Benefit Transfer as Preference Calibration

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Abstract

This paper proposes and illustrates the use of a new approach to benefit transfer for the non-market valuation of environmental resources. It treats transfer as an identification problem that requires assessing whether available benefit estimates permit the parameters of a preference function to be identified. The transfer method proposed uses these identifying restrictions to calibrate preference parameters and bases the benefit estimates on that preference function. The approach is illustrated using travel cost, hedonic and contingent valuation estimates, as well as combinations of estimates. It has three potential advantages over conventional practice: (1) it allows multiple, potentially overlapping estimates of the benefits of an improvement in environmental quality to be combined consistently; (2) it assures the transferred estimates of the benefits attributed to a proposed change can never exceed income; and (3) it provides a set of additional "outputs" that offer plausibility checks of the benefit transfers.

Key Words: benefit transfer, calibration, non-market valuation

JEL Classification Numbers: D61, Q20, H40

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I. INTRODUCTION

Most uses of applied welfare analysis for environmental resources, whether benefit-cost calculations or natural resource damage assessments, rely on adaptations of existing benefit estimates rather than new research for their evaluation of policies or injuries affecting these resources. Almost ten years ago, David Brookshire [1992] organized a set of papers in *Water Resources Research* to focus attention on the practice of benefits transfer. Since then, there has been growing interest in research on the potential for improvement in these techniques, but little methodological progress. The best overall summary of what has been learned about these methods over the past decade is reflected in a common conclusion reached by three recent and independent evaluations--*conventional benefit transfers are very unreliable!*[1] The procedures used in these adaptations of existing research generally seek to measure the benefits from a quantity or a quality increase. However the framework used in most cases arises from an approximation introduced for measuring the consumer surplus associated with price changes. To our knowledge, there has not been an attempt to develop an alternative method.

This paper proposes a new approach to benefit transfer and illustrates how it would work using three empirical examples. Rather than computing a unit value or constructing a statistical function describing how unit values change with the economic or demographic variables associated with the samples used for existing benefit studies, the same existing research can calibrate a specific preference function. The required benefit measures can then be computed from that function. This strategy has three advantages over conventional methods: (1) it allows multiple, potentially overlapping estimates of the benefits of an improvement in environmental quality to be combined consistently; (2) it assures the transferred estimates of the benefits attributed to a proposed change can never exceed income; and (3) it provides a set of additional "outputs" that offer plausibility checks of the benefit transfers.

After describing the steps that usually comprise a conventional benefit transfer and their relationship to Hicks-Harberger approximations for consumer surplus in Section II, we

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[1] The three studies are Loomis et al. [1995], Downing and Ozuma [1996], and Kirchoff, Colby and LaFrance [1997].

outline our preference calibration approach using three examples in Section III. This section closes by discussing some potential shortcomings with our proposal and suggesting how it could be used to modify meta analyses of benefit studies. The last section provides some policy context for our proposal and argues that applied benefit analyses for large scale policy changes, based on some type of transfer, must be restricted to assure consistency with the economic properties of a willingness to pay function if the results are to be taken seriously.

II. BACKGROUND

Benefit transfer is the practice of adapting 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 transferring them to a different context that is relevant for a policy being evaluated. 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 utilize 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 (marginal) WTP estimate for a unit change in lake water quality might be inferred from a hedonic property value 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 lake 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] With this second approach, the entire equation (function) is transferred to the policy context and the benefit estimate is then "tailored" to conform as closely as possible to meet the new population's characteristics as well as to reflect the affected 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.

[2] These benefit functions usually come from one of two sources. The first is from contingent valuation (CV) studies. In these cases, the analyst uses a statistical summary of the design variations included in the primary CV study. These models are derived from the primary data and are reduced form equations describing how changes in the CV choice questions, object of choice, or respondents influenced the estimates of the benefits. The second type involves meta analysis summaries. In this case, the models are based on data that generally uses summary statistics from the original sources and includes 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. For a discussion of the use of meta analytic summaries, see von den Bergh [forthcoming].

Benefit transfers usually proceed in four steps:

1. Translate the policy change into one or more quantity changes for the uses linked to an environmental resource that are permitted because of the policy change for the typical user.[3]
2. Estimate the number of typical users before and after the policy change.
3. Transfer a per "unit" consumer surplus measure, with the unit measure comparable to the index used 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.[4]

When the process is described in this way it resembles the methods used to approximate willingness to pay measures for price changes. Equation (1) translates the steps to a simple relationship describing the typical benefit transfer.

$$CS_P = \frac{CS_T}{\Delta d_T}\left(d_1 \cdot N_1 - d_0 \cdot N_0\right) \tag{1}$$

where

CS_P = estimate of consumer surplus for policy being evaluated.

d_i = the amount of use permitted by policy change (i = 1) and in absence of the policy change (i = 0) by a typical user (e.g. visits per year).

N_i = the number of people engaged in the use with policy change (i = 1) and without (i = 0)

CS_T = consumer surplus gain (for a representative individual) measured in other literature for a change (or set of changes) judged to be comparable to how policy affects d

Δd_T = change presented in existing literature for the measurement of CS_T

The distinction between the use of *benefit value* or *benefit function* arises from what is substituted in equation (1) for ($CS_T/\Delta d_T$). This approach focuses the analysis at the individual

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[3] As we illustrate in the next section, this simplification can be misleading. The unit values derived from some approaches to non-market 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. At a general level, this connection forms the basis for all the revealed preference approaches to recovering Hicksian welfare measures--see Bockstael and McConnell [1993] and more recently Ebert [1998] for further discussion.

[4] Often the estimates are included in discounted measures of aggregate benefits less aggregate costs. We ignore this step in the discussion that follows.

level and will be influenced by the characterization of "*d*." In the recreation context, the measure for d is usually a trip or day. However, in the health context it could be an episode of illness avoided or a reduction in the risk of some acute condition (e.g. asthma). The selection of a measure in the health example alters the relevant benefit concept used for the numerator (CS_T). As a rule the unit value is treated as a constant, regardless of the size of the change experienced by each individual.

CS_P can also be expressed for a quality change but it may not be measured in numerical units that are linked to uses of the resource. In this case the denominator in (1) is replaced by a measure of the amount of use (d_T) rather than the change in quality. This approach requires a one-to-one correspondence in the measurement of quality adjusted quantity. It would be used if the quality change is treated as being experienced over a number of trips (and users). For such applications, d_i and N_i would be reinterpreted as trips (or days) per user and the number of users respectively, the $d_0 \cdot N_0$ term in equation (1) would not be relevant because we would want to estimate the gains experienced by all users under the new quality regime.

This distinction can be subtle for some applications. It arises because CS_T is being reinterpreted as the consumer surplus for a quality change experienced by the "typical" user. d_T is the amount of use per person observed in the study experiencing the improvement. $d_1 \cdot N_1$ is the amount of use that the analysts anticipate would take place with the improved conditions.

Each adaptation to the basic format such as this one changes the mix of assumptions required to interpret the results as consistent with the basic benefit concepts. Such adaptations seem likely to also affect the performance of the approach as an estimator for the benefits associated with the change. However, none of the adjustments is directly linked to the underlying concept of willingness to pay that we would like to measure. All of the transfer strategies appear to start from a Hicksian approximation for consumer surplus associated with price changes. Equations (2) and (3) provide the relationships for the compensating variation (CS_C) and equivalent variation (CS_E) measures for price changes in a single commodity from P_0 to P_1 (with q_0 and q_1 the corresponding quantities demanded).[5]

$$CS_C = P_1(q_1 - q_0) \tag{2}$$

$$CS_E = P_0(q_1 - q_0) \tag{3}$$

Comparing equation (1) to equations (2) and (3), our formulation of the typical benefit transfer method, seems to confirm this conclusion. The logic for transfer provides a direct parallel to the logic originally outlined by Hicks [1942]. When Hicks defined compensating and equivalent variation measures for consumer surplus it appeared that the information

[5] Diewert [1992] outlines the link between the Hicksian definitions and first-order Taylor series approximations as underlying the Hicksian logic.

required to implement these concepts would not be available. Thus, he likely promotes these measures as first order approximations to the original definitions for a price change.[6]

Another approximation commonly used in tax applications is Harberger's [1971] method for measuring deadweight loss, given in equation (4) as CS_H. Harberger's proposal might seem to be an average of the CS_C and CS_E. Actually the logic underlying his strategy is more complex. He assumes that there is no change in the total expenditures on the commodities experiencing the price change.[7]

$$CS_H = \frac{1}{2}(P_0 + P_1)(q_1 - q_0) \tag{4}$$

Conventional benefit transfers assume that the policy reduces bad outcomes (e.g. avoids premature deaths, reduces acute and chronic morbidity effects, reduces maintenance required by accelerated corrosion or soiling, etc.) or increases the capacity to support desirable activities (e.g. increases a lake or river's ability to support fishing or swimming). In this case the unit value serves as a "price" for the specific outcome assumed to change with the policy. As we noted, this parallel extends to the benefit transfers used in cases where policies reduce risks of mortality or morbidity effects. For them, reductions in pollution may reduce the risk of premature death for specific populations. The risk of death at an exposure level is akin to the amount of use at each quality level (d_i) in equation (1) and the population experiencing the change is comparable to the numbers of people (N_i) engaged in the activity. While the value of a statistical life is a simple transformation of an *ex ante* marginal rate of substitution, MRS, and therefore variable with the size of the risk, it is nonetheless usually treated as a constant like a "price."

