ANALYSIS

Environmental value transfer: state of the art and future prospects

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Abstract

The main objectives of the paper are to (1) give an overview of the state of the art of environmental value transfer, (2) discuss its prospects and potential role in CBA as a decision-support tool, and (3) provide further guidelines for proper use and application. Environmental value or benefit transfer is a technique in which the results of studies on monetary environmental valuation are applied to new policy contexts. The technique is controversial, not least because of academic and political reservations over the usefulness and technical feasibility of economic valuation tools to demonstrate the importance of environmental values in project or programme appraisals. Testing of environmental value transfer so far has been unable to validate the practice. Taking into account the conditions set out in the literature for valid and reliable value transfer, most transfers appear to result in substantial transfer errors. This paper discusses why and addresses the question of which factors may have been overlooked. It is argued that the problem is much more fundamental than previously acknowledged. Strict guidelines in terms of quantitative adjustment mechanisms to valid value transfer are meaningless if the more fundamental issue of differences in the very nature of the values elicited is not addressed at the same time. © 2000 Elsevier Science B.V. All rights reserved.

Keywords: Environmental valuation; Value transfer; Validity; Reliability; Protocol

1. Introduction

Environmental value or benefits transfer remains a controversial issue in various policy contexts, not least because of academic and political reservations over the usefulness and technical feasibility of economic valuation tools to demonstrate the importance of environmental values in project or programme appraisal.1

1 The term environmental value transfer is used here instead of the popular term benefits transfer. Existing economic valuation techniques can also measure the benefits foregone, which makes the estimated values costs instead of benefits. The values discussed in this paper refer to the monetised non-market costs or benefits associated with environmental change.
Environmental value transfer is commonly defined as the transposition of monetary environmental values estimated at one site (study site) through market-based or non-market-based economic valuation techniques to another site (policy site). The most important reason for using previous research results in new policy contexts is cost-effectiveness. Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time consuming original research to quickly inform decision making.

Environmental value transfer has been applied extensively in various natural resource policy contexts, ranging from water quality management (Luken et al., 1992) and associated health risks (Kask and Shogren, 1994) to waste (Brisson and Pearce, 1995) and forest management (Bateman et al., 1995).

Especially the work recently by Costanza et al. (1997) in which an attempt was made to extrapolate the monetary values from existing valuation studies to the flow of global ecosystem services and natural capital has brought the validity and reliability of environmental value transfer back in the picture. This transfer exercise evoked a storm of indignation, some of which was published in a special issue of Ecological Economics in April 1998 (volume 25, number 1).

A lot of the controversy surrounding value transfer can be related back to the long-standing debate about the role of CBA in informing decision-making (Turner, 1978). For most critics of CBA, the economic value theory underlying CBA and economic valuation techniques are overly restrictive. Although there are some obvious links, the economic concept of value is only loosely linked to the value concepts found in the philosophy and social-psychology literature (e.g. Perry, 1950; Rescher, 1975). The assumptions underlying the theory are considered too narrow and simple to properly describe the environmental values people hold, the process of value construction, or the way individual values are aggregated into a social value. Other criticism seems to originate from fears that the economic efficiency (net benefit) criterion is promoted as the sole decision-making criterion. Some of the critics (e.g. Sagoff, 1988; Jacobs 1997) consider environmental valuation more as a social process relying upon social agreements and as such only loosely tied, if at all, to technical economic valuation methods and techniques.

In environmental economics, various models and techniques have been developed to measure the value people attach to natural resources and the services these resources provide. Environmental values are measured in money terms through the concept of individuals’ willingness to pay (WTP) or willingness to accept (WTA) compensation in order to make them commensurable with other market values. Of these two, the WTP approach has become the most frequently applied and has been given peer review endorsement through a variety of studies (e.g. Arrow et al., 1993).

