5. Value transfer and environmental policy

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WHY VALUE TRANSFER?

Increased use of economic analyses in the environment, transport, energy, health and cultural sectors has increased the demand for information on the economic value of environmental and other non-market goods by decision-makers. Due to limited time and resources when decisions have to be made, new environmental valuation studies often cannot be performed, and decision makers must rely on transfer of economic estimates from previous studies (often termed 'study sites') of similar changes in environmental quality to value the environmental change at the 'policy site'. This procedure is most often termed 'benefit transfer', but since damage estimates can also be transferred, I will use the more general term 'value transfer'.

Value transfer increases the uncertainty in the estimated environmental value, and a crucial question becomes: what level of accuracy is acceptable, and how does the need for accuracy vary with the policy use of the value? Results from validity tests of value transfer procedures have shown that the uncertainty in spatial and temporal benefit transfer could be quite large. Thus one should be careful in using value transfer in policy uses where the demand for accuracy is high.

The practice of value transfer can be traced back to be the calculation of lost recreational value from the Hell's Canyon hydroelectric project 30 years ago (see Krutilla and Fisher, 1975, chs 5 and 6). The first large-scale user of value transfer, however, was the USDA Forest Service. In preparation for the 1980 Resource Planning Assessment (RPA) the Forest Service launched a large-scale effort to collect data on the economic values associated with recreational use of forestlands, in order to balance these against timber production and other uses. These early examples of value transfer were, however, conducted in an uncritical manner, often lacking sound theoretical, statistical and empirical basis, and did not question the validity and reliability of the transferred values. The validity of value transfer was placed firmly on the agenda about ten years ago in a set of papers in a special section of a 1992 issue of *Water Resources Research* (vol. 28, no. 3). Since then there has been a steady growth in the literature on testing validity of benefit transfer, the development of transfer methods and statistical techniques, and applications of these to environment and health. Reviews of these development can be found in Desvousges et al (1998) and Navrud and Ready (forthcoming).

This chapter is organized as follows. The next section reviews the different types of policy use where value transfer has been used. Then I review environmental valuation techniques, and describe the methods used to transfer value estimates from these techniques. Finally, I discuss tests of validity of value transfers, the main challenges to value transfer and offer my suggestions for future research to address these challenges.

POLICY USE OF ENVIRONMENTAL VALUATION ESTIMATES

Environmental valuation studies have four main types of policy use, where value transfers have also been performed:

1. Cost–benefit analysis (CBA) of investment projects and policies (both *ex ante* and *ex post* analyses).
2. Environmental costing in order to map the marginal environmental and health damages of, for example, air, water and soil pollution from energy production, waste treatment and other production and consumption activities. These marginal external cost can be used in investment decisions and operation (for example as the basis for 'green taxes').
3. Environmental accounting at the national level (green national accounts) and at the firm level (environmental reporting and accounting).
4. Natural Resource Damage Assessment (NRDA)/Liability for environmental damages, that is, compensation payments for natural resource injuries from e.g. oil spills and other pollution incidents.

Environmental valuation techniques are mostly used in CBAs, but have also been used in NDRAs in the USA, for example Carson et al. (2003); and environmental costing of electricity production from different energy sources in both the USA and Europe, for example Rowe et al. (1995), Desvousges et al. (1998) and European Commission – DG XII (1995, 1999). New York State and a few other US states have used valuation for environmental costing to construct 'adders' to their electricity prices. Adders are increments added to the private marginal costs that allow you to get closer to full marginal social cost of electricity production. In this case they were used to make more rational dispatch decisions for electricity generation by using marginal social rather than marginal private costs

(Brennan et al. 1996; Tietenberg, 2003, p. 175). Finally, environmental valuation has been used in green national accounting exercises, for example the Green Accounting Research Project (GARP) of the European Commission (Tamborra, 1999, GARP II, 1999). The UN's statistical division UNSTAT has actively supported the development of resource accounting systems (for example the *Handbook on Integrated Environmental Economic Accounts*).

The need for accuracy in the economic estimates increases, and thus the applicability of value transfer techniques decreases, as we move down the above list of potential policy uses. For NRDAs, and partly also for environmental accounting and costing, there seems to be a more direct link between the outcome of the analysis and policy impact, and the group affected is better defined, than in a CBA (Navrud and Pruckner, 1997). However, even in CBAs the need for accuracy can be large, for example if the estimated costs and transferred benefit estimates of a new environmental policy are very close.

CBA has a long tradition in the USA as a project evaluation tool, and has also been used extensively as an input in decision-making ever since President Reagan issued Executive Order (EO) 12292 in 1981, necessitating a formal analysis of costs and benefits for federal environmental regulations that impose significant costs or economic impacts (that is, Regulatory Impact Analysis). However, some statutes, for example the Clean Air Act, specifically rule out CBA as a basis for setting ambient standards, and resorts to cost-effectiveness analysis (CEA).

In Europe, CBA has a long tradition in evaluation of transportation investment projects in many countries, but environmental valuation techniques were in most cases not applied. There seems to be no legal basis for CBA in any European country, with the exception of the law of interior transport (1982) in France requiring the use of CBA to evaluate public transportation investments, and the UK Environment Act (1995; section 39) requiring a comparison of costs and benefits. Some countries have administrative CBA guidelines for project and policy evaluation, and in a few cases these include a section on environmental valuation techniques. Paragraph 130r of the Maastricht Treaty, which focuses on the EU's environmental goals, environmental protection measures and international cooperation in general, says that the EU will consider the burden and advantage of environmental action or non-action. Furthermore, the 'Fifth Activity Programme for Environmental Protection Towards Sustainability' (1993–2000) says:

> In accordance with the Treaty, an analysis of the potential costs and benefits of action and non-action will be undertaken in developing specific formal proposals within the Commission. In developing such proposals every care will be taken as far as possible to avoid the imposition of disproportionate costs and to ensure that the benefits will outweigh the costs over time. (European Community 1993, p. 142)