Thus, the benefit values are used as if they were virtual prices (i.e. marginal willingness to pay measures) for the quantity or quality change associated with the policy. Moreover, to the extent the unit value is defined in terms of uses allowed (i.e. fishing days) or impacts avoided (i.e. avoided sick days), then there are also implicit assumptions being made about the relationship between the environmental quality of interest and some observable quantity. To see how this is different from a first order approximation for the value of a quantity or quality change, consider a simple version of a Hicksian expenditure function, $e(.)$, with priced goods, (and $\overline{P}$ a vector of prices) one non-priced 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 (5) with U_0, the initial utility level, and Z_0 the level of the non-marketed good:

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6 The concept of consumer surplus as originally defined by Marshall was controversial for over fifty years. Hicks sought to rehabilitate the idea but proposed measures that many economists felt could not be measured. This set of maintained beliefs changed in the seventies with the work of Willig [1976], Burtless and Hausman [1978], Hanemann [1978], and Hausman [1981].

7 Assuming no change in incomes as a result of the price change, equation (4) follows from Harberger's [1971] equation (5′).

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

We expect this difference to be positive. *Z* is a desired 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 and yet attain the same level of well being. This interpretation implies individuals require less to realize the initial level of utility.

Using the virtual price of *Z*, r_z, to approximate a parametric price, the first order approximation for consumer surplus, $\tilde{CS}_C$, for a change from s_0 to s_1 might be written as (5b):[8]

$$\tilde{CS}_C = r_z \cdot (Z(s_0) - Z(s_1)) \tag{5b}$$

$r_z = \frac{\partial e}{\partial z}$ and must be 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 r_Z and Z, will be functions of *s*. As a result, they can be expected to be interrelated through e(.). This outcome contrasts with approximations intended to measure the value of changes in price where we can assume prices are constant and exogenous.

To illustrate how this distinction can be important, consider a simple example that involved a recent EPA commissioned benefit analysis for the benefits from water quality improvements attributed to the Clean Water Act. This analysis is comparable to the types of benefit analyses routinely conducted for Regulatory Impact Analyses. Table 1 summarizes the background assumptions and data used in the benefit transfer. In this case the $d_i \cdot N_i$ are reported as a single aggregate for each type of use--fishing and swimming. The capacity to support different types of fishing activities is assumed to increase due to improvements in the water quality. For swimming the pre-CWA use was assumed to be zero due to bans on swimming in the Williamette River prior to 1972. Figure 1 illustrates the implicit logic underlying the estimates developed for water quality induced increases in fishing. We can use it to explain the difficulties in using Hicksian price-based approximations to describe the benefits due to the quality and quantity changes used in this analysis. D_0 describes the pre-CWA demand for fishing and D_1 the post-CWA demand. Assume 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.

[8] The assumption implies *Z* and *s* are in a separable subfunction from the other arguments in the expenditure function.

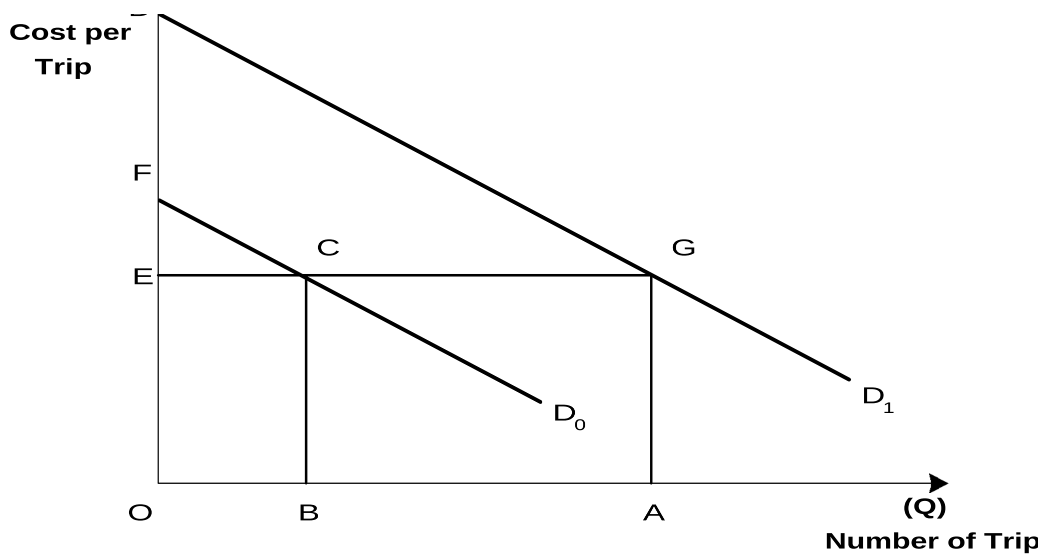
Table 1. Benefit Transfer for the Willamette River Basin[a]

Example Activity	Activity Measures[b]: Without CWA	Activity Measures[b]: With CWA	Unit Value (1995$)	Sources (Location/Author)
Recreational Fishing				
Salmon (trips)	21,302	213,019	$133.70 per trip	Columbia River Oregon and Washington Olsen et al. (1991)
			$86.50 per trip	Oregon Rowe et al. (1985)
Trout (days)	100,218	1,002,182	$31.80 per day	Oregon and Washington McCollum et al. (1990)
			$21.38 per day	Oregon Brown and Hay (1987)
Warmwater (days)	24,207	242,069	$30.47 per day	U.S. Walsh, Johnson, and McKean (1992)
			$16.22 per day	U.S. Bergstrom and Cordell (1991)
Direct Water Contact Recreation				
Swimming	0	1,001,859	$19-$30 per day	Not given
Water skiing	0	244,197	$35-$41 per day	Not given

[a]This material is a partial summary from Tables 5-6 and 5-7 in Bingham et al. (1997).

[b]This designation implies the assumed level of use with and without the regulations associated with the Clean Water Act (CWA).

Figure 1: Quality as a Quantity Change



In our algebraic example, assume 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 values would be estimates of *DEG/OA*, consumer surplus per trip for the desired fishing experience.[9] 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):

$$\tilde{C}S_P = \left(\frac{DEG}{OA}\right)\cdot BA \tag{6}$$

BA is the postulated increased 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 seek *DFCG*. We can use some geometry and the results from Table 1 to illustrate the extent of the mistake. The logic used in this transfer assumes *OB* is a constant multiple of the activities currently observed. In this case, it is 10 percent. Therefore assume:

$$OB = \alpha \cdot OA$$

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.

$$DEG == \frac{1}{2} DE \cdot OA \tag{7}$$

$$FEC == \frac{1}{2} FE \cdot OB = \frac{1}{2}(\alpha DE)(\alpha OA) \tag{8}$$

Simplifying the expression for *DFCG*, we have equation (9) describing the desired benefit measure:

$$DFCG == \frac{1}{2} DE \cdot OA(1-\alpha^2) \tag{9}$$

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

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[9] 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.

$$CS_P = \left(\frac{DEG}{OA}\right) \cdot BA = \left(\frac{\frac{1}{2} DE \cdot OA}{OA}\right) \cdot (1 - a)OA = \frac{1}{2} DE \cdot OA(1 - a) \qquad (10)$$

This geometry implies we have a relationship between the "correct" benefit measure and the simple approximation. The ratio of equation (9) to (10), suggests that the correct measure is $(1 + a)$ times the approximation. In terms of the Williamette study, 10 percent larger (i.e., 1.10 times the estimate reported). As noted earlier, this approximation relies on DD_1 being the correct demand. For the case of swimming activities involving water contact (provided again DD_1 is the correct demand) the approximation in equation (6) is correct because the quantity measure is assumed to be zero with the pre-CWA 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 buried in deriving the components of equation (1) from existing estimates are important to the reliability of the results from simple benefit transfers. Improving the adjustments to unit values for demographic variables (with the *benefit functions* approach to transfer) does not address this issue because these functions do not reflect the assumptions being made about how environmental quality relates to the observable quantity measure (in our example days or trips using the river for different types of recreation).[10]

The difficulties arise because the approximations treat each task as independent. The estimates used to "fill in" relationships comparable to equation (1) are derived from studies that attempt to measure consumer surplus for a specific change. The analyst must adapt them as well as the description of the change to recover plausible units and an associated unit value. Each study can involve a different set of these adjustments with the quantity/quality/marginal willingness to pay interrelationships. Our point is that such adjustments must be coordinated within a single framework. The coordination for policies assumed to affect market prices takes place because the affected commodity and its appropriate price are matched by the market. For environmental quality (and quantity) there is no market. Moreover, there is also another level of complexity in applications of benefit transfers to environmental applications. It stems from the need to "match" the physical/environmental changes derived from specific policies to the benefit measures available.