WTP is measured directly by asking people to state a WTP amount in a social survey format called contingent valuation (CV), and indirectly by assuming that this value is reflected in the costs incurred to travel to specific sites (travel cost (TC) studies) or prices paid to live in specific neighbourhoods (hedonic pricing (HP) studies). The latter two approaches measure environmental use values through revealed preferences, while CV is believed to be able also to measure non-use or passive use values through stated preferences.

Although it is beyond the scope of this paper to go into much detail with respect to existing controversies, they are important when assessing the validity and reliability of environmental value transfer. The search for generally applicable models for valuing non-market environmental goods and services implies more than just looking for a technical solution to the transfer of values over sites, ecosystems and human populations, as often seems to be the overriding opinion in the benefits transfer literature. The question of what these values reflect and hence their potential use in different policy contexts is fundamental to the future social–political acceptability of such models (Vatn and Bromley, 1995).

This paper will address some of the issues relevant to the prospects for valid and reliable value transfers, in particular for contingent values. The main objectives are (1) to give an overview of the state of the art of environmental value transfer, (2) to discuss its prospects and potential role in CBA as a decision-support tool, and (3) to provide...
further guidelines for proper use and application. These objectives are successively addressed in the following sections.

2. State of the art: the search for ‘technical’ criteria for valid value transfer

2.1. Hypothesised criteria

In 1992, the American journal Water Resources Research (volume 28, number 3) dedicated a special issue to the concept and technique of environmental value transfer. In this issue, a number of authors outlined criteria for selecting among studies for value transfer (e.g. Boyle and Bergstrom, 1992; Desvousges et al., 1992). These criteria refer to the environmental goods involved, the sites in which the goods are found, the beneficiaries and study quality.

First of all, studies considered for inclusion must be based on adequate data, sound economic methods and correct empirical techniques. Moreover, studies should contain regression results which describe WTP as a function of relevant explanatory factors. Secondly, the sites must have similar populations. Thirdly, the environmental good and the change in provision levels at the different sites should be similar. Fourthly, the sites in which the goods are found should be more or less the same. Finally, the constructed markets, including the distribution of property rights, have to be the same at each site.

These criteria look fairly straightforward. However, they do not question the validity and reliability of the stated or revealed WTP amounts in valuation studies, nor were they tested empirically for their degree of relevance. In practice, most of the studies used for environmental value transfer are unable to comply with all these criteria and average point estimates are usually unconditionally applied in new policy contexts. Even if the environmental goods are more or less the same, as well as the sites in which they are found, their provision and quality levels may differ significantly across sites. In practice, it is more often than not a coincidence when both the reference and target levels of resource provision and quality are the same at different geographical locations.

Moreover, even if the goods, the sites where they are found and their user groups are similar, the benefits derived from these goods are not necessarily the same if the distribution of the population and their characteristics around the sites are not the same (Loomis, 1992). In view of the different spatial and temporal scales at which environmental problems may operate, and consequently the policy programmes aiming at solving these problems, the proper definition of the beneficiaries or stakeholders involved is of utmost importance since this will have far reaching consequences for the estimation of the aggregate economic values.

In those cases where studies did seem to meet these criteria, no valid value transfer could be established unequivocally. These studies are presented in the next section.

2.2. Looking for criteria using single studies

Very little published evidence exists of studies which test the validity of environmental value transfer across sites. Moreover, in the few studies that have been carried out, the transfer errors are substantial (Table 1).

Table 1 details seven studies which have tested the validity of value transfer empirically so far. The specific economic valuation technique used is shown in the second column of the table, the environmental goods valued in the third column and the transfer errors in the last column. The upper range in the last column refers to the unadjusted average point estimates, while the lower range refers to the average unit values adjusted for factors which significantly influenced value estimates, i.e. based on a value function.

A third value transfer can also be distinguished, namely the transfer of average unit values adjusted for factors which significantly influence value estimates and the extent to which these factors are expected to influence stated WTP amounts in the new policy context. That is, adjusting the parameter estimates in the value function according to new expectations. This third function-based approach is often neglected in the literature. Only a few studies exist which test the
stability of WTP functions through time (e.g. Loomis, 1989; Carson et al., 1997). The transfer of values based on value functions is more robust than the transfer of unadjusted average unit values since effectively more information can be transferred (Pearce et al., 1994).

