S. PATTANAYAK, V. K. SMITH AND G. VAN HOUTVEN

IMPROVING THE PRACTICE OF BENEFITS TRANSFER: A PREFERENCE CALIBRATION APPROACH

1. INTRODUCTION

Most applied welfare analyses for environmental policy evaluations, whether benefit-cost or natural resource damage assessments, rely on adaptations of existing benefit estimates in what is described as benefits transfer rather than new research. Over 10 years ago, David Brookshire organized a set of papers in *Water Resources Research* to focus attention on the practice of benefit transfer (see Brookshire and Neill [1992]. Since then, interest in research on the potential for improvement in these techniques has exploded and this volume reports on a number of innovations relying on refinements in the statistical methods used in meta analyses that often provide empirical benefit functions for transfer. Nonetheless, where evaluations of benefits transfer have taken place, current practice is generally regarded as *very unreliable*![1]

This paper considers a different perspective on the practice of benefit transfers. It is one that interprets the benefit transfer problem as an identification problem. That is, the analyst must calibrate individual preferences for the environmental resources of interest based on the available empirical benefit estimates. Our proposed methodology is general. Here we apply it to one example – the development of consistent measures of the benefits of water quality improvements. To develop this logic we begin with a historical perspective, interpreting Harberger's [1971] approximation using indifference curves and then suggesting this logic seems to have been a conceptual antecedent to the logic used with unit benefit transfers. However, in this case the same desirable properties cannot be assured. Section 2 presents a detailed description of conventional benefit transfer practices and these antecedents. Section 3 illustrates the limitations of such practices through a simple example. In Section 4, we provide a detailed description of our proposed methodology in six steps. Using a case study from the Chesapeake Bay our proposed approach is illustrated in Section 5 with travel cost and contingent valuation (CV) data. We then demonstrate how the calibrated functions can be used to construct benefit estimates for a separate situation. We discuss how the resulting benefit estimates differ from those of a more traditional benefit transfer practice, hereafter labeled "simple approximation." Finally, in Section 6 we present a few methodological conclusions.

S. Navrud and R. Ready (eds.), Environmental Value Transfer: Issues and Methods, 241–260.

Table 1. Recent Evaluations of Conventional Benefit Transfers

Commodity/Service	Authors	Year	Source
Recreation water quality	Loomis, Roach, Ward, and Ready	1995	*Water Resources Research*
Fishing water quality	Downing and Ozuna	1996	*JEEM*
Recreation water quality	Kirchhoff, Colby, and LaFrance	1997	*JEEM*
Health water quality	Barton	1999a	Working Paper
Waste water treatment benefits	Barton	1999b	Working Paper
Rural farm landscape	Santos	1999	Working Paper
Peat meadow amenities	Brouwer and Spaninks	1999	*Environmental and Resource Economics*
Overview	Brouwer	2000	*Ecological Economics*
City air quality	Rozan	2000	Working paper
Forest amenities	Scarpa, Hutchinson, Chilton, and Buongiorno	2000	Working paper

[a] The numbers correspond to the RFF water quality ladder and index (boatable = 2.5, fishable = 5.1, and swimmable = 7).

2. BENEFIT TRANSFER: CONVENTIONAL PRACTICE AND CONCEPTUAL ANTECEDENTS

Benefit transfer adapts available estimates of the economic value for a change in environmental quality (or quantity) to evaluate a proposed policy-induced change in the same or a "similar" resource. In these situations, the analyst is typically taking the results from one or more existing studies (defined in terms of their time frame, the location, the environmental resource, or quality change, and the affected population), and using them to evaluate a different context that is relevant for a specific policy. The new policy context can require changes in both the features of the resource and the characteristics of the people who care about it.

Most benefit transfer methods use either the *benefit value* or the *benefit function* approaches. In the case of a benefit value approach, a single point estimate (usually a mean willingness to pay (WTP) estimate) or value range is typically used to summarize the results of one or more studies that have been developed for another purpose. For example, an average consumer surplus per fishing trip might be taken from a recreational travel cost study, or a mean WTP estimate for an incremental change in water quality may be estimated from a CV study. These values are then used to evaluate the benefits from proposed policies that change water quality at different locations. In these applications, the transfers are intended to assess the economic value of fishing trips or changes in water quality in new areas. In the case of a benefit function transfer, a model has been estimated to describe how benefit measures (from one or more existing studies) change with the characteristics of the study population or the resource being evaluated.[2] Often this function is derived from a meta analysis. With this second approach, the function is "transferred" to the policy context, and the benefit estimate is then "tailored", based on the arguments of the function, to meet the new population's characteristics and the new resource's features. For example, a travel cost demand model from one site might be used with the income, average travel costs, and quality conditions for a policy site to estimate the consumer surplus under different conditions.

Benefit transfers usually proceed in four steps:

1. Translate the policy change into one or more quantity changes related to the affected environmental resource, such as the resulting change in level of use of the resource for a typical user.[3]
2. Estimate the number of individuals linked to the affected resource (e.g., the number of typical users) before and after the policy change.
3. Transfer a per "unit" value (e.g., consumer surplus) measure, with the unit measured in terms of the quantity index selected in Step 1.
4. Combine estimates in Steps 1 through 3 for each year considered in the analysis and compute the discounted aggregate benefit measures.

A typical benefit transfer of this type can also be summarized in a simple relationship, such as the following equation.

$$CS_p = \frac{CS_T}{\Delta d_T}(d_1 \bullet N_1 - d_0 \bullet N_0) \tag{1}$$

where

$CS_P =$ the estimate of consumer surplus gain for policy being evaluated;

$CS_T =$ the consumer surplus gain (for a representative individual) measured in other literature for a change judged to be comparable to that of the policy;

$d_i =$ the quantity index, such as the level of use of the affected resource by a typical user (e.g., visits per year), in the presence of the policy change ($i = 1$) and in the absence of the policy change ($i = 0$);

$\Delta d_T =$ the change in the quantity index that corresponds to the CS_T measures in the literature; and

$N_i =$ the number of individuals linked to the affected resource with the policy change ($i = 1$) and without it ($i = 0$).

