the consumer surplus element of value. Learn more on the difference between value and market price here, p. 7-8).

Valuation approaches

Valuation approaches fall in two main categories, depending on how preferences are inferred: stated and revealed preferences approaches.

Stated preference methods directly elicit individuals' preferences for non-market goods through the use of surveys based on simulated markets. The contingent valuation method and choice modeling experiments are the main forms of stated preference techniques. Learn more on these techniques (description, advantages and disadvantages) here (p. 12-13).

Revealed preference methods infer individual preferences by observing their behavior in markets in which a given environmental good is indirectly purchased (making the assumption that non-market use values are indirectly reflected in consumer expenditure). The most widely used revealed preference methods are: the travel cost method, hedonic pricing, averting behavior and defensive expenditure and the cost of illness and lost output method. Read more on these methods (description, advantages and disadvantages) here (p. 14-17).

Note that each valuation technique has different properties when it comes to valuing parts or whole of the total economic value of environmental assets -where one mainly distinguishes use and non-use (or passive use) values. Learn more on the concept of total economic value and its components here (p. 9-11).

Pricing approaches

Various methods exist as well to infer the price of an ecosystem service: market prices, opportunity cost and replacement cost approaches. Learn more on these pricing approaches here (p. 18-19).

Relation to systems approach

Ecosystem services result from complex interactions at multiple spatial and temporal scales. A systems approach can help understand and quantify these interactions. On this basis can monitoring, measuring and valuing of ecosystem services be done in a meaningful way: any change in the quality or quantity of ecosystem services could have consequences that have an impact upon human well-being.

Since they are based on preferences inferred at one point in time (through surveys or other data collection methods), valuation techniques are not especially well suited to inclusion in dynamic simulation models. If their results are used and compared in a dynamic setting (i.e. considered as state variables of an integrated simulation model and/or converted to its time scale), be aware that you are assuming stable preferences in time.

When this method is especially to be used

Ecosystem services valuation techniques allow expression of the many-faceted benefits derived from ecosystems in one common unit, i.e. money, which facilitates a direct comparison of returns to different uses of ecosystems. This helps to provide a transparent set of information about the human benefits derived from ecosystems which can be used as one aspect in decision-making. However, this process can be complicated: while a number of ecosystem services can be valued in economic terms others cannot because of uncertainty and complexity conditions. The process can sometimes also be controversial, particularly when certain non-marketed ecosystem services are included in the analysis.

The results of valuation techniques can be used within the frame of a cost benefit analysis. If a full assessment is not possible, a partial assessment can be undertaken. These techniques can be used to help shed light on the human well-being benefits/costs derived from ecosystem changes along the studied scenarios or policy options. The analysis should then be augmented by a qualitative description of the changes in ecosystem services that could not be valued and which includes some indication of the importance of these changes and their likely magnitude. Alternatively, a multi-criteria analysis could be performed.

How to best present results to stakeholders?

Ecosystem services valuation techniques help provide monetary figures to stakeholders. Bear in mind that these figures are a translation of human preferences (i.e. welfare) and do not represent money that can be touched upon. Whether these figures are presented within the frame of cost benefit analysis or not, these methods have the benefit of simplicity. However, they are most of the time simplifying reality, hiding uncertainties and resting on strong assumptions. Results often suffer from many biases inherent to survey data collection methods or to the theoretical grounds on which these methods are built. It is important to underline all these limitations while presenting your assessment to stakeholders. Explain thus clearly what your results cover, what they do not cover and discuss possible implications. A qualitative description of the changes in ecosystem services that have not been/could not be valued should accompany the analysis. Encourage stakeholders to use these results with caution, it is important not to give them a distorted view of the impacts of the studied issue/scenario on human well-being.

Example of use of the methods

The team of Himmerfjärden in Sweden designed a tool to assess policy options to mitigate eutrophication/ manage nitrogen loads. In order to ascertain a value to water clarity improvement, they used results from a travel cost method in the Stockholm archipelago -in which Himmerfjärden is part- (Soutukorva, 2005 and Kinell, 2008). This assessment was part of the cost benefit analysis they undertook (read the example section in the CBA specification sheet for more information).

To explain the choice of a recreational site in the archipelago (using a random utility model), three variables were used: the cost of travelling to the site, including the opportunity cost of travel time; the bathing water quality as measured by Secchi depth and accessibility to sites by public ferry. A conditional logit model was used to calculate compensating variation: a change in consumer surplus or monetary measure of the change in human well-being due to a one meter Secchi depth improvement. Learn more on the discrete choice models the team used here (p. 21-22).

The logit model also helped to derive useful results for the simulation model such as how a change in explanatory variables affects the probability of selecting a particular site. Learn more on this travel cost method and its different results here (p. 23-26).

Further information

- A core reference for economic valuation methods: Pearce, D. W., G. Atkinson and S. Mourato. 2006. Cost-benefit analysis and the environment: recent developments, Paris: Organisation for Economic Co-operation and Development. Available online at: http://www.lne.be/themas/beleid/milieueconomie/downloadbare-bestanden/ME11_cost-benefit%20analysis%20and%20the%20environment%20oeso.pdf, accessed 01/2011
- Note that where monetary assessment is not possible or appropriate, there is a number of qualitative valuation methodologies, that are described here (p. 20).
- Benefit transfer is a technique for valuing ecosystem services that employs results from previously existing studies and transfers them into a similar policy context. You will find a paper related to benefit transfer, with examples and discussion on the reliability of results here (p. 27-50). Details on international databases of values for different ecosystem services as well as a tool for quality assessment of economic valuation studies can be found here (p. 51-56).

Welfare, utility or human well-being as benefits or costs

The effects of changes in ecosystem services on human society in terms of increases or decreases in benefits, costs, welfare, utility or human well-being require some definition. When we refer to benefits of a policy or project we mean that there has been (or, will be) some increase in human well-being or welfare associated with implementing that policy or project. Economists measure this increase in human well-being or welfare using the concept of utility. Utility is a measure of satisfaction: the more utility we have the more satisfied we are, or, alternatively the greater is our welfare or well-being.

Costs are the opposite of benefits. If the overall effects of a policy or project represent a cost to society this would mean that implementing that policy or project would result in a decrease in society's welfare or well-being and hence in the overall utility that society enjoys.

The problem with the concept of utility is that it is not directly measurable – so, how then do we compare situations where utility has been changed as the result of the implementation of some project or policy? Consider a simple example where we have one individual who enjoys a particular level of utility – we will call this U_0 – that is attained with an income of Y_0, and which is associated with a given level of environmental quality – E_0. Suppose then that the implementation of a new policy or project causes an improvement in the environmental quality that the individual experiences from E_0 to E_1 and that this improvement increases their utility from U_0 to U_1: so they move from a state $U_0(Y_0,E_0)$ to $U_1(Y_0,E_1)$. As we have said we cannot directly measure this increase in utility, but we can indirectly by considering how much income this individual would be willing to give up in order to bring about this change. Hypothetically, the individual is considering two combinations of income and environmental quality that both give her/him the same level of utility, i.e. U_0. In the first combination, income is reduced and environmental quality is increased, and in the second, income is not reduced and environmental quality is not increased. The reduction in income that is required to make these two combinations equal represents what the individual is willing to pay for the change in environmental quality, i.e.:

$$U_0\,(Y_0 - WTP,\ E_1) = U_0\,(Y_0,\ E_0)$$

Alternatively an individual could be asked to consider how much additional income they would be willing to accept in order to give up the improvement in environmental quality, but still remain at the increased utility level U_1, i.e.:

$$U_1\,(Y_0 + WTA,\ E_0) = U_0\,(Y_0,\ E_1)$$

Similar measures of change in utility can be developed for policy or project effects that cause deteriorations in environmental quality.

The basic principle that is at work here is that utility (or alternatively, welfare or well-being) can be indirectly measured in terms of the income that people are willing to give up in order to achieve some improvement; or, what they are willing to accept in compensation for foregoing some improvement. Willingness to pay (WTP) and willingness to accept (WTA) represent the monetary equivalents of changes in utility.

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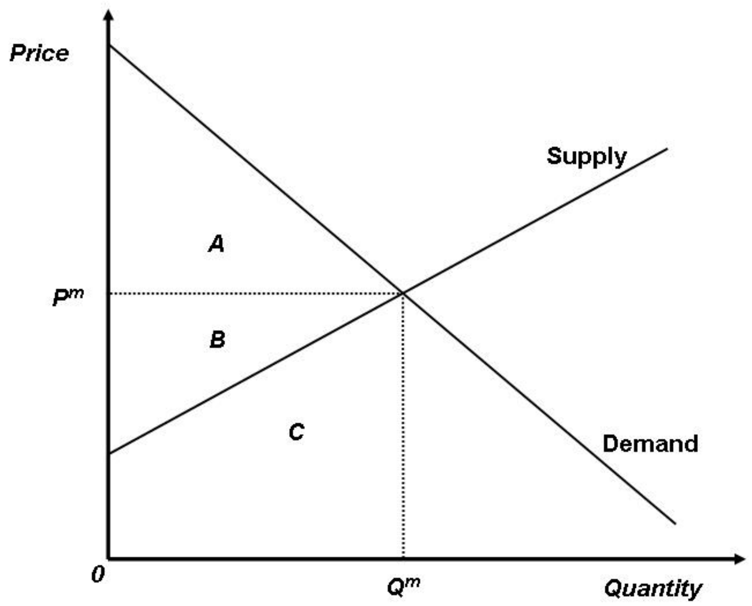
Willingness to pay and the notion of value

Willingness to pay (WTP) equates to economic conceptions of value and it is useful to discuss this by reference to the demand and supply curves for a hypothetical good or service. To simplify things, Figure 1 below represents these curves as straight lines.

Figure 1: Willingness to pay, price and consumer surplus



The slope of the demand curve shows how much consumers are willing to pay for each extra unit of the good or service (i.e. it describes the marginal benefit they derive from each extra unit), and the demand curve slopes downwards because the benefit (utility) they derive from each additional unit declines with increasing quantity (known within economics as the law of diminishing marginal utility). The supply curve slopes upwards as the curve is derived from the costs of production, as more is produced more inputs are required and this increases the costs of each additional unit produced (i.e. the supply curve is directly analogous to the marginal costs of the firm). Hence producers will only supply extra units for a corresponding increase in price.

The area under the supply and demand curves indicates the aggregate supply and demand respectively for the good or service (it is aggregate in the sense that it represents the sum of all the individual demands of all the consumers in this market, and the sum of supply from all the firms in this market). In a competitive, freely functioning market, a quantity Q_m of the good or service is traded at the market price P_m, which is the price at which demand matches supply. If quantities less than Q_m are traded, consumers are willing to pay more than the market price (the demand curve is higher than the level P_m), suggesting that market price alone is only a minimum estimate of the economic value or benefit derived. The area between the market price and the demand curve (triangle A) is the consumer surplus, or the additional utility gained by consumers above the price paid. Therefore, gross social benefits are the expenditure (areas B + C, or price multiplied by quantity) plus the consumer surplus (area A). The total cost of

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producing quantity Q_m is the area below the supply curve (area C). The area above the supply curve and below the market price is the producer surplus; this occurs because producers are willing to sell for less than the market price if the quantity traded is less than Q_m (the supply curve is less than P_m). The net social benefit is the consumer surplus (area A) plus the producer surplus (area B).

The point of this exposition is to make it clear that the price of a good or service and its economic value are distinct and can differ greatly: so, for example, water used for irrigation could have a very high value, but a very low price or no price at all. The price of a given good thus only informs us of the cost of purchasing that good and not its value. Since WTP consists of both the price paid to purchase a particular good, as well as consumer surplus, pricing approaches, or cost based measures are unable to capture the consumer surplus element of value and so must be regarded as only a partial measure. However, whilst valuation approaches may be theoretically correct, pricing approaches are often used to value various aspects of ecosystem value. This is because valuation approaches are often very expensive and time consuming to undertake and so price/cost based techniques are common where time and resources are limited. In addition, pricing approaches can be useful in providing rough monetary estimates of ecosystem services that might otherwise remain unvalued in the absence of other, more difficult to obtain (and often expensive), evidence.

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Total economic value

Ecologists use the term *value* to mean "that which is desirable or worthy of esteem for its own sake; something or some quality having intrinsic worth". Economists use the same term to describe "a fair or proper equivalent in money, commodities, etc", where *equivalent in money* represents that sum of money that would have an equivalent effect on the welfare or utilities of individuals. A number of ecosystem services can be valued in economic terms, while others cannot because of uncertainty and complexity conditions. The notion of total economic value provides an all-encompassing measure of the *economic value* of any environmental asset. It decomposes into use and non-use (or passive use) values. Total economic value does not encompass other kinds of values, such as intrinsic values which are usually defined as values residing "in" the asset and unrelated to human preferences or even human observation. However, apart from the problems of making the notion of intrinsic value operational, it can be argued that some people's willingness to pay for the conservation of an asset, independently of any use they make of it, is influenced by their own judgements about intrinsic value. This may show up especially in notions of "rights to existence" but also as a form of altruism.

Coastal ecosystems provide a wide range of services of significant value to society - storm and pollution buffering functions, flood alleviation, recreation and aesthetic services, and so forth. In valuing a resource such as a coastal ecosystem, it is important to capture the values to society of these characteristic services. The use of the total economic value classification enables the values to be usefully broken down into the categories shown in Figure 1 below. The initial distinction is between *use value* and *non-use value*. Use value involves some interaction with the resource, either directly or indirectly:

- *Direct use value*: involves direct interaction with the ecosystem itself rather than via the services it provides. It may be consumptive use, such as fisheries or timber, or it may be non-consumptive, as with some recreational and educational activities. There is also the possibility of deriving value from 'distant use' through media such as television or magazines, although it is unclear whether or not this type of value is actually a use value, and to what extent it can be attributed to the ecosystem involved.
- *Indirect use value*: derives from services provided by the ecosystem. This might, for example, include the removal of nutrients, thereby improving water quality, or the carbon sequestration services provided by some coastal ecosystems.