Table 1 shows that the transfer errors can be as large as 56% in the case of unadjusted unit value transfer and 475% in the case of adjusted value transfer. A range of transfer errors is presented because the convergent validity of WTP values was tested for at least two sites: transferring the point estimate or function from site A to B and the other way around. In most studies more than two sites were included in the transfer exercise.

Little can be concluded especially from the large differences in upper limit transfer errors between unadjusted and adjusted value transfers. Most of the errors refer to statistically invalid transfers anyway. No study has yet been able to show under which conditions environmental value transfer is valid.

A number of reasons can be given why the studies in Table 1 are unable to produce a valid transfer of environmental values. First, the quantity and quality of control included in the studies are limited, for instance, in the way general population characteristics at different sites have been analysed. Traditional population characteristics are included in the WTP functions such as respondent sex, age, education level and household income. However, the statistical specification of these characteristics is in most cases very simple, relying upon dummy variables to indicate whether or not a respondent had a specific education level, earned a specific income level or was older or younger than 50 years. Also, for site characteristics dummy variables are often used, for example, to indicate whether or not a site is accessible. The specification of these explanatory variables contrasts sharply with the complex continuous response variable (WTP), which is expected to reflect the strength of people’s preferences for specified changes in the provision levels of environmental goods or services.

Secondly, the explanatory power of the models specifying differences in WTP is usually low. In the case of CV studies, statistical models account, on average, only for about 30–40% of the variability found in stated WTP amounts (e.g. Willis and Garrod, 1994). Most of the variability remains unexplained. Hence, it may not come as a surprise that a generally applicable model for valid value transfer has not yet been found.

### Table 1
Transfer errors found in studies testing the validity of environmental value transfer

<table>
<thead>
<tr>
<th>Study</th>
<th>Valuation technique</th>
<th>Environmental good</th>
<th>Transfer error (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Loomis (1992)</td>
<td>TC</td>
<td>Sport fishing</td>
<td>5–40</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>5–15</td>
</tr>
<tr>
<td>Parsons and Kealy (1994)</td>
<td>TC</td>
<td>Water quality improvements</td>
<td>4–34</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>1–75</td>
</tr>
<tr>
<td>Loomis et al. (1995)</td>
<td>TC</td>
<td>Water-based recreation</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>1–475</td>
</tr>
<tr>
<td>Bergland et al. (1995)</td>
<td>CV</td>
<td>Water quality improvements</td>
<td>25–45</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>18–41</td>
</tr>
<tr>
<td>Downing and Ozuna (1996)</td>
<td>CV</td>
<td>Saltwater fishing</td>
<td>1–34</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kirchhoff et al. (1997)</td>
<td>CV</td>
<td>White water rafting</td>
<td>24–56</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>6–228</td>
</tr>
<tr>
<td>Brouwer and Spaninks (1999)</td>
<td>CV</td>
<td>Biodiversity on agricultural land</td>
<td>27–36</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>22–40</td>
</tr>
</tbody>
</table>

* TC, travel cost study; CV, contingent valuation study.

* Minimum–maximum transfer errors found in the studies. The upper range refers to the absolute transfer errors based on unit value transfer and the lower range to the absolute transfer errors based on value function transfer.

* For different types of lake recreation.

* Primarily for use by local residents, such as recreation.
Whilst it is generally believed that equality of the impacts of different factors across sites is a sufficient condition for the construction of a valid value transfer model, such a model should also include more or less equally significant factors across sites in order for the explanatory power of the model at different sites to be the same (Brouwer and Spaninks, 1999). Under these conditions the transfer of environmental values may be valid even though the overall explanatory and hence predictive power of the model is low.

Thirdly, even if statistically adequately specified, the factors included so far 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 in a CV study. Human behaviour as measured in TC and HP 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).

