Confronting Uncertainty and Missing Values in Species Conservation Investment with Environmental Value Transfer

Sonia Akter[*] and R. Quentin Grafton[1*]

[1] Authorship is alphabetical. Corresponding author's e-mail: quentin.grafton@anu.edu.au,

Tel: +61-2-6125-6558, Fax: +61-2-6125-5570

[*]Crawford School of Economics and Government, The Australian National University, Canberra, ACT 2601, Australia.

Running head: Environmental value transfer and species conservation

Key words: environmental value transfer, uncertainty, transfer error, species conservation, non-use values

Category of Paper: Review

Word count: 7,030 including title page, body of paper and references.

21 September, 2009

Abstract

An important category of conservation benefits are non-use (or passive) values that are time-consuming and costly to quantify using direct surveys. In the absence of estimates of these values, there will likely be an underinvestment in conservation actions that generate substantial non-use benefits, such as species and biodiversity conservation. To improve conservation investment decision making, this paper explains why, when and how to use environmental value transfer (EVT) to derive indirect estimates of non-use value while accounting for the uncertainty associated with transferring values from one site to another.

Introduction

A key challenge for decision makers is how best to allocate scarce conservation dollars across competing uses and species (Sutherland et al. 2009). An important input when deciding how to allocate a conservation budget is the non-use (or passive) values of species conservation. Some contend that conservation decisions should only be based on the intrinsic value of species (Soulé 1985; Noss & Cooperrider 1994; McCauley 2006). Monetary values, however, allow for a direct comparison across competing claims and are frequently used in conservation decision-making (Gutman 2002; Balmford et al. 2003; Cicia et al. 2003; MacMillan et al. 2004; Naidoo & Adamowicz 2006).

Non-use values include an existence value that reflects benefits from the current generation from knowing that a species exists (Hageman 1985; Loomis & White 1996), and a bequest value that represents the benefits from ensuring species will be conserved for future generations (Moran & Pearce 1994). The preferred method to obtain monetary estimates of non-use values is via direct surveys using stated preference techniques. Two common approaches to estimate these values are contingent valuation surveys and choice experiments. Both are widely used in terms of endangered species valuation (Hageman 1985; Loomis & White 1996; Hanley et al. 2003; Svensson et al. 2008) and involve public surveys that ask relevant groups of a population their willingness to pay for a policy/conservation action by constructing a hypothetical market or referendum. The development of a stated preference survey instrument, its pre-testing, survey implementation, data collection, and analysis typically requires the time and effort of a number of individuals over several months, and at a substantial cost. In conservation practice, many decision makers neither have the time nor sufficient resources to implement direct surveys. In the absence of direct non-use value estimates, a zero non-use value may be assigned to species conservation (Department of Environmental Heritage 2005). This results in an underinvestment of conservation dollars to projects or actions that generate substantial non-use values (Alexander 2000), such as species conservation.

To help overcome the misallocation of conservation dollars due to missing direct non-use value estimates, decision makers can use environmental value transfer (EVT) to indirectly estimate these values. EVT uses existing data and parameters estimated in settings other than for what they were originally collected (Rosenberger & Loomis 2003) to estimate a monetary value of an environmental good at a particular location. This method 'transfers' both use and non-use values from one location (a study site) to another (a policy site) while accounting for differences at the two sites (Brouwer 2000). The principal motivation for EVT is the need to improve the cost-and-time-effectiveness of environmental policy and decision making. The principal drawback to EVT is the transfer error (TE), the over or underestimation of values at the policy site from not undertaking a direct survey, and the uncertainty it creates when making conservation investments.

In this paper we: (1) explain why, how and when to conduct EVT; (2) describe protocols to reduce TE in terms of study design and method of transfer; and (3) present a framework to account for uncertainty of TE in the decision-making process. In the following section we present a review of the EVT literature and discuss the potential sources of TE. To account for the uncertainty associated with the transfer of values from one site to another, we develop a risk and simulation approach that assigns possible error distributions and performs Monte Carlo simulations to evaluate the possible net benefits of conservation investment. We then provide a step-by-step guide to EVT followed by a decision heuristic to guide conservation practitioners about when to use EVT.

Environmental Value Transfers: Causes of Transfer Error

Environmental value transfer is undertaken using one, or a mix, of the following approaches:

1. Unit value transfer
2. Value function transfer
3. Meta-value function transfer

In unit value transfer, a single point estimate from one study, or an average of multiple point estimates from several studies, is transferred from study site(s) to the policy site. In value function transfer, a value or benefit function and its estimated parameters are used to transfer values while accounting for differences in the independent variables at the study site and policy site. In meta-value function transfer, the value or benefit estimates obtained from a number of relevant studies are first analyzed and synthesized in a meta-analysis to control for differences in study design and sites. Estimated parameters from the meta-value function are then used to transfer the values using sample population characteristics at the policy site.

Measuring Transfer Error

The key criticism of all three approaches to EVT is the validity of the value estimates at the policy site. Many authors, including Loomis (1992), Loomis et al. (1995), Downing and Ozuna (1996), Kirchhoff et al. (1997), Brouwer and Spaninks (1999), Bergland et al. (2002), and Ready and Navrud (2006), have shown that value transfer is associated with significant errors. These errors can only be quantified if direct survey estimates are available at both the study and policy sites (Navrud & Ready 2007b). Where such studies do exist, the error can be quantified as per equation (1),

$$TE = \frac{\left| WTP_{EVT} - WTP_{P} \right|}{WTP_{P}} \tag{1}$$

where WTP is the estimated willingness to pay of the relevant population for a given change in the provision of the environmental good, the subscript $_{P}$ stands for the value that is estimated by a direct study at the policy site and the subscript $_{EVT}$ represents the EVT value obtained by transferring values from a study to the policy site. A lower value of TE implies, all else equal, a more valid EVT. Typically, decision makers will not know the TE when undertaking EVT because they will not have undertaken a direct survey at the policy site.

Sources of Transfer Error

Context Similarity

It is asserted that TE is lower the more similar is the policy context of the study site to that of the policy site. This is known as the 'context similarity condition'. Several empirical studies, including KristòFersson and Navrud (2007) and also Lindhjem and Navrud (2008), conclude that TE is inversely related to context similarity, but others contend that it is a necessary not a sufficient condition to obtain a valid EVT (Loomis & Rosenberg 2006, Navrud & Ready 2007a).