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. Unlike the Hicks/Harberger formulations for approximating the consumer surplus attributed to price changes, these procedures are not derived from a framework that establishes the connections between the quality change to be valued, the quantity of use associated with the new and old quality, and the economic values

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

(total and marginal) 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. There is nothing in the methods that assures the measures of willingness to pay will be consistently related to household income (i.e. the transfers do not necessarily incorporate the restrictions implied by "ability to pay"). For the most part, virtual prices (marginal willingness to pay) 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 others, such as the Costanza et al. [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 results do not satisfy simple comparisons for economic consistency. For Costanza et al. the estimated global annual willingness to pay for these ecosystem services exceeded the global gross domestic product. For the EPA report, the change was treated as creating an asset whose impact on the average household income was implausibly large. The changes being evaluated in each of these studies were so large that these income/willingness to pay relationships became extremely important.

Harberger's approach anticipated this possibility. His approximation for price changes (equation (4)) builds in an assumption that total expenditures on the commodities affected by the price change do not themselves change. As Hines [1998] 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.[11] He sought to evaluate policies after accounting for the

[11] One approach to characterizing the Harberger demand would be to assume utility is maximized subject to two constraints--the budget exhaustion condition as well as a restriction that a subset of the expenditures are held constant as a group. In the Harberger case this constant would be equal to the initial expenditures on this same group. In our analysis this is designated as "k." This framework implies that Harberger demand would be derived from an indirect utility function with a modified form of Roy's identity as:

$$X_i = \frac{-V_{p_i}}{V_m + V_k}$$

where V_j= partial derivative of indirect utility with respect to argument *j*.

We know that $V_{p_i} < 0$, V_m>0 and V_k=0. Rearranging terms in the definition of X_i and then integrating over the path(s) of change in one price, p_i , we have the usual expression for the change in utility attributed to the price change:

$$\Delta V = -\int_s V_{p_i} dp_i = \int_s (V_m + V_k) \cdot X_i dp_i$$

Combining V_m and V_k in this way offers a basis for understanding the Harberger strategy as a method to reduce the effects of assuming a constant marginal utility of income (V_m) on the Marshallian consumer surplus (MCS) as a measure of ΔV . This term was a central focus of most of the early discussions of the properties of the MCS.

One way to see what Hines attributes to Harberger's reasoning is to consider the change in the composite V_m + V_k with respect to p_i. The terms can change in opposite ways for normal goods, i.e. $V_{mp_i} < 0$ and $V_{kp_i} > 0$.

Thus, it is conceivable that we would reduce the sensitivity of consumer surplus measures along a Harberger 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?[12]

The proposal developed in the rest of this paper is only one way to impose consistency conditions comparable to what underlies the approximation used for price changes. Nonetheless, it does illustrate the importance of basing benefit transfer on theoretically consistent approximations.

III. PREFERENCE CALIBRATION

The logic associated with preference calibration is straightforward. (a) specify a "representative" individual's preference function; (b) derive a relationship between the available information on what has been measured in terms of that preference function; (c) include additional information (or preference restrictions) to permit the identification of the preference parameters; (d) calibrate the preference function using the baseline conditions of the study (or studies) in the literature; (e) construct the required conditioning variables to use the calibrated preference function for the policy application;[13] and (f) calculate the Hicksian consumer surplus measure from the adjusted, calibrated preference functions as well as a set of estimates of other response outcomes (e.g. implied price elasticities of demand, levels of use, etc.) to gauge the plausibility of the transfer.

This process is certainly not new. A large part of the logic is inherent in all revealed preference approaches used in estimating of the economic values of changes in environmental resources. What seems to have been overlooked in benefit transfers is the insight first suggested by Cameron [1992] to use specific preference restrictions to link contingent valuation estimates of environmental quality improvements to revealed preference measures for the same, or a closely selected value change, taking place for the same resources. Her analysis recognized that there may be several adjustment margins available to consumers to respond to environmental quality. As a result, with specific assumptions about preferences and constraints, it is possible to isolate restrictions linking the parameters estimated with the different revealed (and stated preference) methods. For Cameron, these parameter restrictions were implemented with joint

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demand function to the effects attributed to V_m changing with p_i by conditioning the surplus measure to hold constant expenditures on a group of commodities.

[12] 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. $r_z^0 Z_0 = r_z^1 Z_1$). $r_z^0 Z_0$ 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: $\frac{1}{2}(Z_1 + Z_2)\cdot(r_Z^0 - r_Z^1)$ with $r_Z^0 \rangle r_Z^1$.

[13] These conditioning variables arise in translating the policy into measures of quality characteristics that are in the calibrated preference function.

estimation of travel cost and contingent valuation models.[14] In our case, these links provide the identifying restrictions to allow calibration of the preference function.

To illustrate how this strategy works, we have focussed on developing benefit estimates for a variety of water quality changes using contingent valuation, travel cost, and hedonic estimates. This approach illustrates how preference calibrated benefit transfers could be implemented right now with existing information. The first example uses travel cost and contingent valuation (CV) results. The second uses CV and hedonic estimates. The last combines all three sources for benefit measures due to water quality changes.

A. Travel Cost and Contingent Valuation

Consider the task of estimating the recreational fishing benefits from water quality improvements with two sources of benefit information.[15] One source uses a contingent valuation estimate that includes all possible uses at all lakes, and the second focuses on recreational fishing and the effects of quality changes for the ability to catch fish. The specific contingent valuation study we selected to illustrate this example has been the "linchpin" of nearly all EPA's water quality benefit estimates--Mitchell and Carson's 1983 survey (reported in Mitchell and Carson [1984], [1986], Carson and Mitchell [1993]). This survey sought to estimate people's willingness to pay to undertake policies that would improve water quality at ninety-nine percent of the nation's waters.[16]

The second study by Englin et al. [1997] relies on a travel cost framework. It has two components. One model links water quality, measured using dissolved oxygen, to total trout catch in New England lakes. These catch models were then used in a second model that describes recreationists' demand for fishing trips. The authors report the average consumer surplus for improvements in dissolved oxygen for a set of lakes used by residents of New York (excluding New York City), New Hampshire, Vermont and Maine during 1989. The illustrative benefit computation in their paper involves an increase in water quality for the poorest lakes to a minimum dissolved oxygen level of 6.0 mg/liter.[17] This scenario is somewhat similar to the logic underlying the Mitchell-Carson CV question which asks about improving water quality in water bodies throughout the U.S.

We focus on the Mitchell-Carson estimates of improvements from boatable to fishable conditions (i.e. conditions suitable to support game fish). Based on the RFF water quality

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[14] There have now been several array of applications of this strategy such as Nikletschek and Lèon [1996], Eom and Smith [1994], and more recently, extensive use of revealed and stated choices as part of conjoint analysis with a RUM framework. See Adamowicz et al. [1997] as an example.

[15] This task would parallel the case studies commissioned by the Office of Water and reported in the benefit-cost analysis of municipal wastewater treatment for the Upper Mississippi and Potomac Rivers by Donlan, et al. [1995].

[16] Mitchell and Carson's question offers an improvement "where 99 percent or more of the freshwater bodies are clean enough so game fish like bass can live in them" [Mitchell and Carson, 1989, p. 385].