Table 2
Value functions based on single and multiple studies

1. Value function based on single studies

\[ WTP_i = a + bX_j + cY_k + \epsilon_i \]

Where

- \( WTP_i \): Willingness to pay of respondent \( i \)
- \( a, b, c \): Parameters
- \( X_j \): Site and good characteristics (\( j \))
- \( Y_k \): Respondent characteristics (\( k \))
- \( \epsilon_i \): Random error

Number of observations is equal to the number of respondents \( \sum_{i=1}^{n} \)

2. Value function based on multiple studies

\[ WTP_s = a + bX_j + cY_k + dZ_l + u_s \]

Where

- \( WTP_s \): Mean willingness to pay from study \( s \)
- \( a, b, c, d \): Parameters
- \( X_j \): Site and good characteristics (\( j \))
- \( Y_k \): Respondent characteristics (\( k \))
- \( Z_l \): Study characteristics (\( l \))
- \( u_s \): Random error

Number of observations is equal to the number of data points (mean WTP) taken from each study \( \sum_{s}^{m} \)

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.

The issue of available and reliable data to supplement estimated value functions is probably one of the biggest barriers to the practical application of environmental value transfer in developing countries. Even if data are available, the quality of existing information systems such as population census or other official statistics is often poor, largely because of the existence of substantial ‘informal markets’ which are not registered in official statistics.

2.3. Looking for criteria comparing and synthesising multiple studies

The testing procedures used in the studies in Table 1 look at the convergent validity of the same environmental valuation study carried out at different geographical sites, in most cases at the same point in time. Site and sample population specific factors are accounted for which are expected to be responsible for differences in valuation outcomes at these different sites. Including an increasing number or different sets of explanatory variables in the testing procedures, one should be able, in principle, to test the amount of control needed to make value transfer a viable valuation technique. These explanatory variables can be put forward via theory, but also by means of previous empirical testing of, for instance, methodological issues regarding the specific valuation technique used.

Another way to test the significance of specific factors in explaining differences in valuation outcomes found in different studies is meta-analysis (MA). MA is commonly defined as a statistical analysis of the summary findings of empirical studies. It looks for study characteristics as potential explanations for consistent results across studies in order to arrive at useful generalizations.
Also MA generates a value function. However, this value function is usually based on global or average statistics instead of individual data (Table 2).

Since the beginning of the 1990s, MA has been playing an increasingly important role in environmental economics research (van den Bergh et al., 1997). In the field of environmental valuation, MA studies have focused on a range of environmental issues from outdoor recreation to urban air pollution and rare and endangered species. The increase in MA research seems to be triggered principally by (1) increases in the available number of environmental valuation studies, (2) the seemingly large differences in valuation outcomes as a result of the use of different research designs, and (3) the high costs of carrying out environmental valuation studies which tend to increase policymaker demand for transferable valuation results.

Table 3 gives an overview of MA research carried out in the field of monetary environmental valuation. This has tended to focus on similar public goods and services, the values of which have been estimated using the same economic valuation technique. HP models have been used to detect the influence of air pollution on house prices. TC-based valuation studies focus on outdoor recreation demand in general and recreational fishing demand in particular. The CV-based MA studies focus on improving the visibility at national parks in the US, woodland recreation in the UK, preservation of rare and endangered species in the USA and the provision of wetland functions across North America and Europe. Only two studies are based on a combination of valuation techniques, one of which furthermore considers three broad classes of quasi-public goods (Carson et al., 1996). The values included in this latter study cover a wide range of environmental values ranging from recreational fishing to occupational health risks expressed in terms of statistical values of life.

Results from these analyses suggest that differences in study design play an important role in explaining the variability in valuation outcomes. In the HP-based studies, besides local property market conditions, the type of data and model specification employed in each study helped to reveal significant negative relationships between house prices and air pollution measures. In the TC-based studies, significant differences were found regarding (1) the overall statistical specification of the TC model, (2) the specification of the opportunity costs of time, and (3) the inclusion of substitute prices.