An important question in terms of context similarity is whether the actual physical locations of the study and policy sites are determinants of the TE. Empirical evidence suggests that the geographical proximity between the study and policy sites does not guarantee a lower TE. Johnston (2007) observes that communities which live in close proximity, but differ in terms of their land use practices, may have substantially different values for the environmental good. Johnston and Duke (2009) support this finding by comparing the TE when undertaken across jurisdictional scale, i.e. transferring values from a community to a county or to a state. They find that, on average, the TE is ten times lower when the study site was located in a different state but the jurisdictional scale was the same as the policy site relative to a transfer where the study site was located in the same state, but differed in terms of the jurisdictional scale. Ready and Navrud (2006) suggest that, if applied carefully, even international transfer can be as valid as intra-country transfer. However, some studies provide conflicting evidence as they find inter-state EVT is more valid relative to cross-state EVT (Loomis et al. 1995; Van den Berg et al. 2001, Piper & Martin 2001).

Methods of Transfer

An on-going debate exists as to whether more a complicated approach to EVT, such a meta-analysis, reduces TE. Engel (2002) compares various EVT studies and shows that the value function transfer method performs better than meta-value function transfer in 14 out of 20 cases. Lindhjem and Navrud (2008) compare meta-value function transfer with unit value transfer and find that a unit value transfer approach yields a TE in the middle of the range of two meta-value transfer models.

Other studies compare unit value transfer to value function transfer and observe the former has a lower TE than the latter (Parsons & Kealy 1994). By contrast, some find that TE is reduced by using value function transfer (Loomis 1992; Kirchhoff et al. 1997; Rosenberger & Stanley 2006) while others find that unit value transfer and value function transfer perform equally well (Barton 2002; Chattopadhyay 2003; Ready et al. 2004, KristòFersson & Navrud 2007).

Temporal Stability

EVT assumes that changes in environmental values over time can be accounted for by transforming past values into current values using an appropriate inflator, such as the consumer price index (Navrud & Brouwer 2007). Zandersen et al. (2007) undertook an EVT using a value function transfer approach over a 20 year period. They show that individual preferences for forest attributes, such as species diversity and age, can change significantly over time. The implication of this finding is that the transfer of values that are further in the past, all else equal, will be less useful for environmental value transfer than more recent direct estimates.

Environmental Value Transfer: Accounting for Uncertainty

TE creates uncertainty as to whether the indirect estimates of values at a policy site are sufficiently accurate enough to inform decision makers when allocating conservation budgets. This is a particular concern for non-use values because they are likely to be a substantial proportion of the total value (Hanley et al. 1999, p. 75), especially in terms species and biodiversity conservation. Two protocols to help account for this uncertainty when transferring non-use values across sites are: (1) an ‘error bounds approach’ developed by Navrud & Brouwer (2007) and (2) a ‘risk and simulation’ method that has not previously been used in EVT.

Error Bounds Approach

To help decision makers account for uncertainty, Navrud and Brouwer (2007) suggest construction of upper and lower bounds of the transferred benefit estimate based on hypothetical TE values. They advocate using different bounds depending on the level of context similarity and propose an error bound of ±20-40 % if the study and policy sites are context similar. If not, they recommend an error bound of ±100 %.

To illustrate this approach, we use the findings of a contingent valuation survey carried out by Rubin et al (1991) that estimates the net benefits of the northern spotted owl conservation in the Pacific northwest. In their study, they estimate the benefit (non-use value) from spotted owl conservation to be USD 1,481 million, the opportunity cost (forgone timber income, job loss) to be USD 497 million and, thus, the net benefit to be USD 984 million. Using the error bounds approach proposed by Navrud and Brouwer (2007) for (1) context similar and (2) context dissimilar cases, estimates of the net benefit from spotted owl conservation are computed (Table 1).

INSERT TABLE 1 HERE

In the context similar scenario, an error bound of ±40 % yields a positive net benefit within the range of USD 392 million to USD 1,576 million. However, for the context dissimilar case, the estimated net benefit lies within the range –USD 497 million and USD 2,465 million. In the latter case, the error is such that it would be difficult, if not impossible, for a decision maker to justify spotted owl conservation on the basis of the estimated net benefits alone.

Risk and Simulation Approach

A more useful approach to account for the uncertainty associated with TE is to assign alternative probability distributions to an uncertain TE, and then use Monte Carlo simulations to construct cumulative density functions (CDFs) for the net benefits. This is an approach that has been used in the context of benefit-cost analysis (Campbell & Brown 2003), but we are the first to apply it in EVT.

In almost all cases decision makers will not know the underlying distribution of transfer errors and, thus, the size of the errors. To account for this distributional uncertainty, various distributions (uniform, triangular, normal) can be used to estimate their effects on the decision to invest across conservation alternatives. Estimates of net benefits are calculated by applying this approach using the spotted owl example and identical minimum and maximum values for TE as used in the error bounds approach (Table 2).

INSERT TABLE 2 HERE

In the context similar case, and given a triangular distribution, the TE is assumed to have a maximum, minimum and a 'most likely' value of +40%, –40% and +40%. The most likely error value (+40%) is based on subjective judgment and can be set to any number within the range of +40% and –40%. Simulating this distribution of TE for the estimated benefits generates a mean net benefit of USD 786 million after 5,000 iterations. The contribution of the 'risk and simulation' approach, however, is not in generating a mean, mode or median value estimate of the net benefits, but in the CDF it generates (Fig. 1). The CDF represents the cumulative probabilities attached to different values of net benefit, conditional upon the chosen error distribution. In this case, the estimated net benefit is strictly positive with a maximum, minimum and mean value of USD 1,562, USD 391 and USD 786 million. Importantly, the approach allows the analyst to develop a confidence interval in terms of the net benefits conditional on the assumed error distribution and error bounds. For instance, there is 90% probability that the estimated net benefit lie within the range USD 400 to 1,400 million (Fig. 1).