[17] They indicate that dissolved oxygen ranged from 0.88 to 11.94 mg/liter in their lakes with a mean of 3.4 mg/liter. 38 of the 61 lakes used in their sample had dissolved oxygen below 6.0.

ladder [Vaughan 1981], this change corresponds to improving dissolved oxygen from about 3.5 mg/liter to 6.0 mg/liter. This increment is approximately the change considered in the Englin et al. [1997] analysis. As a result we treat the two as representing comparable water quality changes for freshwater bodies relevant to users. Englin et al. [1997] measure the Marshallian consumer surplus based on fishing trips and Mitchell and Carson estimate the total Hicksian willingness to pay for the water quality improvement.[18] For this example, we will assume the policy to be evaluated involves improving dissolved oxygen (DO) levels in a set of specific lakes with the primary use of them being recreational fishing. The first step in preference calibration requires selection of 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 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 now to illustrate the approach we have selected simple forms 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 (11):

$$V = \left[\left(P - h(d_1)\right)^{-\alpha} m\right]^{b} \tag{11}$$

P is the round-trip travel costs, m is household income, and α and b are parameters.[19] $h(d)$ is a function that describes how increases in water quality reduce the effective price of a trip. Water quality, as noted earlier, is measured with dissolved oxygen, d. 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 (12):[20]

$$X_1 = -\frac{V_P}{V_m} = \frac{\alpha\, m}{\left(P - h(d)\right)} \tag{12}$$

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[18] Mitchell and Carson also distinguish the willingness to pay for improvement at local sites from quality changes at all sites. Later we illustrate how this might be used. In that case we use their total WTP estimate to calibrate preferences.

[19] α is used here in a different way than in section II. For the calibrated models that follow, it will be interpreted as approximately equal to the budget share for the recreational activity.

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

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.[21] This is given in equation (13):

$$MCS = \alpha\, m \int_{P_0}^{P_C} \frac{1}{(P - h(d))} dP = \alpha\, m \ln(P - h(d)) \Big]_{P_0}^{P_C} \tag{13}$$

Evaluating the integral yields equation (14)

$$\mathrm{MCS} = \alpha\, \mathrm{m}\ \left[\ln\left(\mathrm{P_C} - \mathrm{h}(\mathrm{d})\right) - \ln\left(\mathrm{P_0} - \mathrm{h}(\mathrm{d})\right)\right] \tag{14}$$

The Englin et al. [1997] analysis implicitly evaluates how *MCS* changes with *d*. This relationship is described, for this preference specification, with equation (15):

$$\frac{\partial\, \mathrm{MCS}}{\partial\, \mathrm{d}} = \alpha\, \mathrm{m}\left[-\frac{\mathrm{h}'(\mathrm{d})}{\left(\mathrm{P}_c - \mathrm{h}(\mathrm{d})\right)} + \frac{\mathrm{h}'(\mathrm{d})}{\left(\mathrm{P_0} - \mathrm{h}(\mathrm{d})\right)}\right] \tag{15}$$

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

Simplifying terms, the first term is the product of demand for angling trips at the choke price (P_C) times $(-h'(d))$ 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 Englin et al.'s [1997] measures to our preference specification. More specifically, the increase in Marshallian consumer surplus per angling trip is exactly h'(d) as in (16):

$$\frac{\dfrac{\partial\, \mathrm{MCS}}{\partial\, \mathrm{d}}}{\dfrac{\alpha\, \mathrm{m}}{\left(\mathrm{P_0} - \mathrm{h(d)}\right)}} = \frac{\dfrac{\partial\, \mathrm{MCS}}{\partial\, \mathrm{d}}}{\mathrm{X_1}} = \mathrm{h}'(\mathrm{d}) \tag{16}$$

To use this information in calibrating preferences we must specify *h(d)*. For our example, assume *h(d)* follows a power function with a declining marginal effect of *d* on the price. One expression using this formulation treats $h(d) = d^{\beta}$, where β is assumed to be constant. Englin et al.'s [1997] consumer surplus estimates of the seasonal gain due to quality improvements, scaled by their estimated number of trips, 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

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[21] Setting $X_1 = 0$ in (11) and solving for *P* does not yield a finite choke price because X_1 approaches zero as *P* assumes arbitrarily large values. For current purposes we assume there is some large finite choke price.

described by Englin, et al. [1997], i.e. increasing dissolved oxygen at the worst lakes to approximately fishable conditions--6.0 mg/liter. Using their estimate allows us to calibrate b.

Their estimate of the average per season increase due to this water quality improvement was $29 (in 1989 dollars, $35.64 in 1995 dollars) per household with each recreationist taking 5.06 trips under the improved conditions. Using a series approximations for the derivative of the power function (i.e., $h'(d) = \text{b}d^{\beta-1} \approx \text{b}\,[1 + (\text{b}-1)\log(d)]$, equation (16) can be written as a quadratic (i.e. equation (17)) and solved for its roots:

$$\log(d)\_\ \text{b}^2 + (1 - \log(d))\,\text{b} - h'(d) = O \tag{17}$$

where

$$h'(d) = [(\partial M\hat{C}S / \partial d) / \hat{X}_1]$$

Each of the roots to this equation is a potential solution. We can discriminate between the two roots for β based on their economic properties. There are two components of this task--solving for the roots and then using them in solutions for a. That is, each value of b is used in the expression for a based on the definition of the WTP in equation (18). Using each of the roots derived from (17) in the expression for $\hat{a}$ in (19) allows consideration of the plausibility of the estimates of a as well as the predicted demand for recreation trips. Of course, to implement the process we need an estimate of *h'(d)* and the WTP.

As we noted earlier, Mitchell and Carson's contingent valuation question provides an estimate of the WTP for a change in dissolved oxygen at water bodies with less than fishable conditions. For this calibration we have treated their estimate as the WTP for a water quality change from boatable (d_B) to fishable (d_F) conditions. The WTP derived from our specification for the preference function is given in equation (18):

$$\text{WTP} = m - \left(\frac{P - h(d_F)}{P - h(d_B)}\right)^{a} m \tag{18}$$

Using the roots to (17) to evaluate *h(.)* for different values of *d* and an estimate of WTP, we can solve (18) for α, and we have equation (19). The calculations for (19) use Mitchell and Carson estimates for improving water quality from boatable to fishable conditions--$163 (in 1983 dollars) and $249.41 (in 1995 dollars) and income was $32,659 (in 1995 dollars).[22]

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[22] While we cannot recover an estimate of *b* in equation (11) with this information, this parameter did not enter the WTP function (i.e., equation (17)) and therefore an inability to isolate it with this information does not preclude our calculation of WTP or demand for new sites.

$$\hat{a} = \frac{\ln\left(\frac{m - \hat{WTP}}{m}\right)}{\ln\left(\frac{P - \hat{h}(d_F)}{P - \hat{h}(d_B)}\right)} \qquad (19)$$

Table 2 illustrates how evaluating the demand for trips assists in a selection of a root that implies a parameter value for α.[23] As the table suggests, the negative root for the expression in (17) implies negative trips and a negative value for α. Computations using equation (17) and (19) identify a sufficient number of the parameters for the indirect utility function in equation (11).[24]

Table 2. Solutions to Travel Cost Demand Calibration[a]

ROOT of Equation (17)	$\hat{a}$ [b]	$\hat{X}_1$ [c]	$\hat{WTP}$
2.29	.024	20.14	517.63
-1.91	-16.990	-5,550.63	210.93

[a] Englin et al. [1997] do not report the average travel cost per trips incurred by their sample of recreationists. These computations assume the round trips costs was $100 (including the time costs of travel).

[b] This parameter would correspond to a share of income spent on the recreation commodity for an indirect utility function that omitted the cross-product repackaging assumption.

[c] These estimates correspond to the level of demand at water quality corresponding to fishable conditions (i.e. dissolved oxygen level of 6.0 mg/l).

This example is now complete. To illustrate how the calibrated preference function can be used, consider the task of measuring the per household benefits for different water quality improvements at different lakes. Assume these lakes support recreational fishing and that these fishing trips are the primary reason for the benefits associated with water quality improvements. Table 3 compares the estimated WTP from these calibrated preferences with a more typical approach--computing the average benefit per unit improvement in dissolved oxygen from Englin et al. (or some comparable scaling of Mitchell and Carson) and multiplying by the size

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23 We could also derive an estimate of α from equation (12), given *d, P, m* and X_1, and we use this strategy in our last example.

24 In the studies available to us, they do not report the average income for their households. As a result an estimate for income from their pilot survey (for 1981) Mitchell and Carson (1981) was used and converted to 1995 dollars.

of the change.[25] For convenience we assumed income was at the level implied by converting the Mitchell-Carson estimate described earlier to 1995 dollars using the CPI. An advantage of the calibrated approach to benefit transfer is that we do not have to make this assumption. Income estimates can be set to correspond to local conditions. An estimate of average roundtrip travel cost is also required. This could also be adapted to fit the specific policy application.