The 1994 Communication from the Commission to the Council of the European Parliament, entitled 'Directions for the EU on Environmental Indicators and Green National Accounting – The Integration of Environmental and Economic Information Systems' (COM (94)670, final 21.12.94), states a specific action for 'improving the methodology and enlarging the scope for monetary valuation of environmental damage'. More recently, the European Commission (EC)'s Green Paper, entitled 'For a European Union Energy Policy', states that 'internalisation of external costs is central to energy and environmental policy.' During the last few years the Directorate General (DG) Environment of the European Commission has performed several CBAs of new air quality targets; see, for example, European Commission–DG XI Environment (1997, 1998). These analyses rely heavily on the work done within the ExternE research project (European Commission – DG XII Research, 1995, 1999). EC DG Environment is now in the process of preparing guidelines on benefit assessments for all DG Environment policy and project assessments. The new EC Water Directive explicitly mentions CBA. The EC adopted the White Paper on Environmental Liability on 9 February 2000 (COM(2000) 66 final), and on 30 March 2000 the Environmental Council meeting supported the construction of a community framework directive on environmental liability that covers environmental damage – both contamination of sites (where liability exists in all member states) and damages to biodiversity as well as traditional damage (health and property). EC DG Environment has now started work to assess the applicability and adequacy of environmental valuation and value transfer methods to value biodiversity damages for the purpose of environmental liability.

International organizations such as the OECD, the World Bank and regional development banks and UNEP (United Nations Environment Program) have produced guidelines on environmental and health valuation and value transfer techniques; for example OECD (1989, 1994, 1995); Asian Development Bank (1996), and UNEP (1995, ch. 12). In many cases they have used valuation techniques as an integral part of CBA of investment projects, for example the World Bank's evaluation of water and sanitation projects (Whittington, 1998).

TRANSFER OF INFORMATION

CBAs of new environmental policies or projects with environmental impacts are often based on some kind of damage function approach (DFA) (see Box 5.1). In a DFA, there is not only uncertainty in the final step of economic valuation of environmental and health impacts. Uncertainty in transfer of information is aggregated over all four steps. Thus there is uncertainty in atmospheric and marine dispersion models, dose-response and exposure-response functions, or expert assessments of environmental and health impact where we lack these models and functions. Therefore, several disciplines can provide insights into environmental value transfer, for example environmental economics, statistics/econometrics, decision theory, ecology, geography, sociology, cognitive psychology and philosophy.

BOX 5.1 DAMAGE FUNCTION APPROACH (DFA) APPLIED TO EMISSIONS TO AIR AND WATER

Step 1: *Emissions and other residuals* →
Step 2: \Transport model\ → *Changed concentrations and other conditions* →\Dose-response functions (environment) and exposure-response functions (health)\ →
Step 3: *Physical impacts* →
Step 4: \New environmental valuation study, or existing studies combined with Benefit transfer techniques\ →
Damages or benefits

\ . . . \ = models
* . . . * = output (or input)

Cropper et al. (1997) clearly illustrates the uncertainty in the transfer of information in other parts of the DFA than valuation. They showed that the transfers of exposure-response functions from Philadelphia in the USA to Delhi in India are likely to be misleading. Alberini and Krupnick (1997) reach the same type of conclusion in their study of transfer of exposure-response functions from the USA to Taiwan.

Uncertainty also originates of course from the environmental valuation techniques used in the original valuation studies, and techniques to transfer these monetary values spatially and temporally. To conduct step 4 in the DFA by benefit transfer, detailed reviews and/or databases of valuation studies would ease the job; see Navrud (1992, 1999) for a review and database of all types of valuation studies, respectively, and Carson (forthcoming) for a review of contingent valuation studies worldwide.

REVIEW OF ENVIRONMENTAL VALUATION TECHNIQUES

The economic estimates of environmental, cultural heritage and health impacts used in a benefit transfer exercise would typically be based upon individual preferences, either observed behaviour (revealed preferences; RP) towards some marketed good with a connection to non-market good of interest; or stated preferences (SP) in surveys with respect to the non-marketed good. Table 5.1 provides an overview of the different types of RP and SP valuation techniques.

Revealed preference techniques can be divided into direct and indirect methods. Direct methods include simulated market exercises, that is, constructing a real market for a public good. This is most often impossible, and, when possible, it is usually very time-consuming and costly. In, for example, Switzerland and California, referenda are held regularly over public goods programmes; for example a proposed increase in taxes to pay for a programme to reduce water pollution. Presuming an informed voter, the decision on how to vote is based on the voter's assessment of whether the marginal utility of the programme is greater than the marginal utility of the amount he/she would have to pay. In order to use the results of actual referenda to value a public good, we need data on voting behaviour for different levels of the good at a fixed tax price or for a fixed level of the good at different tax prices. However, in most actual referenda the voters only vote for or against one specified tax rate to reach one specified water quality level. Contingent valuation (CV)

Table 5.1 Classification of environmental valuation techniques

	Indirect	Direct
Revealed preferences (RP)	Household production function (HPF) approach	Simulated markets
	• Travel cost (TC) method • Averting costs (AC)	Actual referenda
		Market prices
	Hedonic price (HP) analysis	Replacement costs (RC)
Stated preferences (SP)	Choice experiments (CE)	Contingent valuation (CV)
	• Conjoint analysis • Contingent ranking • Contingent rating • Pairwise comparisons	

surveys using discrete choice elicitation techniques seek to overcome this by conducting hypothetical referenda. Another advantage of CV surveys over actual referenda is that they secure a more representative sample of the population than actual referenda, which often have low participation rates and are dominated by better-educated and better-off citizens. In CV surveys the respondents are better informed, since information provided to the voters in the actual referenda is often complicated, difficult to understand, and incomplete as it does not address aspects of the programme of concern to the public.

Some environmental impacts can be valued using dose-response functions and market prices, for example impacts on crops, forests and building materials (corrosion and soiling) from air pollution. This approach uses only the physical or biological dose-response relationship to estimate the response to a change in some environmental parameter. The observed market price of the activity or entity is then multiplied by the magnitude of the physical or biological response to obtain a monetary measure of damage. Thus neither behavioural adaptations nor price responses are taken into account. Simple multiplication provides an accurate estimate of economic behaviour and value – in this case changes in gross revenue – only if economic agents are limited in the ways in which they can adapt to the environmental effect, and if the effect is small enough to have little or no impact on relative prices. This combination of circumstances is very unlikely. If, for example, crop damages from air pollution are large enough to change prices, changes in consumer and producer surpluses have to be calculated. If farmers undertake preventive measures, like switching to crops that are less sensitive to air pollution, the simple multiplication approach will overestimate damage costs. Thus other approaches should be used; see Adams and Crocker (1991).