INSERT FIGURE 1 HERE

In the context dissimilar case, and again for illustrative purposes only, we assume a different probability distribution and assign specific probabilities (0.1, 0.1, 0.2, 0.5, and 0.1) to different quintiles of TE within the range of ±100%. These probabilities are assigned subjectively and can, and should, vary depending on the decision-maker's judgment as to the possible distributions for TE. Under these probabilities, and for the assumed probability distribution, the mean net benefit of spotted owl conservation is USD 747 million. The CDF for this case shows that there is a 92% probability that conservation action yields a positive net benefit (Fig. 2).

INSERT FIGURE 2 HERE

The contribution of the risk and simulation approach is that it allows decision makers to make their own judgments as to the appropriate distributions and also lower and upper bounds of TE, and to then simulate the consequences for conservation decision-making. Thus, it does not eliminate uncertainty, but provides a means where decision makers can evaluate the possible risks under different scenarios, and to make useful comparisons across conservation investments.

Environmental Value Transfer: A Step-by-Step Guide

The practice of EVT over the past 20 years or more provides guidance as to how to undertake value transfer in ways that reduce TE. In this section, we provide a step-by-step guide to EVT with a focus on: (1) the policy context; (2) the selection of the primary study (ies) for value transfer; (3) design and reporting criteria of the primary study (ies) and (4) the methods of value transfer.

Policy Context

The policy context is composed of three components (1) the environmental good to be valued; (2) the type and size of the policy action; and (3) the stakeholders or the characteristics and size of the population affected (benefited) by the proposed policy. All three components are helpful when selecting direct studies for value transfer. For example, a practitioner may wish to use direct studies on salmon conservation if the policy context involves an action that may increase a salmon population at a defined policy site. The policy context at the policy site is also important to identify *ex-ante* whether they match the context at the study sites (Loomis & Rosenberger 2006). For example, if the proposed action at the policy site is the establishment of a marine protected area to promote mammal conservation, then the chosen study site should, if at all possible, be of a similar nature (direct survey of value of a marine reserve for mammal conservation).

The policy context also requires detailed information about stakeholder characteristics (age, education, income, attitudes, etc.) to determine the estimated benefit to the population from the conservation policy. Thus, if the non-use value of an endangered bird species is transferred from a study site where the population has a higher level of environmental awareness to a policy site where the population is relatively unconcerned about species conservation, the transfer would (all else equal) be expected to overestimate of true non-use value. Population size is another important characteristic necessary for aggregating values at a policy site where the larger is the relevant population size the greater will be the aggregate benefit of species conservation.

Selection of the Primary Study (ies)

A key step in reducing TE is to identify primary studies at the study sites that are similar to the policy site. Several databases are available for this purpose that, collectively, contain thousands of primary valuation studies and include: The Environmental Valuation Resource Inventory

(EVRI), the Environmental Valuation (Envalue) database, the Ecosystem Services Database (ESD) and the Review of Externality Database (RED). These databases are described in McComb et al. (2006). Relevant datasets may also be collected by contacting the author(s) of the original study.

The most important challenge when selecting suitable study sites is the relevance of the studies to the context at the policy site. This is because the more similar are the contexts between study sites and the policy site, all other factors equal, the lower should be the TE (Navrud & Ready 2007a).

Another important issue in selecting study sites is the quality of the studies. According to Freeman (1984), a direct or primary study must be based on adequate data, sound economic method and correct empirical technique. To quantify these factors, Brouwer (2000) suggests using the notions of internal and external validity. Internal validity refers to whether the study findings correspond to theoretical expectation. For example, we might expect that the willingness to pay for species conservation would increase the higher is a person's income. External validity refers to the interpretability of the actual findings and the estimated values. For instance, a study might find that respondents' willingness to pay increases with income (internally valid) but the amount they are willing to pay almost equals their entire disposable income – an unlikely finding for most conservation outcomes (externally invalid). Sample selection procedures and the response rate of the study site are also important indicators of data quality, as is credibility of the hypothetical valuation scenario, the hypothetical payment vehicle (the method respondents are asked to pay for the policy action), and the treatment of protest responses, among other considerations (Brouwer 2000).

Assessing the quality of direct studies based on technical criteria can be difficult for non-specialists in non-market valuation techniques. In such circumstance, some ‘rules of thumb’ may be helpful. For instance, Costanza et al. (2006) divide the available non-market studies into three different categories: (1) peer-reviewed empirical analyses; (2) non peer-reviewed analyses (technical reports, PhD Theses and government documents); and (3) secondary, summary studies such as statistical meta-analyses of primary valuation literature. The expectation is that peer reviewed studies would, on average, be of higher quality than non peer-reviewed studies (Liu & Stern 2008) while journal rankings, date of the study and experience or reputation of the study authors may also be useful signals of study quality.

Design and reporting criteria of the primary study (ies)

In some instances, the EVT validity suffers due to poor design and insufficient reporting by the original study. The usefulness of a primary study for EVT applications can largely be enhanced if care is taken at the design stage of the original study and some minimum reporting criteria are fulfilled. For instance, the quality characteristics at a study site should be measured using objective and physical units while demographic variables, ideally, should be measured using categories and definitions that are consistent with census definitions to allow for comparison of the characteristics of the population at the policy site with the population of the study site.

Loomis and Rosenberger (2006) recommend some minimum reporting criteria to enhance the study’s usefulness in EVT application. These same criteria can also be used to compare the quality of studies for use in value transfer. First, they suggest that a study should contain detailed description about the good being valued including information related to the study site (location, size, number, accessibility, etc.), type and size of the policy and, most importantly, the exact wording of the valuation scenario. Second, a study should include a description of the demographic characteristics for the study site population and sample (tourists or residents), their knowledge about the good being valued and their level of awareness, concern and attitude towards the conservation problem at hand. Finally, care should be taken in the way the welfare measure is reported such that the unit of the welfare measure (per person or per household, per month or per year) is clearly specified.

The methods of value transfer

Unit Value Transfer

Unit value transfer is the simplest method to undertake EVT. In this approach, the estimate of mean willingness to pay is transferred directly from a study site to the policy site, usually without accounting for differences in socio-demographic or other characteristics between the two groups of populations. To illustrate, we present an example by Loomis (2006).

Non-use values of wild salmon in the Lower Snake River (the policy site) were obtained from transferring values from a study site that estimated the non-use value of an increased salmon population size on the Elwha River (the study site). The proposed policy action for increased wild salmon at the policy site was a dam removal which was the same policy action at the study site. The estimated mean willingness to pay at the study site was $73 dollars per household for an increase of 300,000 wild salmon in the Elwha River.