Table 3. Illustrative Transfers Based on Travel Cost and CV Estimates for Water Quality Improvements

Hypothetical Lake	Travel Cost	Water Quality Baseline	Water Quality New Level	X_1	Calibrated WTP	Conventional Approximation
A	$100	1	4	10.45	$208.71	$147.26
B	$100	3	6	20.14	$627.96	$283.79
C	$100	5	6	20.14	$332.97	$94.60

Several further aspects of these computations should be highlighted. The levels of baseline and improved quality affect the estimated WTP for the change in water quality. In conventional transfer approaches adjustments for differences in most income or baseline conditions are ad hoc. The table also presents one auxiliary measure that can be computed from the calibrated model as a plausibility check. Two predicted trip levels per household are presented to illustrate the computations. Other measures such as the price elasticity of demand, or with more complex preference specifications, other features of demand could be computed for comparison.

This process offers one simple way to demonstrate how the same empirical information used for conventional benefit transfers could be used to calibrate preferences. We deliberately used only the information available in published papers and widely available reports. The choices for how to use the record involved simple compromises. There may well be much better choices for many of these variables. However, this criticism misses the point of our example. With this preference specification (and many others), we do not require new primary research to develop consistent benefit transfers.

B. Hedonic Property Value and CV

Suppose that the benefit transfer used a different set of existing information. This alternative set of benefit estimates for improvements in water quality could arise from site specific amenities rather than from benefits due to fishing or other observable recreation.

[25] More specifically, the Englin et al. [1997] benefit measure, is divided by the change in average dissolved oxygen levels (i.e. 5.0-3.5) to calculate a "per trip consumer surplus per unit of water quality." The quantity is then multiplied times the proposed change in dissolved oxygen and the predicted trips at the highest quality level.

Such amenity benefits would be measured by changes in property values, and, in this situation, the benefit transfer would combine CV results with hedonic estimates. While it could also be suggested that the hedonic approach captures a wider set of benefits, the physical connection between the location of the house and the resource experiencing water quality requires that we rely on water bodies providing owners fairly immediate access. Equally important, because models describing the purchase of a house usually assume an *ex ante* perspective, it is reasonable to assume that specific use patterns are uncertain. In this sense, the hedonic framework could imply values for amenities conveyed simply by the location as well as better access for use of the resource's amenities. In any specific application, it may not be clear how the benefits measured for lakes in close proximity to an individual" home relate to the benefits to the consumers from being able to enjoy quality improvements at more distant sites. However, this concern is not a new one; it has been a part of the qualifications raised with most benefit transfer studies.[26]

Each indirect valuation method implicitly establishes a different spatial link between people and quality improvements in water bodies close to them. Therefore, if we are to use a range benefit estimates (marginal or total willingness to pay for quality changes), the adopted preference structure must explicitly resolve differences in these hypothesized links between a site" water quality and people" preferences.[27] This and the next example illustrate that specific assumptions about the links between people's well being and environmental quality result in specific resolutions of the degree of overlap between different benefit measures that are available.

Linking hedonic and contingent valuation estimates requires that we acknowledge their different connections to preferences, as we did for the example using the travel cost and CV estimates in calibration. Hedonic models provide an estimate for the marginal rate of substitution between environmental quality and a numeraire good (usually money). This estimate can also be interpreted as the marginal willingness to pay evaluated at a specific level for the water quality. It is widely recognized that the ability to estimate this marginal willingness to pay at a point does not necessarily imply it is possible to recover the full marginal willingness to pay schedule [see Freeman 1993, Bartik 1987, and Epple 1987].[28]

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[26] Indeed, it may explain the durability of the Mitchell-Carson study. That is, the format of their CV question makes this distinction explicit by attempting to capture all reasons for valuing water quality improvements at all water bodies in the U.S. In separate questions their survey also considered how respondents would adjust their responses for changes at only local resources and for partial improvements in water quality.

[27] McConnell [1990] recognized this issue in his discussion of the overlap in estimates from hedonic and travel cost methods.

[28] While there are several reasons for this conclusion, important among them is the fact that the analysis assumes consumers have different preferences. This qualification is not relevant to preference calibration because our approach to transfer requires that the analyst assume a specific preference function for the representative individual. Feenstra [1995] is a notable alternative case. In this case, however, a specific form of preference heterogeneity is assumed in order to allow the demand behavior to be represented by the utility of a representative consumer.

The logic parallels what Quigley [1982] used in demonstrating that hedonic models could identify preference parameters.[29]

We adapted our preference specification in equation (11) to conform to Quigley's suggestion. This alternative form for the indirect utility is given in equation (20). With it, our example shows how one can use an estimate of marginal WTP along with an estimate of the WTP for a discrete change in that water quality to calibrate preferences. Our example assumes that all other prices are constant across individuals. We also maintain that the housing choice is the only way to "select" a water quality level.

$$V = \sum_{i=1}^{K} (\theta_i \cdot A_i)^b + (m - R(A_1, \ldots A_K))^b \qquad (20)$$

where

$R(.)$	=	hedonic price function expressed as the annual rent
A_i	=	housing characteristics (assume A_1 = water quality)
m	=	income spent on all other goods
θ_i, b	=	parameters

The first order condition for the choice of site specific water quality, represented here as A_1, yields equation (21):

$$\frac{\partial V}{\partial A_1} = 0 = \theta_1 b \cdot (\theta_1 \cdot A_1)^{b-1} - \left(\frac{\partial R}{\partial A_1}\right) \cdot b \cdot (m - R(A_1, \ldots A_K))^{b-1} \qquad (21)$$

Re-arranging (22) establishes how an estimate of the marginal WTP is expressed in terms of the parameters that need to be recovered. Benefit measures for changes in A_1 can be derived from equation (21) if it is possible to calibrate b and θ,

$$\frac{\partial R}{\partial A_1} = \theta_1^b \cdot \left(\frac{A_1}{m - R(\cdot)}\right)^{b-1} \qquad (22)$$

[29] CES is an abbreviation for a constant elasticity of substitution function. The preference specification given in equation (20) adds a separable contribution for water quality from that observed directly through the housing market to reflect the expanded nature of the Mitchell-Carson question. It is a generalization to that given in equation (11) to include these terms. Water quality benefits arising from fishing do not reflect this separable component. The Mitchell-Carson CV would. We did not use it for our first example because preference calibration would have required more than the three estimates. We illustrate this point below with the use of three methods. The preference specification for that example (given in equation (25)) includes those in (11) and (20) as special cases.

The second estimate assumed to be available is a measure of willingness to pay for improving water quality, as described in the Mitchell-Carson contingent valuation study. Equation (23) defines the willingness to pay for their proposed plan to improve water quality from A_1 to $A_1 + D$ using the preference function defined in equation (20).[30]

$$(m - WTP)^b + \sum_{i=2}^{K} (q_i \cdot A_i)^b + (q_1 \cdot (A_1 + \Delta))^b = m^b + \sum_{i=1}^{K} (q_i \cdot A_i)^b \tag{23}$$

Solving for WTP yields (24) as the Hicksian willingness to pay function for the improvement in water quality.

$$WTP = m - (m^b + (q_1 \cdot A_1)^b - (q_1 \cdot (A_1 + \Delta))^b)^{1/b} \tag{24}$$

Using equation (22) and the estimate of $\partial R / \partial A_1$ we can "eliminate" q_1 from (24). With estimates of WTP from the Mitchell-Carson study, the re-formulated equation (24) can be solved for *b*. The marginal willingness to pay for a small change in water quality and the total willingness to pay for a larger change are defined in equations (22) and (24). With estimates for each benefit measure, we have are two non-linear equations in the two unknowns q_1 and *b*. These can be solved to generate sufficient preference information to use the model to infer the economic value of new policies involving changes in A_1. As in the case of the recreation demand transfer, there are economic plausibility restrictions that follow for the economic plausibility of the calibrated function which can assist in discriminating among multiple solutions to these non-linear equations. q_1 must be different from zero and positive, otherwise water quality is not a positively valued good. *b* has a direct link to the Frisch money flexibility of income [see Freeman 1984] and thus we have a plausible range of values for it as well.

Finding an application to use in implementing this logic with water quality has, until recently, been difficult because few hedonic property value studies that included water quality as a site specific amenity. However, Kevin Boyle has led a series of studies of the role of water quality in Maine lakes for property values. We selected two early reports of these findings [Michael et al. 1997, and Lawson 1997] to illustrate the issues in linking these hedonic results with the Mitchell-Carson CV estimates.

The first distinction that arises in considering a hedonic/CV example of preference calibration is the selection of measure for water quality. With the hedonic property value models, the measure should be based on the features of the water bodies that can be perceived

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30 Note that in this case we have assumed that the housing decision has been made (and thus left out the *R*(.) term from the indirect utility function used to define WTP). This assumption is not essential to the method. It is a simplification to focus on how the assumptions with hedonic estimates contrast with those from travel cost models. When it is included we can also use this framework to consider how the extent of capitalization of gains influences the WTP [see Palmquist, 1988].

by those buying the houses.[31] An important implication of this difference for applications is the point raised by Poe [1998]. There is a clear need to establish consistent links in the various technical measures used in each behavioral model that contributes to a calibration. These links become part of the maintained assumptions that contribute to the performance and plausibility of results from that calibration.