The replacement cost method (also termed restoration cost method) has been used to estimate economic damages from soil erosion by using market prices for soil and fertilizers to calculate what it would cost to replace the lost soils. This approach has also been used to calculate loss of ecosystem functions. Restoration costs are, however, just arbitrary values that might bear little relationship to true social values. Individuals' willingness to pay for the restoration of environmental and cultural amenities may be more or less than the cost of replacement.

The greatest advantage of these direct RP methods is that they are relatively simple to use. But as noted earlier, the methods ignore the behavioural responses of individuals to changes in the environmental amenities. They also obscure the distinction between benefits and costs – there is no guarantee that people are actually willing to pay the estimated cost.

The indirect RP methods entail two main groups of methods: the household production function approach (including the popular travel cost method and averting cost method) and hedonic price analysis.

The household production function (HPF) approach involves investigating changes in consumption of commodities that are substitutes or complements for the environmental attribute. The travel cost (TC) method, used widely to measure the demand for recreation, is a prominent example. The costs of travelling to a recreational site together with participation rates, visitor attributes and information about substitute sites are used to derive a measure for the use value of the recreational activity at the site. Travel can be used to infer the demand for recreation only if it is a necessary part of the visit, or in economic terms is a weak complement. TC models build on a set of strict assumptions, which are seldom fulfilled, and the results are sensitive to the specification of the TC model, the choice of functional forms, treatment of travel time and substitute sites and so on. However, they can be relatively cheap to perform (compared to SP methods), and give reasonably reliable estimates for use values of natural resources (for example recreational fishing, hunting and hiking) for the current quality of a site.

Another example of the use of the household production function approach is averting costs (AC) (also known as defensive or preventive expenditures) to infer value. Averting inputs include air filters, water purifiers, noise insulation, and other means of mitigating personal impacts of pollution. Such inputs substitute for changes in environmental attributes; in effect the quality of a consumer's personal environment is a function of the quality of the collective environment and the use of averting inputs. We measure the value of changes in the collective environment by examining costs incurred in using averting inputs to make the personal environment different from the collective environment. A rational consumer will buy averting inputs to the point where the marginal rate of substitution between purchased inputs and the collective environment equals the price ratio. By characterizing the rate of substitution and knowing the price paid for the substitute, we can infer the price that consumers would be willing to pay for a change in the environment. The common element in household production methods is the use of changes in the quantities of complements to estimate the value of a change in quality.

HPF uses actual behaviour as the basis for valuation, but is limited to use value. Non-use values, which not entail direct consumption, cannot be estimated by looking at complements or substitutes. HPF approaches have mostly been used to value recreational activities, health and material damages.

Hedonic price (HP) analysis refers to the estimation of implicit prices for individual attributes of a market commodity. Some environmental goods and services can be viewed as attributes of a market commodity, such as real property. For example, proximity to noisy streets, noisy airports and polluted waterways; smell from hog operations, factories, sewage treatment plants and waste disposal sites; exposure to polluted air, and access to parks or scenic vistas are purchased along with residential property. Part of the variation in property prices is due to differences in these amenities. Other applications have been to wages for jobs that entail different levels of mortality risks (termed hedonic wage models) to estimate the 'value of a statistical life' (VOSL). HP data can be quite costly to obtain, as there is often no database of residential properties that have data on environmental amenities and other attributes which determining the property price. In addition the second stage of the HP analysis is often impossible to do since we lack socioeconomic data about the buyers of residential properties. The HP function is very sensitive to the specification and functional form, and it is often difficult to find a measure for the environmental amenity where data exist, and in which the bidders for residential properties can recognize marginal changes and has complete information at the time they bid for the property. Two examples: (i) there are often no data on traffic noise levels, and using the annual average number of vehicles on the nearest road or distance to this road as a proxy variable for noise levels could easily value all road-traffic-related externalities (including accident risks, health impacts from air pollution, barrier effects and soiling). (ii) Properties are shown to potential buyers on Sundays when there is little traffic on the nearby road, and thus they place their bid on the property with incomplete information about the road traffic noise level.

While (indirect) RP methods are based on actual behaviour in a market for goods related to the environmental good in question (and hence the value for the environmental goods is elicited based on sets of strict assumptions about this relationship), SP methods measure the value of the environmental good in question by constructing a hypothetical market for the good. This hypothetical nature is the main argument against SP methods. However, no strict assumptions about the relationship between marketed complements or substitutes, or attributes of a marketed good, and the environmental good, have to be made. SP methods also have the advantages of being able to measure the total economic value (TEV), including both use and non-use value (also termed passive use value), of deriving the 'correct' Hicksian welfare measure, and can measure future changes in environmental quality.

The SP methods can be divided into direct and indirect approaches. The direct contingent valuation (CV) method is by far the most used method, but over the past few years the indirect approaches of choice experiments (CE) have gained popularity. The main difference between these two approaches is that while the CV method is typically a two-alternative (referendum) approach, CE employs a series of questions with more than two alternatives that are designed to elicit responses allowing for estimation of preferences over attributes of an environmental state.

A contingent valuation (CV) survey constructs scenarios that suggest different possible future government actions. Under the simplest and most commonly used CV question format, the respondent is offered a binary choice between two alternatives, one being the *status quo* policy, and the other having a cost greater than maintaining the *status quo*. The respondent is told that the government will impose the stated cost (for example increased taxes, higher prices associated with regulation, or user fees) if the non-*status quo* alternative is provided. The key elements here are that the respondent provides a 'favour/not favour answer' with respect to the alternative policy (versus the *status quo*), and that what the alternative policy will provide how it will be provided, and how much it will cost, and how it will be charged for (that is, payment vehicle), have been clearly specified. This way of eliciting willingness to pay (WTP) is termed binary discrete choice (DC). In such a closed-ended version of CV, respondents can also be asked to value multiple discrete choices in double- and multiple-DC WTP questions. An alternative elicitation method is open-ended questions, where respondents are asked directly about the most they would be willing to pay to get the alternative policy. A payment card with amounts ranging from zero to some expected upper amount could be used as a visual aid. Then the data could be treated statistically as interval data; that is, if you say yes to pay 50 euro as the highest amount, but say no to 100 euro, we know that the respondent has a WTP within this interval. One of the main challenges in a CV study is to describe the change in the environmental amenity that the alternative policy will provide, in a way that is understandable to the respondent and is at the same time scientifically correct.