In undertaking the value transfer, the mean willingness to pay was adjusted for inflation because the original study was conducted in 1994 while the transfer was undertaken in 1996. Adjusting for a 5% inflation rate over the two years, Loomis (2006) calculated the mean willingness to pay to be USD 76.48. The original study also revealed that, other things remaining the same, the annual mean willingness to pay (USD 73) of Washington State residents' is significantly larger than the annual mean willingness to pay of the residents from the rest of the country (USD 68). Consequently, a downward adjustment was applied to account for this difference. In the final step of the value transfer, the non-user population size at the policy site was adjusted to obtain aggregate values of the population (Table 3). After discounting (at a discount rate of 3.875% per year) the future stream of benefits over fifty years, the annual non-use values from increased salmon population using EVT generates an annual benefit equivalent of about USD 49 million.

INSERT TABLE 3 HERE

Value Function Transfer

The transfer of a value function can directly account for differences in site characteristics and characteristics of the affected populations (Rosenberger & Loomis 2003) at the study and policy sites. The application of this approach requires a direct study that estimates a willingness to pay function, as shown in equation (2):

$$\mathrm{WTP_s} = f(\beta_s, X_s) \tag{2}$$

where $\mathrm{WTP_s}$ refers to willingness to pay, the subscript s stands for study site, β_s is the vector of estimated parameters at the study site and X_s refers to a vector of relevant independent variables including socio-demographic and environmental good characteristics at the study site.

Using the willingness to pay function from the study site, an analyst would collect information about relevant independent variables from the policy site. At the final stage of the value transfer, the parameter vector estimated at the study site (β_s) are used along with the vector of relevant independent variables at the policy site (X_p) to obtain the willingness to pay at the policy site, as per equation (3)

$$\mathrm{WTP_p} = g(\beta_s, X_p) \tag{3}$$

where the subscript p refers to the policy site and the value function used at the policy site (g) need not be the same as at the study site (f).

To illustrate the approach, we summarize a study by White et al. (1997) who investigate the economic values of two endangered species (the otter *Lutra lutra* and the water vole *Arvicola terrestris*) within the context of the United Kingdom biodiversity action plans. They employ a contingent valuation survey to estimate the willingness to pay for conservation of the two endangered species from about 500 randomly selected North Yorkshire residents via telephone interviews. In addition to providing socio-demographic and attitudinal responses, respondents were asked to pay a pre-specified bid (money) amount in the form of an income tax towards the implementation of conservation action plans for the two species. Using this primary data, the authors estimated a function given by equation (4):

$$Y = 1.94 - 0.14\mathrm{bid} - 0.40\mathrm{age} + 1.07\mathrm{member} + 1.05\mathrm{threat} \tag{4}$$

where Y is the probability of a respondent accepting the offered bid level, bid is the amount of money respondents were asked to pay, age is the respondent's age, member is whether the respondent is a member of an environmental organization and threat indicates if the respondent is aware of the threat to the mammals. Equation (4) can be rearranged to estimate a mean willingness to pay of the sample population for conservation actions as follows,

$$\text{bid(mean willingness to pay)} = -\frac{1.94 - 0.40\text{age} + 1.07\text{member} + 1.05\text{threat}}{-0.14} \qquad (5)$$

Evaluating the willingness to pay function given by equation (5) at the mean value of the sample characteristics (age = 3.325, member = 0.22, threat = 0.55), a mean willingness to pay of £10.12 for the conservation of both species together can be calculated.

To transfer the estimated mean willingness to pay (£10.12) from North Yorkshire, it is necessary to estimate the mean willingness to pay at the policy site while also accounting for differences in the population characteristics at the policy site. For example, if we assume that the mean willingness to pay values are transferred to a location where a half of the residents are members of environmental organizations (member = 0.5), half of the residents are aware of the threats to which the endangered species are exposed (threat = 0.5), and the policy site residents belong to the average age group of 30-39 (which can be categorized as an age category = 3) then equation (6) can be used to estimate willingness to pay at the policy site:

$$\text{bid(mean willingness to pay)} = -\frac{(1.94 - 0.40\times 3 + 1.07\times 0.5 + 1.05\times 0.5}{-0.14} \qquad (6)$$

Equation (6) yields a mean willingness to pay of £12.86 per household per year.

Meta-value Function Transfer

In meta-function value transfer the point estimates (mean willingness to pay values) from a number of existing direct studies are collected and synthesized. Using these point estimate data, multi-variate regression is used to estimate a meta-value function. This meta-value function is then used, as with the value function transfer approach, to provide estimates of value at a given policy site.

To illustrate the approach, we use the findings of Richardson and Loomis (2009) who estimate a meta-analysis function using 31 stated preference studies that vary in terms of methodology, study design (elicitation format, payment vehicle, payment frequency, compliance imperative, study year, survey mode, and response rate), sample characteristics (visitors, local residents) and the characteristics of the endangered species (fish, marine mammals, birds). They estimate several meta-value functions, but recommend the following for EVT:

$$\begin{aligned}&\ln \mathrm{WTP(2006\$)} = -153.231 + 0.870 \ln \mathrm{changesize} + 1.256 \mathrm{visitor} + 1.020 \mathrm{fish} + \\ &0.772 \mathrm{marine} + 0.826 \mathrm{birds} - 0.603 \ln \mathrm{responserate} + 2.767 \mathrm{conjoint} + 1.024 \mathrm{charismatic} \\ &-0.903 \mathrm{mail} + 0.078 \mathrm{studyyear}\end{aligned}$$

(7)

The explanatory variables included in equation (7) along with the values for evaluating the meta-analysis function at a policy site are described in Table 4. Equation (7) can be used to estimate mean willingness to pay for species conservation by substituting in the sample means for the methodological variables and the appropriate values for the policy relevant variables. For example, if the decision maker wishes to estimate the value of a 50% increase in charismatic sea otter populations to non-visitors in the year 2007, the mean willingness to pay value could be estimated as per equation (8):

$$\begin{aligned}&\ln WTP(2007\$) = -153.231 + 0.870 \times 3.912 + 1.256 \times 0 + 1.020 \times 0 + 0.772 \times 1 \\ &+0.826 \times 0 - 0.603 \times 3.894 + 2.767 \times 0 + 1.024 \times 1 - 0.903 \times 0.851 + \\ &0.078 \times 2007\end{aligned} \qquad (8)$$

where the value of the methodological variables in equation (8), such as responserate, conjoint and mail are obtained from the sample mean of the 31 studies that were included in the meta analysis.