For this example we use water quality measured by the secchi disk reading. Estimates of the marginal willingness to pay for water quality and income are taken from one of the groups of towns used in the studies by Michael et al. and Lawson. The group we selected was designated the Lewiston/Auburn market area and includes four lakes--Sabattus Lake, Taylor Pond, Thompson Lake, and Tripp Pond. The average housing price was $105,704 (in 1995 dollars), the water quality reading measured with clarity corresponded to swimmable conditions. It was linked to other measures of water quality based on the RFF water quality ladder. To establish a one-to-one connection between the various pollutants contributing to the index that defines the "rungs" of the ladder, we assume that levels of other pollutants also declined as clarity improved. At a secchi disk reading of 5.66 meters (the water clarity measure), the estimated marginal willingness to pay (using the housing price) is $4,569. The household income reported in Michael et al. is $82,074. Mitchell and Carson reported an estimate of $242 annually (in 1983 dollars) for cleanup of all lakes and rivers from their current conditions to at least swimmable conditions.

To assure consistency with the static framework implied by the model, three adjustments must be introduced to the available benefit information. As in the first example, the benefit estimates must be converted to consistent dollars. In this example, the hedonic model's focus on the cleanup of water quality at local lakes offers a reason for considering another aspect of the Mitchell Carson results. We used their estimate of the portion of the national benefits that can be specifically attributed to *local* improvements. Based on their respondents' answers to a question asking about local improvements, they estimated this fraction to be .67.[32] Second, the housing price and marginal willingness to pay were converted to annual equivalents using Poterba's [1992] annualization factor which reflects the income tax and property tax effects on the rental cost of housing. While his annualization factor was computed for 1990 we assumed it would be relevant for 1995.

Finally, the hedonic model must be linked to the physical interpretation offered for the water quality described in the contingent valuation study. This question was resolved for our earlier use of Englin et al. travel cost and Mitchell-Carson studies by linking them both to

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31 Hanemann [1978] identified this issue at an early stage in his empirical study of water quality and beach recreation. Bockstael et al. [1988] consider how these perceptions are formed and confirmed the importance of perceptions.

32 The Mitchell Carson survey asked about partial improvements in two ways. The first of these considered how respondents would adjust to improving water quality at all water bodies. The second and source of the .67 estimate focused on the spatial features of the improvement. It asked respondents how they would divide their total willingness to pay between their state and the rest of the nation. See Carson and Mitchell [1993] for further discussion.

changes in dissolved oxygen. In this application, Mitchell-Carson's descriptions of water quality changes are linked to the RFF water quality ladder, which are then related to the secchi disk readings used in measuring how water quality is perceived by homeowners in the hedonic model.[33] Although, this step of the process was somewhat ad hoc. Nonetheless, this feature of the calibration does not detract from the overall logic. We identify it here to illustrate the specific types of judgments that must be made about how technical measures of water quality are related to each other. With these estimates, an assumed baseline of swimmable water quality conditions, and the assumption that WTP to avoid the loss of swimmable quality corresponds to the Mitchell-Carson estimate, equations (22) and (24) can be solved for q_1 and b. We did not set up a formal loss function numerically to optimize. Instead we calibrated using a simple grid search over values of q_1.

There is little variation in the results for small changes in q_1. The calibrated estimates of WTP use $q_1 = 0.10$. Table 4 presents the WTP estimates for three *new* alternative water quality changes (DA_1=1,2, and 4 meters) from boatable conditions, a secchi disk reading of $A_1 = 2.96$ meters. The last column reports a simple approximation often used in practice. It assumes the willingness to pay derived from the hedonic model value is approximately constant and assumes it applies for the full extent of the quality change. Our estimate of the marginal value from the hedonic was constructed for swimmable conditions. As a result, the approximation could be interpreted as one form of the linear approximation discussed earlier to define CS_C (see equation (2)).

Table 4. Illustrative Transfers of the Value of Water Quality Changes from Hedonic Price Models: Calibrated Versus Simple Approximation (1995 $)

Water Quality Change (Δ)	New WTP (b=.10)	Simple Approximation $(\partial P / \partial A_1) \bullet (\Delta)$
ΔA_1=1	$534.4	$530
ΔA_1=2	$956.3	$1,060
ΔA_1=4	$1,605.5	$2,120

As the size of the change in water quality increases, the difference between the calibrated WTP and the simple approximation for WTP grows, with a one-third larger estimate for the simple approximation in comparison to the calibrated preference function

[33] The RFF ladder parameters include (among other attributes) dissolved oxygen and turbidity (measured using Johnson Turbidity Units, JTU) such that boatable quality equals 100 JTU and swimmable water quality equals 10 JTU. We adapted information reported in Smith and Desvousges [1986] to estimate a simple conversion relationships that translated the turbidity units used to define boatable to swimmable conditions in the RFF water quality ladder to secchi disk readings in meters. Boating and swimming level water quality are calculated as 2.96 and 5.66 meters respectively.

with the largest water quality increase of 4. Of course, the measure from preference calibration is simply a more complex set of numerical calculations. There is no "true" or correct value for these types of comparisons. Our estimates simply illustrate how the relative size of the transferred benefit values vary with the size of the quality increments. There are a large number of factors that influence the discrepancies between any transferred estimate and what we would be prepared to describe as the "true" value for the willingness to pay. We return to the interpretation of these discrepancies below.

C. Travel Cost, Hedonic, and CV

Using all three sources of benefit estimates requires that the preference function reflect the types of adjustments consumers are assumed to be making in the travel cost and hedonic frameworks in response to different water quality conditions. Calibration of preferences based on estimates from three separate benefit studies requires that a separable role be established for water quality as a site attribute from the role that is attributed to it as a location amenity. This is accomplished using the CES function, and by including in the function, a role for the adjusted travel cost term from equation (11). This alteration yields the indirect utility function given in equation (25):

$$V = \sum_{n=1}^{K} (\theta_i A_i)^b + \left[\left(P - h(A_1)\right)^{-\alpha} \left(m - R(A_1, A_2, \ldots, A_k)\right) \right]^b \tag{25}$$

With this new specification several additional relationships will be important to our calibration strategy and the issues it raises. The first of these is the revised demand function for this preference specification, given in equation (26). A key issue in this formulation is the recognition that income is defined to be net of the expenditures on rent (i.e. *m-R(.)*). As a result, measures of the change in the Marshallian consumer surplus (per recreation trip), due to a change in water quality, must also consider whether the rent is assumed to change when the water quality changes. We assume it does not. This restriction affects how benefit estimates from recreation models are used in calibration and is illustrated in the relationship given by equation (27):

$$X_1 = \frac{\alpha (m - R(A_1, A_2, \ldots, A_K))}{(P - h(A_1))} \tag{26}$$

$$\frac{\partial MCS}{\partial A_1} = X_1 \cdot h'(A_1) \tag{27}$$

The relationship given in (27) is equivalent to assuming that adjustment in the rental market does not affect the disposable income relevant to an individual's observed recreation trips, i.e. the individual does not have the ability to move houses in response to water quality changes or, equivalently, that the available estimates from the travel cost demand analysis reflect the short run benefits associated with water quality improvements. In imposing this restriction, the most important consideration is whether the assumption offers a reasonable description of the conditions associated with the estimate of $\partial MCS / \partial A_1$. Once the preference function is calibrated, it is possible, in principle, to change the assumption for a policy analysis. However, this would require a general equilibrium model describing how the changes water quality influenced the housing price equilibrium.

A second issue associated with calibrating preferences using two revealed preference methods arises from the potential for overlap in the types of gains measured by the two methods. In simple terms, if there are two ways to enhance the environmental amenities, we know people will choose the appropriate mix that minimizes the cost of realizing the required level of the amenity. If each approach provides a perfect substitute for the other and there are no capacity constraints, it is likely that only one mode of adjustment will be used. A mix of uses arises because of imperfect substitution, increasing marginal costs and/or capacity constraints with each approach. McConnell's [1990] discussion of consumer surplus measured through recreation demand or hedonic based marginal willingness to pay measures recognized this relationship. With full adjustment and the assumption that water quality is only valued (in the short term) because of its effect on recreation that takes place on-site, the two approaches (e.g. the change in the value of access measured through the recreation demand or the hedonic analysis) should yield the same results.[34] This conclusion follows from McConnell's equation (9). It suggests that when the motive for living near the resource is access for recreational purposes, then both the travel cost and hedonic property value models measure the economic value for the same concept of access. However, when the models distinguish separate motives, the interconnections become more complex.