This approach focuses the analysis on individual-specific quantity measures, which are characterized by the d term. In the recreation context, d is usually measured as a trip to or visitor-day at a recreation site. However, in the health context it could be measured as an episode of illness avoided or the risk of some acute condition (e.g., asthma). N would then be the number of affected or exposed individuals (through the affected resource, such as contaminated surface water), and Δd_T would be expressed as a *reduction* in the health effect or risk.[4]

CS_P can also be expressed for a quality change. That is, d can be expressed as a quantitative measure of the quality of the resource (e.g., water quality) that is affected by the policy. In this case, CS_T would have to be measured and interpreted as the consumer surplus gain from a specified quality change (Δd_T) for a similar resource that is experienced by a typical individual. In this case, $d_1 - d_0$ would be interpreted as the quality change resulting from the policy number, and N would be the number of individuals experiencing the change ($N_1 = N_0$).

In any adaptation of equation (1), the term $CS_T/\Delta d_T$, serves as a unit value or "price." In recreation-based applications, it might be expressed as a consumer surplus per visitor-day. With the benefit value approach to benefit transfer, the unit value is treated as a constant, regardless of the characteristics of the individual or the size of the change experienced by each individual. The use of a benefit function approach can provide more flexibility in how $CS_T/\Delta d_T$ is defined for different individuals and, in some cases, for different sizes or types of change; however, in practice, nearly all existing transfers follow the basic format of equation (1).

Each adaptation to the basic format described in equation (1) changes the mix of assumptions required to evaluate their consistency with respect to the basic benefit concepts. Although such adaptations are likely to affect the performance of the benefit estimation approach, none of these adjustments is directly linked to the concept of Hicksian willingness to pay (WTP).

When the benefit transfer process is described by the relationship in equation (1), it resembles the methods used to approximate WTP measures for price changes. Figures 1a and 1b illustrate the basic logic. In figure 1a we assume a simple linear demand function and seek to estimate the consumer surplus from P_0 to P_1 (with $P_0 < P_1$). This is given in equation (2) and simplifies to the expression given in equation (3).

$$CS = (P_1 - P_0)q_1 + 1/2(P_1 - P_0)(q_0 - q_1) \tag{2}$$

$$CS = 1/2(P_1 - P_0)(q_1 + q_0) \tag{3}$$

Figure 1b illustrates how it relates to expenditure changes.

$$CS = 1/2(CD + AB) \tag{4}$$

This result is readily established when we recognize that:

$$\begin{aligned} CD &= (m_0 - P_0q_1) - (m_0 - P_1q_1) = (P_1 - P_0)q_1 \\ AB &= (m_0 - P_0q_0) - (m_0 - P_1q_0) = (P_1 - P_0)q_0 \end{aligned} \tag{5}$$

Thus, the consumer surplus is an average of the change in expenditures on all other goods when q_1 is evaluated at P_0 and P_1 versus evaluating q_0 at these prices. Therefore, the logic for transfer provides a direct parallel to the expenditure changes that Harberger considered as approximating consumer surplus. The average quantity is valued by the price change to compute the consumer surplus.

In practice, benefit transfers have used the same general approach to estimate WTP for a quantity or quality change, rather than a price change. The transfer values are used as if they were virtual prices (i.e., marginal WTP measures) for the quantity or quality change associated with the policy. An important limitation of this approach is that it treats these virtual prices as if they were constant, when in