Environmental Value Transfer: A decision heuristic

A review of the methods of EVT, and how to manage the uncertainty associated with TE, can be combined to provide a decision heuristic about when to use EVT. This heuristic is a decision tree that can be used to guide the decision process and illustrates the possible decision branches to determine if, and how, EVT should be applied (Fig. 3).

INSERT FIGURE 3 HERE

Given the possible errors associated with a value transfer, the first question faced by decision makers is whether or not EVT should be used. The response will depend on the level of accuracy required for the conservation problem at hand. If a decision framework requires highly accurate information, such as to settle litigation and to calculate compensation payments due to a conservation action, then a direct study is likely to be preferred. However, if the estimated value is to be used to evaluate the net benefits of alternative conservation investments an EVT, with some TE, can be useful and preferable to assuming a zero non-use value in the absence of a direct study. For instance, if EVT shows that there is likely to be large non-use values associated with a particular conservation policy action, it provides a justification for either further analysis (such as direct survey) or, possibly, investment in the proposed policy action.

Primary or direct survey research is preferred to estimate non-use values (Rosenberger & Johnston 2009) if the conservation decision is not constrained by time and/or financial resources.

These direct costs can be considerable although these will vary by country where the study is being undertaken, the chosen methodology (contingent valuation or choice modeling) and survey mode (mail, personal interview, and internet). For example, according to Whitehead (2006), the minimum cost to implement a mail survey ranges between USD5 to USD10 per respondent in 2003 dollars, and these surveys would typically require several hundred respondents. Direct surveys also require a series of focus group discussions to develop the survey instrument followed by several rounds of pretests before finalizing the survey instrument. This direct survey process, from start to finish, would typically take three to six months or possibly longer. Thus, when a conservation decision cannot be delayed and neither can it support the cost of primary data collection, EVT is the preferred alternative.

Subsequent to the decision to apply EVT, a practitioner needs to determine which value transfer approach is the most appropriate. As a general guide, Navrud and Brouwer (2007) suggest that unit value transfer should not be used except when the policy site and study sites are context similar. In situations when the policy site and the study site are context dissimilar, a value function transfer is recommended, but with two caveats. First, the value function should have sufficient explanatory power. Second, the transfer function should contain variables for which data is available at the policy site, or could be collected from secondary sources. They also suggest using a meta-value function transfer only if the scope of environmental good and policy contexts at the study sites are not too different to the policy site.

Conclusions

Insufficient time, personnel or financial resources make it difficult to undertake direct surveys to estimate the non-use values that are required to make informed decisions about conservation investments. In the absence of such estimates, there will likely be an underinvestment in conservation actions that generate substantial non-use values, such as species and biodiversity conservation. As an alternative to treating such values as zero when direct survey estimates are unavailable, we describe when and how to use environmental value transfer to derive indirect estimates of non-use values so as to ensure a better allocation of funding among conservation alternatives.

A major challenge in the use of environmental value transfer is the uncertainty about the sign and size of the error from transferring direct estimates of non-use values from a study to policy site. All else equal, the greater the similarity of the policy context (environmental good, type and size of the policy, stakeholder characteristics) at the study sites and the policy site, the smaller will be the transfer error. However, even with the best of matches between sites and policy contexts, there will be an unknown transfer error.

To facilitate the use of environmental value transfer and to account for uncertainty in value transfer, a risk and simulation approach is developed that assigns probability distributions to an uncertain transfer error, and uses Monte Carlo simulations to construct alternative cumulative density functions. This method does not reduce the transfer error, but provides a method to account for the effect of transfer error in conservation decision-making. This approach and a decision-based framework about when to use environmental value transfer, offer an improvement to current decision-making about how to compare alternative actions and when to invest in species and biodiversity conservation.

References

Alexander, R.R. 2000. Modelling species extinction: the case for non-consumptive values. Ecological Economics **35:** 259-269.

Balmford, A., K.J. Gaston, S. Blyth, A. James, and V. Kapos. 2003. Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. Proceedings of the National Academy of Sciences USA **100**: 1046–1050.

Barton, D. N. 2002. The transferability of benefit transfer: contingent valuation of water quality improvements in Costa Rica. Ecological Economics **42**: 147–164.

Bergland, O., K. Magnussen., and S. Navrud. 2002. Benefit transfer: testing for accuracy and reliability. Pages 117–132 in R. J. G. M. Florax, P. Nijkamp and K. Willis, editors. Comparative environmental economic assessment. Edward Elgar, Cheltenham, UK.

Brouwer, R., and F.A. Spaninks. 1999. The validity of environmental benefits transfer: further empirical testing, Environmental and Resource Economics **14**: 95–117.

Brouwer, R. 2000. Environmental value transfer: state of the art and future prospects. Ecological Economics **32**: 137-152.

Campbell, H.F and R.P.C. Brown. 2003. Benefit-cost analysis: financial and economics appraisal using spreadsheets. Cambridge University Press, Cambridge, UK.

Chattopadhyay, S. 2003. A repeated sampling technique in assessing the validity of benefit transfer in valuing non-market goods. Land Economics **79**: 576–596.

Cicia G., E. D'Ercole, and D. Marino. 2003. Costs and benefits of preserving farm animal genetic resources from extinction: CVM and bio-economic model for valuing a conservation program for the Italian Pentro horse. Ecological Economics **45**:445–459.

Costanza, R., M. Wilson, A. Troy, A. Voinov, S. Liu, and J. D'Agostino. 2006. The value of New Jersey's ecosystem services and natural capital. Report of New Jersey Department of Environmental Protection, Division of Science, Research, and Technology, Trenton, NJ.

Department of the Environment and Heritage. 2005. Making economic valuation work for biodiversity conservation. Biological Diversity Advisory Committee Land and Water Australia, Canberra.

Downing, M., and T. Jr Ozuna. 1996. Testing the reliability of the benefit function transfer approach. Journal of Environmental Economics and Management **30**: 316–322.

