Improving the Practice of Benefit Transfer: A Preference Calibration Approach

Interim Final Report

Prepared for

U.S. Environmental Protection Agency
Office of Water
Office of Policy, Economics, and Innovation

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DISCLAIMER

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1 Introduction

Economic evaluation of the net worth of proposed policies has been a part of the fabric of policy analysis for several decades. In 1981 a Presidential Executive Order (E.O.) formalized this requirement, and an amended form of the benefit-cost mandate continues to be a part of current regulatory policy. Estimating the benefits of water quality programs instituted under the 1972 Clean Water Act (CWA) is therefore one of the requirements faced by the U.S. Environmental Protection Agency (EPA or Agency). It is also an integral part of the Agency's ongoing process to evaluate the contribution of its water quality programs to society, and it is one of the many ways in which the Agency identifies how it can be more effective in addressing the needs of society. To support these objectives, EPA has initiated a program to improve the data and methods used for estimating the benefits of its water quality programs. This document contributes to this effort by proposing a methodology for improving the way in which available information is used to develop these benefit estimates.

Applying a conventional economic paradigm to evaluate how water quality policies contribute to social welfare first requires that analysts identify and measure how the services provided by water resources are affected (i.e., enhanced) by changes in water quality. It then requires an assessment of how society values the changes in water services attributed to the policies. Water resources provide withdrawal services (e.g., irrigation, process cooling), in-place services (e.g., life support for plants and animals, recreation), and existence services (i.e., environmental stewardship and the altruistic concern for the welfare of others). Extending the application of this paradigm, the most commonly accepted and applied metrics for valuing these services (in benefit-cost analyses of the type required by E.O. 12866) are either individuals' maximum willingness to pay (WTP) in dollars for an improvement in environmental quality or their minimum willingness to accept (WTA) to forego an improvement in environmental quality.

In practice, benefit analyses of water quality programs (or other Agency initiatives) rarely afford enough time or resources for analytical staff to develop "new" WTP/WTA estimates that specifically apply to the policy impact. This is particularly the case when evaluating broad-scale policy initiatives, such as a retrospective benefits assessment of the CWA as a whole. As a result, a variety of pragmatic methods have evolved that use existing benefit (or cost) measures for "similar situations" to develop benefit estimates for policy-specific changes.[1] These methods are commonly referred to under the rubric of "benefit transfer."

Although benefit transfer offers the potential to economize on the time and resources typically needed to perform policy-specific studies, as we discuss below, its implementation is not without challenges, and there is scope for improving and expanding its application.[2] In this report, we propose an adaptation to the more typically applied benefit transfer practices. Economic theory posits that individuals' WTP for environmental improvements is ultimately defined by the structure of their preferences (i.e., a "utility function"). Our proposed benefit transfer approach relies on a more explicit specification of this preference structure. As such, it offers the potential for generating benefit estimates that are more consistent with economic theory.

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[1]As a rule, these estimates come from research studies that may themselves not be intended to estimate benefits but instead focus on a new model, estimator, or hypothesis test.

[2]Some examples of the early focus include Freeman's (1984) comparison of top down versus bottom up approaches to benefit transfer, a special section of *Water Resources Research* edited by Brookshire and Neill (1992) on the topic, and several recent evaluations of benefit transfer in the context of air quality changes (see Desvousges, Johnson, and Banzhaf [1998] and Alberini et al. [1997] as examples).

What is benefit transfer?

Benefit transfer is the practice of adapting available economic value estimates of a quality or quantity change for some environmental resource to evaluate a proposed change in some other "similar" resource. In these situations, the policy analyst is typically taking the results or data from the context of one or several existing studies (defined in terms of their time frame, location, environmental resource, environmental quality change, and/or their affected population) and transferring them to a context that is specifically relevant for a policy of interest.

The original data providing the starting point for this type of analysis may be derived from a natural experiment, or they could be the result of a specific experimental design that has been structured to test a hypothesis that is not directly relevant to the policy of interest. As a result, in conducting these analyses the analyst must carefully consider the similarity of the study context and the policy context. This comparison can involve evaluating their congruence in such factors as the affected resource, the magnitude of damages (or improvements), the existence of substitute resources, and the economic and demographic characteristics of the affected population.

How is benefit transfer typically applied?

Most benefit transfer methods use either the *benefit value* or the *benefit function* approaches to develop estimates. In the case of a benefit value approach, a single point estimate (usually a mean WTP estimate) or value range is typically used to summarize the results of one or more studies. For example, an average consumer surplus per fishing trip might be taken from a recreation 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 can then be transferred to assess the value of fishing trips or changes in lake quality at an alternative (i.e., policy) site. In the case of a benefit function transfer, an equation is typically estimated to describe how benefit measures (from one or many existing studies) change with the characteristics of the study population or the resource being evaluated. With this second approach, the entire equation (function) is transferred to the policy context, and the benefit estimate is then tailored to the population and/or resource affected by the policy.

One recent example and evaluation of the benefit function transfer approach can be found in a study by Downing and Ozuna (1996). They used a contingent valuation (CV) survey to measure a benefit function describing how WTP for a single year's worth of saltwater fishing trips varied statistically for different Texas Gulf Coast bays and time periods. They then used the WTP function to transfer benefit estimates across time periods. They conclude that "...the procedure of utilizing the benefit function transfer approach to determine the terms of appropriate compensation to harmed individuals at a policy site is unreliable" (p. 322). This conclusion may, however, be too strong given the nature of their analysis. In particular, the benefit function they considered in their evaluation did not include demographic or resource quality measures. In other words, it did not incorporate measurable differences in individual or water resource characteristics; consequently, it did not investigate the importance of these factors for the benefit transfer estimates implied by their CV data.

A second example is a study by Kirchhoff, Colby, and LaFrance (1997), which evaluates the statistical properties of estimated benefit functions. They also use a CV study to estimate how WTP for an improvement in river rafting quality (measured in terms of water flow) varied in terms of location, visitor characteristics, and the size of the change in river rafting quality. Using four different recreation sites, they then compared original CV WTP estimates for each site with benefit estimates transferred from other sites using a WTP function. They report findings that are only slightly more encouraging.[3] Their comparisons indicate that the transfer of simple benefit value (mean WTP) estimates from one site to another does not provide "valid" estimates.[4] On the other hand, use of benefit functions *can* provide valid estimates. The conditions for a valid transfer involved similarity of the source site and the recreation transfer site. Where the benefit function transfer was judged invalid, the implicit conditions suggested that the sites were not close substitutes.

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[3]Their appraisal used information on study and policy sites to develop one set of estimates (for the policy site) as the true benefit measure to be compared with various types of transfers (from the study site).

[4]Transfers are interpreted to be "valid" if "the values obtained form benefit transfer are not statistically different from those obtained through site-specific estimation" (p.84).

What are the inherent limitations of these benefit transfer approaches?

The empirical findings of these two studies most likely reflect some of the underlying limitations of these benefit transfer approaches. The benefit value transfer in particular is limited by the fundamental assumption that the benefit measure is essentially a constant. Even in cases where the benefit value is expressed as a WTP *per unit* of a quality or quantity change (e.g., $X per unit increase in a measure of lake water quality), this value ignores how this value might depend on the characteristics of the individual or other site qualities. The benefit function transfer approach, as it has been typically applied, addresses these limitations by assuming and estimating a linear relationship between the benefit measure and these characteristics.

However, importantly, neither of these approaches makes any explicit assumptions about the structure of preferences that are underlying the measured values. In other words, they use either mean benefit estimates or benefit functions in ways that cannot be checked for consistency with the utility maximization framework that is assumed to be at their foundation. For example, they do not explicitly consider how WTP is ultimately limited by income; therefore, they can generate results that are outside the scope of credibility. A notable recent example of the problems posed by the absence of consistency checks can be found in the retrospective component of EPA's (1997) recent benefit-cost analysis of the improvement in air quality attributed to the Clean Air Act regulations from 1970 to 1990. The benefit analyses monetize the effect-specific measures of morbidity and mortality effects attributed to air pollution. The result is an estimated benefit of $22 trillion. It implies that improvements in air quality created an asset worth about $221,000 (in 1994 dollars) for each U.S. household. In annual terms, this would yield income (at 5 percent interest rate) that increases personal income per household by 25 percent. A change of this magnitude is so large that it is outside the range of credible extrapolation.[5]

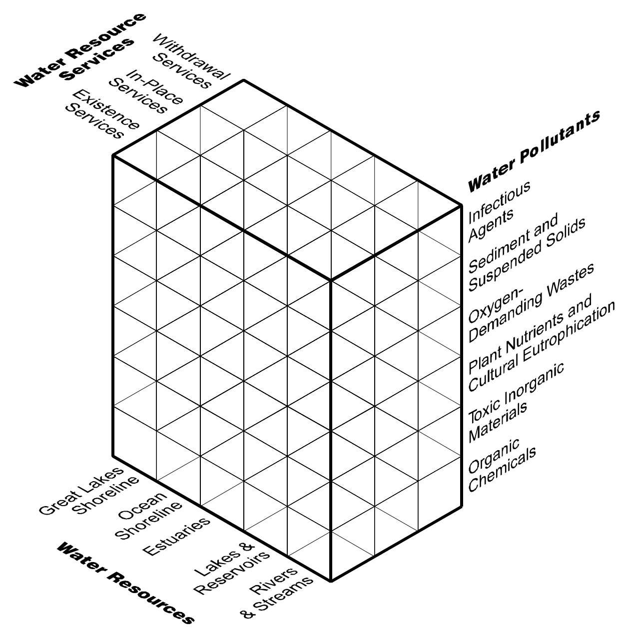
[5]That is, we cannot simply assume the values per health effect would remain constant for such large changes. Yet there is nothing in the conventional partial equilibrium approach to impose the consistency (and adjustment in values) we would expect if households actually had to pay for its composite of changes in morbidity and premature mortality effects.

Furthermore, these approaches do not explicitly consider how the gain in individual well-being from each unit increase in environmental quality may vary depending on the reference level from which the improvement occurs (i.e., each additional unit of improvement may contribute less to individual well-being than the previous unit). Nor do they specify how the increase in well-being associated with one use of the improved resource (e.g., recreational use of a lake) may depend on other uses of the resource or on the quality levels of other related resources.