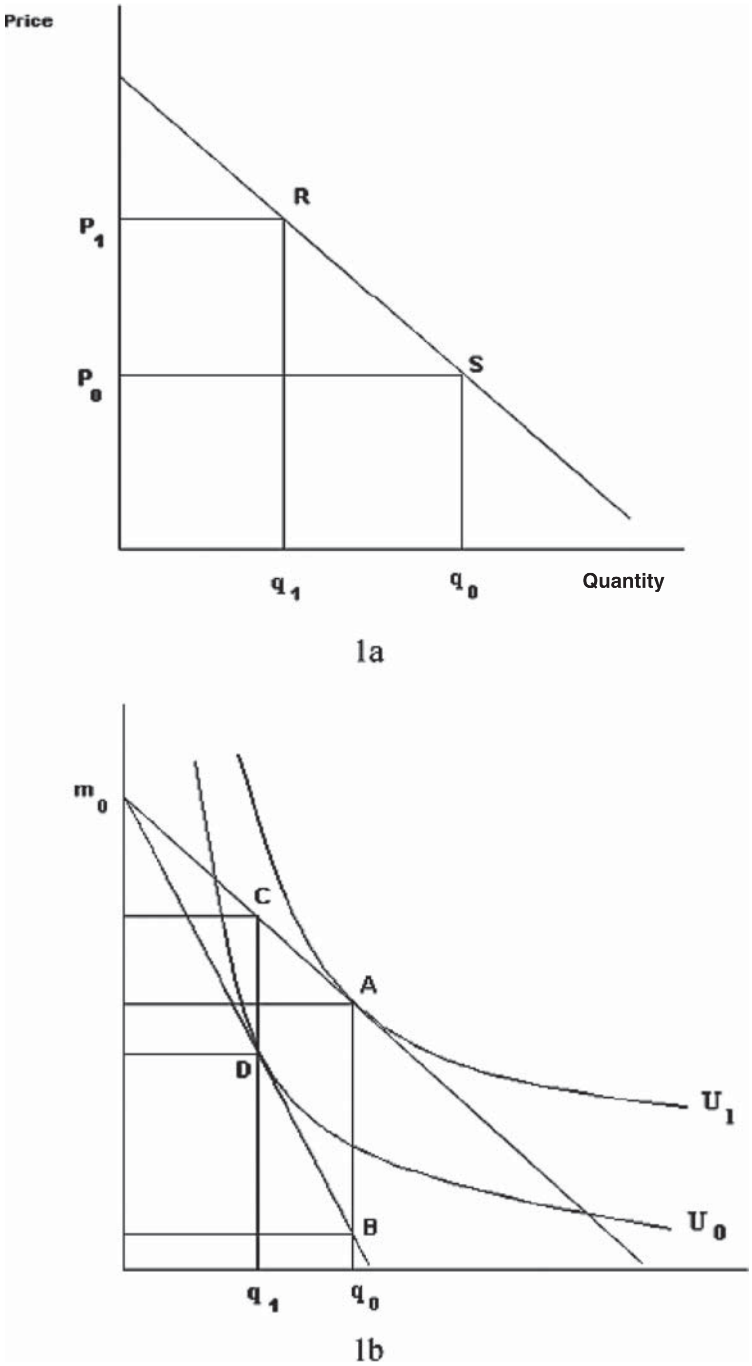

Figure 1. Illustration of Harberger Approximation

fact they are not. This outcome contrasts with approximations intended to measure the value of *price* changes, where we can assume prices are constant. This can be illustrated with a simple version of a Hicksian expenditure function, e(.), with priced goods, (and $\bar{P}$ a vector of prices); one nonpriced good, Z; and a quality feature for Z, measured by s. The Hicksian consumer surplus (WTP) for a change in quality from s_0 to s_1 is given in equation (5a) with U_0 the initial utility level and Z_0 the level of the nonmarketed good:

$$\text{(5a)} \qquad \mathrm{WTP} = e(\bar{P}, Z_0, s_0, U_0) - e(\bar{P}, Z_0, s_1, U_0) > 0$$

We expect this difference to be positive. Z is a desirable environmental service and its quality, s, is measured such that $s_1 > s_0$. Improvements in quality allow an individual to spend less on marketed goods but be able to attain the same level of well being.

Using the virtual price of Z, ρ_z, to approximate a parametric price, the first order approximation for consumer surplus, $\widetilde{CS}_c$, for a change from s_0 to s_1 might be written as equation (5b).[5]

$$\text{(5b)} \qquad \tilde{C}S_c = \rho_z \bullet [Z(s_0) - Z(s_1)]$$

with $\rho_z = \dfrac{\partial e}{\partial z}$, which is evaluated at a value for Z (i.e., $Z[s_0]$ or $Z[s_1]$).

Unfortunately, any set of values we would propose to use for p_z and Z can be expected to be interrelated through e(.) and will be functions of s. The virtual price for the quality change can therefore, strictly speaking, not be treated as exogenous or constant.

3. DIFFICULTIES WITH CURRENT BENEFIT TRANSFER PRACTICES

Some of the limitations associated with conventional approaches to benefit transfer can be illustrated through an example involving water quality changes and recreation-based benefits. This example serves to explain the difficulties in using price-based approximations to describe the benefits due to the quality and quantity changes. In this case, the objective of the transfer is to assess the benefits of policies that have reduced pollutant discharges to a major river system in the U.S. The capacity of this river system to support different types of fishing activities is assumed to increase because of improvements in the water quality.

Figure 2 illustrates the implicit logic underlying a simple benefit transfer approach. D_0 describes the pre-policy demand for fishing and D_1 the post-policy demand, assuming that the change in water quality leads to a parallel shift in the demand function. The benefits from a quality improvement that shifts the demand from D_0 to D_1 would be DFCG, treating OE as the average travel cost to use the site for fishing.

In this algebraic example, it is assumed that $CS_T/\Delta d_T$ is a "perfect match" with conditions at the improved site (i.e., *after* the policy change) so that the unit

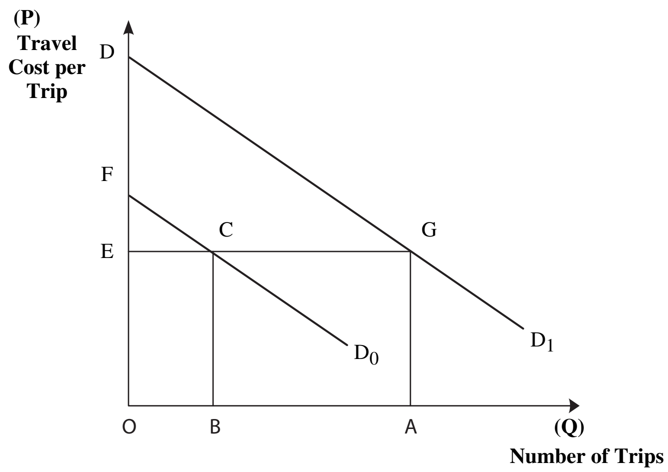

Figure 2. Quality as a Quantity Change

values would be estimates of DEG/OA, consumer surplus per trip for the desired fishing experience.[6] The benefit measure implied when equation (1) is adapted to approximate the value of quality change that has been linked to use $\tilde{C}S_P$ is then given by equation (6).

$$\text{(6)} \qquad \tilde{C}S_P = \left(\frac{DEG}{OA}\right) \bullet BA$$

BA is the postulated increase in fishing trips taken because of the water quality improvement. In contrast to this measure, the appropriate estimate of the value of the water quality improvement would be DFCG. We can use the geometry from Figure 2 to illustrate the extent of the mistake. The logic used in this transfer assumes OB is a constant multiple, α, of the activities currently observed:

$$\text{(7)} \qquad OB = \alpha \bullet OA$$

The "correct" benefit measure, given our assumptions, is DFCG = DEG − FEC, with quality leading to a parallel shift in FD_0 to DD_1. We can simplify matters using the following relationships for areas of the two triangles:

$$\text{(8)} \qquad DEG == \frac{1}{2} DE \bullet OA$$

$$\text{(9)} \qquad FEC == \frac{1}{2} DE \bullet OB = \frac{1}{2}(\alpha\ DE)(\alpha\ OA)$$

Simplifying the expression for DFCG, we have equation (10) describing the desired benefit measure:

$$\text{(10)} \qquad DFCG == \frac{1}{2} DE \bullet OA(1 - \alpha^2)$$

The expression given in equation (6) for the usual benefit transfer method can be expressed in terms of DE, OA, and α as

$$(11) \qquad CS_P = \left(\frac{DEG}{OA}\right) \bullet BA = \left(\frac{\frac{1}{2}DE \cdot OA}{OA}\right) \bullet (1-\alpha)OA = \frac{1}{2}DE \bullet OA(1-\alpha)$$

This geometry implies we have a relationship between the "correct" benefit measure and the simple approximation. The ratio of equation (10) to (11) suggests that the correct measure is $(1+\alpha)$ times the approximation. The approximation in equation (6) would therefore only be correct if the quantity measure is assumed to be zero with the pre-policy water quality conditions.