Engel, S. 2002. Benefit function transfer versus meta-analysis as policy-making tools: a comparison. Pages 133-153 in R. J. G. M. Florax, P. Nijkamp and K. G. Willis, editors. Comparative environmental economic assessment. Edward Elgar, Cheltenham, U.K. and Northampton, Mass.

Freeman, A.M. 1984. On the tactics of benefit estimation under Executive Order 12291. Pages 167-186 in V.K. Smith, editor. Environmental policy under Reagan's executive order: The role of benefit–cost analysis. The University of North Carolina Press, Chapel Hill, NC.

Gutman, P. 2002. Putting a price tag on conservation: cost benefit analysis of Venezuela's national parks. Journal of Latin American Study **34**: 43–70.

Hageman, R. 1985. Valuing marine mammal populations: benefit valuations in a multi-species ecosystem. National Marine Fisheries Service, Southwest Fisheries Centre, La Jolla, California.

Hanley, N., M. Whitby, and I. Simpson. 1999. Assessing the success of agri-environmental policy in the UK. Land Use Policy **16**: 67-80.

Hanley, N., D. Macmillan, I. Patterson, and R.E. Wright. 2003. Economics and the design of nature conservation policy: a case study of wild goose conservation in Scotland using choice experiments. Animal Conservation **6**:123–129

Johnston, R. J. 2007. Choice experiments, site similarity and benefits transfer. Environment and Resource Economics **38**:331–351.

Johnston, R. J., and J. M. Duke. 2009. Willingness to pay for land preservation across states and jurisdictional scale: implications for benefit transfer. Land Economics **85**: 217-237.

Kirchhoff, S., B. G. Colby., and J. T. LaFrance. 1997. Evaluating the performance of benefit transfer: an empirical inquiry. Journal of Environmental Economics and Management **33**: 75–93.

KristÒFersson, D., and S. Navrud. 2007. Can use and non-use values be transferred across countries? Pages 207-226 in S. Navrud and R. Ready, editors. Environmental Value Transfer: Issues and Methods. Springer Verlag, Berlin, Germany.

Lindhjem, H., and S. Navrud. 2008. How reliable are meta-analyses for international benefit transfer? Ecological Economics **66**; 425-435.

Liu, S., and D. I. Stern. 2008. A meta-analysis of contingent valuation studies in coastal and near-shore marine ecosystems. Socio-economics and the environment in discussion (SEED) working paper series 2008-15, CSIRO Sustainable Ecosystems, CSIRO, Canberra.

Loomis, J. B. 1992. The evolution of a more rigorous approach to benefit transfer: benefit function transfer. Water Resources Research **38**: 701–705.

Loomis, J. B., B. Roach, F. Ward., and R. Ready. 1995. Testing transferability of recreation demand models across regions: a study of corps of engineer reservoirs. Water Resources Research **31**: 721-730.

Loomis, J.B., and D. S. White. 1996. Economic benefits of rare and endangered species: summary and meta-analysis. Ecological Economics **18**: 197–206.

Loomis, J.B., and R.S. Rosenberger. 2006. Reducing barriers in future benefit transfers: needed improvements in primary study design and reporting. Ecological Economics **60:** 343-350.

Loomis, J. B. 2006. Stated preference benefit transfer approaches for estimating passive use value of wild salmon, Pages 54-70 in J. Rolfe and J. Bennett, editors, Choice modelling and the transfer of environmental values, Edward Elgar, Cheltenham.

MacMillan, D., N. Hanley, and M. Daw. 2004. Costs and benefits of wild goose conservation in Scotland. Biological Conservation **119**:475–485.

McCauley, D. J. 2006. Selling out on nature. Nature **443**:27–28.

Moran, D., and D. Pearce. 1994. The value of biodiversity. World Conservation Union, Biodiversity Programme, Earthscan Publications, London, U.K.

McComb, G., V. Lantz, K. Nash, and R. Rittmastera. 2006. International valuation databases: overview, methods and operational issues. Ecological Economics **60**:461–472.

Naidoo, R., and W.L. Adamowicz. 2006. Modeling opportunity costs of conservation in transitional landscapes. Conservation Biology **20**: 490–500.

Navrud, S., and R. Brouwer. 2007. Good practice guidelines in benefits transfer of forest externalities, COST Action E45, European Forest Externalities, Euroforex.

Navrud, S., and R. Ready. 2007a. Lessons learned for environmental value transfer, Pages 283-290 in S. Navrud and R. Ready, editors, Environmental value transfer: issues and methods. Springer Verlag, Berlin, Germany.

Navrud, S., and R. Ready. 2007b., Review of methods for value, Pages 1-10 in S. Navrud and R. Ready, editors, Environmental value transfer: issues and methods. Springer Verlag, Berlin, Germany.

Noss, R., and A. R. Cooperrider. 1994. Saving nature's legacy: protecting and restoring biodiversity. Island Press, Washington D.C.

Parsons, G. R., and M. J. Kealy. 1994. Benefits transfer in a random utility model of recreation. Water Resources Research **30**: 2477–2484.

Piper, S., and W. E. Martin. 2001. Evaluating the accuracy of the benefit transfer method: a rural water supply application in the USA. Journal of Environmental Management **63**: 223–235.

Ready, R., and S. Navrud. 2006. International benefits transfer: methods and validity tests. Ecological Economics **60**: 429-434.

Ready, R., S. Navrud, B. Day, R. Dubourg, F. Machado, S. Mourato, F. Spanninks, and M. Rodriquez. 2004. Benefit transfer in Europe: how reliable are transfers between countries. Environmental and Resource Economics **29**: 67–82.

Richardson, L., and J. B. Loomis. 2009. The total economic value of threatened, endangered and rare species: an updated meta-analysis. Ecological Economics **68**: 1535-1548.

Rosenberger, R. S., and J. B. Loomis. 2003. Benefits Transfer. Pages 445–482 in P. A. Champ, K. J. Boyle and T. C. Brown, editors. A primer on nonmarket valuation. Kluwer, Boston.

Rosenberger, R. S., and T. D. Stanley. 2006. Measurement, generalization and publication: sources of error in benefit transfers and their management. Ecological Economics **60**: 372-378.

Rosenberger, R. S., and R. J. Johnston. 2009. Selection effects in meta-analysis and benefit transfer: avoiding unintended consequences. Land Economics **85**: 410–428.