Differences in CV study design have been investigated in much less detail, especially in the first two CV-based MA studies. Whether the WTP question was open-ended or respondents were asked to say yes or no to a given bid amount was the only significant variable included in the analysis. In the study by Bateman et al. (1995) a distinction was also made between current and future use values.

The last two CV-based MA studies in Table 3 include a wider range of explanatory factors. Besides methodological issues, two study quality in-

<table>
<thead>
<tr>
<th>Study</th>
<th>Topic</th>
<th>Valuation technique</th>
</tr>
</thead>
<tbody>
<tr>
<td>Smith and Kaoru (1990)</td>
<td>Outdoor recreation</td>
<td>TC</td>
</tr>
<tr>
<td>Walsh et al. (1992)</td>
<td>Outdoor recreation</td>
<td>CV/TC</td>
</tr>
<tr>
<td>Smith and Huang (1993, 1995)</td>
<td>Air pollution</td>
<td>HP</td>
</tr>
<tr>
<td>Sturtevant et al. (1995)</td>
<td>Fresh water fishing</td>
<td>TC</td>
</tr>
<tr>
<td>Bateman et al. (1995)</td>
<td>Woodland recreation</td>
<td>CV</td>
</tr>
<tr>
<td>Smith and Osborne (1996)</td>
<td>Visibility at national parks</td>
<td>CV</td>
</tr>
<tr>
<td>Carson et al. (1996)</td>
<td>Recreation, environmental amenities, health risks</td>
<td>HP/TC/CVDE/ market prices</td>
</tr>
<tr>
<td>Loomis and White (1996)</td>
<td>Rare and endangered species</td>
<td>CV</td>
</tr>
<tr>
<td>Brouwer et al. (1997)</td>
<td>Wetland ecosystem functioning</td>
<td>CV</td>
</tr>
</tbody>
</table>

* TC, travel costs; CV, contingent valuation; HP, hedonic pricing; DE, defensive expenditures.
dicators were included, one for the quality of the individual studies included in the MA and one for the MA itself. The quality of individual studies is indicated by the study response rates and the quality of the meta-analysis by the so-called scope test. Both indicators are found back in the NOAA’s 'burden of proof' requirements (Arrow et al., 1993).

The WTP functions estimated from single or multiple studies and the nature of most of the explanatory factors included in these functions suggest that full explanation and hence a valid model for environmental value transfer is more or less a 'technical' problem depending on 'correct' research design and model specification. The reason for this is the quantitative nature of the values and the survey format in which they are elicited. However, evaluating the estimated models in terms of their low explanatory and hence predictive power, they have been unable to define the set of requirements for value transfer to be a viable valuation alternative, or maybe even conventional economic valuation itself for that matter.

The next section will discuss which factors may have been overlooked so far in the search for generally applicable models for environmental value transfer. This will be a more qualitative investigation into the underlying nature of the values elicited which is expected to provide more insight into the question why values still vary even though the recommended controls are implemented.

3. What is transferred? Meaning, interpretability and stability of environmental values

3.1. End-state versus process-oriented approaches to environmental value elicitation

The issue of valid and reliable environmental value 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, for instance, with the value elicitation mechanisms used, but the values themselves remain undisputed.

A second perspective, advocated in this paper as complementary to the first one, is more critical about the estimated values. Even though a statistically valid transfer can be established when the explanatory power of the transfer model is low, 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?

Investigating the process of value formation, articulation and elicitation in order to better understand the values themselves means that one of the core assumptions underlying economics has to be revisited, namely, that preferences are given and that it does not matter why people value things. This core assumption deserves much more attention from environmental economists. Differences in underlying reasons and motives may enable us to better explain differences in valuation outcomes and hence come up with a model which has a sufficiently high explanatory power to validly and reliably predict values across sites and groups of people.