Non-use value is associated with benefits derived simply from the knowledge that a particular ecosystem is maintained. By definition, it is not associated with any use of the resource or tangible benefit derived from it, although users of a resource might also attribute non-use value to it. Non-use value is closely linked to ethical concerns, often being linked to altruistic preferences, although according to some analysts it stems ultimately from self-interest. It can be split into three basic components, although these may overlap depending upon exact definitions.

- *Existence value*: derived simply from the satisfaction of knowing that an ecosystem continues to exist, whether or not this might also benefit others. This value notion has been interpreted in a number of ways and seems to straddle the instrumental/intrinsic value divide.
- *Bequest value*: associated with the knowledge that a resource will be passed on to descendants to maintain the opportunity for them to enjoy it in the future.
- *Altruistic value*: associated with the satisfaction from ensuring resources are available to contemporaries of the current generation.

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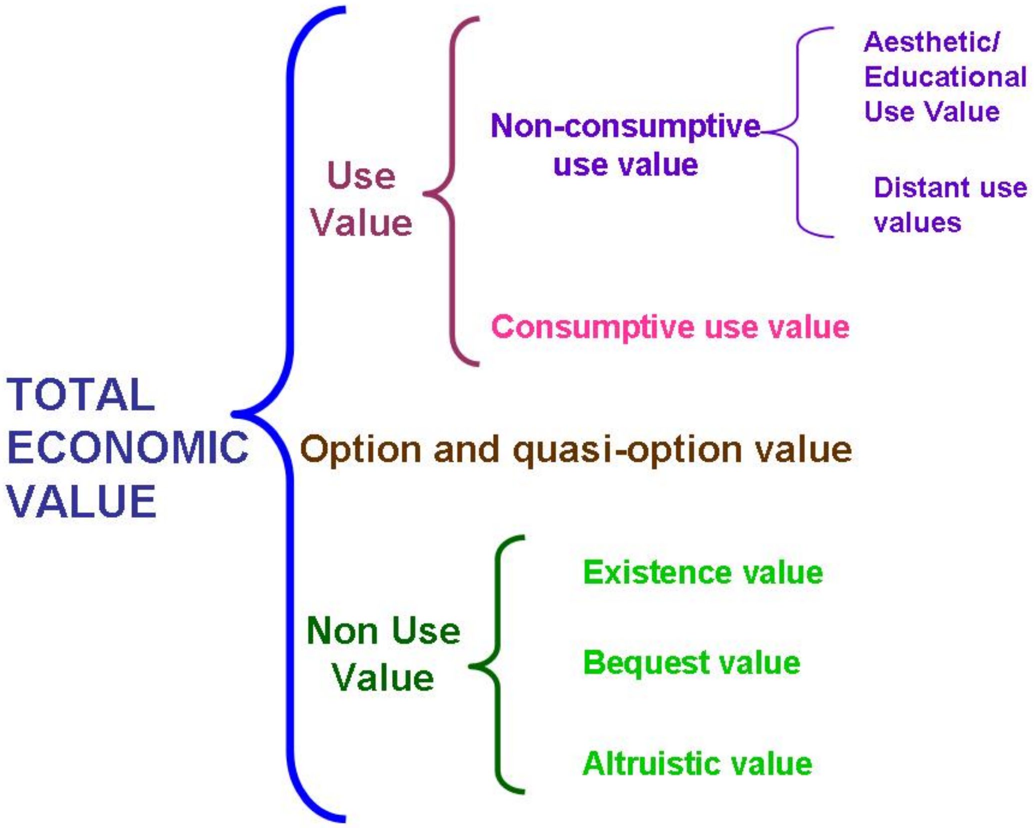
Figure 1: Total Economic Value



Finally, two categories not associated with the initial distinction between use values and non-use value include:

- *Option value*: an individual derives benefit from ensuring that a resource will be available for *use in the future*. In this sense it is a form of use value, although it can be regarded as a form of insurance to provide for possible future but not current use.
- *Quasi-option value:* associated with the potential benefits of waiting for improved information before giving up the option to preserve a resource for future use. In particular, it suggests a value of avoiding irreversible damage that might prove to have been unwarranted in the light of further information. An example of an option value is in bio-prospecting, where biodiversity may be maintained on the off-chance that it might in the future be the source of important new medicinal drugs. Potentially, quasi-option value could make up a sizeable proportion of total economic value, although measurement of its magnitude could be problematic.

These various elements of total economic value are assessed using economic valuation methods, and some of these elements are more easily valued than others, especially those with easily identifiable uses (usually the use type values). Non-use values are usually more difficult to assess. The main problem when including the full range of ecosystem services in economic choices is that many of these services are not valued in markets. There is a gap between market valuation and the economic value of many ecosystem functions.

Total economic value is derived from the preferences of individuals. When goods and services are exchanged in actual markets, individuals express their preferences via their purchasing behaviour. In other words, the price they pay in the market reflects how

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much, at the very least, they are willing to pay for the benefits they derive from consuming that good or service. For environmental resources which are not traded in actual markets, such behavioural and market price data are missing. Hence these resources generate non-market or external benefits. In addition to interpreting the market data, the methods of economic valuation provide several tools that may be employed to value benefits that are derived from non-market goods and services.

Choices between different policy options usually involve *marginal* changes in the provision of ecosystem services. It is the marginal value of ecosystem services, i.e. the value yielded by an additional unit of the service, all else held constant, that will determine the consequence of trade-offs, i.e. the costs of losing or the benefits of preserving a given amount or quality of a service (Daily, 1997). In other words, the methodologies for estimating economic value relate to relatively small changes in ecosystem services, not to the totality of the functions themselves. Clearly the value of the latter is infinite, as without this stock of natural capital, there would be no life on earth.

References

Daily, G. C. 1997. Nature's Services, Island, Washington, DC.

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Stated preference methods

Stated preference methods directly elicit individuals' preferences for non-market goods through the use of surveys based on simulated markets. In contrast to other valuation approaches, these methods can also estimate the non-use component of total economic value (as well as other components). In the case of ecosystem services non-use value may be significant, particularly for irreversible impacts.

The main forms of stated preference technique are as follows:

Contingent valuation

Contingent valuation methods employ a questionnaire format where respondents are asked how much they would be willing to pay (WTP) or willing to accept (WTA) for a specified gain or loss of a given good or service. Economic value estimates yielded by contingent valuation surveys are 'contingent' upon the hypothetical market situation that is presented to respondents and allows them to trade off gains and losses against money. WTP/WTA questions may be asked in a number of ways, including an open-ended format where the respondent is simply asked to state their maximum WTP/WTA, and a dichotomous choice format, where the respondent is required to answer yes or no to a 'bid' (e.g. are you willing to pay €x?). Although this method is considered to be controversial in some quarters, the contingent valuation method has gained increasing acceptance in recent years amongst many academics and policy makers as being a versatile and powerful methodology for estimating the monetary value of the non-market impacts of projects and policies.

An example of a contingent valuation study that is directly relevant to coastal zone management is Georgiou *et al*. (1999). This study asks respondents what they are WTP to reduce the perceived risk of falling ill after bathing at two beaches with differing water quality in East Anglia in the UK. The survey asked the question, "what is the maximum amount of money that you would be willing to pay per year in the form of higher water rates to ensure that the bathing water at this beach passes the EC standard (does not fall below the EC standard)". Results showed that over the whole sample the mean WTP was £12.32 and £14.64 per year for the two study sites.

Advantages of contingent valuation:

- can estimate use and non-use values;
- a widely used and much researched environmental valuation technique;
- applicable to a wide range of ecosystem services.

Disadvantages of contingent valuation:

- like many questionnaire techniques can suffer from a wide range of biases. Questionnaires need to be very carefully designed and pre-tested;
- very resource intensive. Reliable surveys need large sample sizes and hence consume manpower and finances;
- depending on the bid format used can be statistically complex to analyse.

Other issues:

- Most reliable when used to estimate the value of environmental gains and where the good or service of concern is reasonably familiar to respondents.

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Choice modelling

Choice modeling approaches involve respondents making choices between goods which are described in terms of their various attributes, offered in different amounts, or levels. There are two main choice formats: contingent ranking and choice experiments. In a contingent ranking exercise, respondents rank a set of alternative scenarios of good or service provision in order of preference. In a choice experiment, exercise respondents are presented with a series of scenarios along with their associated costs or prices and asked to choose their most preferred option. Survey results are then analysed statistically to arrive at the values of WTP that correspond to each scenario.

See Hanley, *et al.* (2006) for an example of the choice experiment approach applied to the issue of valuing improvements in the ecological status of the Rivers Wear and Clyde in the UK. Respondents to the survey were asked to choose between a number of different options which were defined in terms of differing levels of certain ecological attributes (river ecology, aesthetics and the state of river banks) and associated costs (in terms of increased water rates to consumers). The value of a change in these attributes from a "fair" to a "good" ecological status was then determined from statistical analysis of the choices made.

Advantages of choice modelling:

- as above for contingent valuation;
- more flexible than contingent valuation as it enables the attributes of an environmental gain scenario to be valued rather than just the overall scenario.

Disadvantages of choice modelling:

- as above for contingent valuation, but even more attention needs to be paid to design issues and analysis can be even more complicated.

References

Georgiou, S., I.H. Langford, I.J. Bateman and R.K. Turner. 1998. Determinants of individuals' willingness to pay for perceived reductions in environmental health risks: a case study of bathing water quality. Environment and Planning A 30 (4):577-594.

Hanley, N., R.E. Wright and B. Alvarez-Farizo. 2006. Estimating the economic value of improvements in river ecology using choice experiments: an application to the water framework directive. Journal of Environmental Management 78 (2):183-193.

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Revealed preference methods

Revealed preference methods infer individuals' preferences by observing their behaviour in markets in which a given environmental good is indirectly purchased. These approaches are reliant upon the assumption that non-market use values are indirectly reflected in consumer expenditure. Note that while these methods are grouped under the same overall category they differ in having slightly different conceptual bases and in being applicable to the valuation of different environmental resources.

Travel cost method

The travel cost method enables the economic value of recreational use (an element of direct use value) for a specific site to be estimated. The method requires that the costs incurred by individuals travelling to recreation sites - in terms of both travel expenses (fuel, fares etc.) and time (e.g. foregone earnings) - is collected. The basic assumption is that these costs of travel serve as a proxy for the recreational value of visiting a particular site.

An interesting application of the travel cost method is described in Font (2000). The study applies the travel cost method to international tourist visits to a set of 10 protected natural areas in Mallorca. The results obtained from the model allows Font to predict that over the course of a year tourists would be WTP a lower-bound figure of 30.21 billion pesetas (in 1997) for the option of being able to visit these sites.

Advantages of the travel cost method:

- a well established technique;
- based on actual observed behaviour.

Disadvantages of the travel cost method:

- can only estimate use values;
- really only applicable to specific sites (usually recreational sites);
- difficult to account for the possible benefits derived from travel, multipurpose trips and competing sites;
- very resource intensive. Reliable surveys need large sample sizes and hence consume manpower and finances;
- statistically complex to analyse.

Hedonic pricing

Hedonic pricing may be applied to the valuation of ecosystem services such as landscape amenity, air quality, and noise. The technique involves isolating the effect of these services on the demand for a marketed good. In most cases price data from the housing market are used. Analysis of the data estimates the implicit price which individuals are willing to pay for the relevant environmental characteristics. By trading these market goods, consumers are thereby able to express their values for the intangible goods, and these values can be uncovered through the use of statistical techniques. This process can be hindered, however, by the fact that a market good can have several intangible characteristics, and that these can be collinear. It can also be difficult to measure the intangible characteristics in a meaningful way.

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The hedonic pricing method has been mainly applied to data from housing and labour markets and especially the former with respect to valuation of environmental attributes. Research has been carried that has studied the effect on housing prices of proximity to landfill sites, or to aircraft noise, or air pollution. Leggett and Bockstael (2000) use hedonic pricing to estimate the effect on waterside property prices of a reduction in faecal coliform counts in Chesapeake Bay in the USA. Their results suggest that the increase in property price associated with this reduction in pollution amounts to up to 2% of average overall property value.

Advantages of hedonic pricing:

- a well established technique;
- based on actual observed behaviour and (usually) existing data.

Disadvantages of hedonic pricing:

- can only estimate use values;
- really only applicable to environmental attributes likely to be capitalised into the price of housing and/or land;
- confined to cases where property owners are aware of environmental variables and act because of them;
- market failures may mean that prices are distorted;
- data intensive and appropriate data may be difficult to obtain;
- statistically complex to analyse.

Averting behaviour and defensive expenditure

These approaches are similar to the travel cost method and hedonic pricing, but they differ as they use as a basis individual behaviour to avoid negative intangible impacts as a conceptual base. For example, people buy goods such as safety helmets to reduce accident risk, and double-glazing to reduce traffic noise, and in doing so reveal their valuation of these bads. However, the situation is complicated (again) by the fact that these market goods might have more benefits than simply that of reducing an intangible bad. Averting behaviour occurs when individuals take costly actions to avoid exposure to a non-market bad (which might, for instance, include additional travel costs to avoid a risky way of getting from A to B). Again, we need to take account of the fact that valuing these alternative actions might not be a straightforward task, for instance, if time which would have been spent doing one thing is instead used to do something else, not only avoiding exposure to the non-market impact in question, but also producing valuable economic outputs.

Bresnahan *et al.* (1997) use an averting behaviour model to study how people change their behaviour (by spending less time outside) in response to increasing air pollution levels in Los Angeles. They do not estimate any values resulting from this but do discuss how economic value may be affected by increased use of air conditioning and by the inconvenience of having to spend time indoors.

Advantages of averting behaviour:

- has a sound theoretical basis;
- uses data on actual expenditures and data requirements can be modest.

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Disadvantages of averting behaviour:

- not a widely used methodology;
- can only estimate use values;
- limited to cases where households spend money to offset environmental hazards/nuisances;
- confined to cases where those affected are aware of the environmental issue and act because of them;
- appropriate data may be difficult to obtain.