Our specification assumes that there are gains to be realized from obtaining amenities from the different sources. Access (or water quality at the site) is separately valued from the recreation. A_1 reduces the effective price of recreation <u>and</u> separately contributes to individual well-being through the term $\sum_{i-1}^{K} (\theta_i A_i)^b$. This specification implies that the hedonic model (with adjustment) will reflect both the incremental recreation gains due to water quality improvement and the separate amenity value of living near a lake with improved water quality. McConnell also analyzed a case like this one. Equation (28) recognizes this type of overlap.

$$\frac{\partial R}{\partial A_1} - \frac{\partial MCS}{\partial A_1} = \frac{\theta_1 (\theta_1 A_1)^{b-1}}{(P - h(A_1))^{-\alpha b} (m - R(.))^{b-1}} \tag{28}$$

[34] This statement assumes that the consumer surplus concept (e.g. Marshallian versus Hicksian) is constant.

The right side of (28) defines the change in value of access because quality has changed. It is reflected in the change in the site rent with A_1, but the $\partial MCS / dA_1$ (the change in seasonal consumer surplus, $X_1 \cdot h'(A_1)$) must also be taken into account. Consistent calibration of preferences from existing travel cost and hedonic estimates requires recognition of the role of amenities realized through the two adjustment margins and some resolution of how this information contributes to preferences.

Our analysis assumes the travel cost measures do not reflect changes in rent, but the hedonic rent (for those adjusting their locations) does take into account the enhanced value of recreation associated with water quality improvements. The reason for the difference is due to the conditions likely to characterize the studies being used to calibrate preferences. The hedonic model assumes people are adjusting their locations in response the differences in environmental amenities. Thus, it is reasonable to assume that studies reporting hedonic estimates would be cases where individuals recognized the full gains from adjustment.

A short run orientation for the recreation component of the estimates used in calibration and a long run for the hedonic is not necessarily a contradiction. This reasoning recognizes that the estimates come from different people. The calibration strategy should assume that these people have a common preference function but does require that the constraints they face have to be the same. In fact, it is probably more reasonable to assume that they will be different as we consider the various modes people use to adjust their behavior in response to differences in the water quality available and what they would like to have.[35]

This approach contrasts with conventional joint estimation of models based on two or more responses to environmental quality changes. In those types of applications, the constraints facing each individual are assumed unchanged across revealed preference methods. This assumption is plausible because the model focuses on the multiple adjustments of the same person (or household). The changes are represented as taking place within the same decision horizon and as a result the constraints would be unchanged.[36] Estimates of the preference parameters are possible because the analyst is able to observe variations in responses to the different constraints across people in the sample face. This information allows the models to be estimated. In our case the framework recognizes that calibration relies on incomplete information from different sets of people. We include all the ways of adjusting by consistently combining these separate adjustments in the calibration of a common preference function.

The last component of the information used in this example involves incorporating the results in equation (28) into the definition of willingness to pay and using it along with the

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[35] Our discussion of the long run versus short run distinction is artificial. Both models are static descriptions of behavior. One holds a set of potential choice variables constant. The other does not. Our terminology labels the first short run and the second long run. The difference between them is intended to provide some insight into the role of adjustment for monetary measures of tradeoffs defined from individuals' choices.

[36] With contingent valuation studies, it is conceivable that the constraints facing an individual would be altered as well as the environmental quality. Indeed, this is exactly what the discrete response format does when offering a tradeoff between higher cost and quality improvement.

Mitchell-Carson estimates to complete the identification of the model's parameters. We assume that respondents to the Mitchell-Carson question believed that their rents would not change with water quality improvements, but that the improvement would reduce their effective costs of recreation trips. Equation (29) defines the WTP from preferences defined by equation (25):

$$WTP = m - \frac{\left[(\theta_1 A_{10})^b - (\theta_1 A_{11})^b + \left[(P - h(A_{10}))^{-\alpha} \, (m - R(A_{10}, \ldots, A_k)) \right]^b \right]^{\frac{1}{b}}}{\left(P - h(A_{11}) \right)^{-\alpha}} - R(A_{10}, \ldots, A_k) \qquad (29)$$

Comparing the definition of the WTP function in equation (29) with the example used to calibrate preferences based on hedonic and CV estimates (i.e. equation (24)) we see that the assumption of constant rents in that case caused rent to drop out of the expression for WTP. In this formulation the effect of water quality on the effective price of recreation prevents such a simplification. Calibration is decomposed into three steps to simplify our explanation of how the restrictions identify the parameters.

First, we use the same assumption as in our first example to incorporate the effects of water quality on the effective price of recreation (i.e. $h(A_1) = A_1^{\beta}$). Substituting the series approximation we can solve for ß and for α using the demand for recreation trips. This requires two sets of information--average travel cost and income net of rent. The level of recreation demand is used to estimate α, given an estimate of β and $h(.)$. We used the estimated trips with improved water quality (5.06 trips per season based on Englin et al.'s [1997] for fishable conditions). The Englin et al. estimates of travel cost reflect 1989 dollars and a time cost corresponding to income levels substantially lower than what was observed with the Boyle survey of the owners in his hedonic analysis. As a result we adjust travel costs proportionately using the CPI and income differential between the two studies.[37]

In our simple travel cost/CV calibration, the Mitchell-Carson CV estimate provided the information to recover α. Now the form of the preference function implies that WTP is affected by the two ways water quality contributes to individual well-being. Our preference specification in the first example maintained that water quality was only important because of its effect on recreation. Once these additional roles are introduced, the identification of preference parameters requires that the analyst introduce other observable variables that are affected by water quality's contribution to recreation. In our example, the two sets of information calibrating this component of preferences are the estimate of consumer surplus and the number of trips under a specified set of water quality conditions.

[37] The Englin et al. average round trip travel and time cost estimate per trip was 20.42 (in 1989 dollars). Adjusting it by the CPI to 1995 dollars and increasing the result by an estimate of net income (after rent) in the two studies yields the travel cost used to calibrate the α parameter in equation (29). The income adjustment was an approximate strategy for reflecting the opportunity costs of time.

$$\alpha = \frac{X_1\left(P - \hat{h}(A_{11})\right)}{\left(m - R(.)\right)} \tag{30}$$

where

X_1	=	the number of trips
m-$R(.)$	=	Michael et al. sample's income estimates net of annual rent at fishable water quality
$P - \hat{h}(A_{11})$	=	adjusted travel cost less computed water quality effect
A_{11}	=	fishable water quality measured with dissolved oxygen

The second step follows from equation (28). This can be solved to provide an expression for θ_1^b, in terms of available estimates--

$$\partial R \big/ \partial A_1, \quad \partial MCS \big/ \partial A_1, \quad P - h(A_1), \quad A_1, \quad m - R(.)$$

together with the calibrated values we derived for β and α from equation (30) and the series used for $\hat{h}(.)$:

$$\frac{\dfrac{\partial R}{\partial A_1} - \dfrac{\partial MCS}{\partial A_1}}{\left(P - h(A_1)\right)^{\alpha b}\left(\dfrac{A_1}{m - R(.)}\right)^{b-1}} = \theta_1^b \tag{31}$$

Of course, to use these diverse sets of information, A_1 must be measured in consistent units. To do so in our example, we converted dissolved oxygen (measured in mg/l) into secchi disk readings (in meters) using an approximate conversion factor (1.11).[38] With Mitchell and Carson's estimates of WTP for the equivalent water quality change, adjusted to 1995 dollars, the remaining parameter to be identified in the expression for WTP is *b*.

The income substituted in the expression for WTP to calibrate *b* is derived from Mitchell and Carson. Because we do not know rent paid by household's responding to this survey, the rent is estimated as a fraction of their reported income, not the annualized rent from the housing prices observed for households in Maine (or for recreationists interviewed by Englin et al.). The logic for these adjustments is that the income, water quality, and travel

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[38] The conversion factor was estimated from available data in Smith and Desvousges [1986] linking dissolved oxygen, secchi disk and turbidity. We do not claim that this is accurate, but note that the conversion must be made in order to call attention to the general need for consistent measures of water quality that can be linked across the methods for benefit measurement.

cost used at each stage must be consistent with each study's conditions. The "take away message" is that calibration must represent the constraints relevant to each set of valuation measures.

Table 5 uses the calibrated function to estimate the benefits from three different water quality changes. All three realize the final endpoint of approximately swimmable conditions. They differ in the starting point. The size of the water quality increment declines with each row in the table. The ratio of the simple hedonic approximation ($\frac{\partial R}{\partial A_1} \cdot \Delta A_1$) to the WTP estimate is in the range of 0.7 to 0.8.