3.2. Economic starting points

Neo-classical economic theory starts from the presumption that it does not matter why people value goods. Preferences are assumed given and the analysis of why preferences vary between (groups of) individuals, so it is argued, falls outside the domain of economics. Contrary to the claim that preferences are value neutral data, economic (value) theory is not value free for a number of reasons and assumptions. First, it implicitly values preferences by assuming (the possibility of) ordinal comparison between them (i.e. commensurability) ex ante and their cardinal representation in utility ex post (Aldred, 1997). Secondly,
the intensity of a preference counts, not the reason(s) for a preference (O’Neill, 1997). Thirdly, utility (welfare) maximisation is chosen as the ultimate good and preference satisfaction as the means to achieve it (Sagoff, 1988). Fourthly, since Edgeworth (1881) all utilitarian thinking is based on pursuing self-interest, also altruism. Finally, the private goods market is also considered an appropriate and procedural just (Rawls, 1971) institution for revealing environmental values and Pareto optimality in their allocation the highest good, where market behaviour and private WTP are used as indicators of people’s preferences and values, respectively.

The moment a common measurement unit is used (the money people are willing to pay), it does not matter anymore why people value things or have certain preferences since underlying reasons and motives are subsumed within this single common measure. Differences in WTP amounts reflect the different values people hold, irrespective of the question why these values vary across individuals. They simply do and the question why becomes more or less irrelevant in view of the established indicator’s presumed ability to make the reasons or motives underlying values commensurable and hence comparable. This implies that the various motives people have to value things (preferences) can be weighted and exchanged (traded off). The indicator focuses on the end-result of the valuation process in terms of individual WTP in the real or hypothesised (as in CV) market place: the higher someone’s WTP, the more he/she values something. From this point of view, the single measure facilitates a straightforward transfer of environmental values across individuals without any need to look at reasons or motives why people value things differently.

The assumption of commensurability remains an important unresolved issue in the debate surrounding monetary environmental valuation. It underlies some of the strongest criticism. Many critics are appalled by the idea that, for instance, aesthetic judgements about a site or painting are treated on a par with preferences for a specific ice-cream flavour. Environmental economics has tried to somewhat overcome this critique by breaking down the concept of total economic value, not to be confused with total ecosystem value, measured by WTP in various use and non-use related reasons and motives for people to value environmental change. However, these different reasons and motives are still assumed commensurable and hence comparable a priori because they are expressed in the same unit of WTP.

How legitimate is it to assume that use and non-use values are commensurable? Do we not make what Levi (1986) calls ‘hard choices’ every day, deciding whether or not to develop or grow economically further, to leave some natural inheritance to our children, to save lives by increasing private and/or public spending or to allow other species a right to co-exist with us on this planet? Or do choices like these activate and reflect fundamentally different underlying levels of considerations, values and beliefs in choice processes, undermining a straightforward comparison? How are basically conflicting motives, for example, in those cases where use will result in exhaustion, depletion or extinction, weighted and traded off? Or, how can someone be compensated, even hypothetically as economic theory suggests, for something which is considered basically priceless?

3.3. Practical problems in environmental value transfer

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 constant across social groups and environmental domains. If more or less constant, these values would be easily transferable without the need to look at motivations underlying such WTP values. Besides the fact that values often differ substantially in practice from case to case, there may be other arguments against this proposition as well.

First, contrary to many of the market-based costs and benefits included in CBA, these environmental values are often not well defined. This undermines their legal-political acceptability in CBA, especially in those cases where they seri-
ously inflate total benefits (costs) for green (economic development) programmes. In the case of TC and HP studies, it is usually fairly clear what is measured: a use value revealed through the amount of money people actually paid to enjoy an environmental good or service. On the other hand, in the case of CV stated 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.

CV assumes that people are willing and capable to think like consumers about public environmental issues, caring about their private interests mainly by considering their disposable income. If individuals take a broader social perspective in public choice contexts as has been claimed by a number of critics (e.g. Blamey, 1995; Burgess et al., 1998) and behave more as citizens, setting private interests aside, stated money amounts may become fictive and merely symbolic because of feelings of social responsibility and/or moral commitment. In those cases these values may become meaningless and seriously misinform policy or decision-making since they do not represent what they are intended to from an economic point of view, i.e. the strength of private preferences and commitment to satisfy these preferences through WTP.