Concerns raised by CV critics over the reliability of the CV approach led the US National Oceanic and Atmospheric Administration (NOAA) in to convene a panel of eminent experts co-chaired by Nobel Prize winners Kenneth Arrow and Robert Solow to examine the issue. In January 1993, the Panel, after lengthy public hearing and reviewing many written submissions issued a report which conclude that 'CV studies can produce estimates reliable enough to be the starting point for a judicial or administrative determination of natural resources damages – including lost passive use value' (Arrow et al. 1993). The Panel suggested guidelines for use in Natural Resource Damage Assessment (NRDA) legal cases to help ensure the reliability of CV surveys on passive use values including the use of in-person interviews, a binary discrete choice question, a careful description of the good and its substitutes, and several different tests should be included in the report on survey results. Since the Panel has issued the report, many empirical tests have been conducted and several key theoretical issues have been clarified. The simplest test corresponds to a well-known economic maxim, the higher the cost the lower the demand. This price sensitivity test can easily be tested in the binary discrete choice format, by observing whether the percentage favouring the project falls as the randomly assigned cost of the project increases, which rarely fails in empirical applications. The test that has attracted the most attention in recent years is whether WTP estimates from CV studies increase in a plausible manner with the quantity or scope of the good being provided. CV critics often argue that insensitivity to scope results from what they term 'warm-glow', by which they mean getting moral satisfaction from the act of paying for the good independent of the characteristics of the actual environmental good. There have now been a considerable number of tests of the scope insensitivity hypothesis (also termed 'embedding'), and a review of the empirical evidence suggests that the hypothesis is rejected in a large majority of the tests performed (Carson 1997).

Producing a good CV survey instrument requires substantial development work, typically including focus groups, in-depth interviews, and pre-test and pilot studies to help determine whether people find the good and scenario presented plausible and understandable. The task of translating technical material into a form understood by the general public is often a difficult one. Adding to the high costs of CV surveys is the recommended mode of survey administration – in-person interviews (Arrow et al., 1993). Mail and telephone surveys are dramatically cheaper, but mail surveys suffer from sample selection bias (that is, those returning the survey are typically more interested in the issue than those who do not) and phone surveys have severe drawbacks if the good is complicated or visual aids are needed. CV results can be quite sensitive to the treatment of potential outliers. Open-ended survey questions typically elicit a large number of so-called protest zeros and a small number of extremely high responses. In discrete choice CV questions, econometric modelling assumptions can often have a substantial influence on the estimated mean and median WTP. Any careful analysis will involve a series of judgemental decisions about how to handle specific issues involving the data, and these decisions should be clearly noted.

According to Carson (2000), the recent debate surrounding the use of CV is, to some degree, simply a reflection of the large sums at stake in major environmental decisions involving passive use and the general distrust that some economists have for information collected from surveys. The spotlight placed upon CV has matured it; its theoretical foundations and limits to its users are now better understood. The CV method has still not reached the routine application stage, and all CV surveys should include new research/tests. Carson (2000) concludes that perhaps the most pressing need is to establish how to reduce the costs of CV surveys while still maintaining a high degree of reliability, and suggests combined telephone–mail–telephone surveys to reduce survey administration costs, and implementation of research programmes designed to solve some of the more generic representation issues such as low-level risk and large-scale ecosystems.

Choice experiments (CEs) have been employed in the marketing, transportation and psychology literature for some time, and arose from conjoint analysis, which is commonly used in marketing and transportation research.

CEs differ from typical conjoint methods in that individuals are asked to choose from alternative bundles of attributes instead of ranking or rating them. Under the CE approach respondents are asked to pick their most favoured out of a set of three or more alternatives, and are typically given multiple sets of choice questions. Because CEs are based on attributes, they allow the researcher to value attributes as well as situational changes. Furthermore, in the case of damage to a particular attribute, compensating amounts of other goods (rather than compensation based on money) can be calculated. This is one of the approaches that can be used in Natural Resource Damage Assessments (NRDAs). An attribute-based approach is necessary to measure the type or amount of other 'goods' that are required for compensation (Adamowicz et al., 1998). This approach can provide substantially more information about a range of possible alternative policies as well as reduce the sample size needed compared to contingent valuation (CV). However, survey design issues with the CE approach are often much more complex due to the number of goods that must be described and the statistical methods that must be employed.

Although results from original studies using all these valuation techniques could be used for value transfer; studies using CV, TC and HP methods dominate. More recently, CE studies have been used for value transfer (Hanley and Wright, 2003, Scarpa et al., 2003). Since CE has the potential to decompose the economic value into values for each component/attribute of an environmental good (and value more changes in environmental quality in the same survey), it could be better suited for value transfer than CV studies. However, more research is needed on the role and validity of CE in value transfer. Value transfer based on any of these four methods can be performed in several ways.

TYPOLOGY OF VALUE TRANSFER METHODS

There are two main approaches to benefit transfer:

1. Unit value transfer
   - Simple unit transfer
   - Unit transfer with income adjustments
2. Function transfer
   - Benefit function transfer
   - Meta analysis

Unit Value Transfer

Simple unit transfer is the easiest approach to transferring benefit estimates from one site to another. This approach assumes that the well-being experienced by an average individual at the study site is the same as will be experienced by the average individual at the policy site. Thus we can directly transfer the benefit estimate, often expressed as mean WTP/household/year, from the study site to the policy site.