The limitations of these approaches are only exacerbated as the scope of the policy scenario to be evaluated expands. Such an increase in scope may require value information from a variety of studies regarding a variety of resources and resource uses. Figure 1-1 illustrates the problems raised in attempting to integrate diverse sources of estimates and information needs. Consider the case of a policy intended to improve water quality on a very broad scale.[6] There are a number of ways of classifying the sources of benefits from such a water quality improvement. On one axis, labeled water resources, we could envision separating the economic gains based on the types of resources—rivers and streams versus lakes, wetlands, or estuarine resources. We might also consider the source of the water quality improvement—which of a set of pollutants was reduced. We could take this decomposition further by asking to isolate which sources—point or nonpoint, municipal or industrial—were responsible for the reductions. Finally, the cube in Figure 1-1 illustrates that, partitioning values according to the various sources of economic gains identified in the literature, water quality improvements can generate multiple measures of value. For example, water quality improvements can enhance withdrawal services by improving the role of water as an input to both production and consumption activities. It may enhance on-site uses, such as recreational fishing, for some and simultaneously enhance "nonuse" values for others (i.e., through the provision "existence services").

[6]This discussion abstracts from the spatial and temporal dimensions of benefit estimates. These may well be equally important. Improvements in some of the components (e.g., lakes, wetlands, river tributaries) or a watershed may well imply improvements in "downstream" resources as well as changes over time.

Figure 1-1. Illustration of the Potential Scope of CWA Benefits



As noted earlier, often the focus of a primary research study that is a candidate for use in a benefit transfer is on one subset or component of the cube described in Figure 1-1. Other candidate studies to be used in part of a transfer may consider separate issues. One research study could, for example, consider the effects of water quality on housing prices in an estuarine area, and a second study might evaluate the recreational benefits associated with the improvement. There are inevitable overlaps in the two studies; therefore, how do we combine and reconcile existing results? This is a critical first step in developing methods for a benefit transfer. To adequately address these connections it may be important to, as explicitly as possible, recognize the interrelationships between the economic benefit measures and between physical systems involved.

What is an alternative approach to benefits transfer?

This report considers an alternative approach and a somewhat different perspective on the practice of benefit transfers. The method that is developed here treats the benefit transfer problem as one requiring the identification of individual preferences for the environmental resources of interest. The most important practical insight from the approach is a requirement that each source of benefit estimates and each desired decomposition of these estimates should, in principle, link to a common specification for individual preferences. This type of overall framework describes how the environmental resources and their quality contribute to individual well-being. Moreover, it also summarizes how other changes in an individual's (or a household's) circumstances might change their economic valuation of the resource change.

In practice, this means that the analyst must first be willing to make explicit assumptions about the functional form of an individual's utility function, as it relates to the resource and environmental quality change of interest. A utility function, in this case, is one that expresses how the consumption of a particular good or service (C_1) is related to the environmental quality of a specific resource and to other goods and services (C_2), and how these factors jointly contribute to the well-being (i.e., utility [U]) of an individual. In general terms it can be expressed as:

$$U = U(C_1, C_2; Q, \alpha) \qquad (1.1)$$

α represents parameters that help to define the "shape" of this function, and Q represents a measure of environmental quality. A specification such as this should allow the analyst to derive the corresponding indirect utility function (V), or alternatively the analyst could begin with an assumption about the functional form of the indirect utility function. In either case, V represents the maximum level of utility achievable, given the income (m), relative prices for C_2 and C_1 (P), and level of environmental quality faced by the individual:

$$V = V(m, P, Q; \alpha) \qquad (1.2)$$

α, again, represents parameters that help to define the "shape" of this function.

WTP for a change in environmental quality from $Q_{\hat{0}}$ to $Q_{\hat{1}}$ can therefore be expressed as the reduction in income that would exactly offset the improvement in Q (i.e., $Q_1 > Q_0$) and leave utility unchanged.

$$V(m, P, Q_{\hat{0}}; \alpha) = V(m - WTP, Q_{\hat{1}}; \alpha) \tag{1.3}$$

Assumptions about the functional form of utility should then allow the analyst to express WTP as a function of the change in environmental quality $(Q_{\hat{1}} - Q_{\hat{0}})$, income, prices, and α.

$$WTP = f\,((Q_{\hat{1}}, Q_{\hat{0}})\, m, P; \alpha) \tag{1.4}$$

This function is, in essence, a benefit transfer function; however, the key feature that distinguishes it from other benefit transfer functions is that, by definition, it is derived from, and thus consistent with, the specification of preferences (i.e., the utility functions).

The second element of this approach is that, rather than using existing studies or evidence to measure WTP directly, it uses these studies (or in some cases careful assumptions) to estimate the parameters in α. In other words, it uses existing studies to "calibrate" a preference structure and, therefore, a WTP function as well. The WTP function can, in principle, be transferred and applied to evaluate different degrees of environmental quality changes that are relevant for policy purposes.

The process described above illustrates the fundamental steps and logic of the proposed alternative approach. The framework can be expanded to include more alternative uses of water and different motivations (or individual-specific characteristics) that underlie why a consumer is willing to pay for water quality improvements. One important objective of this report is to provide a more detailed description of the approach, in part by presenting illustrative applications and by demonstrating how the process and results of this approach contrast with those of more traditional benefit transfer practices.

What are the main advantages and disadvantages of this alternative approach?

The proposed approach offers a more systematic way to construct benefit measures under the time and resource constraints typically facing policy analysts. As described above, the primary advantage of this approach is that it provides a means of generating benefit estimates that are more consistent with individual behavior. This is because the estimates are designed to take explicit account of the assumptions regarding individuals' preferences and the constraints they face. As such, this approach permits the integration of multiple estimates of the value of nonmarket resources and helps to ensure consistency between economic benefit measures for different resource uses. As a practical matter, however, increases in the diversity of benefit measures incorporated into the analysis will necessarily add to the difficulties posed for applying the approach.

The approach also makes explicit the roles of analyst judgment in developing the connections between what has been measured and what is needed for each policy task. Analysts must gauge whether the existing literature contains sufficient information to link what is known about the economic worth of different types of environmental resources to what is needed for evaluating some change to one or more of them. The approach does place a burden on the analyst to specify the structure of preferences. This structure must be specified in such a way that the critical utility parameters (the components of α) can be reasonably inferred from existing data and studies. There is also an important strategic element to selecting the functional form of the utility function so that it can be mathematically manipulated to derive an applicable WTP function.

To illustrate the general logic of our analysis, this report provides several algebraic examples. However, the method is not simply a matter of detailed algebra. Rather it is based on a recognition that a set of economic consistency conditions should be a part of the methods used to transfer benefit estimates.[7] While this report is the first time (to our knowledge) this strategy has been used, elements of the logic are implicit in most benefit transfers. Thus, even if the algebraic details and assumptions are considered too demanding,

[7]In principle the same types of arguments would apply to cost transfer studies.

the framework may prove a useful way to organize and evaluate simpler methods in practice.

In Section 2, we begin with a basic model valuing a water quality improvement that is primarily related to outdoor recreation. In Section 3, we provide more detailed examples of the proposed methodology. These examples are designed to demonstrate how information from a hedonic study, a travel cost study, and a CV study can be selectively combined to calibrate specifically defined utility functions. We then demonstrate how the calibrated functions can be used to transfer benefit estimates to a separate context and how the resulting benefit estimates differ from those of a more traditional benefit transfer practice, hereafter labeled "simple approximation."

By describing and illustrating our proposed approach, this document reports on research in progress. There are a number of ways in which this research can and will be extended. In particular, the proposed approach raises at least three important issues for future advances in the application of benefit transfer. First, what are the advantages of using more (or less) complex specifications for the assumed underlying preference structure? Second, how can multiple benefit estimates best be integrated into a preference calibration process? Third, how can the advance specification of preferences be used to evaluate the benefit transfer process and results? Each of these issues is addressed in more detail in Section 4, which discusses next steps in the application of the preference calibration approach.

2 An Introduction to the Deductive Approach to Benefit Transfer

When it is not possible to conduct new research to evaluate the benefits of a proposed policy, the usual practice involves translating the anticipated effects of that policy into changes in the prices, quantities, or qualities of commodities that people want. When the policy applications involve price or quantity changes for marketed commodities, components of market exchanges are observed and the primary focus of policy analysis is to use the observed exchanges in approximating a change in consumer surplus. The situation is more complex for policy changes associated with environmental applications. Even in cases where the affected environmental resources have direct uses, they do not as a rule have prices. A common practice in policy applications is to use some measure of consumer surplus that is reported in the existing literature to compute an average consumer surplus per unit of the change evaluated. This per-unit value is then multiplied with the amount of change implied by the policy. Both the mean benefit and the benefit function approaches to transfer rely on the conceptual validity of per-unit consumer surplus measures. Probably the most common example of this per-unit approach can be found in land management agencies' use of unit values for planning. As a result, considerable effort has been devoted to estimating consumer surplus measures per trip to provide the unit values for different types of recreational activities (see Bergstrom and Cordell [1991] as an example).

The logic underlying these types of computations likely stems from approximations frequently used for marketed goods and introduced by Hicks (1940-41) and Harberger (1971). Unfortunately, the properties of these approximations do not easily transfer to situations where we cannot assume that people are making choices on a unit price and quantity basis. By converting consumer surplus estimates to this format, analysts can make significant errors. One way to avoid these problems is to consider a different approach for transferring benefit estimates from existing studies to policy applications. This method involves using the available estimates in a way that entails calibrating a function intended to describe consumer preferences. Calibration in this context means using the estimates to establish numerical values for parameters (e.g., α) that shape a specified preference function (usually an indirect utility function). With such a calibrated function it should then be possible to develop the required benefit measures for each new policy to be evaluated.

The purpose of this section is to explain the logic of the simple approximations often used in practice and why they do not easily "fit" the context of most environmental applications. Following that discussion, Section 2.2 illustrates how one can improve the consistency of transfers using a case study for fishing benefits in the Willamette Basin. This approach is compared with the logic implied by the Hicks-Harberger approximation.

The third part of this section extends this reasoning by illustrating how the simple approximations adopted in developing estimates of the Marshallian consumer surplus attributed to a quality change may be inconsistent with any underlying preference function. The objective of this discussion is not ultimately to discourage benefit transfer. Instead we suggest that, for large changes where the restrictions on "ability to pay" or the effects of simultaneous price changes may be important, developing transfers that incorporate these restrictions may be necessary. As proposed in Section 1, the most direct way to meet this objective is to use available information to calibrate an indirect utility function.