Cost of illness and lost output

Finally, methods based on cost of illness and lost output calculations are based on the observation that intangible impacts can, through an often complex pathway of successive physical relationships, ultimately have measurable economic impacts on market quantities. Examples include air pollution, which can lead to an increase in medical costs incurred in treating associated health impacts, as well as a loss in wages and profit. Davies (2006) provides a nice example of this type of methodology with respect to calculating the cost of environmental contaminants by their effects on child health in Washington State in the USA. Air pollution can also negatively affect the yields of agricultural crops and if the relationship between the pollutant and the response (the loss of yield) can be established then a subsequent value of lost output can be calculated. Kuik *et al.* (2000) use this approach to estimate the benefits of reducing low level ozone pollution to the Netherlands in terms of increased agricultural output. The difficulty with these methods is often the absence of reliable evidence, not on the economic impacts, but on the preceding physical relationships.

Advantages of cost of illness and lost output:

- theoretically sound;
- very useful where there is a clearly established exposure-response relationship;
- can be a relatively simple exercise where exposure-response relationships have already been established and data on exposure and response is available;

Disadvantages of cost of illness and lost output:

- can only estimate use values;
- uncertainty regarding exposure-response:
    - are there threshold levels before damage occurs?
    - are there discontinuities in the exposure–response relationship?;
- market failures may mean that the prices of market impacts are distorted;
- can be a very complex and resource intensive exercise where exposure-response relationships have not been established and where data on exposure and response is not readily available.

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Pricing approaches

Market prices

Market prices data from ecosystem services that are traded, either in local or international markets, offer perhaps the most visible indication of value. Products such as timber and crops are obvious examples. However, it may be necessary to adjust prices to account for government subsidies or taxes in order to obtain real or so called shadow prices.

A recent example of a study that uses market price data in this context (along with other valuation approaches) is Croitoru (2007). This study estimates the value of non-timber forest products in the Mediterranean region and arrives at a figure of €39/ha of forest.

Advantages of market prices:

- relatively simple.

Disadvantages of market prices:

- can only estimate direct use values;
- prices can be distorted by market failure;
- all pricing approaches are only a partial measure of value.

Opportunity cost

The opportunity cost approach estimates the benefits that are foregone when a particular action is taken. For example, foregone revenues from timber sales and the loss of benefits from foregoing other forest products may be viewed as the opportunity cost of a forest conservation project that prevents extractive activities. In the strictest sense, opportunity cost should be viewed as the next best alternative use of a particular resource. Also opportunity cost allows estimation of the net value of a particular resource. For instance non-timber forest products typically entail a harvesting cost: time and effort spent that could be applied to some other activity if non-timber forest products were not collected. This approach is also used in Croitoru (2007).

Advantages of opportunity cost:

- can be relatively simple;
- can be very useful where a policy precludes access to an area – for example estimating forgone money and in-kind incomes from establishment of a protected area.

Disadvantages of opportunity cost:

- can only estimate direct use values;
- may require detailed household surveys to establish economic and leisure activities in the area in question;
- all pricing approaches are only a partial measure of value.

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Replacement costs

The replacement cost (or substitute goods) approach entails estimating the provision of an alternative resource that provides the function of concern. A wetland that provides protection against flooding could, for example, be valued, at the very least, on the basis of the cost of building man-made flood defences of equal effectiveness.

Shadow project costs consider the cost of providing an equal alternative ecosystem service at an alternative location. Such an approach may also be termed as a 'replacement cost' approach, which measure environmental value by applying the cost of reproducing the original level of benefit.

Advantages of replacement costs:

- can be relatively simple.

Disadvantages of replacement costs:

- can only estimate direct use values;
- all replacement costs approaches are only a partial measure of value.

For further understanding, read Pearce *et al.* (2006) that give a good overview of all these approaches as well as EPA (2000).

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Methods for eliciting non-economic values

There may be occasions where economic valuation is either not appropriate or not possible. This could be due to the nature of the ecosystem service, the degree of uncertainty surrounding environmental change, or because of objections to monetary valuation from stakeholders and/or the researchers involved in the study. In this situation a variety of qualitative valuation methodologies can be undertaken. Some of these are briefly summarised below:

Focus groups, in-depth groups. Focus groups aim to discover the positions of participants regarding, and/or explore how participants interact when discussing, a pre-defined issue or set of related issues. In-depth groups are similar in some respects, but they may meet on several occasions, and are much less closely facilitated, with the greater emphasis being on how the group creates discourse on the topic.

Citizens' juries. Citizens' juries are designed to obtain carefully considered public opinion on a particular issue or set of social choices. A sample of citizens is given the opportunity to consider evidence from experts and other stakeholders and they then hold group discussion on the issue at hand.

Health-based valuation approaches. The approaches measure health-related outcomes in terms of the combined impact on the length and quality of life. For example, a quality-adjusted life year combines two key dimensions of health outcomes: the degree of improvement/deterioration in health and the time interval over which this occurs, including any increase/decrease in the duration of life itself.

Q-methodology. This methodology aims to identify typical ways in which people think about environmental (or other) issues. While Q-methodology can potentially capture any kind of value, the process is not explicitly focused on 'quantifying' or distilling these values. Instead it is concerned with how individuals understand, think and feel about environmental problems and their possible solutions.

Delphi surveys, systematic reviews. The intention of Delphi surveys and systematic reviews is to produce summaries of expert opinion or scientific evidence relating to particular questions. However, they both represent very different ways of achieving this. Delphi relies largely on expert opinion, while systematic review attempts to maximise reliance on objective data. Delphi and systematic review are not methods of valuation but, rather, means of summarising knowledge (which may be an important stage of other valuation methods). Note that these approaches can be applied to valuation directly, that is as a survey or review conducted to ascertain what is known about values for a given type of good.

For more information on these and other non-monetary valuation methodologies plus detail on other forms of assessment refer to Stagl (2007). SDRN Rapid Research and Evidence Review on Emerging Methods for Sustainability Valuation and Appraisal: Final report to the Sustainable Development Research Network. Available at:

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On the use of discrete choice models for modelling non-market behaviour

What are discrete choice models?

Dependent variables in models are often discrete rather than continuous, which implies that there are many cases where conventional regression analysis is not suitable to apply. By "discrete dependent variables" we refer to cases when the dependent variable takes values 0,1,2,… Such values are sometimes meaningful in themselves, for example, when a dependent variable y is the number of persons in a family. But most often the values 0,1,2,… are instead codes for some qualitative outcome. Greene (1997, p. 872) gives the following examples:

- "**Labor force participation:** We equate "no" with 0 and "yes" with 1. These are qualitative choices. The zero/one coding is a mere convenience.
- **Opinions of a certain type of legislation:** Let 0 represent "strongly opposed", 1 "opposed", 2 "neutral", 3 "support" and 4 "strongly support". These are **rankings**, and the values chosen are not quantitative but merely an ordering. The difference between the outcomes represented by 1 and 0 is not necessarily the same as that between 2 and 1.
- **The occupational field chosen by an individual:** Let 0 be clerk, 1 engineer, 2 lawyer, 3 politician, and so on. These are merely categories, giving neither a ranking nor a count."

The typical approach to statistical analysis of models involving discrete dependent variables is similar to conventional regression analysis in the sense that these models try to relate the discrete outcome to a number of explanatory variables. This is done by applying various probability models where the probability that y takes a particular value j, i.e. $P(y=j)$, is viewed as a function of a vector of explanatory variables ($\boldsymbol{x}$) and their associated parameters (β), i.e. $P(y=j) = F(\beta'\mathbf{x})$. A specification of this function requires an assumption of some probability distribution such as the normal distribution and the logistic distribution.

The random utility model

The estimation of the discrete choice model might be made *ad hoc* by simply selecting a probability model that fits the data available. However, it could also be based on more explicit behavioural assumptions such as the random utility model (RUM). For example, a RUM setting is often a point of departure for environmental valuation methods such as the travel cost method and various stated preferences methods including the contingent valuation method and choice experiments (e.g., Haab and McConnell, 2002, Hensher et al., 2005).

In a RUM, an individual is viewed as choosing between J alternatives, which is described by a vector of attributes ($\boldsymbol{a}$). This means that the indirect utility of alternative i for individual k can be written as $v_{ik} = V_{ik}(\mathbf{a}_i, M_k - p_i)$, where M_k is the income of individual k and p_i is the cost incurred when selecting the ith alternative. Given that the individual is characterized by a utility maximizing behaviour, alternative i is chosen if and only if:

$$V_{ik}(\mathbf{a}_i, M_k - p_i) > V_{jk}(\mathbf{a}_j, M_k - p_j) \text{ for all } j \neq i$$

An individual is assumed to know her preferences and to maximize her utility in every choice made. However, these preferences are not known by the researcher, for whom utility therefore appears to be a random variable. An error variable (ε) is included in the utility function in order to capture this randomness, which means that the condition above can be written as:

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$$V_{ik}(\mathbf{a}_i, M_k - p_i, \varepsilon_{ik}) > V_{jk}(\mathbf{a}_j, M_k - p_j, \varepsilon_{jk}) \text{ for all } j \neq i$$

The introduction of randomness implies that it is now adequate to express the condition in terms of the probability that individual *k* chooses alternative *i*:

$$P_{ik} = P(V_{ik}(\mathbf{a}_i, M_k - p_i, \varepsilon_{ik}) > V_{jk}(\mathbf{a}_j, M_k - p_j, \varepsilon_{jk});\ \forall\ j \neq i)$$

An empirical version of this RUM model requires a specification of the probability distribution of the error term and the functional form of the utility function. Some common assumptions are the following:

1. ε is entered into the utility function as an additive term
2. ε has an extreme value type I distribution
3. the utility function is a linear function of the attributes, e.g. $v_{ik} = \beta_1 a_{1i} + \beta_2 a_{2i} + \beta_M (M_k - p_i)$ in a case with two attributes and $M_k - p_i$ as a third explanatory variable

These assumptions constitute the basis for the conditional logit model, i.e. the probability that individual *k* chooses alternative *i* can be computed as:

$$P_{ik} = \frac{\exp(\beta_1 a_{1i} + \beta_2 a_{2i} + \beta_M (M_k - p_i))}{\sum_{j=1}^{J} \exp(\beta_1 a_{1j} + \beta_2 a_{2j} + \beta_M (M_k - p_j))}$$

where the parameters can be estimated through applying standard statistical software packages. However, some packages such as LIMDEP and NLOGIT (see http://www.limdep.com), include particularly many pre-defined estimation procedures for various types of discrete choice models, i.e. there is no need for the users to specify own likelihood functions even for quite advanced and complicated models.

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A travel cost study applied to case study Himmerfjärden

The simulation model for the case study Himmerfjärden considers, for example, the results of various policy options related to reductions of nutrient loadings to the sea. One probable result is an increased Secchi depth. The benefits of such an increase are obtained from applying an earlier travel cost study of the Stockholm archipelago, of which SSA Himmerfjärden is a part. Using a random utility model (RUM) setting and a conditional logit model, Soutukorva (2005) estimated the value of a one-metre Secchi depth improvement in the Stockholm archipelago to 9-29 million EUR (85-273 million SEK) per year (1 EUR = 9.4 SEK). This study was based on a mail questionnaire survey sent to a random sample of residents in the two counties of Stockholm and Uppsala. The vector of attributes ***a*** consisted of three variables considered to explain the respondents' choices of recreational sites in the archipelago: (i) the cost of travelling to the sites including the opportunity cost of travel time, (ii) the bathing water quality at sites as measured by Secchi depth, and (iii) the accessibility to sites by public ferry.

A common problem in travel cost studies is the presence of multi-purpose trips, i.e. respondents have more than one purpose when visiting a recreational site, such as both bathing and visiting a restaurant. Soutukorva (2005) approached this problem by letting the respondents in the survey mark the importance of water clarity for their site choice on a continuous scale. For respondents who put a mark on the right end of the scale ("vital importance"), 100 per cent of the travel cost was included in the estimation. When water clarity was of less importance, travel costs were adjusted accordingly. For those respondents who stated that water clarity was of no importance for their choice of site, travel costs were set to zero in the estimation.

Using the part of the survey data that concerned the case study Himmerfjärden, Kinell (2008) also estimated a conditional logit model, which gave the results reported in Table 1. Model A and B refer to a specification excluding and including the accessibility by public ferry variable, respectively. c is the intercept, and β_{tctime}, β_{sd} and β_{ferry} refer to the parameters associated with the three explanatory variables of travel cost, Secchi depth and accessibility by public ferry.

The estimates in Table 1 are the basis for calculating the compensating variation as a monetary measure of the change in human wellbeing due to a Secchi depth improvement in case study Himmerfjärden. Compensating variation is a measure of the change in the (Hicksian) consumer surplus. An individual's consumer surplus is equal to the difference between the maximum amount of money that he/she is willing to pay for consuming a particular amount of a good and what he/she actually has to pay. The change in consumer surplus is therefore used in economics as a measure of change in wellbeing. See also, e.g., Freeman (2003).

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Table 1: Estimated coefficients (p-values in parentheses)

	Model A	*Model B*
c	-4.539590 (0.00)	-4.506779 (0.00)
β_{tctime}	-0.000960 (0.01)	-0.002184 (0.00)
β_{sd}	0.078781 (0.00)	0.056435 (0.00)
β_{ferry}		0.079149 (0.00)
LR statistics	56.9 (0.00) 2df	245.02 (0.00) 3df

Compensating variation for a changed Secchi depth is obtained as (see, e.g., Haab and McConnell, 2002, p.224):

$$CV = \frac{\ln\left\{\sum e^{\left(v_i^1\right)}\right\} - \ln\left\{\sum e^{\left(v_i^0\right)}\right\}}{\gamma}$$

where superscript 0 (1) denotes the initial (final) Secchi depth level and γ is the marginal utility of income. In the case of an increase in water clarity, compensating variation is the maximum willingness to pay for obtaining such an improvement. For example, computing compensating variation for the particular case of a one-metre Secchi depth improvement in case study Himmerfjärden results in the estimates presented in Table 2 below (1 EUR= 9.4 SEK). This is an example of the results that have also been produced in the simulation model.