Rubin, J., G. Helfand, and J. Loomis. 1991. A benefit-cost analysis of the northern spotted owl. Journal of Forestry **89**: 25–30.

Soulé, M.E. 1985. What is conservation biology? BioScience **35**: 727-734.

Sutherland, W.J., W.M. Adams, R.B. Aronson et al. 2009. One hundred questions of importance to the conservation of global biological diversity. Conservation Biology **23**(3): 557-567.

Svensson, P., L. D. Rodwell, and M. J. Attrill. 2008. Hotel managed marine reserves: A willingness to pay survey. Ocean & Coastal Management **51**: 854–861

VandenBerg, T.P., G. L. Poe, and J. R. Powell. 2001. Accessing the accuracy of benefits transfer: evidence from a multisite contingent valuation study of groundwater quality. Pages 101-120 in J. C. Bergstrom, K. J. Boyle and G. L. Poe, editors. The economic value of water quality. Edward Elgar, Cheltenham.

White, P.C.L., K. W. Gregory, P. J. Lindley, and G. Richards. 1997. Economic values of threatened mammals in Britain: a case study of the otter Lutra lutra and the water vole Arvicola terrestris. Biological Conservation **82**: 345–354.

Whitehead, J.C. 2006. A practitioner's primer on the contingent valuation method. Pages 92-115 in A. Alberini and J.R. Kahn, editors. Handbook on contingent valuation. Edward Elgar, Northampton, MA.

Zandersen, M., M. Termansen, and F. S. Jansen. 2007. Testing benefits transfer of forest recreation values over a 20-year time horizon. Land Economics **83**:412-440.

Table 1 Construction of error bound for net benefit under context similarity and dissimilarity

	Study Site Values	Scenario 1: Context similar		Scenario 2: Context dissimilar	
Error values (%)		+40	-40	+100	-100
Benefit (million USD)	1481	888	2073	0	2962
Cost (million USD)	497	497	497	497	497
Net Benefit (million USD)	984	392	1576	-497	2465

Table 2 Estimated net benefits conditional on error distributions

	Study Site Values	Scenario 1: Context similar	Scenario 2: Context dissimilar
Error distribution		Triangular [a]	Histogram[b]
Benefit (million USD)	1481	1284	1244
Cost (million USD)	497	497	497
Net Benefit (million USD)	984	787	747

Note: [a] Transfer error is assumed to follow a triangular distribution with minimum, maximum and most likely value of -40%, 40%, 40% respectively.

[b] Transfer error has assumed to take different values within the range of ±100% with different probabilities. The probability distribution function {value (probability)} is as follows: {-1 (0.1), -0.5 (0.1), 0 (0.2), 0.5 (0.5), 1 (0.1)}.

Table 3 Example of a Unit Value Transfer

Policy site	Size of non-user population (in millions)	Aggregate benefit = mean willingness to pay X population size (in USD)
Washington	1.4	76.48 X 1.4 = 107.1 million
Rest of the population	11.1	71.12 X 11.1 = 789.4 million
Total economic benefit at the policy site from 300,000 additional salmon		= 896.5 million
Marginal benefit from salmon at the policy site		896.5 million/300,000 = 2, 988
Annual benefit (B) from 37,000 salmon at the policy site		2,988 X 37,000 = 111 million
Total discounted benefit (PV) from 37,000 salmon over 50-year life of the project		=2,427 [a] million
Average discounted benefit from 37,000 additional salmon		= 48.8 million

[a] This values has been obtained by using the formula: $PV = \frac{B}{i} * [1 - \frac{1}{(1+i)^n}]$ where i = 3.875% and n=50.

Table 4 Description of variables used in Richardson and Loomis (2009)

Variable	Variable description	Values
changesize	The percentage change in the species population proposed in the survey	50[a]
visitor	Whether the survey respondents were visitors (visitors=1, households=0)	0
fish	If the endangered species valued in the study is classified as fish (fish=1, otherwise=0)	0
marine	If the endangered species valued in the study is classified as marine mammals (marine=1, otherwise=0)	1 [a]
birds	If the endangered species valued in the study is classified as birds (birds=1, otherwise=0)	0
responserate	The mean survey response rate of all the 31 studies included in the meta-analysis (a continuous variable)	50 [b]
conjoint	Sample mean of methodology used in the 31 studies included in the meta-analysis	0
charismatic	A species' 'charisma' or high profile status on the public's valuation.	1 [a]
mail	Sample mean of survey mode used in the 31 studies included in the meta-analysis	0.85 [b]
studyyear	The year the study was performed (a discrete variable)	2007 [a]

[a] These values correspond to a hypothetical policy site
[b] These values are average of all the studies used in estimating the meta-analysis function.

Figure 1 Cumulative probability distribution of net benefit under context similarity condition and triangular error distribution.