Section 3 illustrates this idea with two applications. The first uses the Mitchell-Carson estimates of the value of water quality with travel cost demand-based estimates of the recreation benefits arising from water quality improvements. The second also begins with the Mitchell-Carson estimates and limits them to hedonic property value models. In both situations, alternative simple transfer approximations are also used to illustrate the potential differences. Before turning to the specifics, it is important to add a caveat. Our numerical computations are intended to be illustrative—a number of simplifying assumptions and approximations were made to permit the use of readily available information. For a full-scale transfer using the preference calibration methodology, each of these assumptions for convenience would need to be revisited. For our purpose, they are not crucial because none of them is a requirement to use the logic implicit in the proposed method.

2.1 APPROXIMATING CONSUMER SURPLUS MEASURES

Following Harberger's (1971) overview, a common approach to measuring the consumer surplus for price changes in one (or more goods) has been to use the *observed* change in the quantity demanded for the good(s) (in response to the price change[s]) weighted by the average of the two price values (for each good if there is more than one.) For example, if P_0 is the initial price and P_1 the new price with q_0 and q_1 the corresponding quantities demanded, then an approximate measure of the consumer surplus for this price change is given by Eq. (2.1):

$$CS_1 = \frac{1}{2}\left(P_0 + P_1\right)\left(q_1 - q_0\right) . \tag{2.1}$$

As Diewert (1992) recently explained, first-order approximations to compensating (WTP) and equivalent variation (WTA) measures of the consumer surplus changes can also be expressed in similar terms. Eq. (2.2) provides the compensating variation (CS_2) and Eq. (2.3) the equivalent variation (CS_3) approximations:

$$CS_2 = P_1 \left(q_1 - q_0\right) \tag{2.2}$$

$$CS_3 = P_0 \left(q_1 - q_0\right) . \tag{2.3}$$

As a result, it is straightforward to see why CS_1 can be interpreted as an average of these two approximations.

This logic relies on the fact that benefit measurement is focused on some policy-induced change in prices *and* the ability to observe the quantity associated with each of the old and new prices.[1] What is important about this background for the use of benefit transfer is the general logic. Analysis is focused on measuring quantity changes and then valuing them by some per-unit "value." The process was intended to fit the case of price changes.

Unfortunately, benefit transfer adopted the same logic for a wider range of applications. As noted at the outset, policy changes affecting access or quality were translated into quantity changes and consumer surplus measures used to compute per-unit benefit values. These average consumer surplus measures or per-unit benefits were then applied to the estimates of quantity change. Ideally, for cases where there is not a per-unit price one would want to use the virtual price (or the price that would make the individual choose exactly the level of the nonmarket good he or she actually receives). However, the rule is never met in practice.

Benefit transfers usually proceed in four steps:

1. Translate the policy change into one or more resulting quantity changes for uses that are linked to an environmental resource.
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.

[1]It is also possible to apply them to multiple market price changes. See Smith (1987) for a comparative evaluation.

4. Combine estimates in Steps 1 through 3 for each year considered in the analysis and compute the discounted aggregate benefit measures.

Sometimes Steps 1 and 2 are combined. Notice that if we isolate the process in this way the result can be rearranged to resemble an approximation to a WTP measure. Eq. (2.4) translates the steps to an equation:

$$CS_P = \frac{CS_T}{\Delta d_T}\left(d_1 \bullet N_1 - d_0 \bullet N_0\right) \qquad (2.4)$$

where

d_i = the amount of use permitted by policy change (i = 1) and in absence of the policy change (i = 0),

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, and

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

The connection between Eqs. (2.4) and (2.2) arises when we interpret $d_i \bullet N_i$ as an aggregate counterpart to q_i. This is probably reasonable given the link (left out of our discussion to this point) of the policy to q and d in the first place.[2] What is not as easily justified is the connection between $CS_T/\Delta d_T$ and P_1. At best, $CS_T/\Delta d_T$ is an average value for a representative person per day (or per trip) depending on how d is measured. A measure that is

[2]As a rule, quantity changes are assumed to increase the amount or the quantity of a particular type of use that is supported by a specific environmental resource. For example, improving water quality at a specific river or lake is assumed for the purpose of benefit transfer to increase the quantity of a specific type of recreation that a resource can support. Table 2-1 illustrates this point with improvements in the water quality for the Williamette River, increasing the amount of different types of fishing and allowing uses that involve contact with the water (e.g., swimming, water skiing).

Table 2-1. Benefit Transfer for the Willamette River Basin[a]

Example Activity	Activity Measures: Without CWA	Activity Measures: 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, Richards, and Scott (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)
Warm water (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).

theoretically consistent would be the marginal value of the quality change provided that change is measured in the same effective units as d.[3] With this amendment Eq. (2.4) would be a first-order approximation of the exact benefit measure. Of course, in practice the relevant question is how much do these differences matter. We now turn to the steps required to evaluate this issue and illustrate them with an example.

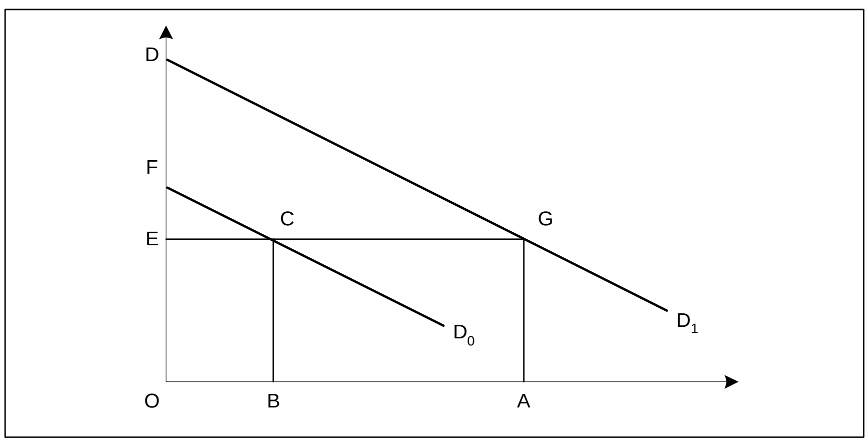
2.2 DIFFICULTIES WITH SIMPLE APPROXIMATIONS

The benefit analysis reported in Bingham et al. (1997) and developed by Industrial Economics, Inc., for the Williamette River Basin fits the basic logic outlined above. Water quality

[3]As Morey (1994) has suggested, there are *not* simple connections that can be made in these situations with quality changes. Smith (1992) also discusses the issues in using such averages as approximations.

improvements attributed to the CWA were assumed to increase the fishing trips for different species tenfold, from relatively low levels to high levels. Estimates of consumer surplus per trip (or per day) were used to value the changes in activity levels. A comparable strategy was used to estimate the economic benefits attributed to other forms of recreation (e.g., swimming, windsurfing, water skiing). In this use the pre-CWA use was assumed to be zero due to bans on swimming prior to 1972. Table 2-1 summarizes a few of the selected estimates reported in that earlier analysis. Figure 2-1 illustrates the implicit logic underlying the estimates developed for water quality induced increases in fishing. We can use it to explain the difficulties posed with the adaptation of approximations intended for price changes. D_0 describes the pre-CWA demand for fishing and D_1 the post-CWA demand. We assume here 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, assuming that OE is the average travel cost to use the site for fishing.

Figure 2-1. Illustration of Logic of Willamette Analysis



The benefit transfer logic interprets estimates of DEG/OA, consumer surplus per trip for the desired fishing experience, as the equivalent of a marginal value (or virtual price.) The benefit measure given in Table 2-1 is then

$$\left(\frac{DEG}{OA}\right) \bullet BA \qquad (2.5)$$

where BA is the increased fishing trips taken because of the water quality improvement. Notice that this assumes we have been able to identify a site that provides "exactly" the same recreation experiences as the improved Williamette will offer. Its demand is DD_1. As the discussion in Section 1 suggested, based on the Kirschhoff, Colby, and LaFrance (1997) evaluation of benefit transfer, differences in site characteristics between the study and the policy sites can be quite important to the validity of transferred estimates. 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 treating consumer surplus per unit as the equivalent of a price.

At the simplest level, consistent transfer would seek DFCG and not the expression given in Eq. (2.5). We can use geometry and the results from Table 2-1 to illustrate the extent of the mistake.

Suppose we assume that DD_1 is completely appropriate for the demand for the fishing activities provided by the quality improvement. The logic used in the calculations reported in Bingham et al. assumes OB is a constant multiple of the activities currently observed. In this case it is 10 percent. To keep the analysis somewhat general, we assume $OB = \gamma \bullet OA$.

The desired benefit measure is DFCG = DEG – FEC. Assuming that quality leads to a parallel shift in FD_0 to DD_1, we can simplify matters using the following relationships for the areas of the two triangles:

$$DEG = \frac{1}{2} DE \bullet OA$$

$$FEC = \frac{1}{2} FE \bullet OB = \frac{1}{2} (\gamma DE)(\gamma OA) .$$

Simplifying the expression for DFCG, we have Eq. (2.6) expressing the desired benefit measure:

$$DFCG = \frac{1}{2}\ DE \bullet OA\left(1-\gamma^2\right) \quad . \tag{2.6}$$

The expression given in Eq. (2.5) for the usual benefit transfer method can be expressed in terms of DE, OA, and γ as

$$\left(\frac{DEG}{OA}\right) \bullet BA = \left(\frac{\frac{1}{2}DE \bullet OA}{OA}\right) \bullet \left(1-\gamma\right) OA = \frac{1}{2} DE \bullet OA\left(1-\gamma\right) \quad . \tag{2.7}$$

This geometry implies we have a relationship between the "correct" benefit measure and the simple approximation. Taking the ratio of Eq. (2.6) to (2.7), we see that the correct measure is $(1 + \gamma)$ times the approximation or 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 activities involving water contact (provided again DD_1 is the correct demand), the approximation in Eq. (2.5) is correct because the quantity measure is assumed to be zero with the pre-CWA water quality conditions.[4]

This development illustrates how, if we are prepared to make assumptions, it is possible to develop transferred benefit estimates that are more consistent (in logical terms) with the changes that are assumed to be provided by the policy. In the next section, we take this argument a step further to illustrate how the quality-quantity link implicit in the shift of the demand function can be made explicit. However, the consistency issue does not stop here because the only requirement imposed by this example is that the quality improvement causes a parallel shift in the demand function. If we hypothesize that quality reduces the effective price (a

[4]Demonstration that the swimming estimates would be correct (given that DD_1 is correct) follows directly. The transferred benefit measure is

$$\frac{DEG}{OA} \bullet OA$$

and the desired measure is DEG.

common assumption in hedonic models), then we must go further to include this requirement. Equally important, the analysis to this point has been "vague" on whether DD_1 is a Marshallian or Hicksian demand. It has not explicitly included substitutes or the role of income. It does not recognize that prices are to be measured relative to those for other goods and services. While the importance of each of these considerations will vary with the application being considered in a benefit transfer, it is desirable to develop the underlying logic and associated framework so that they are capable of accommodating these added details.