For the past few decades this procedure has routinely been used in the United States to estimate the recreational benefits associated with multipurpose reservoir developments and forest management. The selection of these unit values could be based on estimates from only one or a few valuation studies considered to be close to the policy site (both geographically and in terms of the good valued), or based on an average WTP estimate from literature reviews of many studies. Walsh et al. (1992, table 1) presents a summary of unit values of days spent in various recreational activities, obtained from 287 CV and TC studies. The US Oil Spill Act recommends transfer of unit values for assessing the damages resulting from small 'Type A' spills or accidents using the National Resource Damage Assessment Model for Coastal and Marine Environment. This model transfers benefit estimates from various sources to produce damage assessments based on limited physical information about the spill site.

The obvious problem with this transfer of unit values for recreational activities is that individuals at the policy site may not value recreational activities the same as the average individual at the study sites. There are two principal reasons for this difference. First, people at the policy site might be different from individuals at the study sites in terms of income, education, religion, ethnic group or other socioeconomic characteristics that affect their demand for recreation. Second, even if individuals' preferences for recreation at the policy and study sites were the same, the recreational opportunities might not be.

Unit values for non-use values of, for example, ecosystems from CV studies might be even more difficult to transfer than recreational (use) values for at least two reasons. First, the unit of transfer is more difficult to define. While the obvious choice of unit for use values is consumer surplus (CS) per activity day, there is greater variability in reporting non-use values from CV surveys, in terms of both WTP for whom, and for what time period. WTP is reported both per household or per individual and as a one-time payment, annually for a limited time period, annually for an indefinite time, or even monthly payments. Second, the WTP is reported for one or more specified discrete changes in environmental quality, and not on a marginal basis. Therefore, the magnitude of the change should be close, in order to get valid transfers of estimates of mean, annual WTP per household. Also the initial levels of environmental quality should be close if one is to expect non-linearity in the benefit estimate or underlying physical impacts.

For health impacts the question of which units to transfer seems somewhat simpler. With regard to mortality the unit could be the value of a statistical life (VOSL). For morbidity, it is more complicated since several units of value are used. For light respiratory symptoms such as coughing, headaches and itchy eyes, symptom days (defined as a specified symptom experienced one day by one individual) are often used (for example Navrud, 2001). Values for more serious illnesses are reported in terms of value per case. However, the description of these different symptoms and illnesses varies in terms of, for example, severity. A better alternative would therefore be to construct values for episodes of illness defined as type of symptoms, duration and severity (described in terms of restrictions in activity levels, whether one would have to go to the hospital and so on); see, for example, Ready et al. (1999).

On the issue of units to transfer, one should also keep in mind that the valuation step is often part of a damage function approach (see Box 5.1). Therefore a linkage has to be developed between the units of the endpoints of dose-response functions, and the unit of the economic estimates. This has been done successfully, for example, for visibility changes at national parks (measured as percentage change in miles of visibility); see Smith and Osborne (1996) and health impacts (for example RSD – respiratory symptoms day, VOSL – value of statistical life, VOLYL – value of life years lost; see Alberini and Krupnick, 2002), but is much more difficult for complex changes in environmental quality and natural resources.

The simple unit value transfer approach should not be used for transfer between countries with different income levels and costs of living. Therefore, unit transfer with income adjustments has been applied. Since most of the environmental valuation studies have been conducted in developed countries, this has become the general practice when conducting CBAs of infrastructure projects in developing countries, for example in the World Bank.

The adjusted benefit estimate B_p' at the policy site can be calculated as

$$B_p' = B_s \, (Y_p/Y_s)ß,$$

where B_s is the original benefit estimate from the study site, Y_s and Y_p are the income levels at the study and policy site, respectively and ß is the income elasticity of demand for the environmental good in question. There is, however, little empirical evidence on how the income elasticity of demand ß for different environmental goods and health impacts varies with income.

The primary assumption in adjusting WTP values in proportion to some measure of income is that the income elasticity (of demand for environmental quality) is 1.0. Krupnick et al. (1996, p. 320) note in their benefit transfer exercise for impacts of air pollution in Central and Eastern Europe (CEE) that there is no reason to think that WTP for environmental quality varies proportionally with income. They note that empirical evidence from the USA shows that the premature mortality risk (from, for example, air pollution) is an inferior good. The relative income approach (that is, assuming an income elasticity of 1.0) will then understate the WTP of lower-income countries. Thus Krupnick et al. use an income elasticity of 0.35 (with 1.0 as a sensitivity analysis) when transferring US mortality values to CEE. Alberini and Krupnick (2002) conclude from their value transfer comparison that assuming an income elasticity of WTP of 1.0, or even making other adjustments, does not appear to be reliable for valuing morbidity and mortality risks in developing countries.

When we lack data on the income levels of the affected populations at the policy and study sites, gross domestic product (GDP) per capita figures have been used as proxies for income in international benefit transfers. However, Barton (1999) clearly shows how this approach could give wrong results in international value transfers when income levels at the study and/or policy site deviate from the average income level in the individual countries.

Using the official exchange rates to convert transferred estimates in US dollars to the national currencies does not reflect the true purchasing power of currencies, since the official exchange rates reflect political and macroeconomic risk factors. If a currency is weak on the international market (partly because it is not fully convertible), people tend to buy domestically produced goods and services that are readily available locally. This enhances the purchasing powers of such currencies on local markets. To reflect the true underlying purchasing power of international currencies, the US International Comparison Program (ICP) has developed measures of real GDP on an internationally comparable scale. The transformation factors are called purchasing power parities (PPPs).

The Asian Development Bank manual on economic valuation of environmental impacts (ADB, 1996) provides monetary values for health and environmental impacts that are adjusted in proportion to per capita gross domestic product (GDP). They note that it would be more appropriate to use PPP estimates of per capita GDP because these estimates have been adjusted to reflect a comparable amount of goods and services that could be purchased with the per capita national income in each country.

Even if PPP-adjusted GDP figures and exchange rates can be used to adjust for differences in income and cost of living in different countries, it will not be able to correct for differences in individual preferences, initial environmental quality, and cultural and institutional conditions between countries (or even within different parts of a country).