Table 2: Aggregate CV per year for a one-metre secchi depth improvement in Himmerfjärden

Explanatory variables included in the model	**CV, EUR/year (SEK/year)**
A: Secchi depth, travel cost (including value of time)	170 151 (1 599 420)
B: Secchi depth, public ferry and travel cost (including value of time)	33 784 (317 566)

While the compensating variation estimate is of great interest because it can be included in an economic evaluation (through cost benefit analysis) of various policy options for reducing the nutrient load to Himmerfjärden, the logit model can also produce other useful results. For example, since the model relates the probability of selecting a site to a number of explanatory variables, it can also predict how a change in an explanatory variable affects this probability. This means that the estimated model can be used for saying something about how a change in Secchi depth is likely to affect the number of visitors to case study Himmerfjärden.

This issue was approached by estimating a quality elasticity of demand or, more precisely, the following elasticity of the probability of a visit to Himmerfjärden as the

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Secchi depth improves (see Ben-Akiva, 1994, or equation (24) in Kinell, 2008, for further explanations):

$$E_{a_{sd,i}}^{P_i} = \frac{\partial \ln P_i}{\partial \ln a_{sd,i}} = [1 - P_i] a_{sd,i} \beta_{sd}$$

This elasticity of the probability of a visit to Himmerfjärden as the Secchi depth improves was computed as a mean of the elasticities estimated for the recreational sites belonging to case study Himmerfjärden. The elasticity indicates a positive relationship between Secchi depth improvement and number of visits to Himmerfjärden.

The next step is to compute the probability of a visit to Himmerfjärden. This probability is estimated by computing the number of visits to Himmerfjärden as a share of the total number of visits to the whole of Stockholm archipelago. This gives a probability of about 0.06, which corresponds to about 231 000 visits[1] per year to Himmerfjärden. Recall that all estimations are based on results from the survey.

The estimated elasticity was subsequently used for computing the increase in the number of visits to Himmerfjärden because of a small (0.1-metre) Secchi depth improvement; see Table 3 for results for the models A and B. The additional number of visits was calculated by multiplying the annual number of visits to Himmerfjärden (about 231 000) by the increase in the probability of a visit to Himmerfjärden after a 0.1-metre Secchi depth improvement.

Table 3: Change in the number of visits to Himmerfjärden following a 0.1-metre Secchi depth improvement

Model	**Number of additional visits**
A	3040
B	4180

Note: The calculations are based on the coefficients estimated in the models (A-B).

The fact that a Secchi depth improvement tends to result in more people visiting Himmerfjärden introduces a feedback loop in the simulation model because it influences aggregate compensating variation.

References

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[1] Note that this number of visits constitutes a lower boundary of the actual number of visits, because the travel cost study only collected data on visits actually involving a travel to Himmerfjärden. For example, visits to Himmerfjärden that take place by simply walking from one's summer house to a beach are not included.

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Environmental benefits transfer

1. Introduction

Environmental benefits transfer is a technique in which the results of previous environmental valuation studies are applied to new policy or decision-making contexts. In the literature, benefits transfer is commonly defined as the transposition of monetary environmental values estimated at one site (study site) to another site (policy site). The study site refers to the site where the original study took place, while the policy site is a new site where information is needed about the monetary value of similar benefits.

In the field of environmental valuation, benefits transfer has been applied extensively in various contexts, ranging from water quality management (e.g. Luken et al., 1992) and associated health risks (e.g. Kask and Shogren, 1994) to waste (e.g. Brisson and Pearce, 1995) and forest management (e.g. Bateman et al., 1995). Costanza et al. (1997) have extrapolated the monetary values of existing valuation studies to the flow of global ecosystem services and natural capital, and have thereby raised a number of questions as well as heavy criticism about the validity and reliability of benefits transfer.

A number of criteria have been identified in the literature for benefits transfer to result in reliable estimates (e.g. Desvousges et al., 1992; Loomis et al., 1995). These are summarised in Brouwer (2000):

- sufficient good quality data;
- similar populations of beneficiaries;
- similar ecosystem services;
- similar sites where these services are found;
- similar market constructs;
- similar market size (number of beneficiaries);
- similar number and quality of substitute sites where the ecosystem services are found.

Study quality is an important criterion, which can be assessed in a number of ways. Above all, one can look at the internal validity of the study results, i.e. the extent to which findings correspond to what is theoretically expected. This internal validity has been extensively researched over the past three decades in valuation studies. Studies should contain sufficient information to assess the validity and reliability of their results. This refers, among others, to the adequate reporting of the estimated willingness to pay (WTP) function. The reporting of the estimation of the WTP function should also include an extensive reporting of statistical techniques used, definition of variables and manipulation of data.

The most important reason for using previous research results in new policy contexts is that it saves a lot of time and money. Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time consuming original research to inform decision-making.

In practice, several approaches to benefits transfer can be distinguished, which differ in the degree of complexity, the data requirements and the reliability of the results. In principle, these approaches are all related to the use of either average WTP values or WTP functions (Box 1). The first approach is most frequently applied, as it requires relatively little data or expertise, and is not very time consuming.

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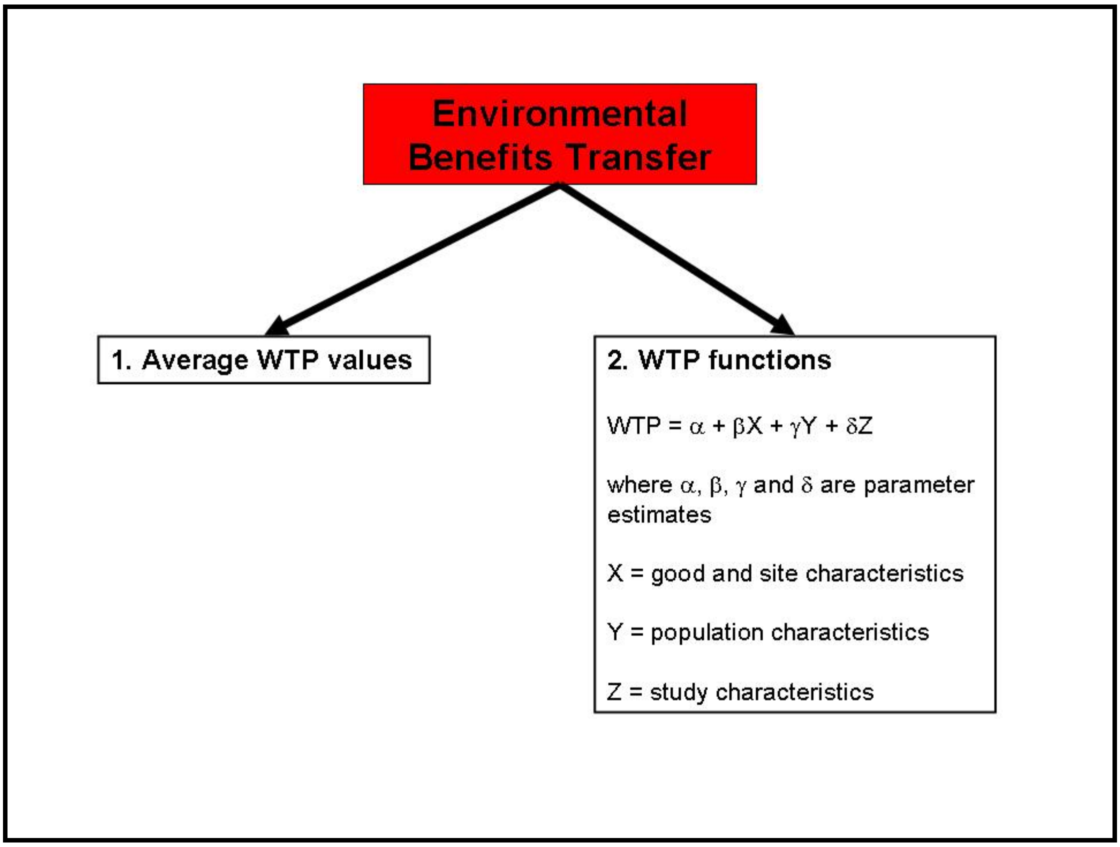
Box 1: Main approaches to benefit transfer



A first approach is where the unadjusted mean WTP point value is used from another study to predict the economic value of the benefits involved at the policy site. Ideally, this study focuses on the same ecosystem services, but was carried out at a different location or at the same location at a different point in time.

A second approach is to use and average the unadjusted mean WTP estimates from more than one study, if available, instead of using the result from one study only. These are the two most frequently applied approaches to benefits transfer in practice. They are relatively data extensive and not very time consuming. However, although a quick and cheap alternative, especially compared to original valuation research, the results may be unreliable if circumstances and conditions in the new decision-making context in which they are used are very different from the ones prevailing in the original research.

A third approach is to use one or more mean WTP values adjusted for one or more factors which are, often based on expert judgement, expected to influence the value estimates at the policy site. For instance, mean WTP is sometimes adjusted for differences in income levels at the study and policy site, based on existing information about the income elasticity of WTP for the service in question, usually taken from the estimated WTP function in the original study.

A fourth approach is to use the entire WTP function from an original study to predict mean WTP at the policy site. Whereas the three previous approaches are referred to in the literature as 'unit value' or 'point estimate' transfers, this fourth approach is usually called 'function transfer'. The estimated coefficients in the WTP function are multiplied by the average values of the explanatory factors in the new policy context to predict an adjusted average WTP value. It has been argued that the transfer of values based on estimated functions is more robust than the transfer of unadjusted average unit values, since effectively more information can be transferred (Pearce et al., 1994). However, this

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approach is usually more data intensive than the first three as information about all the relevant factors have to be ready available or collected.

A fifth approach is to use a WTP function, which has been estimated based on the results of various similar valuation studies. The difference between this approach and the fourth approach is that the WTP function is in this case estimated on the basis of either the summary statistics of more than one study or the individual data from these studies. In the literature, this approach is usually referred to as meta-analysis. Formally, meta-analysis is defined as the statistical analysis and evaluation of the results and findings of empirical studies (e.g. Wolf, 1986).

Finally a sixth approach can be identified. That is the use of a value function - either one which was estimated in a single previous study (fourth approach) or one which was estimated based on multiple previous studies (fifth approach) - in which the coefficient estimates are adjusted when transferring the estimated value function to a new policy context based on prior knowledge. This approach corresponds to a more Bayesian oriented approach to benefits transfer (e.g. León et al., 2002).

The fourth and fifth function approaches assume that the estimated coefficients remain constant, through time, across groups of people and across locations. However, based on previous knowledge and expert judgement, for instance from previous research at similar study sites or previous research at the new policy site, one may find a reason to adjust coefficient estimates. For example, available information about increases in income level in an area and available information about previously estimated income elasticities of WTP at different income levels, the coefficient estimate in the value function can be modified to better fit the new situation. This approach is expected to become especially relevant when functions are used in benefits transfer exercises, which were estimated a long time ago. Obviously, preferences reflected in stated WTP change as a result of changing circumstances. The fifth and sixth approach can be referred to as an 'adjusted function' approach, because in both cases a new WTP function is used, either based on the adjusted original function or a re-estimated function in a meta-analysis of multiple studies.

Thus, while benefit transfer provides a quick and cheap alternative to original valuation research, some conditions must be met if it should provide reliable results. Above all, the local circumstances and conditions in the new decision-making context need to be close enough to the ones prevailing in the original research. The risk of obtaining misleading results may be controlled and reduced by integrating more explaining variables into the transfer. However this also increases the data requirements and the complexity of the analysis. Also, the possibilities of conducting a sound and reliable benefits transfer hinge on the number, quality and diversity of valuation studies available – the larger, the better and the more diverse the existing set of studies is, the more likely will there be a primary study that is "close enough" to the policy site for results to be transferable.

2. Uncertainty and transfer errors

The extent to which non-market economic valuation methods are subject to uncertainties and produce estimation errors has not been subject to systematic analysis. In general, a distinction is made in the economic valuation literature between validity and reliability. Validity refers to the question to what extent a method measures what it is intended to measure. This is often called the 'true' economic value of the ecosystem services involved. Since this true economic value is unknown (the reason why it is being measured through different valuation methods), the validity of economic valuation research is tested in practice by looking at the consistency of research findings compared

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to the theoretical starting points[2]. Reliability concerns the replicability of findings, for example with respect to the extent to which the method is able to produce the same outcomes at different sites across different groups of people at different points in time. Reliability is usually associated with the degree to which variability in contingent valuation (CV) responses can be attributed to random error.

According to Bateman and Turner (1993), reliability is related to two potential sources of variance: variance introduced by the sample and variance introduced by the method. The usual solution to the former is to use large samples. The general approach in the literature for examining the latter has been to assess the consistency of CV estimates over time in so-called 'test-retest' studies (e.g. Loomis, 1989; McConnell et al., 1998). To date test-retest studies have only considered relatively short periods, ranging from two weeks (Kealy et al., 1988 and 1990) to two years (Carson et al., 1997). These have supported the replicability of findings and stability of values across such modest periods[3]. In a recent test-retest study covering a time period which is more than double that considered in previous test-retest analyses (Brouwer and Bateman, 2005), average WTP values and WTP functions appear to be significantly different across this longer time period for a number of reasons, including those expected from standard economic theory (changes in preferences and incomes).