In the next section, we illustrate the general logic considering the link between Marshallian and Hicksian measures of the value of a quality change. The analysis largely presumes that the evaluation is done within the context of travel cost demand, but it does not require this approach to nonmarket valuation. It can be readily generalized to CV or hedonic models.

2.3 RECOGNIZING THE IMPORTANCE OF CALIBRATION

The previous section illustrated the importance of how we represent environmental quality changes. If they are assumed to shift the demand function for a recreation site whose use depends on that quantity, and the baseline level of recreation use is not zero, then simple Hicks-Harberger approximations of consumer surplus can be misleading. This conclusion follows from the properties of partial equilibrium demand functions that shift with changes in environmental quality.

The two transfer approaches illustrated with the Willamette case also resemble situations where a per-unit benefit measure is transferred rather than a benefits function. In this case, however, it is the demand function for recreation, and in particular knowledge of how it shifts with water quality, that is transferred. Many transfers referred to as using a benefit function approach in fact rely on a "reduced form equation" describing how the consumer surplus measure varies with demographic characteristics.[5]

[5]That is, all the information necessary to define the preference structure is not available. Therefore, the analyst employs a reduced form rather than a structural equation.

Both approaches make assumptions that become progressively more important as the scale of the change increases. Whether we use the consumer surplus per unit or information on the quality effects on the demand for recreation, we implicitly hold income constant and especially any role "ability to pay" has in limiting monetary measures of the value of quality change.[6]

Efforts to reconcile existing benefits to a consistent behavioral structure are important for additional reasons. They force the analyst to consistently account for the role of quality in behavior that can be observed. To develop this point, consider an example where quality is assumed to enhance the "effective" services provided by a recreation site. Eq. (2.8) uses this augmentation form in describing the direct utility function for a representative individual. This Cobb-Douglas specification assumes the individual's well-being is related to recreation (C_1) and all other goods (C_2) as in Eq. (2.8):

$$U = \left(A(W)C_1\right)^{\alpha} C_2^{\,1-\alpha} \quad . \tag{2.8}$$

In this specification A(W) is the augmentation function. It describes how enhancements to water quality, W, increase the effective services provided by the recreation site through C_1. The introduction of A(W) assumes that the quality improvement increases the effective amount of C_1 available. The explanation behind Figure 2-1 and the analysis of the Willamette River is somewhat different from what the utility specification implies in Eq. (2.8). As quality improves (i.e., realized through increases in A), the amount of C_1 required by an individual to maintain her overall well-being at a constant level (i.e., $A(W) \bullet C_1$) actually declines. Thus, the Hicksian or compensated demand for C_1, describing what is required to maintain utility, decreases with increases in W. It is possible to establish this result by deriving the indirect utility function and expenditure function that correspond to

[6]This issue is seen directly in one of the Willig (1978) conditions for relating Marshallian and Hicksian measures for the value of a quality change. The change in consumer surplus due to a quality change per unit of the linked good must be independent of income.

constrained utility maximizing behavior. Eq. (2.9) describes the indirect utility function and Eq. (2.10) the expenditure function:

$$V = \left(\frac{P}{A(W)}\right)^{-\alpha} m \bullet a \tag{2.9}$$

where P = the relative price of C_1 to C_2, with the latter normalized to unity; m = income; and a = constant scaling factor:

$$m = \left(\frac{P}{A(W)}\right)^{\alpha} \frac{V}{a} \;. \tag{2.10}$$

We know that the partial derivative of the expenditure function with respect to the price of C_1 yields the compensated demand function as in Eq. (2.11):

$$\frac{\partial m}{\partial P} = C_1^* = \left(\frac{\alpha}{a}\right) P^{\alpha - 1} \bullet (A(W))^{-\alpha} \bullet V \;. \tag{2.11}$$

In logarithmic form, this suggests Hicksian demand for C_1 shifts in as W increases. This is seen in Eq. (2.12):[7]

$$\ln C_1^* = \ln\left(\frac{\alpha \bullet V}{a}\right) + (\alpha - 1)\ln P - \alpha \bullet \ln(A(W)) \;. \tag{2.12}$$

Although this seems to contradict Figure 2-1, to interpret what it means consider the behavior described by the compensated demands associated with this utility function: the amount of effective services of C_1 the consumer realizes is larger for each P. Re-arrange Eq. (2.11),

[7]In general, α is the share of total expenditures on C_1. As a result, we can assume it is less than one, and $(\alpha\text{-}1) < 0$.

$$\text{at}\quad W_0:\quad A(W_0)^{\alpha}\bullet C_1 = \left(\frac{\alpha}{a}\right)P^{\alpha-1}\bullet V$$
$$\text{at}\quad W_1:\quad A(W_1)^{\alpha}\bullet C_1 = \left(\frac{\alpha}{a}\right)P^{\alpha-1}\bullet V \quad . \tag{2.13}$$

If $A(W_1) > A(W_0)$ and nothing else changes, then we see that to maintain a constant utility level, C_1 is reduced because the consumer receives a greater amount of effective services from the same unit of C_1. The Marshallian demand function for C_1 does not reveal an effect for W. This is important to our example because it is commonplace in benefit transfer to make some fairly specific assumptions about substitution between price and quality. To describe how they are made and relate them to the suggestion that calibrating to a consistent behavioral function is desirable, consider first the form of Marshallian demand.

Applying Roy's identity[8] to Eq. (2.9), we derive the Marshallian demand in Eq. (2.14) and the Marshallian consumer surplus for trips to a recreation site with W* water quality in Eq. (2.15). Neither function includes water quality. The analyst must recognize the difference in site conditions. With the augmentation specification for preferences, quality effects are seen through the Hicksian demand but not the Marshallian. Nonetheless, analysts often "build in" quality effects in the ways the demand functions are used:

$$C_1 = \alpha\bullet\frac{m}{P} \quad . \tag{2.14}$$

The logic of this process assumes a recreationist has an array of possible recreation sites near his (or her) home at different distances. As a rule, we assume higher quality sites can sustain other activities at all quality levels below their existing quality conditions. Thus, a lake that supports swimming can also support

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[8]Roy's identity provides the link between the Marshallian demand and the induced utility function:

$C_1 = -(V_P/V_m)$, with V_i = partial derivative of V with respect to element.

game fishing and boating because the water quality conditions required for these activities are less than that required for swimming.

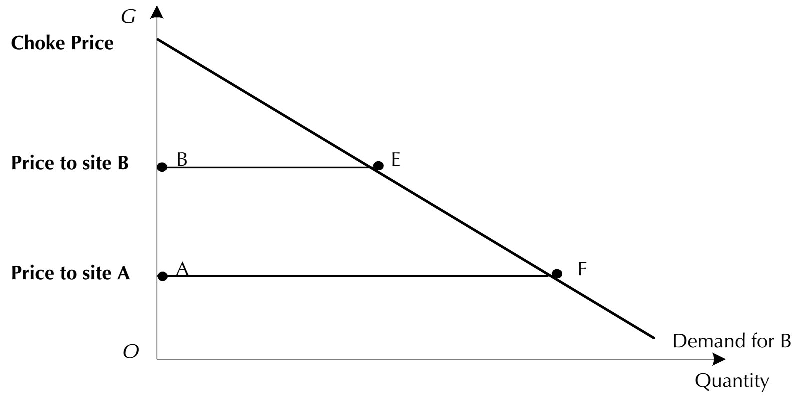
Recognizing this assumption, we assume that people's travel behavior embodies a desire to obtain the required water quality (for the activities they plan to undertake) at least cost. This logic also maintains that the water quality is the only difference in recreation sites. In this context, we are assuming an individual is adjusting the travel costs to reflect the water quality. This argument is consistent with what is generated by the Hicksian demand under an augmentation format (e.g., prices are adjusted up or down based on the quality of the services a site provides). Benefit estimates derived using this logic can be seen as any simple approximation of the Hicksian measure for the value of a quality improvement.

Here is how the specific case works. When confronted with the need to value a quality change, analysts often suggest that improving quality at a specific site is equivalent to reducing the "price" of using a higher quality site. This logic is what the augmentation model implies for price in the context of Hicksian demand. In practice, the concept is often approximated by describing how a set of consumers' choices would change with a quality change.

Suppose we have two lakes: A has water quality level sufficient to allow boating but not swimming, and B has a water quality level that permits swimming (and therefore also boating). In the absence of other differences (such as congestion at each site), recreationists who want to both fish and swim would likely use site B and not A. When we observe them using site A, it is usually because A is closer to their homes.

With this background, then, the logic of the transfer associated with improving water quality at site A is described by a process that suggests the quality change is "like" moving site B closer to their homes. That is, the price (travel cost) of using site B has reduced to the costs to visit A, meaning the higher quality conditions are now available at lower cost. This is a specific substitution assumption because it assumes the improved A is a perfect substitute for B; thus, the gain is measured as a price change along B's demand function, as shown in Figure 2-2.

Figure 2-2. Quality Treated as a Price Change



The consumer surplus associated with the quality change is then measured as ABEF. This is often operationalized by considering the area under the Marshallian demand for site B (i.e., the one with the initially higher water quality) for a price change from P_{OB} to P_{OA}.

At this point, a reader might ask—why undertake this type of approximation if we know the water qualities at the two sites? The answer is direct. There may not be sufficient information about quality conditions and how users perceived them to measure their role in demand for the sites. Or, alternatively, the analyst may simply have two demand studies and recognize that this quality distinction is what gives rise to the difference between the sites. The area ABEF is measured as the difference in two triangles (AGF – BGE): the consumer surplus for recreation at price OA less the consumer surplus for price OB. In terms of our Marshallian demand function (derived from the Cobb-Douglas utility function), this is the gain for the price change OB to OA as in Eq. (2.15):

$$MCS = \int_{P_{OA}}^{P_{OB}} \frac{\alpha m}{P} dP = \alpha m \left[\ln P_{OB} - \ln P_{OA} \right] . \qquad (2.15)$$

At this stage, the logic implicit in Eqs. (2.9) or (2.10) is being used since quality improvement is treated as the equivalent of a reduction in the effective price. The expenditure function implies that improvements in W serve to reduce the effective price, because P/A(W) enters the indirect utility function Eq. (2.9). This reasoning suggests that the effects of price changes (or well-being) depend on quality.