Of course, this example is just one possible way water quality could change the demand for recreation. Nonetheless it illustrates why the details that are often not made explicit in deriving the components of equation (1) from existing estimates are important for assessing the reliability of results from simple benefit transfers. Adjusting unit values to reflect different demographic characteristics (as might be done with a typical benefit functions transfer approach) would not address this issue because assumptions regarding how environmental quality relates to the observable quantity measure (in our example days or trips using the river for different types of recreation) would still not be made explicit.[7]

Thus, this graphical example illustrates two important problems with the current practices of benefit transfer. The first arises because the procedures are not consistently linked to the concept we wish to measure. The Harberger formulations for approximating the consumer surplus attributed to price changes are developed within a consistent economic framework. In contrast, conventional benefit transfer practices are not derived from a framework that links the quality change to be valued to the quantity of use associated with the quality change and to the economic values that people would place on the change. Morey's [1994] critique of using consumer surplus per trip in recreation applications is an example of this larger problem.

The second problem is much less apparent from most of the applications of benefit transfer techniques. Nothing in the methods assures the measures of WTP will be consistently related to household income. In other words, the transfers do not necessarily incorporate the restrictions implied by "ability to pay." For the most part, virtual prices (marginal WTP) are treated as constants, regardless of the scale of the changes being evaluated. With small, localized changes, the income effects may not be large. In other cases, such as Costanza et al.'s [1997] effort to measure the annual value of the earth's ecosystems and EPA's [1997]] retrospective analysis of the net benefits of the Clean Air Act, the large scope of the changes evaluated raises serious questions about the economic consistency of the results.[8]

Harberger's approach anticipated the potential problems associated with evaluating such large-scale changes. His approximation for price changes (equation (3)) can also be interpreted (when more than one good's price is changing) as building in an assumption that total expenditures on the commodities affected by the price changes do not themselves change. As Hines [1999] has suggested, Harberger's alternative to ordinary and compensated demands was an effort to form a simple general equilibrium demand function that recognized the importance of the income effects for large policy-induced changes in prices. He sought to evaluate policies after accounting for the income effects of the transfers that can arise from policy. In his applications the issue was with the disposition of the tax revenues. For our applications, where improvements in previously unpriced goods are to be paid for, the central issue underlying this type of closure condition is where will the money come from?[9]

4. A PREFERENCE CALIBRATION PROPOSAL

Preference calibration refers to a specific logic that specifies a preference function (usually an indirect utility function) and then defines each of the available benefit measures in terms of this function. The challenge of calibration is to determine whether the available information is sufficient to identify the parameters of the preference function and derive numerically calibrated values for them. The most direct practical insight from the approach is a requirement that each source of benefit estimates and each desired decomposition of these estimates should, in principle, link to a common specification for individual preferences. This type of overall framework describes how the environmental resources and their quality contribute to individual well-being. Moreover, it also summarizes how other changes in an individual's (or a household's) circumstances might change their economic valuation of the resource change.

In practice, this means that the analyst must first be willing to make explicit assumptions about the functional form of an individual's utility function (or indirect utility function). If V represents the maximum level of utility achievable, given the income, relative prices for marked goods (P), and Q level of environmental quality faced by the individual, κ represent parameters describing the "shape" of the function. WTP for a change in environmental quality can therefore be expressed as the reduction in income that would exactly offset the improvement in Q (i.e., $Q_1 > Q_0$) and leave utility unchanged.

$$V(m, P, Q_0; \kappa) = V(m - WTP, Q_1; \kappa) \tag{12}$$

Assumptions about the functional form of utility allow the analyst to express WTP as a function of the change in environmental quality, income, prices, and κ as in equation (13).

$$WTP = f(Q_1, Q_0, m, P; \kappa) \tag{13}$$

This function can be treated as a benefit transfer function. However, a key feature distinguishing it from statistical benefit functions is that it is derived from, and thus consistent with, the specification of preferences.

The second element of this approach is that it uses these existing benefit measures to estimate the parameters in κ. That is, the measured values for WTP and/or

Marshallian consumer surplus are used to "calibrate" the specified preference structure. The WTP function can then be derived from the preference function.

The process described above summarizes the basic logic of preference calibration as a strategy for benefit transfer. One way to offer more detail on the approach is through an illustration. Thus, we demonstrate the process for a specific example and contrast it with more traditional benefit transfer practices.

5. AN APPLICATION TO WATER QUALITY IN THE CHESAPEAKE BAY

To illustrate the logic we selected a contingent valuation study and a recreation demand analysis for water quality changes in the Chesapeake Bay area. The Chesapeake Bay application offers a situation with a well-defined water resource where large-scale changes in water quality are of policy interest and different nonmarket valuation methods have been applied previously to evaluate water quality-related benefits. The policy relevance of the Bay arises from the fact that its water quality has improved significantly over the last three decades because of Clean Water Act (CWA) initiatives. These changes have prompted EPA and federal and state agencies to consider evaluating the achievements of water pollution control policies in the region by comparing their benefits and costs (Owens and Morgan [2000]). Unfortunately, as described below, the regulatory processes giving rise to the cleanup did *not* include a systematic effort to collect the economic data over this time that would allow the evaluation to take place using observed behavioral responses to the improved conditions. Thus, because of the data limitations described below, this application of the benefits calibration approach to the Chesapeake Bay should be viewed primarily as an illustration of our methodology rather than as a rigorous assessment of CWA policies in the Bay.