Function Transfer

Transferring the entire benefit function is conceptually more appealing than just transferring unit values because more information is effectively taken into account in the transfer. The benefit relationship to be transferred from the study site(s) to the policy site could be estimated using either revealed preference (RP) approaches like TC and HP methods or stated preference (SP) approaches like the CV method and choice experiments (CE). For a CV study, the benefit function can be written as:

$$WTP_{ij} = b_0 + b_1 G_j + b_2 H_{ij} + e, \tag{5.1}$$

where WTP_{ij} = the willingness to pay of household i at site j, Gj = the set of characteristics of the environmental good at site j, and H_{ij} = the set of characteristics of household i at site j, b_0 , b_1 and b_2 are sets of parameters, and e is the random error.

To implement this approach the analyst would have to find a study in the existing literature with estimates of the constant b_0 and the sets of parameters, b_1 and b_2. Then the analyst would have to collect data on the two groups of independent variables, G and H, at the policy site, insert them into equation (5.1), and calculate households' willingness to pay at the policy site.

The main problem with the benefit function approach is due to the exclusion of relevant variables in the WTP (or bid) function estimated in a single study. When the estimation is based on observations from a single study of one or a small number of recreational sites or a particular change in environmental quality, a lack of variation in some of the independent variables usually prohibits inclusion of these variables. For domestic benefit transfers researchers tackle this problem by choosing the study site to be as similar as possible to the policy site.

Instead of transferring the benefit function from one selected valuation study, results from several valuation studies could be combined in a meta-analysis to estimate one common benefit function. Meta-analysis has been used to synthesize research findings and improve the quality of literature reviews of valuation studies in order to come up with adjusted unit values. In a meta-analysis, several original studies are analysed as a group, where the result from each study is treated as a single observation in a regression analysis. If multiple results from each study are used, various meta-regression specifications can be used to account for such panel effects.

The meta-analysis allows us to evaluate the influence of a wider range in characteristics of the environmental good, the features of the samples used in each analysis (including characteristics of the population affected by the change in environmental quality), and the modelling assumptions. The resulting regression equations explaining variations in unit values can then be used together with data collected on the independent variables in the model that describes the policy site to construct an adjusted unit value. The regression from a meta-analysis would look similar to equation (5.1), but with one added independent variable; C_s = characteristics of the study *s* (and the dependent variable would be WTP_s = mean willingness to pay from study *s*).

Smith and Kaoru's (1990) and Walsh et al.'s (1990, 1992) meta-analyses of TC recreation demand models using both TC and CV studies for the USDA Forest Service's resource planning programme were the first attempts to apply meta-analysis to environmental valuation. Later there have been applications to HP models valuing air quality (Smith and Huang, 1993), CV studies of both use and non-use values of water quality improvements (Magnussen, 1993), CV studies of groundwater protection (Boyle et al., 1994), TC studies of freshwater fishing (Sturtevant et al., 1995), CV studies of visibility changes at national parks (Smith and Osborne, 1996), CV studies of morbidity using quality of life years (QUALY) indexes (Johnson et al., 1996), CV studies of endangered species (Loomis and White, 1996), CV studies of environmental functions of wetlands (Brouwer et al., 1997), HP studies of aircraft noise (Schipper et al., 1998), CV studies of landscape changes (Santos, 1998), CV studies of WTP for wastewater treatment in coastal areas (Barton, 1999), and outdoor recreation (Shrestha and Loomis, 2001). The last five studies are international meta-analyses, including both European and North American studies. All the others, except Magnussen (1993), analyse US studies only.

Many of these meta-analyses of relatively homogeneous environmental goods and health effects are not particularly useful for benefit transfer even within the USA, where most of these analyses have been conducted, because they focus mostly on methodological differences.[1] Methodological variables such as 'payment vehicle', 'elicitation format' and 'response rates' (as a general indicator of quality of mail surveys) in CV studies, and model assumptions, specifications and estimators in TC and HP studies, are not particularly useful in predicting values for specified change in environmental quality at the policy site. This focus on variables describing the methodological choices made is partly due to the fact that some of these analyses were not constructed for benefit transfer (for example Smith and Kaoru, 1990; Smith and Huang, 1993, Smith and Osborne, 1996). Another reason is that inadequate information was reported in the published studies with regard to characteristics of the study site, the change in environmental quality valued, and income and other socio-economic characteristics of the sampled population. In particular, the last class of variables would be necessary in international benefit transfer, assuming cross-country heterogeneity in preferences for environmental goods and health effects.

In most of the meta-analyses secondary information was collected on at least some of these initially omitted site and population characteristics variables, or for some proxy for them. These variables make it possible to value impacts outside the domain of a single valuation study, which is a main advantage of meta-analysis over the benefit function transfer approach. However, often the use of secondary data and/or proxy variables introduces added uncertainty, for example using income data for the population in the region instead of income data for the fishermen at the study site. On the other hand, these secondary data are more readily available at the policy site without having to do a new survey.

Most meta-analyses caution against using them for adjusting unit values due to potential biases from omitted variables and specification/measurement error of included variables. To increase the applicability of meta-analysis for value transfer, one could select studies that are as similar as possible with regard to methodology, and thus be able to single out the effects of site and population characteristics on the value estimates. However, the problem is that there are usually so few valuation studies of a specific environmental good or health impact that one cannot do a statistically sound analysis.

VALIDITY AND RELIABILITY OF BENEFIT TRANSFER

While there are very detailed guidelines, although disputed, on how to carry out high-quality original valuation studies, for example Arrow et al. (1993) for contingent valuation (CV) surveys, no such (universally accepted) guidelines exist for benefit transfer. Although Smith (1992) called for the development of a standard protocol or guidelines for conducting benefit transfer studies more than ten years ago, no such generally accepted protocol exists. More recent studies, however, comparing benefit transfers with new CV studies of the same site to test the validity of benefit transfer, provide valuable input in the development of such guidelines.

Loomis (1992) argues that cross-state benefit transfer in the USA (even for identically defined activities) is likely to be inaccurate, after rejecting the hypotheses that the demand equations and average benefits per trips are equal for ocean sport salmon fishing in Oregon versus Washington, and for fresh-water steelhead fishing in Oregon versus Idaho. Bergland et al. (1995) and Downing and Ozuno (1996) used the benefit function transfer and unit value approaches. Downing and Ozuno only looked at use value, while Bergland et al. also cover transfer of non-use value.