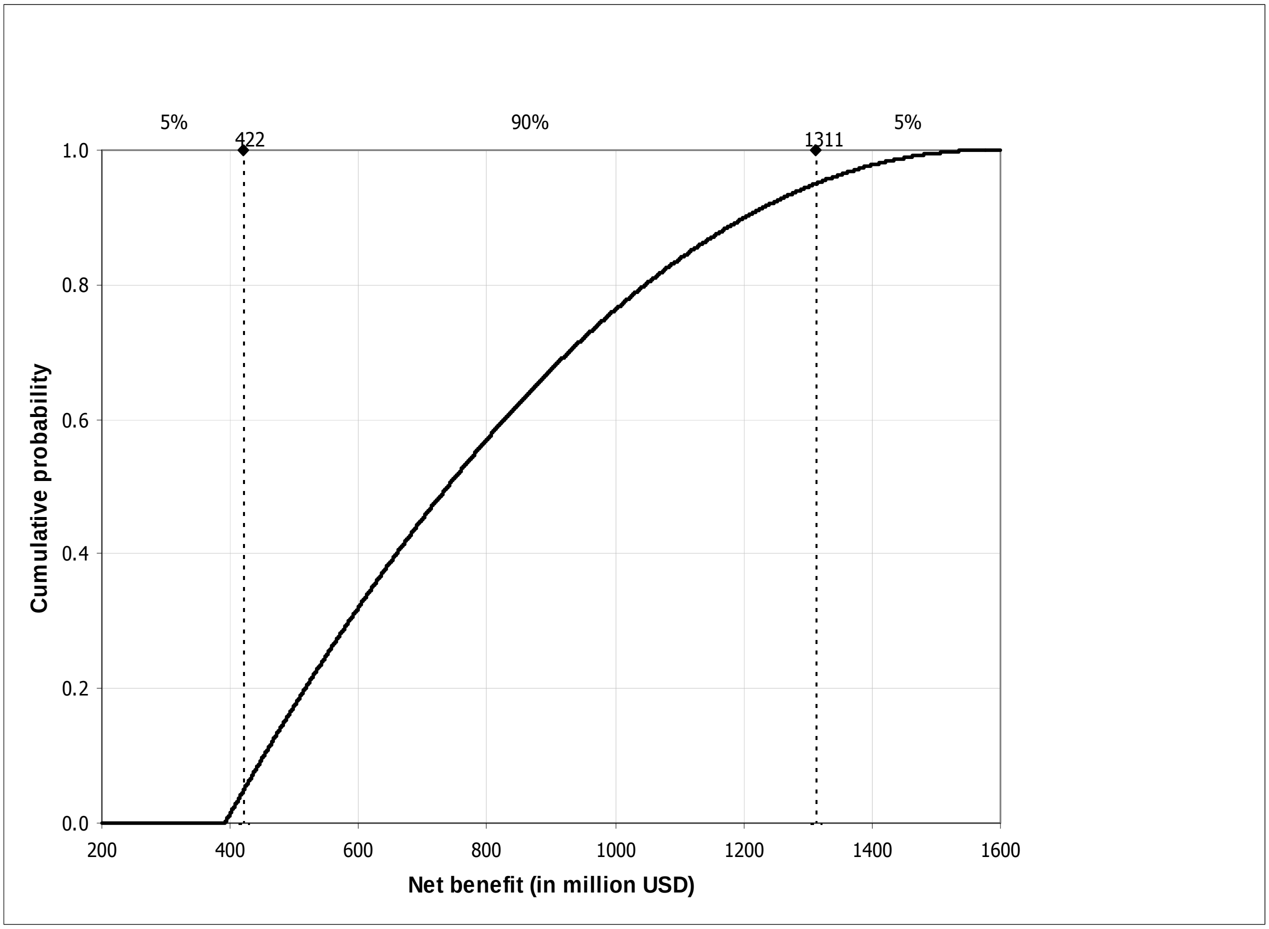

Figure 2 Cumulative probability distribution of net benefit under context dissimilarity condition and uncertain error distribution.

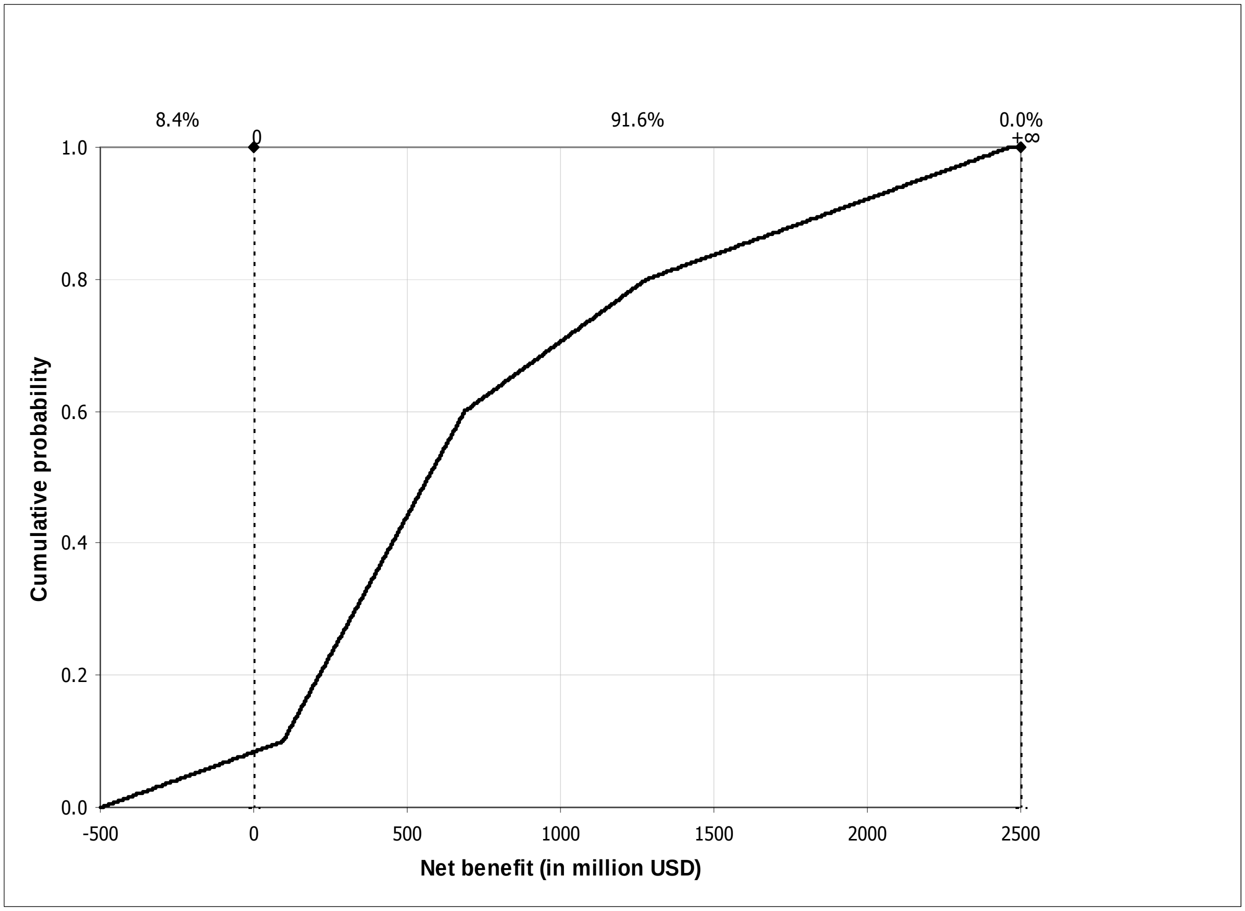

Figure 3 Decision tree to perform value transfer for species conservation