Although benefits transfer is used extensively in practice, very little published evidence exists about its validity and reliability. Table 1 gives an overview of water related studies, which tested the reliability of the transfer of WTP values. Although not complete, Table 1 shows that most studies tested the reliability of transferring contingent valuation results. Three studies investigate the transferability of travel cost studies. The estimated benefits in these studies are related to different types of water use, such as recreational fishing, boating or other recreational water use (also the study by Bergland et al. (1995) and Parsons and Kealy (1994) look at water quality improvements for recreational use). The last column presents the range of transfer errors found in these studies, i.e. the absolute error when using the estimated economic value of a specific water use or water quality deterioration from another study in a new policy context. So, a transfer error of 50% means that the value from the previous study used in the new policy context is 50% higher or lower than the 'true' value in the new policy context. A range of transfer errors is presented as the reliability of benefits transfer was tested for at least two sites (transferring a WTP value from say site A to site B and the other way around) and for both WTP values and WTP value functions (see Brouwer (2000) for more details).

From Table 1, it is difficult to say how large the errors can be expected to be on average when using existing economic value estimates in new decision-making contexts. In some cases they can be very low, in other cases they can be as high as almost five times the value, which would have been found if original valuation research was carried out. No distinct differences can be found based on Table 1 when comparing transfer errors for contingent valuation and travel cost studies.

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[2] In the contingent valuation literature a distinction is made between four different validity concepts (e.g. Mitchell and Carson, 1989): content validity, criterion validity, convergent validity and construct validity. It is mainly the last two validity concepts, which have been tested most in the existing literature. A number of studies have compared, for instance, the outcomes of contingent valuation studies with those from travel cost or hedonic pricing studies or other valuation studies (e.g. Smith et al., 1986; Carson et al., 1996) or the outcomes of different WTP elicitation formats in CV such as open ended or dichotomous choice WTP questions (e.g. Desvousges et al., 1983; Bateman et al., 1995).

[3] An overview of studies investigating the reliability of CV estimates is found in McConnell et al. (1998).

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Table 1: Errors found in water related economic valuation studies testing benefits transfer

Study	*Valuation method*	*Estimated benefits*	*Transfer errors (%)*
Loomis (1992)	TC	sport fishing benefits	5 – 40
Parsons and Kealy (1994)	TC	water quality improvements	1 – 75
Loomis et al. (1995)	TC	water based recreation	1 – 475
Bergland et al. (1995)	CV	water quality improvements	18 – 45
Downing and Ozuna (1996)	CV	saltwater fishing benefits	1 – 34
Kirchhoff et al. (1997)	CV	white water rafting benefits	6 – 228
Brouwer and Bateman (2005)	CV	flood control benefits	4 – 51

Source: Adapted from Brouwer (2000).
Notes: TC= Travel Cost, CV = Contingent Valuation

Another illustration of the accuracy underlying the use of existing economic estimates as proxies for environmental values is presented in Table 2.

Table 2: Break-down of average economic values found in the literature for wetlands in temperate climate zones in US$ per household per year (price level 1995)

	Mean WTP	*Standard error*	*Min WTP*	*Max WTP*
Wetland type				
Saltwater	84.3	40.8	28.5	205.5
Freshwater	88.4	9.2	1.5	400.5
Wetland function				
Flood water retention	138.9	36.6	36.0	265.5
Water recharge	32.3	10.2	4.5	88.5
Pollutant retention	78.8	8.9	13.5	261.0
Wildlife habitat	114.2	19.2	1.5	516.0
Wetland value				
Use value	102.2	12.6	13.5	516.0
Non-use value	53.3	7.2	18.0	117.0
Use and non-use	95.7	19.4	1.5	400.5
Continent				
North America	106.2	11.7	4.5	516.0
Europe	49.2	12.6	1.5	265.5

Source: Adapted from Brouwer et al. (1999).

Table 2 presents an overview of the results of a meta-analysis of 30 CV studies of wetlands in temperate climate zones. The CV studies focus on different issues related to wetland conservation and were carried out at different points in time (in the 1980s and 1990s) in different places (different countries in Europe and North America). A statistical meta-analysis of the findings of the different CV studies produced the summary statistics shown in Table 2.

The summary statistics (average WTP values) show a high degree of variability (measured through the minimum (Min) and maximum (Max) average WTP values found in individual studies). Standard errors, measures of the accuracy of the estimated average values, range between 10 and 50 percent of the summary statistics' average value (i.e. variation coefficient). The 95 percent confidence interval around these estimates is almost two times higher. For instance, the 95 percent confidence interval around the average economic value of freshwater wetlands is US$ 70.4 – 106.4, whereas the 95 percent confidence interval around the average economic value of saltwater wetlands is US$ 4.3 – 164.3. Together with the hydrological function water recharge, floodwater retention has the highest variation coefficient. The variation coefficient related to the economic value of the ecological function wildlife habitat provision is about half the

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size of that. However, the range of values found in the existing literature is highest for this latter ecological function, varying between one and five hundred US dollars per household per year.

The errors reported in Table 1 have to be considered in the light of the purpose the user wishes to use previous valuation results for. In some cases the user may find a transfer error of 50 percent too high, in other cases such an error may be acceptable. The extent to which the transfer errors reported in Table 1 are considered a problem depends upon the acceptability of these errors by the user (policy or decision maker) of the results. User acceptability of these errors will depend upon subjective judgement by the user self, but also on the purpose and nature of the cost-benefit evaluation and the phase of the policy or decision-making cycle in which the evaluation is carried out. The reliability (and corresponding errors) of pre-feasibility studies carried out in an early stage of policy formulation to aid policy development is usually much lower (and errors larger) than the reliability of detailed cost-benefit studies which are looking at the practical implementation of concrete policy measures on the ground. In general, the further the policy or decision-making process has moved forward towards practical implementation, the higher the reliability of the evaluations based on increasing quantity and quality of information. Large errors and low reliability as a result of unresolved uncertainties and lack of information will become less and less acceptable the closer the project moves towards the practical implementation of policy measures on the ground.

3. Trying to explain non-transferability and large transfer errors

A number of reasons have been suggested in the literature why the test results found so far are ambiguous (Brouwer, 2000). First, contrary to many of the market based costs and benefits included in cost benefit analysis (CBA), environmental values are not always well defined, especially in situations where the complexity of the environmental issue extends the complexity of the valuation process beyond reasonable expert and/or public comprehension. This undermines their political and legal acceptability in CBA, especially in those cases where they seriously inflate total benefits (costs) for green (economic development) programmes. In the case of travel costs and hedonic pricing studies, it is usually fairly clear what is measured: a use value revealed through the amount of money people actually paid to enjoy an ecosystem service. On the other hand, in the case of CV expressed WTP values may have a variety of meanings, related to (potential) use and non-use. In fact, they may be so diverse that attempts to aggregate them across individuals to produce a total economic value ultimately obscure what exactly is measured. The problem of correctly interpreting findings on the basis of underlying motivations has sometimes been referred to as a 'technical' survey problem of proper definition of the good being valued. However, it may also reflect people's inability to express much more than a general moral commitment to help financing environmental programmes (Vadnjal and O'Connor, 1994).

A wide range of values produced by a black box undermines the argument put forward to include those values, especially non-use values, which reflect some kind of overall moral commitment to environmental causes and which are expected to stay more or less the same across social groups and environmental domains. If more or less constant, these values would be easily transferable without a need to look at motivations underlying such WTP values. However, values often do differ substantially in practice from case to case.

Secondly, as a result of unclear definition, there is a real risk of double counting when aggregating these values across different stakeholder groups.

Thirdly, instead of solving the problem of aggregation (i.e. the number of stakeholders and the values they hold to be included in the analysis), the inclusion of especially non-use values only seems to aggravate the problem. They show that also non-users may

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attach a value to the ecosystem services involved, but without identifying the boundaries of this specific 'market segment'. On the other hand, CV values elicited in a very specific local context based on a sample of local residents or visitors may also reflect more than simply current and future use values. The historical-cultural context in which these values have come about may be a significant determinant of the elicited WTP values. Also in those cases where stated values seem to reflect upon well-defined local issues, it is important to carefully investigate the broader applicability of these values which may be embedded in specific local conditions when aiming to transpose these values across sites.

Finally, it is perhaps also important to point out that especially CV results reflect a one-time snapshot of people's preferences. Evidence furthermore suggests that CV surveys evoke constructed rather than well-articulated preferences, especially in situations where people are unfamiliar with a specific environmental issue or are asked for a maximum WTP for public goods. Preference and value formation on the basis of the information supplied is not specific to CV, but a more general phenomenon in communication consistent with findings in socio-psychological research of decision-making (Schkade and Payne, 1994). However, the question is how stable constructed preferences and subsequently people's stated WTP in a say 15 to 30 minute interview remain through time and subsequently how legitimate it therefore is to put them together and make them comparable with other value statements at different points in time in a discounted CBA. One could argue the same for market based costs and benefits reflecting existing market prices. Also these costs and benefits are assumed to stay the same through time. However, for these prices often time series are available, which can be analysed and extrapolated.

Finally, the explanatory power of most benefit functions usually does not exceed 30 percent. Although R-squared statistics have to be interpreted with the necessary care in view of the nature of the panel data collected in economic valuation research, most of the variability in stated WTP amounts remains unexplained. Therefore, perhaps not surprisingly, a generally applicable model has not yet been found. The quantity and quality of control included in most models is very limited in terms of the way general site and population characteristics are specified statistically, for instance as dummy variables which merely indicate whether or not a site is accessible to the public or someone earns a specific amount of income. This simple specification of explanatory factors is in sharp contrast with the complex continuous response variable, which is expected to reflect the strength of people's preferences for specified changes in provision levels of ecosystem services.

Furthermore, even if statistically specified adequately, most factors included in these models do not explain why respondents from the same socio-economic group may hold different beliefs, norms or values and hence possess different attitudes and consequently state for instance different WTP amounts, especially in a CV study. Human behaviour as measured in travel cost studies and hedonic pricing studies and behavioural intentions as measured in CV are liable to several influencing factors, as can be learned from the related socio-psychological literature (e.g. Brown and Slovic, 1988). Attitudes are considered an important key to the understanding of people's preferences in terms of WTP (Fishbein and Ajzen, 1975). However, if accounting for attitude variables provides a valid basis for value transfer, then this is bad news for its practical viability since it suggests the need for data collection of such variables alongside people's socio-economic characteristics at the policy site. The data needed to calculate adjusted average value estimates based on a value function at the policy site has to be easily accessible for value transfer to remain a cost-effective valuation alternative.

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4. Towards a protocol of good practice

In principle, the reliability of benefits transfer can be approached from two main perspectives. The first one, which has been dominating the value transfer literature so far, does not question the environmental values themselves. The monetary values found are taken as valid and reliable outcomes of people's valuation. The variability found in valuation outcomes is attributed to differences in study design, good and population characteristics and to some extent value types (use and non-use values). Hence, there may be something wrong with, for instance, the value elicitation mechanisms used, but the values themselves remain undisputed.

A second perspective, advocated in Brouwer (2000) as a complementary approach to the first one, is more critical about the estimated values. Even though a valid transfer can be established when the explanatory power of the transfer model is low (Brouwer and Spaninks, 1999), the question is whether users of environmental valuation results are happy with the numbers they are given from a 'black box'. How can environmental values be reliably predicted across sites and people if currently much if not most of the variability of the values in original studies cannot be explained? This second perspective is focusing much more on the process of value formation, articulation and elicitation in order to better understand the values themselves.

Based on these premises, a number of steps will be highlighted which are considered important to the practice of environmental benefits transfer and monetary valuation of environmental change in general. If previous study results are questionable in terms of validity and reliability, their use in new policy contexts will only result in more controversy. The steps are summarised in Box 2.

Box 2: Practical steps towards a protocol for good practice

Step 1: Defining the ecosystem services to be valued
Step 2: Identifying stakeholders and/or beneficiaries
Step 3: Identifying the various values held by different stakeholder groups and/or beneficiaries
Step 4: Stakeholder involvement in determining the validity of monetary environmental valuation
Step 5: Study selection
Step 6: Accounting for methodological value elicitation effects
Step 7: Stakeholder and/or beneficiaries involvement in value aggregation

Step 1: Defining the ecosystem services

An essential part of the first step is the identification of the relevant ecological functions which underpin the supplied services and the importance of these functions for sustaining ecosystems and hence human systems. Obviously, this requires scoping of the problem in terms of the geographical and temporal scales involved.

Ecosystem services provide different kinds of benefits to different kinds of people. In order to keep the analysis transparent and to avoid double counting, the benefits preserved or foregone have to be identified first, for example in terms of direct and indirect extractive and non-extractive benefits. Examples of direct extractive benefits from renewable natural resources are fish and wood, while examples of direct non-

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extractive benefits are recreational activities in forests, rivers or lakes. Indirect benefits are often found off-site. An example of an indirect extractive benefit from renewable resources is clean drinking water, while an example of an indirect non-extractive benefit is the provision of landscape diversity.

For the purpose of a valid and reliable benefits transfer, the identification of the various economic benefits is not enough. The provision and quality levels of these benefits in the reference and desired target situation are equally important (Fischhoff and Furby, 1988). In practice, reference and target situations in the old and new policy context may differ significantly, seriously limiting the application of previous study results across different policy contexts. Most CV studies lack information about preferences for a variety of reference and target levels; hence the recent increase in popularity of multi-attribute utility based choice models. In the case of CV, no adjustment mechanism is available to account for possible differences. Random utility travel cost models and contingent choice experiments seem to be the only tools available at present which are able to meet this problem.

Step 2: Identifying stakeholders

Different benefits usually accrue to different groups of people. After the various benefits preserved or foregone have been identified, the people who value these benefits for what they are, the beneficiaries, have to be identified. Although this step identifies beneficiaries, not the reasons why these beneficiaries value ecosystem services (see the next step), they are interdependent. To clarify this, an analogy with market goods and services can be drawn. When estimating the economic value of market goods and services, an important step is to look at their market size in order to determine which prices should be used in the value calculation, for example local market prices or world market prices. In principle, one could argue that the same applies to non-market goods and services.

Step 3: Identifying values held by different stakeholder groups

The same good or service may hold different values to different people. An analogy can be made again with market goods and services: within the market place different market segments may exist where different prices prevail. When identifying the benefits of ecosystem services, the reasons why these benefits are considered benefits by various stakeholders have to be addressed at the same time. Benefits can only be identified as such if their value is known. Whether or not this value can be monetised is another question (see the next step).