To calibrate a preference function using the benefit estimates available in the literature, we propose six steps:

1. Specify a "representative" individual's preference function.
2. Specify information required to ensure that the parameters of this preference function can be identified from the estimates.
3. Define explicitly the relationships between the available benefits measures and the specified preference function.
4. Adapt the available information to assure cross-study compatibility.
5. Calibrate the preference function using these benefit measures.
6. Estimate benefits using calibrated WTP function.

5.1. Specify a "Representative" Preference Function

The first step in preference calibration requires selecting a function to describe the representative individual's preferences. This decision is itself a tradeoff. Complex functions may well capture a wider range of behavioral responses, but they will also be difficult to calibrate with limited data. Simple expressions may be so restrictive that they are inconsistent with findings in the existing literature. As in the case of the functions used in computable general equilibrium models, the forms that will ultimately gain acceptance (if our proposed strategy for transfer is adopted) will no doubt balance these types of considerations. For this illustration we selected a simple form with fairly transparent implications.

To introduce quality in a simple way that is linked to the use of a resource, we follow Willig [1978] and Hanemann [1984] and adopt a preference specification that is consistent with what Willig labels "cross-product repackaging." This implies that the indirect utility function is structured so that the water quality measure reduces the effective "price" of using the recreation site, as in equation (14).

$$V = [(P - h(d))^{-\alpha} m]^{b} \tag{14}$$

P is the round-trip travel costs, d is the water quality measure, m is household income, and α and b are parameters. h(d) is a function that describes how increases in water quality reduce the effective price of a trip.

5.2. Specify Information to Identify Preference Parameter

Consider the task of estimating the recreational fishing benefits from water quality improvements with two sources of benefit information for the Chesapeake. One source uses a contingent valuation (CV) estimate that includes all possible uses in the Bay, and the second focuses on recreational fishing. The specific CV study we selected to illustrate this example is Lindsey et al. 1989 survey (reported in Lindsey, Paterson, and Luger [1995]). This survey sought to estimate people's WTP to undertake storm water control programs to help achieve Chesapeake Bay nutrient-reduction objectives.

The second study by Bockstael et al. [1989] relies on a travel cost model, which has two components. One model links water quality, measured using nitrogen and phosphorous loadings, to striped bass catch levels in the Chesapeake Bay. These catch models were then linked to Maryland fisher's demand for fishing trips during 1980. The authors report the average consumer surplus for a 20% improvement in nitrogen and phosphorous loadings in the Chesapeake Bay.

We selected the Lindsey et al. estimate for a fourpercent improvement from fishable conditions (i.e., conditions suitable to support game fish). The estimates from the Bockstael study can be adapted to consider an approximately equal change in water quality. Therefore, the two studies can be interpreted as providing comparable water quality improvements in ways that are relevant to users. Bockstael et al. [1989] measure the Marshallian consumer surplus based on fishing trips, and Lindsey et al. estimates the total Hicksian WTP for the water quality improvement.

5.3. Define the Relationships between the Available Benefit Measures and the Specified Preference Function

To calibrate the preferences with the empirical record in the literature, each benefit measure must be related to this common preference structure. Using Roy's identity, the demand for trips, X_1, can be expressed as equation (15).[10]

$$X_1 = -\frac{V_p}{V_m} = \frac{\alpha m}{(P - h(d))} \tag{15}$$

The Marshallian consumer surplus, MCS, associated with access to the recreation sites providing these fishing opportunities at travel costs corresponding to P_0 can be found from the area under this demand between P_0 and the choke price (labeled here as P_C). This is given in equation (16).

$$(16) \qquad MCS = \alpha m \int_{P_0}^{P_c} \frac{1}{(P - h(d))} dP = [\alpha m \ln(P - h(d))]_{P_0}^{P_c}$$

Evaluating the integral yields equation (17).

$$(17) \qquad MCS = \alpha m \, [\ln(P_C - h(d)) - \ln(P_0 - h(d))]$$

The Bockstael et al. [1989] analysis implicitly evaluates how MCS changes with d. This relationship is described, for this preference specification, with equation (18).

$$(18) \qquad \frac{\partial MCS}{\partial d} = \alpha m \left[-\frac{h'(d)}{(P_C - h(d))} + \frac{h'(d)}{(P_0 - h(d))} \right]$$

where $h'(d) = dh/dd$.

Simplifying terms, the first term is the product of demand for fishing trips at the choke price (P_C) multiplied by ($-h'(d)$) and the second is the demand at P_0 multiplied by $h'(d)$. The definition of the choke price (even if it cannot be expressed in closed form) implies the first of the terms on the right side of equation (15) should be zero. The second term offers one approach to linking Bockstael et al.'s [1989] measures to our preference specification, but first we must specify h(d). For our example, assume h(d) follows a logarithmic function with a declining marginal effect of d on the price: $h(d) = \beta \ln(d)$, where β is assumed to be constant.[11] More specifically, the increase in Marshallian consumer surplus per fishing trip is exactly $\beta/d(= h'(d))$, as in equation (19).

$$(19) \qquad \frac{\dfrac{\partial MCS}{\partial d}}{\dfrac{\alpha m}{(P_0 - h(d))}} = \frac{\dfrac{\partial MCS}{\partial d}}{X_1} = h'(d) = \frac{\beta}{d}$$

Bockstael et al.'s [1989] consumer surplus estimates of quality improvements, scaled by their estimated number of trips and the ratio of water quality changes in the two studies, offers an estimate of the left side of equation (16). This is the effect of a quality adjustment on incremental consumer surplus per trip, h′(d). We interpret it as the Marshallian surplus estimate for the water quality change as described by Bockstael et al. (i.e., improving water quality by four percent in the Chesapeake Bay). Thus, their estimate allows us to develop a calibrated estimate for β.