Table 5. Benefit Measures from Preferences Calibrated with Travel Cost, Hedonic, and CV Estimates[a]

Water Quality[b] BASELINE	FINAL	Hedonic Approximation	Marshallian Consumer Surplus	Calibrated WTP
3 (2.56)	6 (5.26)	$1,431	$324	$1,954
4 (3.46)	6 (5.26)	$954	$296	$1,277
5 (4.36)	6 (5.26)	$477	$234	$688

[a] Calibrated parameters are β=2.29, α=.0029, and b=.05. The last of these parameters is an approximate solution. The hedonic marginal price is taken from Larson [1997]. The travel cost information relies on Englin, Lambert, and Shaw [1997] and Englin and Lambert [1995]. Based on the RFF Water Quality ladder DO=6.5 is swimmable, DO=4.0 is rough fishing, and DO=3.5 is boatable conditions.

[b] The numbers correspond to dissolved oxygen (DO) levels. The numbers in parentheses are secchi disk readings in meters.

It might seem that the results in the table contradict the arguments we made initially for undertaking calibration to develop a consistent set of benefit transfers. That is, it appears the sum of the hedonic approximation and the Marshallian consumer surplus measure of the enhanced value of recreation trips due to the water quality improvements approximates the "consistent" WTP reasonably well. This correspondence contrasts with what would seem to be implied by our formal model as well as by McConnell's discussion of a comparable issue. However, there is no contradiction in the two sets of results. The models we specified demonstrate relationships between the marginal values not the integrals of these marginal values. So, our approximation is not the area under the Marshallian willingness to pay derived from a hedonic price function. It is a linear extrapolation. Thus, the fairly close proximity between calibrated WTP and the sum is an accident of the numerical example we selected.

D. Lessons from the Calibration Examples

Our examples are more than a collection of "tricks with algebra." They highlight four underappreciated aspects of benefit transfers. First, the task associated with developing benefit estimates to evaluate a new policy should be interpreted as an *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. 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 that compatible indexes of environmental quality. 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.[39] Third, to reconcile multiple, overlapping measures of people's incremental benefits from changes in non-market resources we must specify how each of these adjustment margins is interrelated to the others. Finally, the observed economic tradeoffs that people make to obtain increases in non-market resources are constrained by their available incomes. None of the existing approaches to benefit transfer meet the simple Harberger test. That is, when transferring benefits, we must ensure that measured WTP are affordable, i.e. well within people's disposable income.

Our comparisons of the differences between the common approximations for benefit measures and the calibrated measures avoided describing one as "correct." Differences between them do not reflect one method's errors. Both embody errors. The correct measures of willingness to pay would consider how individuals experiencing a change in environmental quality would actually respond to that change. This ideal can never be measured.Analysts always face compromises. By definition, evaluation implies analyzing outcomes that have not taken place. Benefit transfer adds to the questions about any benefit analysis of a proposed change by using *different* people's choices for *different* resources (than the one(s) considered for the policy) to measure the benefits of the policy. The accuracy of these transfers depends on how our assumptions about the people and the resources compare to what is actually the case. For each set of assumptions we can no doubt readily derive different conclusions for the available benefit transfer practices.

The primary argument that can be made for preference calibration as a strategy is that it imposes economic consistency conditions on the ways the existing information is used, and therefore avoids simple contradictions in the transferred results. Experience and experiments (with CGE models) comparable to what has taken place for nearly fifty 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 fifty years of experience to consider revising current practices. Rather, it implies that we have a clear roadmap to follow in evaluating preference calibration. In the meantime, while we wait

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

for such numerical experiments, we can use preference calibration to judge whether the imposition of these types of consistency conditions would change conclusions.

E. Calibration and Meta Analysis

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 of the existing literature, the logic of preference calibration implies that there may exist sufficient information to *identify* the parameters of preferences. When there are such multiple sets of information it is possible to treat the relationships necessary to identify the parameters as a system of equations defining moment conditions. In this context the basic logic would be implemented using multiple sets of estimates as "data." Moment conditions can then be used to estimate the preference parameters. We can illustrate the logic with an outline of what was done for our calibration examples--indicating how it can be adjusted for estimation.

In simple terms, equation (32) defines an indirect utility function:

$$v = V(P, m, A_1, \beta) \tag{32}$$

Roy's identity links $V(.)$ to observable Marshallian demands and duality theory link it to Hicksian and Marshallian consumer surplus measures. With a specific parameterization of $V(.)$, estimates of demand and benefits can be expressed in terms of these preference parameters as well as the exogenous variables assumed to influence individual's choices (as in equations (12) – (14)). Each of these relationships defines a moment condition. That is, if Z_i designates the measure being observed, identifying restrictions assure there is a way to connect each estimate to the parameters in the preference function. This link implies that with enough measures describing how a_i's relate to V(.) and with changes in observable, exogenous variables (i.e., p, m, A_1) it is possible, in principle, to estimate β. Our examples used identifying restrictions to estimate the single value of the parameters in β that would be consistent with the measures of Z_i at one point. Multiple measures of the Z_i's over a number of points allows formal estimation. This task requires adding an error, ϵ_i, to the conditions defining the Z_i's. By assuming the errors are centered at zero, the framework provides the basis for defining moment conditions. Using the independence of the ε_i from a set of instruments defines the methods of moments estimator (see Greene [1997] for a simple introduction). This strategy implies we can go beyond a single point to all points that provide sufficient information, in order to calibrate preferences.

IV. IMPLICATIONS

Conventional practices used in benefit transfers are analogous to approximations that were developed to measure the consumer surplus associated with price changes. They have been used in ways that do not assure they will be consistent with the economic concepts underlying the definition of willingness to pay for quantity or quality changes. Moreover, the larger the change being evaluated, the greater is the likelihood of serious biases.

While it is relatively easy to identify errors with the implausible estimates associated with the Costanza et al. [1997] estimate of the global aggregate of the annual willingness to pay for the services of the earth's ecosystems or EPA's [1997] retrospective evaluation of the Clean Air Act, it is also possible to find large inconsistencies in small scale, simple transfers. We illustrated this possibility with the Williamette case study. Unfortunately, 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 benefits transfer as analogous to an identification problem. That is, consistent transfers require sufficient information to recover either the marginal willingness to pay function (or enough of it) to estimate the benefits of proposed policies. This strategy is similar to the logic proposed by Ebert [1998] and can be applied with limited data. It can calibrate a preference function to a single baseline point or provide the basis for specifying a set of relationships as estimating equations (when there exists a large number of benefit estimates for the same aspect of environmental quality).

In measuring the welfare effects of policy-induced price changes, it is possible to avoid the inconsistencies that can arise with environmental benefit transfers. This can be done by imposing consistency as part of the approximation (e.g. measuring changes along Harberger demands) or by using computable general equilibrium models. However, for most environmental resource applications, neither strategy has been possible.[40] In this paper we impose similar consistency checks on the calculation of policy induced benefits by calibrating existing benefit information to a preference function.[41] Our examples illustrate this approach can work where there are sufficient estimates to permit identification of simple preference (or marginal willingness to pay) functions.

Congressional mandates now call for benefit-cost analyses to evaluate the performance of regulatory programs [see Office of Management and Budget, 1998]. If responses to these

[40] The task associated with introducing non-market goods into CGE models is more complex than the limited literature on the topic has acknowledged. There are at least two aspects of this added complexity. First, all revealed preference methods assume specific forms of nonseparability which result in linkages between marketed goods and the environmental media giving rise to the nonmarketed environmental quality (e.g. air pollution depends on both the emissions and the diffusion system). This connection affects the conventional practices using social accounting matrices in calibration. Second, the models usually represent economic activity as if it takes place at a point. Thus, the spatial variation that is a central element for these processes is not reflected. For an example that proposes one way to address these questions see Smith and Espinosa [1996]. A more detailed discussion of the methodological issues is in Espinosa and Smith [1999].

[41] In so doing, we follow McConnell’s [1992] argument that "the power of economics lies principally with the logic of theory, and then with the strength of empirical evidence…[because there is] no single way to mechanically transfer a model."

requirements are to avoid discrediting the practice of benefit cost analysis, 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. Given the scale of these policies, it may not be possible to meet this goal without a complete general equilibrium framework. There exist real concerns about a sequence of implausible benefit-cost analyses developed with off-the-shelf estimates and current transfer practices is real. Avoiding these implausible and inconsistent estimates requires considering the importance of the consistency achieved with calibration. Using a benefit transfer strategy that identifies and calibrates a preference function (or its equivalent) is a first step in developing that more completely account for the effects of large scale policies.

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