Step 4: Stakeholder involvement in determining the validity of monetary environmental valuation

One of the underexposed areas in monetary and non-monetary environmental valuation is the assessment of the appropriateness of different valuation procedures in different environmental domains based on their underlying axioms and assumptions. Like traditional economic theory, alternative approaches to environmental valuation based on social processes of deliberation may be questioned on their implicit value judgments regarding the legitimacy of the social-political organisation of the process of value elicitation. Instead of making assumptions a priori, research efforts should perhaps focus more on the processes by which actual public attitudes and preferences towards the environment can best be facilitated and fed into environmental or other public policy decision-making.

One way of making sure that the transfer (valuation) exercise generates socially and politically acceptable results is to get the stakeholders involved who are (going to be)

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affected by environmental change and whose values the researcher and decision-maker(s) are interested in. This stakeholder consultation process provides the researcher with an external validation mechanism of the monetary environmental valuation exercise and helps defining the boundaries of monetary environmental valuation. Stakeholder groups or their representatives can be asked for their most preferred form of public consultation in general and environmental value elicitation in particular before any value elicitation structure is imposed on them. If there is agreement about the monetisation of certain environmental values present in a specific policy context, stakeholder involvement can be very useful in determining what these monetary values should reflect (e.g. in terms of individual WTP). It is then up to the researcher to look into previous studies and see to what extent these values have been estimated in a valid, reliable and, if possible, replicable way.

There usually is increased difficulty in computing monetary economic values from direct extractive to indirect non-extractive benefits. Monetary values for direct extractive benefits (e.g. fish, reed etc.) can often be computed from available market data. In some cases, market data will also be available for indirect extractive benefits (e.g. water consumption off-site). In other cases, one can rely upon non-market valuation techniques. Direct non-extractive values (e.g. recreational benefits) are more difficult to calculate since market data will be absent unless one relies upon some complementary relationship between the non-market benefit and for example actual expenditures made to enjoy the good (as in travel cost studies). Finally, indirect non-extractive benefits are usually the most difficult benefit category to estimate in money terms. Market data will not be available and there may exist a whole range of diverse reasons why people value these benefits, which may be difficult to accommodate in money. CV is usually the only way to estimate these benefits.

Step 5: Study selection

After going through steps 1 to 4, appropriate studies have to be selected. If possible or available, a meta-analysis of these studies will provide a useful tool to synthesise previous research findings, for example by identifying different outcomes as a result of different research design formats. Otherwise, a number of criteria have been identified in the literature to select among studies (Desvousges et al., 1992; Loomis et al., 1995). These criteria are generally applicable (see section 0). Often the selection of existing studies will be based on a qualitative assessment. Study quality is an important criterion, which can be assessed in a number of ways.

First, one can look at the internal validity of the study results, i.e. the extent to which findings correspond to what is theoretically expected. This internal validity has been extensively researched over the past three decades in valuation studies. Studies should contain sufficient information to assess the validity and reliability of their results. This refers, among others, to the adequate reporting of the estimated WTP function. The reporting of the estimation of the WTP function should also include an extensive reporting of statistical techniques used, definition of variables and manipulation of data.

Secondly, the appropriateness of monetising environmental values in a specific context through individual WTP, i.e. their external validity, can be assessed by looking at the actual meaning and interpretability of the values found. Contrary to travel cost (TC) and hedonic pricing (HP), CV allows assessment of the external validity of stated WTP values through the social survey format itself: i.e. via response rates, protest bids and reasons why respondents are willing and able to state a specific payment.

Response rates are often ill-defined in the reporting of CV results. A high non-response, either to the entire survey instrument or the valuation question, raises concern regarding the study’s representativeness, and questions the validity of the survey design employed

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and the extent to which the valuation scenario in the questionnaire was comprehensible and credible (Arrow et al., 1993).

Criteria to determine whether or not a respondent is a legitimate zero bidder to a WTP question or a protest bidder are often arbitrary. A lot of studies do not report these criteria at all. No guidelines exist as to how much protest responses invalidate a survey. It is common practice to exclude them from further analysis, classifying them as 'non-usable response' without providing detailed information why respondents protested. Protest responses reveal much more useful information than they have been given credit for in CV research. They can be used as an indicator of the acceptability of the use of the monetary environmental values by different stakeholder groups.

Asking respondents for the reasons why they protest against the WTP question or why they are willing and able to state a specific payment is considered of paramount importance to assess the appropriateness of the survey and the actual meaning of their replies. Understanding the meaning of answers, especially to the valuation questions, is a prerequisite to define the appropriate context in which the survey results can be used and how they should be interpreted. Therefore, besides thorough pre-testing of survey formats, it is recommended that post-survey debriefings of interviewers and respondents are used, individually or in a group, to discuss the actual meaning of the answers given in the questionnaire.

Step 6: Accounting for methodological value elicitation effects

Different research designs in environmental valuation methods such as TC, HP and CV have resulted in different results. In TC and HP models, most of the differences seem to originate from the specific model used, the statistical estimation method, the inclusion or exclusion of specific explanatory variables, the definition of these variables and data quality. It is difficult to recommend adjustment mechanisms for these differences in research design. For instance, which statistical model specification is expected to provide the most robust results? Random utility models provide certain advantages over the traditional zonal TC models, but at the same time there is an increase in complexity with respect to the statistical models used, the assumptions underlying the computational heuristics of these models and their data requirements. This also applies to most contingent choice experiments and CV studies using iterative bidding formats.

In CV different survey elements have been shown to result in different WTP values. A number of research design effects have been investigated in the past, of which payment mode, elicitation format, the level of information, sensitivity to scope and/or embedding effects are probably the most important ones. As in TC and HP models, it is often hard to tell how CV findings should be modified based on the specific research design used. In accordance with best practice recommendations, generally a conservative approach seems to be preferable (Arrow et al., 1993).

Step 7: Stakeholder involvement in value aggregation

After one or more studies have been selected and values are found which reflect the values policy or decision-makers are looking for under the specific circumstances, these values can be adjusted, if necessary and secondary data at the policy site are available, for differences in site and population characteristics using the estimated WTP function or average WTP value. These modified values can then be discussed again with (representatives of) different stakeholder groups to which they relate before they are extrapolated over the relevant population which is (going to be) affected by the environmental change. Also this should be discussed with the stakeholder groups involved. Finally, the economic aggregate is included in a CBA together with other

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economic costs and benefits, which can then play its part in the facilitation of the overall, real world, multi-criteria decision-making process.

5. Examples

Technical approach

This first example looks at the transferability of visitor valuation of the recreational and amenity benefits provided by the Broads National Park, one of the most extensive freshwater wetlands in the UK. The example is based on Brouwer and Bateman (2005). More specifically, this example illustrates the stability of WTP values and WTP functions over an extensive period of time. The example considers a time period between surveys which is more than double that considered in previous test-retest analyses. Whereas such previous studies have reported stable values over relatively short time periods, the example presented here finds a statistically significant decrease in real WTP over this more extended time period. The issue of temporal stability over extended periods is one of more than academic interest. CBAs frequently employ values estimated some considerable time prior to those analyses. Temporal stability is therefore implicitly assumed rather than explicitly tested. Yet there is no reason to suppose that values for non-market goods should remain constant over extended periods.

Temporal stability is addressed through the application of two matching surveys, concerning the same case study area (the Norfolk Broads in the UK), focusing on the same ecosystem service and valuation scenarios (flood protection and conservation of freshwater wetland habitat and associated recreational amenities), using the same payment vehicle (coercive taxation), the same sampling frame (random in-person interviews) applied to the same sample population (visitors to the area), but sampling at different points in time, namely in the summers of 1991 and 1996.

The Norfolk Broads are a large freshwater wetland area located in East Anglia, UK. The area consists of a system of shallow lakes, marshes and fens, linked by low-lying rivers. The site is of national and international wildlife importance, being a designated Environmentally Sensitive Area and containing twenty-four Sites of Special Scientific Interest, including two sites notified under the international RAMSAR convention. The area is also a major focus for recreation, attracting more than one million visitors a year, of which 200,000 spend their holidays on boats hired for a week or longer (Broads Authority, 1997).

The character of the low-lying landscape of the Broads depends upon 210 km of reinforced river embankments for protection from saline tidal waters. However, at the time of the surveys these flood defences were increasingly at risk from failure, because of their age, erosion from boat wash and the sinking of the surrounding marshlands. Thus, the standard of flood protection afforded by these man-made defences was decreasing over time. If flood defences were breached, the ensuing saline inundation would fundamentally and enduringly alter the nature of the area both in terms of its habitat capabilities and in respect of the recreational opportunities currently afforded.

In 1991 the National Rivers Authority, later named the Environment Agency, initiated a wide-ranging 'Flood Alleviation Study' to develop a cost-effective strategy to alleviate flooding in the Norfolk Broads for the next 50 years (Bateman et al., 1992). The flood alleviation study consisted of five main components: hydraulic modelling; engineering; benefit-cost assessment; environmental assessment; and public consultations. The item of most relevance here is the benefit-cost assessment, which compared benefits of undertaking a scheme to provide a particular standard of flood protection to the corresponding costs. Although market benefits of flood alleviation have been considered

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in terms of agriculture, industry, living conditions and infrastructure (Turner and Brooke, 1988), the value of the non-market benefits from the area was uncertain.

As part of the benefit-cost assessment, a large CV study was mandated in 1991 (Bateman et al., 1994; 1995) and a follow-up carried out in 1996 (Powe, 1999; Powe and Bateman 2003; 2004), in order to assess user valuations of conserving the area in its current state. The studies aimed, among other things, to provide a valid and reliable monetary estimate of the current recreational and amenity benefits enjoyed by visitors to the Broads. Findings were used to inform a CBA of various flood defence options (Brouwer et al., 2001). The cost-benefit ratio found ranged between 0.98 and 1.94 (National Rivers Authority, 1992). The results, including the findings from the 1991 CV study, were submitted to the relevant Ministry of Agriculture, Food and Fisheries as part of an application of central government funding support for the proposed flood alleviation strategy. Following lengthy consideration of this application, the Environment Agency received conditional approval for a programme for bank strengthening and erosion protection in 1997 (Environment Agency, 1997). The actual scheme was taken forward in 2000 on the basis of a long-term private-public partnership scheme between the EA and relevant government support ministries and a private engineering firm consortium.

Temporal reliability of the dichotomous choice CV models estimated in this study was tested by examining the statistical equality of unadjusted average WTP values (hypothesis 1) and the dichotomous choice WTP functions (hypothesis 2). An iterative approach was developed in order to see how much control is needed to produce transferable models of WTP. These models are generated by progressively blending theoretically expected determinants of WTP with additional ad-hoc variables, which may be more transitory in their effect. This approach involves a gradual expansion in the number of explanatory variables added to a model of WTP. At each addition of a variable temporal transferability is assessed by applying the model to both the 1991 and 1996 data and undertaking various tests. This progressive expansion approach should in principle allow the identification of the optimal level of control for transferability. This approach is compared to that obtained by estimating a statistical best-fit model for a given dataset and transferring this to the other survey period and vice-versa.

For each model transferability is assessed both forward in time (from 1991 to 1996) and back (from 1996 to 1991) using statistical tests for coefficient stability as per Brouwer and Spaninks (1999). A further test of the transferability of each specification is obtained by pooling the data and assessing transferability through application of the Likelihood Ratio test as per Downing and Ozuna (1996) and Carson et al. (1997). For this latter test data from the two surveys are pooled and a dummy variable included to represent the year in which the study was undertaken. If study year has a significant impact on respondent WTP, this implies that the study results are not transferable. The pooled regression results are the same as the outcomes of the Likelihood Ratio test.

Mean WTP values based on parametric and non-parametric estimation approaches are presented in Table 3. In order to be able to compare the 1991 and 1996 WTP values, the 1996 values are corrected for intervening differences in purchasing power.

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Table 3: Mean real WTP values from the 1991 and 1996 surveys (£ p.a. in 1991 prices) obtained from the parametric logistic model and (lower bound) non-parametric Turnbull model

	Parametric Linear-Logistic		***Non-parametric Turnbull***	
	1991	**1996**	**1991**	**1996**
Mean WTP (£)	248.1	215.8	54.2	37.8
Standard error	23.3	29.3	2.9	2.4
95% CI {1996 – 1991}	{-34.3 ; -30.3}		{-16.6 ; -16.2}	
Min-max values	-∞ - +∞	-∞ - +∞	0-200	0-200
N	1747	1108	1747	1108

The results from the linear-logistic and Turnbull models suggest that visitor valuation of the recreational and amenity benefits provided by the Broads has decreased across the period between the two surveys. In constant prices, mean WTP calculated from the linear-logistic model is 13 percent lower in 1996 than in 1991, and 30 percent in the case of the Turnbull model. The observed difference in income levels between the 1991 and 1996 visitors is one possible explanation for this decrease.

Although the Turnbull model is known to provide a lower bound for mean WTP, the large difference between the Turnbull and linear-logistic model is striking. The parametric estimates are about five times higher than the non-parametric estimates. No big differences exist in terms of the accuracy of the estimates. In relative terms the standard errors of the linear-logistic estimates are only slightly higher than the standard errors of the Turnbull estimates. The differences in mean WTP are statistically significant as can be seen from the 95 percent confidence interval constructed around their difference based on the standardised normal variable (z). The estimated differences indicate that the real value of the recreational amenities in the Broads have decreased by 3 to 6 percent per annum over the study period. This significant decrease in real WTP is in contrast to the non-significant changes noted over shorter periods and may well be a consequence of the longer interval under consideration in this example.