The Lindsey et al. study uses his CV question to provide an estimate of the WTP for a four percent improvement from fishable water quality (d_F). Specifying this question using the indirect utility function we have WTP defined by the indifference condition in equation (20a).

$$[(P - h(1.04d_F))^{\alpha}(m - WTP)]^{b} = [(P - h(d_F))^{\alpha} m]^{b} \tag{20a}$$

Simplifying the equations and solving for WTP we have equation (20b). This relation is the WTP for a four percent change in water quality from a baseline of fishable conditions that is implied by our preference function.

$$WTP = m - \left(\frac{P - h(1.04^{*}d_F)}{P - h(d_F)}\right)^{\alpha} m \tag{20b}$$

Values from the Lindsey et al. study for WTP can be combined with values for m, P, and $h(d_F)$ to calibrate the remaining parameter, α, as described below.

5.4. Adapt the Available Information to Assure Cross-Study Compatibility

With our benefit measures defined within a consistent economic framework, the next task is to convert the data relevant to each study's estimates in compatible units. We begin by adjusting the Bockstael et al.'s [1989] per trip consumer surplus estimates of the gain due to water quality improvements. The estimates of travel cost reflect 1980 dollars and a time cost corresponding to income levels that are substantially lower than what was observed with the Lindsey et al. study. Therefore, the travel cost should be adjusted to reflect a higher opportunity cost of time.[12] Second, the adjusted measures of travel cost and per-trip-consumer surplus ($2) need to be placed in comparable dollars with Lindsey et al., reflecting the effect of the overall price level. This is consistent with an implicit restriction in the preference function—homogeneity of degree zero in *all* prices and income. The consumer price index (CPI) is used to adjust monetary measures of prices, incomes, and consumer surplus from 1980 to 1998 dollars. We make a similar adjustment to the WTP amount of $42 in Lindsey et al., converting 1989 to 1998 dollars.

Finally, water quality, d, is characterized so that it is consistent between the two studies. The RFF water quality ladder and index (Vaughn [1981]) is used to establish this correspondence in the water quality measures.[13] The Bockstael et al. study measures water quality in terms of nitrogen and phosphorous. Recognizing that the Bay waters must be of fishable condition for the Maryland recreationists to be fishing for bass, we use the water quality ladder to assign this condition a water quality index value of 5.1. Similarly, we represent the baseline water quality in the Lindsey et al. study—again of fishable conditions—using a water quality index value of 5.1. Thus, the four4 percent improvement in fishable water quality described in Lindsey et al. study corresponds to improving the index value from 5.1 to 5.3, and the 20% improvement described in the Bockstael et al. study corresponds to improving the index from 5.1 to 6.1.

5.5. *Calibrate the Preference Function Using these Benefit Measures*

Equations(19) and (20b) allow us to calibrate the two unknown parameters—β, and α. Here we describe how these results can be reproduced. Specifically, from equation (19), we calibrate β to be

$$\hat{\beta} = \left(\frac{\frac{\partial \mathrm{MCS}}{\partial \mathrm{d}}}{\mathrm{X}_1} \right) \bullet \mathrm{d}. \tag{21}$$

We can take this calibrated value of β and use it in equation (20b), to calibrate α as

$$\hat{\alpha} = \frac{\mathrm{In}\left(\frac{\mathrm{m} - \mathrm{W\hat{T}P}}{\mathrm{m}} \right)}{\mathrm{In}\left(\frac{\mathrm{P} - \hat{\beta}(1.04 \bullet \mathrm{d_F})}{\mathrm{P} - \hat{\beta}(\mathrm{d_F})} \right)}. \tag{22}$$

Using the estimates for travel cost, income, consumer surplus change, and baseline water quality from the Lindsey et al. and Bockstael et al. studies, we calculated the calibrated parameters as $\alpha = 0.29$ and $\beta = 10.33$.

5.6. *Estimate Policy Benefits in the Chesapeake Bay Using the Calibrated WTP Function*

To illustrate how the calibrated preference function can be used, consider the task of measuring the per household benefits of water quality improvements in the Chesapeake Bay as the result of CWA policies. An assessment of water quality models for the Chesapeake Bay indicates that the CWA improved bay-wide water quality by 60% (Owens and Morgan [2000]). Using two different reference points to characterize conditions without the CWA, the calibrated values from equations (21) and (22) are used in equation (20b) to estimate the per household benefits for

- a 60% improvement in water quality that *results* in fishable water quality (i.e., a change in the water quality index from 3.2 to 5.1), and
- a 60% improvement in water quality that *starts* from fishable water quality (i.e., a change in the water quality index from 5 to 8.2).

As reported in column 5 of Table 2, the calibrated per-household WTP estimates for the two improvements are \$627 and \$696 respectively.

For the sake of comparison, Table 2 also presents per household values that would be used in a simple, conventional benefit transfer: a travel cost-based value and a CV-based value. These values were estimated in two stages. First, per-unit (of the water quality index) values were estimated for each case (each expressed in 1998 dollars). Travel cost estimates were calculated by taking the adjusted per-trip consumer surplus estimate from Bockstael et al., multiplying it by the average

Table 2. Individual Benefit Estimates for a 60 percent Improvement in Chesapeake Bay Water Quality Using Simple Approximations and the Proposed Preference Calibration Approach[a] (1998 dollars)

Water Quality[b]		Approximation Using Travel Cost Study[c]	Approximation Using Contingent Valuation Study[d]	Calibrated WTP
Baseline	Final			
3.2	5.1	$213	$527	$627
5.1	8.2	$339	$840	$696

[a] Calibrated parameters are β=10.33, and α=.29. The travel cost information relies on Bockstael et al. [1989]. The CV estimate applies an adjustment of the Lindsey et al. estimate (reported in Lindsey, Paterson, and Luger [1995]). The consumer price index was used to convert into 1998 dollars.

[b] The numbers correspond to the RFF water quality ladder and index (boatable=2.5, fishable=5.1, and swimmable=7).

[c] The consumer surplus approximation is calculated by multiplying the per-trip per unit of dissolved oxygen consumer surplus estimate (from Bockstael et al. [1989]) by the size of the proposed policy change in water quality and by the average number of trips.