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).

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 attach a value to the environmental goods and 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 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. Although 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), the question is how stable constructed preferences and subsequently people’s stated WTP in a, say, 15- to 30-minute interview remain through time. How legitimate is it to put them together and make them comparable with other value statements at different points in time, as in discounted CBA or MA to generate a constant assumed value function? 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.

4. Prospects for environmental value transfer

4.1. Using values from travel cost and hedonic pricing models

TC and HP models seem to have circumvented much of the controversy that CV has evoked.
They provide a straightforward technical approach to the monetary valuation of natural or environmental characteristics based on people’s actual behaviour, avoiding many of the biases found in expressed or stated preference approaches such as CV.

However, these models rely upon a number of assumptions the validity of which can be questioned. People’s preferences for the natural or environmental characteristics of recreation sites or specific neighbourhoods are measured indirectly by assuming that their value is reflected in the costs incurred to travel to these sites or the price paid to live in these neighbourhoods. These monetary prices (assumed values) are broken down according to the characteristics identified as relevant by the researcher. Hence, these characteristics are taken as the reason why people visit a site or live in a specific urban area, avoiding the use of social survey formats such as CV to investigate people’s own stated reasons.

Recently, more sophisticated statistical models have been developed, such as random utility models (RUMs), which model, for example, the probability that someone decides to visit a specific site as a function of site characteristics. These models are based on the same theoretical premise as choice experiments where respondents are asked to rank bundles of characteristics, including price, from which the researcher then infers their monetary value (e.g. Hanley et al., 1998). However, people’s own perception and valuation of these characteristic features of sites (or houses in specific neighbourhoods in HP models) may differ from the characteristics as identified by the researcher and hence remain outside the scope of the model.

Furthermore, the extent to which these characteristics can be valued individually and out of context, for example, by transferring them across sites in models based on geographical information systems (e.g. Lovett et al., 1997), is open to dispute, especially for more complex environmental goods and services where these individual characteristics are highly interdependent, either as perceived from a social or natural science point of view.

The availability of these more advanced statistical estimation techniques has the advantage that estimates become more accurate. On the other hand, there is a risk that more fundamental issues or problems in the elicitation of WTP values are pushed to the background or ignored. Advanced statistical techniques may better fit the data, resulting in higher explanatory powers than before, but they will be unable to repair flaws incurred in earlier stages of the research.

Finally, the potential use of travel cost and hedonic pricing models seems to be limited to specific environmental concerns such as local or regional natural resource management involving substantial recreational use, or the assessment of compensation payments in liability regulation and legislation, which is expected to increase in urban pollution contexts.

4.2. Using contingent values

CV findings point out the complexity of eliciting values people hold for environmental goods and services. Even well-defined user groups may appreciate these goods and services for other reasons than (potential) use only. The extent to which this diversity of subjective values can be legitimately aggregated into a single measure over the various interdependent good characteristics as is done in CV and associated research such as contingent ranking or choice experiments is questionable.

Strict guidelines for valid and reliable value transfer based on contingent values will be hard to produce. There are still too many variables for which we are unable to control. For the purpose of environmental value transfer, it is important to recognise the diversity of motivations underpinning valuations and the context in which they are elicited. However, attempts to extend current technical restrictions to include numerical indicators for the diversity of underlying motivations or beliefs do not seem to provide much promise for the practical application of environmental value transfer. They are part of the problem, not the solution. Here the operational boundaries of quantitative tools seem to come to a stop. A quantitative approach to what seems to
be essentially a qualitative issue will be unable to adequately address the wider interpretability and hence applicability of the values elicited.

What potential role is there then for contingent values in transfer exercises if adjusting these values numerically on the basis of technical criteria such as the study’s elicitation format or differences in stakeholder income levels does not address the more fundamental issue of possible differences in the very nature of the values elicited through CV? How does one account for differences between policy contexts in terms of the complexity of people’s engagement with their natural environment and consequently the existing plurality of values?