Results from our various analyses of model transferability are shown in Table 4. From Table 4 it can be observed that, using the Likelihood Ratio test, all models appear transferable. However, adopting the Wald test (which is more stringent) yields a more mixed result, but one from which a clear pattern emerges. Focusing upon these latter tests, both models relying solely upon variables suggested by economic theory (models using the Bid variable alone or those supplementing this with the household Income variable) are transferable. However, when such models are extended through the addition of more ad-hoc variables, not derived from theory, transferability becomes sporadic. Here, those models using the binary Local variable (identifying those respondents who live near to the study site) do transfer, whereas those substituting in the continuous Distance variable (the number of miles travelled to reach the site) fail Wald tests of transferability, questioning the usefulness of more sophisticated distance-decay relationships in models of WTP for transfer purposes. Statistical best-fit models also fail Wald transferability tests. This reflects the differing determinants, which enter each of these models.

Hence, while previous studies considering shorter periods have shown no significant difference in real WTP values, the analysis presented here reveals a significant difference across a longer period of time. Tests of model transferability indicate that simple models, based solely upon variables derived from economic theory, are transferable across this period. This suggests that underlying relationships for such key determinants are stable even across this longer period. However, expanding models by including theoretically

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unanticipated factors brings ad-hoc and possibly transitory factors into the models, which consequently prove non-transferable.

Table 4: Transfer test results from the dichotomous choice CV models

		Model specification							
Transfer	***Test***	***Bid***	***Bid Income***	***Bid Income Distance***	***Bid Income Local***	***Bid Income Distance Scenery***	***Bid Income Local Scenery***	***Best fit 1991***	***Best fit 1996***
Transfer of the estimated 1991 models to 1996	Wald	0.93	3.71	9.70	3.51	13.20	5.88	20.50	15.03
	$\chi^2_{critical}$	5.99	7.81	9.45	9.49	11.07	11.07	14.07	12.59
	LR	0.58	2.19	6.19	2.07	7.97	3.23	11.49	10.40
	$\chi^2_{critical}$	5.99	7.81	9.45	9.49	11.07	11.07	14.07	12.59
Transfer of the estimated 1996 models to 1991	Wald	1.64	5.31	15.98	4.98	19.92	7.45	26.35	30.61
	$\chi^2_{critical}$	5.99	7.81	9.45	9.49	11.07	11.07	14.07	12.59
	LR	0.58	2.19	6.19	2.07	7.97	3.23	11.49	10.40
	$\chi^2_{critical}$	5.99	7.81	9.45	9.49	11.07	11.07	14.07	12.59

Notes: Critical values at 5%.
☐ = null hypothesis of model equality cannot be rejected (model is transferable)

Using commonly used testing procedures in the benefits transfer literature, it can be shown that also dichotomous choice models extended with these ad-hoc factors are transferable, even though the residual variance in these statistically best fit models is significantly different in the two survey years. Contrary to previous findings, this seems to suggest that the unobserved determinants of preference embedded in the stochastic components of utility over time is not stable in this study. The 1996 model explains less of the variability in the dependent variable than the estimated 1991 model. Hence, important determinants of WTP, which have stayed unobserved, may have been overlooked.

In conclusion, this study suggests that over extended periods real WTP for public goods such as the flood protection and wetland conservation scheme considered here can change by statistically significant amounts. However, the analysis suggests that underlying economic theoretic determinants of WTP remain stable over such periods. Nevertheless, ad-hoc changes in determinants other than those predicted by theory can result in non-transferability of extended (and statistically best-fit) models. This suggests that transfer exercises might usefully focus upon models with firm theoretical underpinnings rather than incorporating more transitory factors.

Interactive approach

In this second example, the application of benefits transfer will be illustrated with the help of a Dutch case study based on Brouwer and van Ek (2004). For centuries the Dutch have reclaimed and drained land and raised dikes to keep their feet dry. Protection against flooding has always been the Government's primary water policy objective in a country of which approximately two thirds is situated below sea level. Dikes have always been the most important means to achieve this. Since the 1990s Government policy is focusing on alternative ways to maintain existing flood protection and safety levels, such

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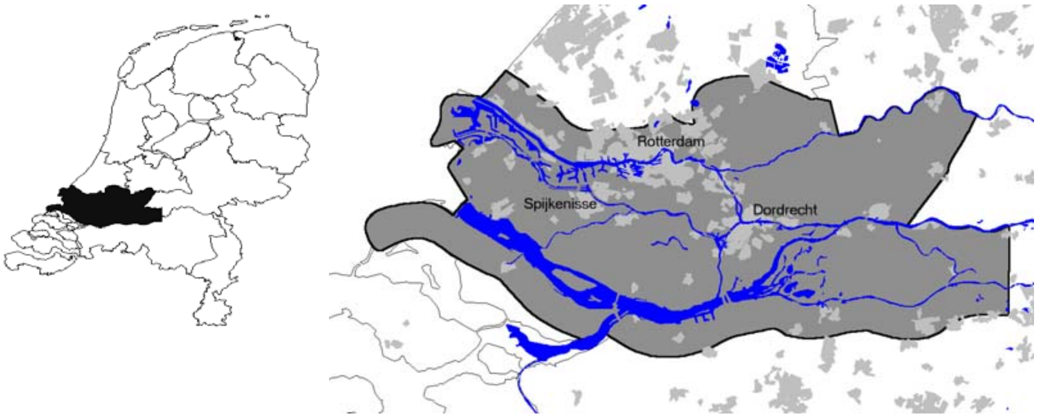
as land use changes in spatial development plans and the restoration of the natural resilience of water systems, including wetlands and floodplains, to absorb excess water[4]. The natural dynamics and flexibility of water systems have been severely undermined in the past through normalisation of rivers, drainage of land and an increase in the built-up area in traditional wetlands and flood plains.

From 1998 until 2000 a Government Working Group investigated in a pre-feasibility study various options for land use changes and floodplain restoration in the Lower River Delta along the rivers Lek, Merwede, Meuse and Waal in the Netherlands (Figure 1). The Lower River Delta is the estuary of the Rhine and the Meuse in so far as these rivers are influenced by the tides. The critical situations during the floods of 1993, 1995 and 1998 when polders were threatened and tens of thousands of people had to be evacuated prove how topical the danger of flooding is in this region. Awareness is growing that alternative measures have to be taken besides raising dikes to prevent the Lower River Delta from flooding in the future.

Figure 1: Location of the Lower River Delta in the Netherlands



Following the floods in 1993 and 1995, existing dikes were quickly strengthened. However, this measure was largely taken to catch up with necessary maintenance and strengthening of dikes to ensure public safety levels in the short term. To maintain present safety levels and anticipate expected water level rises between twenty centimetres and one metre and fifteen centimetres over the next fifty years (based on different climate change and sea level rise scenarios), alternative land use change and floodplain restoration measures (hereafter referred to as managed realignment) were identified in the area, which provide the same safety levels. These measures will be implemented stepwise between 2000 and 2005, 2006 and 2015, and subsequently from 2016 until 2050. Based on the legally defined safety norms in the area, these measures are part of a planning strategy that is designed to prevent, where possible, new rounds of dike reinforcement and encourage multi-functional use of land and the development of biological diversity present (de Jong et al., 2000). Examples of these measures are shown in Figure 2, Figure 3 and Figure 4.

As part of the Working Group's task, the aggregate effects of these sets of measures in the long term were examined and assessed in detail. Besides an environmental impact assessment, also an economic analysis was carried out. However, as often is the case, an integrated assessment based on these two separate studies was lacking. The expected impacts of the proposed managed realignment measures are shown in Table 5.

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[4] This new policy is laid down in the fourth National Water Policy Document published in December 1998 and more recently in the recommendations of the Commission looking at important water management issues in the twenty-first century (Commissie Waterbeheer 21^{e} Eeuw) published in August 2000 and the Government's policy paper with respect to these recommendations published in December 2000, titled "*A different approach to water*".

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Figure 2: Deepening of rivers

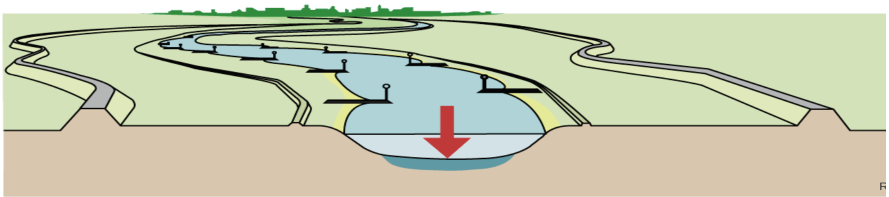

Figure 3: Deepening floodplains

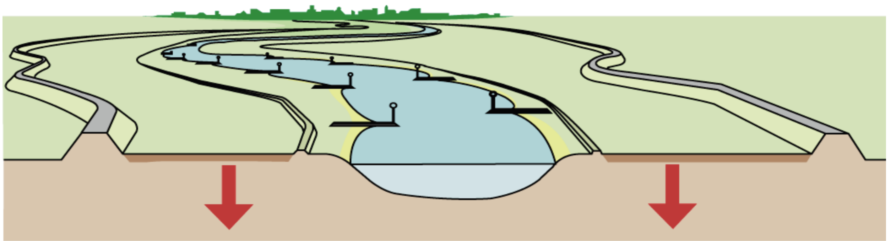

Figure 4: Realignment and floodplain restoration

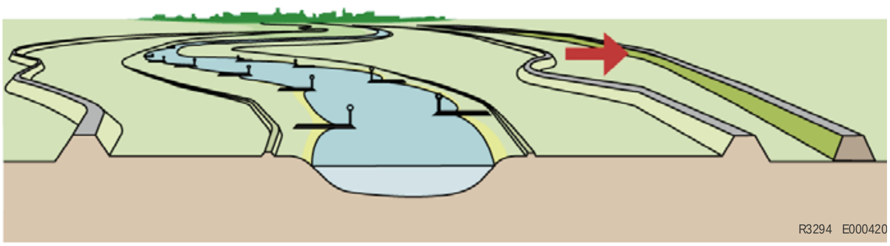

Table 5: Expected impacts of managed realignment compared to 'holding the line'

		Priced		***Non-priced***	
		Efficiency	*Redistribution*	*Efficiency*	*Redistribution*
Direct	*Principal Users*	Investment costs Damage costs		Discharge capacity Public perception safety	
	Third parties	Benefits from sand and grit extraction	Income losses in agriculture and industry	- Biodiversity conservation - Public perception dislocation	Employment in agriculture and industry
Indirect		- Recreational benefits - Commercial shipping benefits		Change in water infrastructure	

A distinction is made between priced and non-priced effects, and direct and indirect effects. The most important non-priced positive effects in the case of the proposed managed realignment measures are changes in the discharge capacity of the water system, public (perception of) safety and biodiversity restoration. The investment costs needed to implement the managed realignment measures and consequently the damage costs avoided are examples of direct priced effects. The investment costs are borne by

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the principal who carries out the project (the Government). Important user groups in the region are people who live and own houses in the area, farmers and industry. Their properties and current and future economic interests will be protected by the proposed measures (at the expense of the relocation of a smaller number of houses and businesses). Third parties who benefit from the proposed managed realignment measures are the sand and grit exploitation companies in the area and, consequently, the construction industry, and possibly dredging companies as a result of increased sedimentation.

In view of the positive effects on nature and landscape, the area is expected to become more attractive for recreational activities. The attraction of extra visitors is expected to create more income in the region. These recreational benefits are considered an important indirect effect. The possible effects of the proposed alternative flood control measures on commercial shipping are also indirect effects, which can be relatively easily valued with the help of market prices. The net effect on commercial shipping can be positive or negative. On the one hand, the deepening of river beds and floodplains and the creation of additional water courses is expected to increase commercial and recreational shipping possibilities, while the change in the water infrastructure may also enhance the accessibility of the area. On the other hand, widening the rivers also lowers water levels throughout the river basin, in which case the shipping possibilities decrease.

Another distinction is made between efficiency and redistribution effects. Efficiency effects are included in the economic CBA, while redistribution effects are excluded. Redistribution effects refer to effects which may have important institutional or financial consequences, but which do not influence the economic output of a country, measured in terms of national income or value added. Examples are the loss of income and employment in agriculture and industry in one area or region as a result of the implementation of the proposed land use changes and floodplain restoration measures, which are off-set by income generation elsewhere as a result of the re-location of farms and businesses.

The structure of Table 5 is based upon the manual for Cost-Benefit Analysis published in 2000 by the Dutch Ministry of Transport, Public Works and Water Management and the Ministry of Economic Affairs (Eijgenraam et al., 2000). This manual was developed to encourage a more integrated assessment of the various impacts of large infrastructure projects in the Netherlands. Effects which cannot be valued in money terms are included, where possible in quantitative terms, in the balance sheet as so-called 'pro memoriam' items. However, in this case, the Working Group's question was to explicitly value the non-priced social and environmental effects of the proposed alternative flood control measures in money terms in order to assess their effect on social welfare. A preliminary assessment of the economic costs of managed realignment in a cost-effectiveness analysis showed that this option was much more expensive than traditional dike strengthening (holding the line). The total costs of holding the line were approximately £500 million, while the total economic costs of managed realignment were estimated at about £4 billion (Brouwer et al., 2001). The most important reason for these high costs for managed realignment was the fact that the measures are proposed in one of the most densely populated and economically developed areas in the Netherlands with an enormous complex infrastructure, which is expected to be affected significantly by the proposed managed realignment measures.

The Working Group expected that economic (monetary) estimation of the non-priced benefits of managed realignment compared to holding the line might be decisive in concluding whether managed realignment is preferred compared to holding the line. Hence, an important first step was to get the necessary authorisation to carry out an economic valuation study of the main non-priced benefits.

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The assessment of the economic value of the expected non-priced social and environmental benefits (public safety and biodiversity restoration) was based on the meta-analysis carried out by Brouwer et al. (1999) based on 30 international studies looking at the economic values of various wetland ecosystem functions (see Table 1 in section 2). In the Netherlands, no valuation research exists with respect to managed realignment. The 30 studies investigated by Brouwer et al. (1999) produced just over 100 willingness to pay (WTP) values. These values were examined in detail and related to the four main hydrological, geochemical and biological ecosystem functions performed by wetlands: flood water retention, surface and ground water recharge, nutrient retention and export and nursery and habitat for plants, animals and micro-organisms and landscape structural diversity. The economic values associated with these four functions are presented in section 2.