[d] The CV approximation is calculated by dividing the estimated WTP by the change in water quality (from Lindsey, Paterson, and Luger [1995]) for the original estimate and then multiplying by the size of the proposed policy change.

number of trips, and dividing it by the water quality increment evaluated in the study. The result is $111 per unit of water quality improvement. The CV values were calculated by dividing the Lindsey et al. WTP estimate by the water quality increment evaluated in his study. The result in this case is $275 per unit. In the second stage, each of these per unit values was multiplied by the 60% water quality improvements (1.9 and 3.1 index units, respectively). The per person value estimates from the travel cost and CV-based approaches are reported in Table 2 in columns 3 and 4 respectively.

The travel cost-based values can be interpreted as incomplete measures of the per household benefits of the water quality improvements (nonuse value, for example, are not captured in these values). The CV-based estimates would be expected to be more comprehensive. Thus, it should not be surprising that the CV-based approximations are closer than the travel cost-based approximations to the calibrated WTP estimates.[14] For the water quality change starting below the fishable level, the calibrated WTP value exceeds the CV-based value. The opposite is true above the fishable level. A further point that is not shown explicitly by the results in Table 2 should be noted. The differences between the calibrated and CV-based values increase in absolute value as one evaluates water quality changes further away from our point of calibration (in the 5.1 to 5.3 range). This distinction arises from the fact that the WTP function can be approximated with a linear function around the calibration point. The marginal welfare gains using the calibrated approach are larger at lower water quality levels (below fishable) and smaller at higher water quality level (above fishable). This pattern also is consistent with the logic of declining marginal WTP for improved water quality, but it would not be reflected in the linear estimates derived using the simpler benefit transfer approaches.

6. DISCUSSION

Conventional benefit transfer practices at best are approximations that have been developed to measure the consumer surplus associated with price changes. They have been used in ways that do not ensure they will be consistent with the economic concepts underlying the definition of WTP for quantity or quality changes. The larger the change being evaluated, the greater the likelihood of serious biases. Moreover, it is possible to find inconsistencies in smaller-scale, simple transfers. We illustrate this possibility with our graphical example in Section 3. Harberger approximations to control for income effects do not apply when the goods involved are not priced. To meet these shortcomings, we have proposed treating benefit transfer as a generalized estimation task—where the available information is linked to a preference function. Benefit estimates can be derived when the maintained assumptions and available empirical information together are sufficient to identify the preference parameter. This approach implies that consistent transfers require sufficient information to recover the parameters of a WTP function. This strategy was illustrated with an example application for water quality improvements in the Chesapeake Bay.

6.1. What Are the Main Advantages of this Proposed Alternative to Conventional Benefit Transfer?

The proposed approach offers a more systematic way to construct benefit measures under the time and resource constraints typically facing policy analysts. The primary argument for preference calibration is that it imposes economic consistency conditions on the ways the existing information is used. Experience and experiments (with computable general equilibrium models), comparable to what has taken place for nearly 50 years with Harberger approximations for the deadweight losses, provide the only basis for judging whether this strategy will be uniformly better. This conclusion does not imply we need to wait for 50 years of experience to consider revising current practices. While we wait for such numerical experiments, we can use preference calibration to judge whether imposing these types of consistency conditions would change conclusions.

The Chesapeake example highlights three underappreciated aspects of benefit transfers. First, the task associated with developing benefit estimates to evaluate a new policy should be interpreted as a type of *identification problem*. That is, when transferring benefits we must judge whether there is sufficient information to develop a *theoretically consistent* measure of the benefits for the changes being considered. This process makes explicit the roles of analyst judgment in developing the links between what has been measured and what is needed for each policy task. The analyst must also specify the structure of preferences in such a way that the critical preference parameters can be inferred from existing data and studies. There is also an important strategic element to selecting the functional form for preferences so that it is tractable. Second, benefit estimates assembled from studies that used different methods will often require that the same aspect of environmental quality be represented with different technical measures. Consistent use of these benefit estimates requires indexes of environmental quality that can be made compatible.

Differences in how this is accomplished may well be as important to discrepancies in transferred estimates as any distinctions in economic assumptions underlying those estimates.[15] Finally, the observed economic tradeoffs that people make to obtain increases in nonmarket resources are constrained by their available incomes. None of the existing approaches to benefit transfer meet this simple ability to pay test. That is, when transferring benefits, we must ensure that measured WTP values are affordable (i.e., well within people's disposable income).

6.2. How Can We Evaluate Benefit Transfers?

Most efforts to evaluate transfer methods have compared "direct estimates" of the benefits provided by some improvement in environmental quality in one location to a "transferred value." The latter is simply a different estimate. Random error alone would imply discrepancies. While sampling studies offer the prospect to control the standard used in evaluation, the assumptions required for describing preferences, true parameter values, characteristics of available data, etc., seem to offer so many combinations of alternatives that this also seems unlikely to offer many practical insights for evaluating transfer practices.

Because benefit measures are never observed, their estimates are unlikely to be evaluated in a context that will be fully satisfactory. That is, there is no "true benefit estimate" that could be found to serve as a measuring stick for the transferred estimates.

Preference calibration offers some advantages for evaluating benefit transfer. With a numerical characterization of the preference function, it is possible to consider estimating other observable "quantities" at the same time as the benefits are measured. For example, we could predict the number of recreation trips per household, the expenditure shares, and price and income elasticities. Such "indexes" may be easier to use in gauging the plausibility of a benefit function than a consumer surplus measure for an unobserved quality change. These types of estimates are not available with other transfer methods because they are not consistently linked to preferences. Large discrepancies between the predictions for the linked private good or elasticities that are judged to be implausible signal a need to evaluate the assumptions being used.[16]

6.3. What Next?

Clearly, what has been proposed here was done in the context of simple specifications to illustrate the logic of a different strategy for conducting benefit transfers. More complex functional forms are possible and numerical calibration analogous to what is used with numerical computable general equilibrium models is also possible. However, the desirability of pursuing such larger-scale efforts depends on the success of experimentation with smaller applications of the method and comparisons with current practice. It would be relatively easy to consider an exercise where recent benefit transfers were "redone" using the calibrated preference logic and compared with the approach used in the policy analysis. This would seem to offer a next step in evaluating the usefulness of the logic and could easily precede more extensive efforts at numerical calibration.