Thus, to consider the specific water quality conditions we assumed for the two sites, this price change describes the quality improvement from boatable (W_B) to swimmable (W_s) and is represented as

$$\frac{P_{OB}}{P_{OA}} = \frac{\overline{P}/A\left(W_B\right)}{\overline{P}/A\left(W_S\right)} \tag{2.16}$$

or

$$\ln P_{OB} - \ln P_{OA} = \ln\left(A\left(W_S\right)\right) - \ln\left(A\left(W_B\right)\right) . \tag{2.17}$$

Substituting into Eq. (2.15), we get Eq. (2.18)—we complete the logic implied by the approximation. However, this description for valuing water quality improvement is not behaviorally consistent. The size of the mistake depends on the importance of C_1 in the individual's consumption. The importance of C_1 can be derived by comparing Eq. (2.18) with the Hicksian measure for the same quality change:

$$MCS = \alpha \bullet m\left[\ln(A(W_S)) - \ln(A(W_B))\right] . \tag{2.18}$$

The term in brackets is approximately the percentage change in the adjustment to recreation services (i.e., the effective units discussed earlier) that is attributed to the water quality change. Thus, we could approximate the Marshallian surplus as

$$MCS \approx \alpha \bullet m \left[\frac{A\left(W_S\right) - A\left(W_B\right)}{A\left(W_B\right)}\right] . \tag{2.19}$$

The Hicksian measure of the WTP for a quality improvement from W_B to W_S is given in Eq. (2.20):

$$HCS = \left(\left(\frac{P}{A(W_B)} \right)^{\alpha} - \left(\frac{P}{A(W_S)} \right)^{\alpha} \right) \frac{V}{a} \quad . \tag{2.20}$$

Expressing HCS per unit demanded (with the Hicksian demand function, C_1^H) we have Eq. (2.21) for Hicksian surplus per trip:

$$\frac{HCS}{C_1^H} = \frac{P}{\alpha} \left[\frac{(A(W_S))^{\alpha} - (A(W_B))^{\alpha}}{(A(W_B))^{\alpha}} \right] . \tag{2.21}$$

Comparing Eq. (2.21) with Eq. (2.19) is difficult because the base income level (and prices) will influence the utility level that can be realized. It may be easier to highlight the difference by considering the surplus measure per unit of C_1.[9]

The two expressions for consumer surplus per unit of C_1 indicate that this approximation does maintain that the consumer surplus gain per unit of C_1 is independent of income, as required by the Willig consistency requirement. It does not include all the preference conditions correctly.[10]

The MCS approximation per unit of C_1 is given in Eq. (2.22):

$$\frac{MCS}{C_1} = P \left[\frac{A(W_S) - A(W_B)}{A(W_B)} \right] . \tag{2.22}$$

[9]Characteristics of the Willig condition (identified in footnote 6) suggest that a key consideration (in addition to weak complementarity) in the relationship between Marshallian and Hicksian measures of quality change is the degree to which the Marshallian consumer surplus per unit C_1 changes with income (i.e., how does [MCS/C_1] change with m).

[10]This formulation also does not impose weak complementarity.

Comparison of the Marshallian and Hicksian measures (per unit of use) suggests that the differences will depend on the importance of C_1 in the overall budget, which is represented by the parameter α. Table 2-2 provides a few examples for one potential change in water quality where there is a 20 percent improvement in water quality (i.e., assuming $A(W_S) = 1.2$ and $A(W_B) = 1.0$). While the differences seem rather small for the applications, when scaled by the number of affected individuals errors ranging from 8 to 9.1 percent can translate into large differences in the aggregate benefits.

Table 2-2. Illustration of Approximation Errors Due to Simple Transfer

Travel Cost	Fraction of Budget (α)	MCS/C_1	HCS/C_1
Middle TC			
$100	.02	$20	$18.26
$100	.04		$18.29
$100	.10		$18.40
Low TC			
$25	.02	$5	$4.57
$25	.04		$4.57
$25	.10		$4.60
High TC			
$250	.02	$50	$45.66
$250	.04		$45.75
$250	.10		$46.00

2.4 IMPLICATIONS

Consistency in benefit transfers requires that the measures of benefits incorporate the limitations imposed by income and other constraints on what a person can pay for some quality improvement. In addition, when a quality change is treated as equivalent to increased capacity for recreation (e.g., as an increase in the quantity of services that can be provided by a recreation site) or as a price reduction (e.g., higher quality services are now closer to users), the methods for introducing these approximations into benefit measurement should be consistent with the way quality is hypothesized to influence consumer preferences.

As Section 2.2 illustrated, this can be as simple as recognizing the properties of a demand function shifting with a quality change. That is, if the baseline level of use is not zero, simple approximations can introduce errors. This is also true when quality change is treated as equivalent to a price change. Using a Cobb-Douglas example, we illustrate errors ranging from 8 to 9 percent due to simple approximations. Of course, at this level we do not know how to adjust the value measure for differences in income levels or the access conditions to available substitutes.

Indeed, the overall logic of multiplying a quantity change by a price is actually an approximation defined for cases involving price changes, not the quality changes. To develop further insight into the size of the errors introduced by such simple approximations, we must be explicit about consumer preferences, how and what we observe of these preferences, and finally how we use these measures to develop benefit estimates for policy.

3 Implementing the Logic with Two Examples

As the earlier Sections suggested, a deductive strategy for benefits transfer requires the analyst to parameterize, in specific terms, how environmental resources enter consumer preferences or a utility function. That is, a deductive strategy develops a model that describes the economic choice assumed to underlie the valuation measure. This definition is the first step in determining the additional information required to identify calibrated estimates as a function of the preference parameters. Such estimates of calibrated transfers will be based on these parameter estimates as well as the prior information that describes the resource or environmental quality attribute(s) and the household characteristics in the study site. The goal then is to identify an indirect utility function that can be used to define the WTP for water quality changes related to alternative policy scenarios. This calibration strategy assumes, of course, that the individual's preferences conform to those in the calibrated indirect utility model.

The models in Section 2 considered different types of errors that arise from some of the simple approximations used in transfer. The first of these focused on evaluating how the use of a link between a quality change and the assumed amount of recreation makes the benefit approximations sensitive to the assumptions made about baseline resource quality and baseline recreation use. The second model considers the use of price changes as proxies for quality changes. Here, too, errors can arise because the simplifying assumptions are often not consistent with the role of quality in reduced-form behavioral models (e.g., either indirect utility or the expenditure functions) that are assumed to underlie the simplifications in logic. This Section takes a more direct approach to extending the basic logic. Instead of highlighting the sources of mistakes in current practice, the Section describes how the information usually available can be used to calibrate indirect utility functions so that they can provide the basis for estimating a representative individual's WTP for a quality change that is different from what was considered in the original source study. The examples include Marshallian consumer surplus associated with different types of recreation use (derived from a travel cost study) or the marginal WTP for water quality attributes of housing (derived from a hedonic property study). Either of these two sets of estimates would be sufficient to "construct" consumer preferences if the analyst were willing to make some assumptions and impose some restrictions on preference parameters. On the other hand, the process of selecting multiple estimates from the available literature, in principle, reduces the restrictions necessary to specify an indirect utility function and to use this function to infer the value of a policy alternative.

To illustrate the process of what we have labeled calibrated benefit transfers, we develop two numerical examples in this Section. The first involves valuing water quality improvements that are related to recreation attributed to those increments in water quality, and the calibration is implemented using the Mitchell-Carson (1984) estimates of the value of water quality changes along with estimates in a recreation demand model by Englin et al. (1997).[1] The second uses a study by Michael, Boyle, and Bouchard (undated) that evaluates the effect of water quality on lakefront property value in Maine along with the Mitchell-Carson estimates. In each case, we also compare the implications for benefits estimation of using the proposed calibrated preference function versus a simple

[1]The estimates and interpretation used in this analysis are taken from Carson and Mitchell (1993) who summarize the features of their 1983 survey and from the questionnaire reported in Mitchell and Carson (1981; 1984).

approximation to benefits transfer that uses a point estimate of the water quality benefits together with the proposed quality change to measure the incremental value.

3.1 TRANSFERRING RECREATION VALUES FOR WATER QUALITY CHANGES: A RECREATION DEMAND EXAMPLE

The first example uses two sets of benefits estimates to calibrate the parameters of an indirect utility function. As our discussion in Section 2 suggested, the selection of a specification for water quality in consumer preference has important implications. So while we would like to keep the logic as simple as possible, our desire must be tempered by the need to recognize how simplifications can lead to a set of behavioral functions that seem to contradict one or more of our beliefs about how changes in water quality influence economic behavior. Given this caveat, we followed Willig (1978) and Hanemann (1984) and adopted a specification that is consistent with what Willig labels "cross-product repackaging." This implies that the indirect utility function is structured so that the role of the water quality measure is restricted to serve as a reduction in the price of the related market commodity as in Eq. (3.1):

$$V = \left[\left(P - h\left(d_1\right)\right)^{-\alpha} m\right]^{b} \quad . \tag{3.1}$$

Because our example combines a recreation travel cost demand-based measure with the early Mitchell-Carson (1989) CV estimate, we treat P as the round-trip travel costs. h(d) is the function that describes how increases in water quality reduce the effective price of a trip. We assume that the recreation involves freshwater fishing and quality is measured with dissolved oxygen, d.

While several possible studies could provide estimates of the Marshallian consumer surplus change, we use the recent Englin et al. (1997) study that develops a link between dissolved oxygen, total trout catch in New England lakes, and a travel cost demand model. These authors' econometric analysis recognizes the count variable structure of both the trip and the catch measures. For our purpose, what is important is that they specify (indirectly) dissolved oxygen as a quality measure (through its influence on catch) in a recreation demand model.