The economic values associated with the various wetland ecosystem characteristics are expressed in average willingness to pay (WTP) per household per year. Very often mean values are related to the size of an environmental asset and expressed accordingly, for example in pounds sterling per hectare. This suggests that the average values can be transferred freely and unconditionally over large and small sites irrespective of the number of people who benefit from these sites. An example is the study carried out by Costanza et al. (1997), where based on average values per hectare the total economic value of the world's ecosystem services was estimated. It is not only the average value used to estimate the value of non-priced environmental benefits which has caused discussion about the 'right' prices, also the determination of the 'market size', i.e. the number of beneficiaries, has proven to contribute to a large extent to the controversy of using monetary estimates in CBA (e.g. Bateman et al., 2000). Expressing average values per household per year implies that the user of the average values has to think carefully about the exact market size in order to be able to calculate a total economic value, which can be used in the CBA.

The values presented in Table 1 in section 2 show an average WTP ranging from 18 pounds sterling for the wetland function surface and ground water recharge to 77 pounds sterling for flood water retention. The fact that the function flood water retention is valued highest conforms to expectations considering the possible risks to life and livelihood as a result of flooding and the capacity of wetlands to reduce this risk. No significant difference exists between average values for fresh and saltwater ecosystems. Use values for wetland ecosystems are significantly higher than non-use values (because of the high value attached to flood water retention). Table 1 also shows that use and non-use values cannot simply be added, as suggested in the literature (Hoehn and Randall, 1989) in order to get a total economic value.

In view of the fact that no valuation results are available in the Netherlands to estimate the economic value of the non-priced benefits of the proposed managed realignment measures, the Working Group agreed to use the values examined in the meta-analysis as the basis for the estimation of a total economic value to be used in the CBA. The results from the meta-analysis were considered the best guesses available. Hence, an important second step was to get the authorisation to use the available information about the estimated economic values of wetland ecosystem functions. The fact that these values were based on not one, but thirty international economic valuation studies, most of which were published in internationally renowned journals, is expected to have played an important role in the acceptance of the estimated average values.

The total economic value of the non-priced benefits (i.e. the public perception and valuation of safety, biodiversity preservation and landscape change) is calculated based on the economic values for flood water retention (£77/household/year) and wildlife habitat and landscape diversity (£63/household/year). These values are adjusted for the income differences found between countries, and the fact that use and non-use values

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cannot simply be added. These corrections result in an average WTP for both flood water retention, wildlife and landscape amenities of £53/household/year[5].

Next, the market size was determined in terms of number of households which are expected to benefit from the proposed managed realignment measures. Together with the Working Group, it was agreed that more or less the whole population of the South-Holland province will benefit. South-Holland contains approximately 1.5 million households. Multiplying this by an average value of £53/household/year results in a total economic value of £80 million per year. Discounted at the prescribed 4% discount rate by the Dutch Treasury over the next 50 years gives a present value of the total economic value of £1.8 billion. The inclusion of this economic value in the CBA still results in a net welfare loss of £900, see Table 6.

Table 6: Present value of costs and benefits of managed realignment in billion pounds sterling (2002 prices)

Costs		**Benefits**	
Investment costs	1.8	Economic risks avoided	0.8
Production loss agricultural land	1.3	Revenues sand extraction	0.3
Maintenance costs	0.7	Economic value public safety and biodiversity preservation	1.8
Total	3.8	Total	2.9

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[5] First, the average values are multiplied by 0.61 (based on estimated regression coefficient) to correct for income differences. Secondly, the income adjusted average values are added and multiplied by 0.62 ([use and non-use]/[use] + [non-use] = 53/85=0.62) to account for the fact that use and non-use values cannot simply be added.

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An instrument for screening existing valuation studies

Introduction

Carrying out environmental valuation studies might sometimes be expensive and time-consuming. An obvious question is therefore whether results from earlier valuation studies can be generalized to new policy settings. For example, could existing results concerning the benefits of an improved water quality in a Polish coastal area be used for saying something about the benefits of such an improvement in a coastal area in France? Such a generalization of valuation results are referred to as benefit transfer, which usually consists of three steps:

1. Identification of environmental valuation studies being potentially suitable for benefit transfer by searching in the scientific and grey literature and/or using databases, among which the Environmental Valuation Reference Inventory (www.evri.ca) is the most comprehensive. There are also smaller and less international databases available, such as the Nordic Environmental Valuation Database (www.norden.org/pub/sk/showpub.asp?pubnr=2007:518), the Australian ENVALUE (www.epa.nsw.gov.au/envalue), and the Swedish ValueBase^{SWE} (www.beijer.kva.se/valuebase.htm). See also McComb et al. (2006) for an overview of international valuation databases.

2. Evaluation of the quality of the studies identified in step #1. This is a very important step in the process. Studies must be screened to identify those which are of a sufficiently good quality to make them suitable for use in benefit transfer. Whilst studies published in peer-reviewed journals might be expected to be of good quality, studies in the grey literature might not have been subject to any quality control. Quality is a multi-faceted feature and it is therefore difficult to create practical quality assessment instruments (QAIs) for valuation studies. One of the few that has been produced is downloadable as a Swedish EPA report from http://tinyurl.com/6phn4p.[6] This QAI is briefly described below. A form to be used by an evaluator of a valuation study is available at http://tinyurl.com/5twq62.

3. Transfer of benefits from the studies considered in step #2 to be of acceptable quality. This procedure entails the choice of different transfer methods and their application is an extensive issue which is thoroughly presented in section 0 of this report. The remaining part of this section therefore consists of a description of step #2 only.

[6] Söderqvist, T., Soutukorva, Å. (2006) *An instrument for assessing the quality of environmental valuation studies*. Report, Swedish Environmental Protection Agency, Stockholm. Available at: http://tinyurl.com/6phn4p.

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Step #2: Screening existing valuation studies

One of the objectives of the QAI in the Swedish EPA report is to communicate crucial aspects of quality to potential evaluators who might not be experts in environmental valuation but have some basic knowledge of environmental economics and statistics/econometrics. This QAI therefore aims at being based on the objectively observable characteristics of valuation studies in order to avoid the kind of subjective assessments that only evaluators equipped with expert knowledge are able to make.

Figure 1 shows the procedure for using the QAI. As indicated by the figure, the QAI is based on an identification of a number of factors related to quality for:

a. valuation studies in general, irrespective of what valuation method was used (see section 3.1 in the QAI), and;
b. the application of particular valuation methods (see sections 3.2 to 3.9 in the QAI).

The valuation methods considered in the QAI include revealed and stated preference methods as well as other methods that are less firmly rooted in welfare economics theory. The quality of a valuation study is thus assessed partly through the quality factors in (a) and partly through the quality factors that according to (b) are relevant for the valuation method(s) used in the valuation study. In order to provide an overview, all these quality factors are listed in Box 1 and Box 2.

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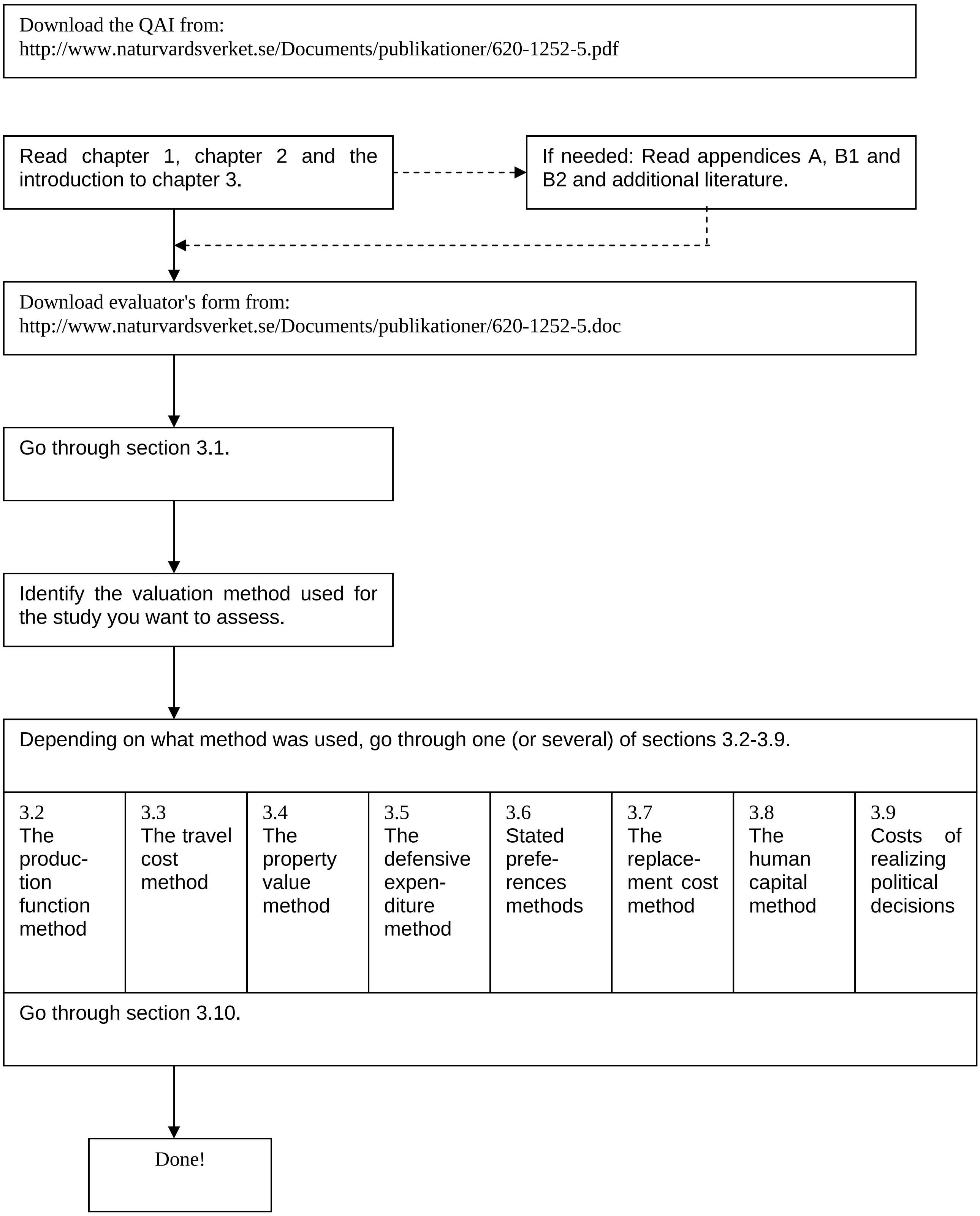
Figure 1: How to use the QAI



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In the QAI, each quality factor is subject to a short discussion, which is followed by one or several check-list questions associated with each quality factor. The purpose of the check-list questions is to make the quality factors more concrete. Most of the questions can be answered by "yes", "no" or "don't know" and they were framed so that "yes" answers are an indicator of good quality. However, "no" or "don't know" answers are not necessarily an indicator of bad quality; this depends on the context and the QAI therefore includes fields for filling in comments that supplement the answers to the check-list questions (e.g. comments about whether a "no" implies a serious weakness of the valuation study or not). Other check-list questions relate to information associated with quality, such as, for example, the non-response rate to a mail questionnaire or interview survey.

Box 1: Quality factors for all valuation studies irrespective of valuation method employed

1. Earlier reviews
2. Principal/funder
3. Valuation method
4. Sensitivity analyses related to results from statistical/econometric analyses
5. Are future values discounted?
6. Primary data or secondary data?
7. Data collection
   - Survey, population and sample
   - The design of the data collection work
   - Data collection method
   - Non-response
   - Survey instrument
8. Access to data
9. Validity test
10. Natural scientific/medical basis

Finally, the QAI is concluded by an opportunity for an evaluator to give an overall quality assessment, based on the answers to the check-list questions and all other considerations that the evaluator might have (section 3.10 in the QAI). It should be emphasized here that the most important feature of the QAI might not be to find a precise answer to a particular check-list question or to arrive at an unambiguous conclusion on overall quality, but that the QAI simply gives hints to an evaluator on what to look for in a study in order to get an idea of its quality. If no major concerns about the quality of the study arise, it should be safe to proceed to step #3, i.e. the actual benefit transfer procedure. In Söderqvist and Soutukorva (2009), the QAI is applied to two valuation studies, and it might be helpful to have a look at how this was done before applying the QAI for the first time.

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Box 2: Quality factors for particular valuation methods employed

The production function method

1. Natural scientific basis
2. Estimation of changes in producer surplus
3. Modelling of the whole market including dynamic effects

The travel cost method

1. Definition of site(s)
2. Sampling strategies
3. Model specification
4. Calculation of travel costs
5. Opportunity cost of time
6. Multipurpose trips
7. Selection of environmental quality variable

The property value method

1. Property values
2. Property attributes
3. Selection of environmental quality variable
4. Choice and estimation of model

The defensive expenditure method

1. Properties of the good
2. Procedure for estimation of the economic value

Stated preference methods

1. Acceptance and understanding of the valuation scenario
2. Description of effects of the environmental change
3. Information on the null alternative
4. Winners or losers?
5. Payment and delivery conditions
6. Willingness to pay or willingness to accept compensation?
7. Valuation function
8. Test for hypothetical bias
9. Specific quality factors for the contingent valuation method
10. Specific quality factors for choice experiments

The replacement cost method[a]

1. The performance of the man-made system as a substitute
2. The cost-effectiveness of the man-made system
3. Willingness to pay for replacement costs?

The human capital method[a]

1. Theoretical considerations
2. Technological development
3. To estimate the value of lost productivity

Valuation based on the costs of realising political decisions[a]

1. Cost-effectiveness
2. Willingness to pay the costs?

[a] These methods are less firmly rooted in welfare economics than the other methods, but are still included in the QAI because they are often used for environmental valuation.

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