Bergland et al. (1995) conducted the same CV study of increased use and non-use values for water quality improvements at two Norwegian lakes (let us call them A and B for simplicity), constructed benefit functions for A and B, and then transferred the benefit function of lake A to value the water quality improvement in lake B, and vice versa. The mean values were also transferred and compared with the original CV estimate, since the two lakes are rather similar with regard to size and type of pollution problem. When selecting the independent variables for the demand function two different approaches were used: (i) selecting variables which give the largest explanatory power, and (ii) selecting variables for which it is possible to obtain data at the policy site without having to do a costly survey. The last approach would ease future transfers, but could give less reliable estimates. Several tests for transferability were conducted, but all indicate lack of transferability statistically speaking (that is, transferred and original values are significantly different at the 5 per cent level). However, the transfer error, defined as the difference between predicted (transferred) mean WTP and observed mean WTP (in the original study), as a percentage of the observed mean WTP, is 'only' 20–40 per cent. In one lake the transferred values were higher, in the other they were lower than the estimate from the original study. Thus, from this study one cannot conclude which procedure would produce the highest values.

While Bergland et al. (1995) test benefit transfers spatially by conducting two CV studies at the same time, Downing and Ozuno (1996) test benefit transfer both spatially and intertemporally through CV and TC models of recreational angling at eight bays along the Texas coast. Using a 5 per cent significance level, they found that 91–100 per cent of the estimates were not transferable across bays, but 50–63 per cent of within-bay estimates were transferable across time. Like Bergland et al. (1995), they conclude that geographical benefit transfer is generally not statistically reliable. Transfer errors are, however, 'only' 1–34 per cent.

Brouwer and Spaninks (1999) reached the same conclusion in their CV studies of use and non-use values of biodiversity (meadow birds and bankside flowers) of two Dutch peat meadow sites. The original CV study gave significantly higher estimates than transferred CV estimate from the other peat meadow area, but the transfer error is again relatively low: 22–40 per cent.

Scarpa et al. (2003) estimated the use value of 26 recreational forests in Ireland using a discrete choice CV survey. As opposed to Downing and Ozuno (1996), they find that benefit function transfer (spatially) is reliable. Fifty-one and 62 per cent of the median and mean WTP estimates, respectively, are transferable; that is, the original and transferred estimates are not significantly different at the 10 per cent level.

Kirchhoff et al. (1997) found in a comparative survey of recreationists in the states of Arizona and New Mexico that out of 24 comparisons of benefit measures, only two involved errors exceeding 100 per cent, and 16 out of 24 indicated errors of less than 50 per cent, which is a magnitude of error that still might be acceptable in a preliminary assessment of a proposed policy's impacts.

All the validity tests of benefit transfer mentioned above look at different sites within the same country. Ready et al. (1999) conducted the same CV study in five European countries (the Netherlands, Norway, Portugal, Spain and the United Kingdom). They found that the transfer error in valuing respiratory symptoms (that could be caused by air pollution) was ±37–39 per cent in terms of predicting mean willingness to pay (WTP) to avoid the symptom in one country from the data of the other countries. The observed transfer error should be compared with the variability in the original estimate within a country of ±16 per cent (estimated using Monte Carlo simulations). Unit value transfer, unit value transfer with income adjustment (using PPP indexes for the cities in which the studies were conducted, since national PPP indexes were not representative for these specific cities) and benefit function transfer performed equally well (or poorly). The remaining differences in valuation between countries are due to other factors than income/purchase power, education level, age, sex, number of children in the household and health status variables. Thus cultural and attitudinal factors seems to be important in explaining differences in valuation across countries.

In another benefit transfer validity test across countries, Rozan (1999) found transfer errors of 15–30 per cent. She conducted the same CV survey in Strasbourg, France and the neighbouring city of Kehl, Germany; and asked respondents to state their WTP for a specified improvement in air quality. Barton and Mourato (2003) and Chestnut et al. (1997) are examples of benefit transfer validity tests between developed and developing countries. These studies show transfer errors often larger than between developed countries.

To conclude, results from validity tests show that the uncertainty in value transfers both spatially and temporally could be quite large. Thus benefit transfer should be applied to environmental valuation where the demand for accuracy is not too high. Kristofersson and Navrud (2002) propose equivalency tests for validity of value transfer. Equivalence testing is better adapted to this task than traditional testing because it combines the concepts of statistical significance and policy significance into one test, by defining an acceptable transfer error before conducting the validity test. The level of acceptable transfer error will depend on the intended policy use.

CHALLENGES IN VALUE TRANSFER

Value transfer is less than ideal, but so are most valuation efforts in the sense that better estimates could be obtained if more time and money were available. Analysts must constantly judge how to provide policy advice in a timely manner, subject to the resource constraints they face. Analysts should compare the cost of doing a new, original valuation study with the potential loss of making the wrong decision when using the transferred estimate. Decision theory and Bayesian analysis could be used to assess the need for further information about both monetary values and other steps in the damage function approach (DFA); see Barton (1999) for an application. Studies that can reduce the uncertainty in transfer of information in other stages of the DFA (see Box 5.1) should also be conducted.

Transfer methods may be particularly useful in policy contexts where rough or crude economic benefits may be sufficient to make a judgement regarding the advisability of a policy or project. Thus value transfer could be used in cost–benefit analyses of projects and policies, but one should be more careful in using transferred values in environmental costing and accounting exercises at the national and firm levels, and particularly in calculating compensation payments for natural resource injuries.