Equally important, they report the average consumer surplus for improvements in dissolved oxygen for a set of lakes used by sampled residents of New York (excluding New York City), New Hampshire, Vermont, and Maine during 1989. The specific scenario we use involves an increase in the poorest lakes to a minimum dissolved oxygen level of 6.0 mg/liter.[2] This scenario is somewhat similar to the logic underlying the Mitchell-Carson CV question which asks about improving water quality in a group of lakes. 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 Resources for the Future (RFF) water quality ladder, which was the vehicle used to describe the implications of a quality improvement, this change corresponds to improving dissolved oxygen from about 3.5 mg/liter to 6.0 mg/liter. This is approximately the change considered in the Englin et al. (1997) analysis. Mitchell and Carson describe what is offered as 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).

As a result of this approximate correspondence, 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 Hicksian WTP.

To calibrate the preferences defined by Eq. (3.1), we need to relate each of these benefit measures to this common preference structure. Using Roy's identity, the demand for trips, C_1, can be expressed as Eq. (3.2) using Eq. (3.1):

$$C_1 = -\frac{V_P}{V_m} = \frac{\alpha m}{(P - h(d))} \quad . \tag{3.2}$$

[2]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. Thirty-eight of the 61 lakes used in their sample had dissolved oxygen below 6.0.

The Marshallian consumer surplus, MCS, associated with access to 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.[3] This is given in Eq. (3.3):

$$MCS = \alpha m \int_{P_0}^{P_c} \frac{1}{(P - h(d))} dP = \alpha m \ \ln(P - h(d)) \Big]_{P_0}^{P_c} . \tag{3.3}$$

When we evaluate the integral, the result is Eq. (3.4):

$$MCS = \alpha m \left[\ln\left(P_C - h(d)\right) - \ln\left(P_0 - h(d)\right) \right] . \tag{3.4}$$

The Englin et al. (1997) analysis implicitly evaluates how MCS changes with d. To evaluate what this would look like analytically with our preference specification, consider $\frac{\partial MCS}{\partial d}$ as in Eq. (3.5):

$$\frac{\partial MCS}{\partial d} = \alpha m \left[-\frac{h'(d)}{\left(P_C - h(d)\right)} + \frac{h'(d)}{\left(P_0 - h(d)\right)} \right] \tag{3.5}$$

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

If we bring αm into the bracket, the first term is seen as the demand for angling trips at the choke price 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 that the first of the terms on the right side of Eq. (3.5) is zero. The second offers a basis for linking one interpretation of the Englin et al. (1997) measures to our preference specification. More specifically, the increase in Marshallian consumer surplus per angling trip is exactly $h'(d)$ as in Eq. (3.6):

[3] Setting $C_1 = 0$ in Eq. (3.2) and solving for P does not yield a finite choke price because C_1 approaches zero as P assumes arbitrarily large values. For our purposes, we assume there is some large finite choke price.

$$\frac{\frac{\frac{\partial MCS}{\partial d}}{\alpha m}}{\left(P_0 - h(d)\right)} = \frac{\frac{\partial MCS}{\partial d}}{C_1} = h'(d) \quad . \tag{3.6}$$

To use this information, we need to specify h(d). For our example, we assume it follows a power function because the shape implies a declining marginal effect of d on the price, when $h(d) = d^{\beta}$ and β is a constant. Englin et al.'s (1997) consumer surplus estimates of the seasonal gain due to quality improvements, scaled by their estimates per trips, offer an estimate of the left side of Eq. (3.6). With the power function specification we can write $h'(d) = \beta d^{\beta-1}$. This is the effect of a quality adjustment on incremental consumer surplus per trip. We interpret $h'(d)$ as the Marshallian surplus estimate for the water quality change as described by Englin et al. (1997) (i.e., increasing dissolved oxygen at the worst lakes to approximately fishable conditions—6.0 mg/liter). This allows us to to use their estimate to recover an estimate of β. 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 taking 5.06 trips under the improved conditions. Using a series approximation for the derivative of the power function (i.e., $\beta d^{\beta-1} \approx \beta\,[1 + (\beta-1)\log(d)]$), we can express Eq. (3.6) as a quadratic, as in Eq. (3.7), and solve for the roots:

$$\log(d) \bullet \beta^2 + (1 - \log(d))\,\beta - \hat{a} = O \tag{3.7}$$

where $\hat{a} = [\ (\partial MCS/\partial d)/X_1\ ]$.

Each of the roots is a potential solution. We discriminate between the two roots for β based on their economic properties. This task is completed by solving for α from the expression for the WTP in Eq. (3.8) below, using each of the roots derived from Eq. (3.7) and then evaluating the predicted demand and the estimates of α. The latter should approximate the share of income spent on recreation.

As we noted earlier, Mitchell and Carson's CV question also corresponds to a WTP for a change in dissolved oxygen at water bodies with less than fishable conditions. We describe this water quality as a change from boatable (d_B) to fishable conditions (d_F). The WTP derived from this preference function (Eq. [3.1]) is then given in Eq. (3.8):

$$WTP = m - \left(\frac{P - h(d_F)}{P - h(d_B)} \right)^{\alpha} m \ . \tag{3.8}$$

Eq. (3.8) defines implicitly the WTP as the maximum exogenous income that can be taken away in the presence of a water quality improvement (from d_B to d_F) such that the recreator is equally well off with less income and better water quality as she was with more income and poorer water quality.

The roots to Eq. (3.7) provide estimates of β that allow h (.) to be evaluated for different values of d. As a result, with an estimate of WTP from the literature we can solve Eq. (3.8) for α. This result is given in Eq. (3.9):

$$\hat{\alpha} = \frac{\ln\left(\frac{m - W\hat{T}P}{m} \right)}{\ln\left(\frac{P - \hat{h}(d_F)}{P - \hat{h}(d_B)} \right)} \ . \tag{3.9}$$

Computations using Eqs. (3.7) and (3.9) identify a sufficient number of the parameters for the indirect utility function in Eq. (3.1).[4] The calculations for Eq. (3.9) 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).[5]

[4]While we cannot recover an estimate of b in Eq. (3.1) with this information, this parameter did not enter the WTP function (i.e., Eq. [3.8]; therefore, an inability to isolate it with this information does not preclude our calculation of WTP or demand for new sites.

[5]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.

This process yields two parameter estimates (one for each of the roots of Eq. (3.7) as given in Table 3-1).

Table 3-1. Solutions to Travel Cost Demand Calibration[a]

Root	$\hat{\alpha}$	$\hat{X}_1$	$\widehat{WTP}$
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 trip cost was $100 (including the time costs of travel).

The selection of an economically plausible root for Eq. (3.7) is clearest using $\hat{\alpha}$ and $\hat{X}_1$. Negative predicted trips are clearly implausible, as is a large (in absolute magnitude) value for $\hat{\alpha}$. In contrast, the first root provides a quite plausible estimate for both. The importance of this type of cross-checking is highlighted by the last column in the table. It reports a new estimate for the WTP to improve water quality from a baseline dissolved oxygen level of 4 mg/liter to 6 mg/liter. Notice that without the economic interpretation of α and the computation of $\hat{X}_1$ predicted trips, it would not have been possible to discriminate between the two solutions based on WTP alone, because each seems to offer a plausible WTP estimate. However, the second WTP estimate is based on obviously incorrect economic parameters.

Having thus calibrated all the necessary parameters (α and β), we are now in a position to compute WTP for alternative water quality changes. Table 3-2 reports some other illustrative computations varying the quality change. For comparison purposes, the last column in the table reports a simple approximation for estimated benefits using the Englin et al. (1997) measure per unit of dissolved oxygen and per trip as the unit benefit measure. In this approach, we divide $\hat{\alpha}$ (defined by Eq. [3.6]) 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." This quantity is then multiplied by the proposed change in dissolved oxygen and the predicted trips at the highest quality level. The difference (understatement) in benefit measure is clear from the results in Table 3-2.

Table 3-2. Illustrative Transfers of the Value of Water Quality Changes from Recreation Demand Models: Calibrated Versus Simple Approximation (1995 $)

Baseline Dissolved Oxygen (d_0)	New Dissolved Oxygen (d_1)	Trips at New Quality	WTP	Approximate Benefit
3	6	20.15	627.96	283.79
4	6	20.15	517.63	189.20
5	6	20.15	332.97	94.60
1	4	10.45	208.71	147.26
2	4	10.45	177.01	98.17

[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 trip cost was $100 (including the time costs of travel).

3.2 TRANSFERRING PROPERTY VALUES FOR IMPROVEMENTS IN THE WATER QUALITY ATTRIBUTE: A HEDONIC PRICE EXAMPLE

In this second example, we assume that the study site has two available sets of information. The first is a measure of the WTP for a proposed plan to improve surface water quality. We use, once again, Mitchell-Carson (1984) for this component; however, in this case we use a different physical measure of water quality and a different size of change in water quality. The second set of information is an estimate of the marginal WTP for water quality that is based on the results of a hedonic property value study by Michael, Boyle, and Bouchard (undated).

In considering the use of information from a hedonic property value model, we must use a different approach from the models developed in the recreation context because hedonic models generally provide an estimate for the marginal rate of substitution for environmental quality relative to some numeraire good (usually money). This estimate is the marginal WTP evaluated at a point. However, the ability to estimate this marginal WTP at this point does not necessarily imply it is possible to identify the full marginal WTP schedule. There are several reasons for this conclusion. Important among these is the fact that the analysis assumes consumers have different preferences and generally does not assume a specific form for the preference function.[6]

When we consider transfer from hedonic estimates, the approach must build in more assumptions. Following Quigley's (1982) argument, it is possible to use one estimate of marginal WTP to recover enough features of preferences (for the case of the CES as a specified preference function).[7] This calibrated preference function allows consistent benefits transfer, particularly because this function permits variations in income and water quality to be incorporated. With more information (than one estimate), it is possible to relax some of the restrictive assumptions.

The form of the CES function used by Quigley is itself specialized: the case of several attributes is given in Eq. (3.10). Note that, in this case, we assume 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:

$$V = \sum_{i=1}^{K} (\theta_i \bullet A_i)^b + (m - P(A_1, \ldots\ldots A_K))^b \tag{3.10}$$

where

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

[6]Feenstra (1995) is a notable alternative case. In his case, however, a specific form of preference heterogeneity is assumed to allow the demand behavior to be represented by a representative consumer's utility.

[7]CES is an abbreviation for the constant elasticity of substitution function. It is also possible to show a relationship between this specification as a generalization to the one used in our first example. This will be developed in future work.