First of all, it is important to point out that certain values will be more easily computable, more meaningful and transferable than others. In contexts where people are already paying for environmental goods and services and these payments are embedded in existing institutional systems, the use of private WTP values in CBA may be more suitable to evaluate environmental improvement programmes, a recommendation already made in one of the first state-of-the-art reports on CV (Cummings et al., 1986). However, in those contexts where (1) these constructs are absent, (2) the complexity of the environmental issue extends the complexity of the valuation process beyond reasonable public comprehension, or (3) the decision is part of a set of culturally evolved and politically negotiated fundamental rights or obligations which are not ‘for sale’, the appropriateness of private WTP values may be significantly less.

It is possible to learn from previous CV studies to enhance the transferability of certain values across policy contexts and hence reduce the need for original studies. However, this involves more than just the reporting of a technical value function in CV studies as recommended in the literature. Much more attention will have to be paid to (1) the relevant spatial and temporal scales at which the specific environmental problem plays, (2) the extent to which human values can be scaled up in a natural scientific sound way from single environmental functions to multifunctional ecosystems and from local to global ecosystems, (3) the break-down and distribution of the associated economic costs and benefits over multiple stakeholders, (4) the diversity of motivations underpinning valuations and the context in which they are elicited, and (5) an assessment of the extent to which (1) to (4) are expected to correspond in different policy contexts. This will be further detailed in the final section.

## 5. Towards a protocol for good practice

A number of steps will be highlighted which are considered important to the practice of environmental value 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 Table 4.

### 5.1. Step 1: defining the environmental goods and services

Environmental goods and 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. One way of doing this is to break them down into 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-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 valid and reliable environmental value transfer, the identification of the various economic benefits is not enough. The provision and quality levels of these benefits in the reference and target situation is 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. In these cases, no adjustment mechanism is available to account for possible differences. Currently, the RUMs and choice experiments mentioned in the previous section are the only tools available which seem to be able to meet this problem.

An essential part of the first step is the identification of the relevant ecological functions which underpin the supplied goods and 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 ecological and social scales involved.

5.2. 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 stakeholders or beneficiaries, have to be identified. Although this step identifies beneficiaries, not the reasons why these beneficiaries value environmental goods and services (see the next step), they are interdependent. To clarify this, an analogy with market goods and services can perhaps 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.

5.3. Step 3: identifying values held by different stakeholder groups

The same good or service may hold different values to different people. As above, 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 environmental goods and services, the reasons why these benefits are considered benefits by various stakeholders has 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).

5.4. Step 4: stakeholder involvement in determining the validity of monetary environmental valuation

One of the underexposed areas in monetary and non-monetary environmental valuation so far 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 judgements regarding the legitimacy of the social–political organisation of the process of value elicitation. Instead of making assumptions a priori, more research efforts should be focused 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) 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 his/her monetary environmental valuation exercise and helps define 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 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 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 often can be readily computed from available market data. In some cases, market data will also be available for indirect extractive benefits. In other cases, one can rely upon non-market valuation techniques. Direct non-extractive values are also 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. 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.

5.5. Step 5: study selection

After going through steps 1 to 4, appropriate studies are selected. If available, a MA 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.

In Section 2.1, a number of criteria were given to select among studies. These criteria are generally applicable. However, it has to be seen to what extent these criteria can be or have been formalised in a WTP value function. 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, that is 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 other things, 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 or their external validity can be assessed by looking at the actual meaning and interpretability of the values found. Contrary to TC and 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 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 and how 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 defining 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 (Brouwer et al., 1999).

5.6. 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. Section 2.3 reported some of the main results found in previous MA research looking at the effects different research designs have on study outcomes.

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? RUMs 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 dichotomous choice or 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 (Arrow et al., 1993), generally a conservative approach seems to be preferable.

5.7. 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. Finally, the economic aggregate is included in a CBA together with other economic costs and benefits, which can then play its part in the facilitation of the overall, real world, multi-criteria decision-making process.

References