The possibilities for using the calibration logic do not stop with alternative spreadsheet computations under a wide array of judgments about which combination of point estimates to use to identify the preference parameters. Preference calibration also offers a strategy for using the existing literature as data to "estimate" the preference function used for benefit transfer. Instead of using reduced-form response functions in meta summaries, the logic of preference calibration implies that sufficient information can exist from the available literature to *identify* the parameters of preferences. When there are multiple sets of information, it is possible to treat the theoretical conditions to identify the parameters (e.g., marginal WTP from hedonic studies) as a system of equations that define moment conditions. That is, preference parameters could be estimated using the multiple sets of benefit measures as "data" in the form of moment conditions.

To summarize, mandates in the U.S., European Union, and in Development Agencies are increasingly calling for benefit-cost analyses to evaluate the performance of regulatory programs. In the absence of the time and resources needed for full studies of each new policy or a "non-market" general equilibrium model, analysts are likely to rely extensively on current transfer practices and off-the-shelf estimates. We have suggested that these practices pose real concerns. If responses to the mandates for economic information about policy tradeoffs are to avoid discrediting the practice of benefit cost analyses, they must recognize the need for imposing internal consistency measures of the gains (and losses) attributed to interrelated (from the consumer's perspective) but independently administered regulatory policies. Calibration offers a first step for avoiding inconsistent benefit estimates and for more completely accounting for the effects of large-scale policies.

Subhrendu Pattanayak and George Van Houtven are Senior Economists, RTI International, Research Triangle Park, NC, USA.. V. Kerry Smith is University Distinguished Professor, Department of Agricultural and Resource Economics, North Carolina State University, Raleigh, NC, USA and University Fellow, Resources for the Future, Washginton, DC, USA..

7. NOTES

[1] The basic approach for evaluation is to conduct benefit analyses at two or more sites and compare the transferred benefits (based on a function derived for Site 1) with actual benefits at the other site. See Table 1 for a list of recent studies in this category.

[2] These benefit functions usually come from one of two sources. The first is from CV studies. In these cases, the analyst uses a statistical summary of the design variations included in the primary CV study. The second type involves meta analysis summaries. In this case, the models are based on data that generally use summary statistics from the original sources and include characteristics of the resources, individuals (whose WTP is being estimated), and methods used. The three recent evaluations of benefit transfer have relied on the first type of benefit function and have considered benefit measures based on CV studies only.

[3] As we illustrate in the next section, this simplification can be misleading. The unit values derived from some approaches to nonmarket valuation are actually transformations of the consumer surplus attributed to price changes. In other situations, the model implicitly forms a specific "price equivalent" of a quality change so that the consumer surplus attributed to the quality change would have a price

change equivalent. For example, in the case of reductions in mortality risks, it is the *ex ante* marginal rate of substitution.

4 The benefit measure in the health context can be different from a consumer surplus gain.

5 The assumption implies Z and s are in a separable subfunction from the other arguments in the expenditure function.

6 By assuming the demand is known, our example ignores this source of error and focuses instead on the error introduced by what the analyst does in constructing a transferred benefit. This error arises from using the approximation for a price change and treating consumer surplus per unit as the equivalent of a price.

7 We could include environmental quality in the model and explicitly adjust for the responsiveness of demand to site quality.

8 For Costanza et al. the estimated global annual WTP for these ecosystem services exceeded the global gross domestic product. For the EPA report, the per capita benefits of air quality improvements relative to average household income strain credibility.

9 One could use a logic comparable to Harberger, requiring that the share of a fixed income attributed to the amenity was fixed before and after a changed (i.e., $\rho_z^0 Z_0 = \rho_z^1 Z_1$). $\rho_z^0 Z_0$ does not correspond to the expenditures on Z_0. It is an arbitrary concept that uses virtual prices to mimic what a consumer's expenditures would be under two different conditions. Using them, together with the conventional measure of incremental WTP for a change from Z_0 to Z_1, the Harberger restriction would imply that this measure should be approximated as $1/2(Z_1 + Z_2) \bullet (\rho_z^0 - \rho_z^1)$ with $\rho_z^0 \rangle \rho_z^1$.

10 Note that in the simplified case without the quality term $V = [P^{-\alpha}{}_m]^b$, $X_1 = \alpha\left(\frac{m}{p}\right)$, and $\alpha = \frac{PX_1}{m}$, the share spent on X_1. With cross-product repackaging, $\alpha = (P - h(d)) \bullet X_1/m$. For small values of h(d), the two measures will be close to each other.

11 The functional form for h(d) by itself is not critical, even though it represents another judgment by the analyst. We did consider an alternative form such as a power function with a declining marginal effect of d on the price and found different quantitative and qualitative solutions using $h(d) = d^{\square}$.

12 The adjustment to travel cost attempts to take account of differences in the opportunity cost of time as a result of differences in income between the Bockstael et al. [1989] and the Lindsey et al. studies. This is accomplished by scaling travel cost by an adjustment for CPI differences and the relative income from the two studies.

13 We do not claim that this water quality conversion is entirely accurate or ideal. It is used in our example as a simple and convenient approximation. It also highlights the general need for defensible conversion procedures and for consistent measures of water quality, which are essential to linking results from different benefit studies.

14 By design of the calibration approach, the calibrated WTP estimate and the CV-based WTP estimate are the same for the water quality increment evaluated in the CV study. In other words, both approaches generate a WTP estimat of $56 for the water quality index increment examined in the Lindsey et al. study (from 5.0 to 5.2).

15 This conclusion is supported by the recent Desvousges et al. [1998] meta analysis for environmental costing.

16 The logic resembles the use of calibration in marketing research where the results of stated preference or conjoint surveys are calibrated based on a variety of other types of information before they are then considered relevant for a market analysis task.

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