The first order condition with respect to A_1 yields Eq. (3.11):

$$\frac{\partial V}{\partial A_1} = 0 = b \bullet (\theta_1 \bullet A_1)^{b-1} - (\frac{\partial P}{\partial A_1}) \bullet b \bullet (m - P(A_1, ...A_K))^{b-1}. \quad (3.11)$$

Re-arranging Eq. (3.11) shows that a point estimate of the marginal WTP, together with Mitchell-Carson estimate of WTP, allows the calibration of the b and θ_1 from the results of a single hedonic model. Conventional practice (e.g., Freeman [1974]; Smith and Huang [1995]) has proposed using simply the marginal value for benefit transfers. This practice is possible because the use of the $\partial P/\partial A_1 \bullet \Delta A_1$ does not require the knowledge of θ_1 and b. Of course, this approximation also assumes that the marginal benefit function is locally constant. As Eq. (3.12) indicates, the slope of the hedonic price function offers a point estimate of a composite of the parameters in the indirect utility function:

$$\frac{\partial P}{\partial A_1} = \theta_1^b \bullet (\frac{A_1}{m - P(\bullet)})^{b-1} . \quad (3.12)$$

Recognizing the role of θ_1 and b in Eq. (3.12) is the first step in recovering the parameters necessary to consistently transfer the value of nonmarginal changes in water quality,[8] measured as changes in A_1 in this hedonic price model.

The second estimate assumed to be available from a CV study is a measure of WTP for improving water quality, as described in the Mitchell-Carson CV study. We summarized the key elements in their question earlier. Eq. (3.13) defines the WTP for their proposed

[8]While analogous to Freeman's (1974) early suggestion for transfers, his framework focused on assumptions about the local shape of the marginal WTP in quality space. This strategy assumes individual preferences are identical and allows consideration of differences in income and price levels as well as the quality effects.

plan to improve water quality from A_1 to $A_1 + \Delta$ using the preference function defined in Eq. (3.10):[9]

$$(m - WTP)^b + \sum_{i=2}^{K} (\theta_i \bullet A_i)^b + (\theta_1 \bullet (A_1 + \Delta))^b \tag{3.13}$$

$$= m^b + \sum_{i=1}^{K} (\theta_i \bullet A_i)^b .$$

Eq. (3.13) defines implicitly the WTP as the maximum exogenous income that can be taken away in the presence of a water quality improvement (from A_1 to $A_1 + \Delta$) such that the property owner is equally well off with less income and better water quality as she was with more income and poorer water quality.

By rearranging terms, Eq. (3.14) is the Hicksian WTP for the improvement in water quality, the benefits measure that we seek:

$$WTP = m - (m^b + (\theta_1 \bullet A_1)^b - (\theta_1 \bullet (A_1 + \Delta))^b)^{1/b} . \tag{3.14}$$

We can use Eq. (3.12) to eliminate θ_1 from Eq. (3.14) and solve for b. With this estimate for b, it is possible to recover sufficient information about the indirect utility function to develop benefit estimates for proposed changes in A_1 for new applications. This process defines WTP in Eq. (3.14) in terms of the marginal hedonic price and b. With estimates of WTP from the Mitchell-Carson study, we solve for the implied estimate of b. As in the case of the recreation demand transfer, there are economic plausibility restrictions that can assist in discriminating among multiple solutions to these nonlinear equations. θ_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]); thus, we have a plausible range of values for it as well.

[9]Note that in this case we have assumed that the housing decision has been made (and thus left out the P(•) 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]).

Overall, then Eqs. (3.12) and (3.14) are two nonlinear equations in two unknowns θ_1 and b, which can be solved to generate sufficient preference information to use the model to infer the value of a policy alternative. Once again, the calibration strategy has used existing benefit estimates and included the restrictions implied by economic theory. This process of selecting multiple estimates from the available literature in principle reduces the restrictions that need to be imposed on the model.

In comparison, we could implement a simple approach (i.e., multiplying the marginal value from the hedonic property value model by the size of the water quality change). This approach does not require the resulting estimates to be consistent with the individual's available income or other constraints. Because the hedonic model provides the marginal WTP for a quality change, this method would approximate the estimated benefit. Instead, our approach would be to calculate a new WTP estimate using Eq. (3.14) with the calibrated values of parameters for the particular ΔA. The numerical example presented below illustrates our central message: for large changes in water quality the difference between the simple approach and this deductive approach can be large and the bias resulting from approximations can be significant. It is important to bear in mind that these numerical computations are meant to be illustrative because simplifying assumptions were made to allow the use of readily available information.

From the Michael, Boyle, and Bouchard (undated) hedonic study, we obtain the following information for a group in their sample:

m = income = \$82,074

$\partial P / \partial A_1$ = marginal rental price = \$4,569

P (•) = \$105,704

A_1 = the measured baseline level of water quality = 2.96 meters measured using secchi disk

The Mitchell and Carson CV study provides the following information:

WTP = \$242 annually for water quality improvements

Because Mitchell and Carson asked respondents for their WTP for national water quality improvements, measured on the water quality ladder from boating to swimming levels, the following three adjustments were made. First, we multiplied the $242 figure by 0.67 to calibrate WTP for national water quality down to WTP for local water quality. This is the proportion of WTP for national quality changes that respondents felt should be set aside for local water quality improvements.

Second, we needed to establish a correspondence between the water quality measures used in the two studies. This is essential because the water quality measure assumed in the hedonic model must be linked to the physical interpretation offered for the water quality described in the CV study. Recall we resolved this question for the travel cost and Mitchell-Carson studies by linking them both to 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 secchi disk measures that were used to gauge the water quality perceived by homeowners in the hedonic model.[10] Clearly, this step of the process was somewhat *ad hoc* since it was constrained by the information at hand. However, it can be easily modified and does not impinge on the calibration logic. It is discussed because the process of establishing the consistency between the physical units involved in different benefits measures is important. Thus, the secchi depth measures necessary to support boating and swimming are calculated as 2.96 and 5.66 meters, respectively.

Third, the WTP had to be adjusted to account for price level changes based on consumer price indices (152.4/99.6). Moreover, the housing rent and marginal rental price were converted into annual terms incorporating tax differences, and the annualizing factor is 0.116. This adjustment factor uses Poterba's (1992) analysis of income tax and property tax effects on the rental cost of housing. These adjustment factors are constructed for 1990. As with the physical conversions for water quality, a full-scale analysis

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[10]The RFF ladder parameters include (among other attributes) dissolved oxygen and turbidity (measured using Johnson Turbidity units) such that boatable quality equals 100 JTU and swimmable water quality equals 10 JTU. We adapted information reported in Smith and Desvouges (1986) to estimate a simple conversion relationship that translated the turbidity units used to define boatable to swimmable conditions in the RFF water quality ladder to secchi disk readings in meters.

for policy purposes would update these to the years relevant for the policy.

With these adjustments, we can solve Eqs. (3.12) and (3.14) for θ_1 and b. Because of the nonlinearity of the system, there is no analytical solution and numerical iteration is used. Although we will not get unique solutions for θ_1 and b, every pair of θ_1 and b presents sufficient information needed to calculate WTP for alternative water quality changes using Eq. (3.13) because together they characterize the indirect utility function from which the WTP measure is derived. Table 3-3 presents the WTP estimates for three alternative water quality changes (ΔA = 1, 2, and 4 meters, from boatable conditions) and three values for b using the proposed deductive approach. These estimates are compared with the result of using a simple approximation (multiplying the marginal rental price with the amount of water quality change).

Table 3-3. Illustrative Transfers of the Value of Water Quality Changes from Hedonic Price Models: Calibrated Versus Simple Approximation (1995 $)

Water Quality	New WTP			Simple Approximation
Change (Δ)	b = .09	b = .10	b = .20	$(\partial P/\partial A_1) \bullet (\Delta)$
$\Delta A = 1$	534.4	534.4	533.7	530
$\Delta A = 2$	955.4	956.3	966.4	1,060
$\Delta A = 4$	1,601.7	1,605.5	1,651.5	2,120

The bias resulting from the simple approximation varies in proportion with the size of the water quality change, and this result is robust to the selection of a calibrated value of b (all of which fall within the plausible range for b, based on its relation to the money flexibility parameter). This result underscores the message that simple approximations can generate biased estimates of the value of nonmarginal water quality changes because there is nothing in such a calculation that ensures that the estimate reflects how consumers with constrained budgets respond to changes in water quality. In contrast, the deductive approach builds the structure (Eq. [3.13]) to explicitly address quality changes, given income and price information.

4 Next Steps

All approaches to nonmarket valuation can be interpreted as providing information that offers a partial measure of consumer preferences.[1] In the case of market choices, there is a long tradition using (and testing) the restrictions implied by constrained utility maximization in interpreting observed behavior. Moreover, Hausman's (1981) analysis demonstrated that one could use the restrictions implied by theory to estimate Hicksian consumer surplus (the appropriate economic welfare measure) for price changes based on Marshallian demand (the observable data). The use of Hausman's logic implies that observed behavior (the demand function) can be combined with the restrictions implied by an economic model that describes the source of that behavior to measure unobservable WTP. Preference calibration as a strategy for developing benefit transfers alters the practices of benefit transfer in a way that is broadly consistent with this basic logic. That is, the method relies on using existing benefit estimates (e.g., consumer surplus, marginal hedonic price, and WTP) from specific applications to calibrate a constrained preference model. The analyst first defines the functional form of the (indirect) utility function and then uses information from existing studies to estimate parameters of this function. Knowledge of the form and parameters

[1]See Smith (1997) for a simple sketch of the linkage between WTP functions, indirect utility functions, and what is measured by hedonic, travel cost, and averting behavior models.

of the utility functions allows the analyst to specify a WTP function that can then be used to estimate WTP for different (i.e., policy relevant) changes in environmental quality. This practice assures that the WTP estimates will be consistent with the utility maximization process that is assumed to form their foundation. It also assures that if there are differences in other important factors to individual choices (e.g., the prices of other goods or income), they will be consistently reflected in the transfer values.

Roy's identity for price changes defines a partial differential equation that underlies Hausman's logic. That is, it links the demand function to the constrained utility maximization model. With nonmarketed environmental resources, this process will differ depending on the method used to estimate them and the type of resource change being evaluated. For example, in the case of a hedonic model, the measure available is a point estimate of a marginal rate of substitution. While in the case of travel cost models, the relationships usually estimated involve demand functions for recreational trips or indirect utility functions for recreation site-choice occasions (random utility models). Environmental quality may well affect each available estimate differently. The primary issue posed in using calibrated benefit transfers is that the method requires the analyst to be explicit about how the benefit measure selected from the literature is connected to a specific preference function (and implied decision process). This process requires additional assumptions that are then combined with the available benefit measure in the process of a benefit transfer.