Five main difficulties and, thus, challenges in value transfer were identified:

1. Difficult access and low quality of many existing studies from which to transfer values.
2. Valuation of new policies or projects are difficult in that:
   - expected change resulting from a policy is outside the range of previous experience;
   - most previous studies value a discrete change in environmental quality; how to convert these values into values for marginal changes resulting from a new policy or project;
   - most previous studies value a gain in environmental quality; how to convert these values to losses in order to value loss in environmental quality of, for example, energy and transportation projects.
3. How to adjust for differences in the study site(s) and policy site that are not accounted for in the specification of the valuation model (function transfer) or in the procedure used to adjust the unit value (adjusted unit values).
4. How to determine the 'extent of the market'. To calculate aggregated benefits, the mean benefit estimate has to be multiplied by the total number of affected households (that is, households that find their well-being affected by the change in the quality of the environmental good). Guidelines on how to determine the size of the affected population are needed.
5. While original valuation studies can be constructed to value many benefit (or cost) components simultaneously, benefit transfer studies often involve transfer and aggregation of individual components. Simply adding them assumes the independence in value between the components (that is, the independent valuation and summation (IVS) procedure). If components are substitutes or complements, this simple adding-up procedure would over- and underestimate the total benefits (or costs), respectively. Thus correction factors to take these interdependencies into

account have to be applied. It remains to see whether it is possible to construct general sets of correction factors for groups of environmental goods.

FUTHER RESEARCH

The response to these main challenges in benefit transfer could be development of:

1. Improved benefit transfer techniques, establishment of correction factors for interdependencies of transferred benefit and cost factors, and a protocol for benefit transfer including recommendations for how to determine the size of the 'affected population', and correct for interdependencies among components of the environmental good.
2. A database of environmental valuation studies.

Recently, there have been great advances in both these issues, but they all need to be explored further: new approaches to value transfer have been suggested by Smith et al. (2002) (preference calibration approach), and Atkinson et al. (1992), León and Vásquez-Polo (1998) and Barton (1999) (Bayesian statistics and statistical decision theory). Smith et al. propose a preference calibration approach to value transfer to ensure that the transferred values are consistent with budget-constrained utility maximization behaviour. The first and foremost advantage of their approach is that the resulting benefit or damage estimate can never be inconsistent with household income, as they have been in some high-profile applications such as Constanza et al. (1998).

Bayesian statistical techniques have been introduced both for the estimation of contingent valuation models in original valuation studies (León and Vásquez-Polo, 1998; McLeod and Bergland, 1999) and for explicit use in value transfer (Atkinson et al., 1992) The general idea is that we have some prior information about the benefits or costs from a project. If we decide to do a new valuation study specifically for that project, the statistical analysis should recognize the existence of prior information (Bayesian statistics). The decision whether or not to do a new survey should explicitly recognize the uncertain information available (statistical decision theory).

On the issue of correction factors for interdependencies, Santos (1998) demonstrated that the frequently used independent valuation and summation (IVS) procedure for multi-attribute landscape changes would overestimate the benefits by 48 to 80 per cent in two case studies. This confirms that the error IVS introduces in policymaking cannot be ignored, as previously stressed by Hoehn and Randall (1989), Hoehn (1991) and Hoehn and Loomis (1993).

Ruijgrok (2001) tests the use of ecological classification in transferring values for nature reserves on the Dutch coast, and concludes that this approach offers new opportunities for transferring economic benefits within a limited geographical area. Thus more research is needed on the appropriate spatial scale of the classification.

Based on a review of value transfer studies and validity tests of transfer, Brouwer (2000) propose a seven-step protocol as a first attempt towards good practice for using value transfer techniques in CBA. Step 1 involves the identification of the relevant ecological functions of the goods and services under consideration, and their importance for sustaining ecosystems and hence human systems. Step 2 focuses on identifying beneficiaries of the ecological functions/services preserved or foregone, and is interdependent with Step 3, which identifies values held by different stakeholder groups in order to be able to sketch out the reasons why they value the environmental good/service under consideration. Step 4 assesses the scope, acceptability and legitimacy of the valuation process(es): monetary and/or deliberative. In Step 5 appropriate studies are selected, and study quality assessed by looking at their internal and external validity. Step 6 looks at the research design of the selected studies, and tries to assess comparability between them, and what kind of adjustment may be chosen to account for differences in design/approach for each chosen study. In Step 7 values as obtained through the previous six steps are discussed with (representatives of) stakeholders, before they are extrapolated over the relevant population affected by the environmental change under consideration. Finally, the economic aggregate is included in a CBA.

The web-based database EVRI (Environmental Valuation Reference Inventory, www.evri.ec.gc.ca/EVRI/) now contains more than 700 valuation studies. Since the database was initiated by Environment Canada, in cooperation with the US Environmental Protection Agency, the majority of these studies are from North America, and only about 10 per cent are European valuation studies. Based on a European user survey, Navrud (1999) evaluated the EVRI database, and found it well suited for European conditions. Few studies outside North America and Europe are registered in the database. Thus there is a need to increase the number of existing valuation studies captured in this database for all countries.

Since many valuation studies are old and use outdated methodology and there are few studies for many environmental goods, there is also a great need for new, original valuation studies using state-of-the-art methodology, and designed with value transfer in mind. There is also a need for comparative studies in terms of conducting the same valuation studies of environmental amenities, cultural assets and health impacts in many countries at the same time, and testing the validity of transfer to produce calibration factors which would improve value transfers between countries and especially between developed and developing countries. Ready et al. (1999), Barton and Mourato (2003), and Krupnick et al. (1996) are examples of studies testing transfer of values between countries. They also demonstrate the practice of using purchase power parity (PPP)-adjusted exchange rates when transferring between countries with different currencies (and preferably PPP indices from the study area within each country, instead of national averages). Studies testing the validity of transfer over time are also scarce. Downing and Ozuno (1996), McLeod and Bergland (1996), and Carson et al. (2003, p. 277) are some of the very few studies in this area. Carson et al. (2003) found that the results of conducting the same survey (that is, their *Exxon Valdez* oil spill CV study) at three different points in time over the course of a year yielded WTP distributions that were statistically indistinguishable (Carson, 1997). However, there is a great need for testing validity of transfers over more than one year as a basis for constructing a procedure to replace the current practice of assuming that the WTP for environmental goods and health effects increases at a rate equal to the consumer price index.

NOTES

In addition to the individual papers cited in this chapter, I would like to recommend two books as sources for new ideas for value transfer research: Desvousges et al. (1998), and an edited collection of papers on recent advances in value transfer (Navrud and Ready, forthcoming).

1. Carson et al. (1996) is an example of a meta-analysis of different environmental goods and health effects, which was performed with the sole purpose of comparing results from valuation studies using both stated preference (CV) and revealed preference methods (TC, HP, defensive expenditures and actual market data).

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