As a rule, we know that the information available from an individual demand function or from estimates derived using another approach to nonmarket valuation will not be sufficient to identify all the parameters in a preference function. Thus, completion of the task requires assembling other information to permit identification of the preferences parameters so that an analyst can develop a "new" benefit measure. This process imposes a discipline that requires defining exactly what was measured (e.g., Marshallian consumer surplus, Hicksian WTP, or marginal hedonic price. Moreover, the baseline and new levels of water quality must be defined in units that are consistent with those assumed to enter the specified utility function.

The examples developed in Sections 2 and 3 of this report use a simplified preference function and a very specific characterization of how environmental resources affect it. The two applications illustrate how even with a simple (and restrictive) case it is possible to adapt the numerical implementation to take account of the different situations associated with each benefit transfer. A number of questions need to be considered in evaluating extensions to this approach. In the balance of this Section, three will be introduced and discussed briefly. This discussion is not intended to be complete. Rather, it highlights some of the next questions to be considered in developing a system of stand-alone calibration procedures. Such a system would allow the analyst to readily calibrate a consistent preference function. These issues for further research include

- using more complex (and presumably more "realistic") specifications for the preference functions,
- integrating benefit estimates from multiple sources into the calibration process, and
- evaluating different transfer strategies.

4.1 PREFERENCE SPECIFICATION

In specifying the preference structure to be applied, one must first ask: will the focus of analysis be a small number of priced commodities and one or more nonmarket environmental resources or does the analysis require a more complete description of an individual's expenditures? Usually the first alternative (e.g., one or at most a few goods are considered with environment quality) has dominated the literature. In this situation, it seems reasonable to argue that developing quasi-indirect utility functions that are consistent with the empirical estimates will be easier than beginning with a more flexible overall preference function. This adopts the logic of Hausman's (1981) approach to benefit measurement and uses the derived incomplete preference relationship for the transfer.[2]

Recently Ebert (1998) has offered a general summary of the issues associated with nonmarket valuation. He deliberately adopts a system approach to describing the tasks posed in nonmarket valuation. In his summary, the analyst wishing to estimate the value of one or more environmental resources combines a conditional demand system for market goods (i.e., demand functions conditional to the levels of public or quasi-public environmental resources outside the individual's direct control) with marginal WTP functions for the nonmarketed goods. Economic theory implies some specific restrictions for each type of behavioral function. The combination can be used to recover estimates of the full set of preferences. This strategy overcomes some of the problems associated with the partial or incomplete approaches that generalize the Hausman logic for valuing environmental quality changes.[3] Of course, it also significantly increases the informational requirements imposed on the modeling process.

While Ebert's objective was to consider the tasks of estimating new benefit measures, it is equally relevant as a general description of the strategy being advocated here for benefit transfer and offers a compact description of one strategy for linking existing benefit estimates to market demand models.

As the number of priced goods increases, the desirability of the strategy diminishes because the ability to solve for closed form expressions for the quasi-indirect utility functions requires simple demand specifications. However, one could easily adapt the results from existing derivations with common demand functions to fit the logic implied here. Table 4-1 reproduces a table from

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[2]Hanemann (1984) proposed this strategy for econometric modeling of consumer demand with mixed discrete/continuous applications. This is also the logic Dubin and McFadden (1984) adopted to merge estimates of the demand for electric appliances with the demand for electricity. It is also a common approach used in the joint estimation of revealed preference and CV models (see Eom and Smith [1994] and more recently Nikletschek and León [1996]).

[3]See Bockstael and McConnell (1993) and Larson (1991; 1992) for a discussion of the difficulties in recovering Hicksian measures of WTP for quality changes using Marshallian demand functions.

Table 4-1. Utility Theoretic Measures Related to Common Demand Specifications[a]

	Linear	Semi-log	Log-linear
Marshallian demand	$a + bp + cm$	$\exp(a + bp + cm)$	$e^a p^b m^c$
Compensated demand	$c\exp(cp)U - \frac{b}{c}$	$\frac{-b\exp(a+bp)}{c(bU+\exp(a+bp))}$	$\frac{e^a p^b}{1-c}\left[(1-c)\left(U+\frac{e^a p^{1+b}}{1+b}\right)\right]^{\frac{c}{1-c}}$
Expenditure function	$\exp(cp)U - \frac{1}{c}\left(bp+\frac{b}{c}+a\right)$	$-\frac{1}{c}\ln\left[-cU-\frac{c}{b}\exp(bp+a)\right]$	$\left[(1-c)(U+\frac{e^a p^{1+b}}{1+b}\right]^{\frac{1}{1-c}}$
Indirect utility	$\exp(-cp)\left(m+\frac{1}{c}\left(bp+a+\frac{b}{c}\right)\right)$	$\frac{\exp(-cm)}{c}-\frac{\exp(bp+a)}{b}$	$\frac{-e^a p^{1+b})}{1+b}+\frac{m^{1-c}}{1-c}$
Direct utility	$\frac{cq_1+b}{c_2}\exp\left[\frac{c(a+cq_2-q_1)}{cq_1+b}\right]$	$\frac{b+cq_1}{-cb}\exp\left[\frac{c(aq_1-bq_2-cq_1\ln q_1)}{b+cq_1}\right]$	
Consumer surplus	$\frac{[(q')^2-(q^0)^2]}{(-2b)}$	$\frac{(q'-q^0)}{(-b)}$	$\frac{(p'q'-p^0q^0)}{(b+1)}$
Compensating variation	$\left(\frac{q'}{c}+\frac{b}{c^2}\right)-\exp\left[c(p'-p^0)\right]\left(\frac{q^0}{c}+\frac{b}{c^2}\right)$	$\frac{1}{c}\ln\left[1+\frac{c}{b}(q^0-q')\right]$	$m-\left[\frac{1-c}{(1+b)m^c}(p'q'-p^0q^0)+m^{1-c}\right]^{\frac{1}{1-c}}$
Equivalent variation	$\exp\left[c(p^0-p')\right]\left(\frac{q'}{c}+\frac{b}{c^2}\right)-\left(\frac{q^0}{c}+\frac{b}{c^2}\right)$	$-\frac{1}{c}\ln\left[1+\frac{c}{b}(q'-q^0)\right]$	$\left[\frac{1-c}{(1+b)m^c}(p^0q^0-p'q')+m^{1-c}\right]^{\frac{1}{1-c}}-m$
Integrability condition	$q \le \frac{-b}{c}$	$b + cq \le 0$	$\frac{qp}{m} \le \frac{-b}{c}$

[a] Price (p) and income (m) are normalized on the price of the Hicksian good. U is a constant of integration, which is a function of utility. Formulas hold only for values of $p \le \tilde{p}$, where $\tilde{p} = p \mid \lim_{p \to \tilde{p}} q = 0$. For the rows other than the direct utility function, we assume that the quantity measure is q_1 and $p = p_1/p_2$; $m = m/p_2$. q_2 is the Hicksian composite good. q' corresponds to the quantity demanded at p' and q^0 the quantity demanded at p^0.

Source: Bockstael, Nancy E., W. Michael Hanemann, and Ivar E. Strand, Jr. 1984. *Measuring the Benefits of Water Quality Improvements Using Recreation Demand Models*. Vol. II. Department of Agricultural & Resource Economics, University of Maryland. Report to U.S. Environmental Protection Agency.

Bockstael, Hanemann, and Strand (1984) illustrating the logic for price changes. The distinction in the current proposal from the difficulties encountered in using Hausman's logic to recover estimates of the WTP for quality change is that the current proposal calls for using the Hausman logic to provide a specification for preferences that is *maintained as "true"* for the purpose of calibrated transfers.

Applications of our proposed approach are not limited to studies with demand functions, drawn typically from travel cost studies. Starting with a specific quasi-indirect utility function, one could "rationalize" the link to a hedonic price function's marginal rate of substitution between some measure of environmental quality and a numeraire. We illustrated the logic of this case with our CES example.

Important advantages of our proposed approach are parsimony of parameters that need to be determined and increased "realism" of the function's implications for measurable economic parameters such as price and income elasticities. Recall that the Cobb-Douglas example used in Section 2 assumes the price elasticity is unity (in absolute magnitude) and the income elasticity is unity. As the number of priced goods to be considered along with environmental quality increases, it would be preferable to follow the strategy used in numerical computable general equilibrium models. These studies generally adopt nested CES or Stone-Geary specifications for the direct utility function (in the hedonic example in Section 3 we adopt a CES).[4] An important and unresolved issue for calibrated transfers is that only one of these studies (Espinosa and Smith [1995]) has considered the role of nonmarketed goods in the preference specification that permits the nonmarketed good to enter preferences as a nonseparable argument. Their approach assumed there was a perfect substitute private good to mitigate the negative effects of deterioration in the environmental resource. Relaxing this assumption complicates the calibration of the full economy to a baseline set of conditions.

[4]See Rutherford (1997) and Perroni and Rutherford (1996) for a discussion of calibration under different preference specifications.

4.2 MULTIPLE BENEFIT MEASURES

The literature beginning with Cameron's (1992) first application of joint estimation linking revealed and stated preference estimates offers the basic logic that could be used in the process of combining multiple estimates from different studies in the literature. One needs to define how the existing estimates relate to a common preference specification. As will be demonstrated in future work, multiple benefits could be incorporated for example by generalizing the hedonic formulation in Eq. (3.2) to include a recreation component in the CES preference function, thereby addressing joint recreation and housing benefits of water quality improvements. As a rule, the problem with multiple estimates is usually not conflicts between them or difficulties in connecting them to a common preference structure. Rather, the problem most often encountered is incomplete information on the characteristics of the sample of individuals whose behavior is being described. This limits the ability to use variations in estimates of common benefit concepts as reflections of "observable" heterogeneity in individuals' preferences.

When the multiple estimates from the literature relate to "exactly" the same benefit concept, then there may be the opportunity to introduce estimation uncertainty (see Chapter 4 in Desvousges, Johnson, and Banzhaf [1998] as an example). Developing multiple estimates of the same benefit concept was the strategy used in meta analyses of past benefit studies.[5] It is important to note that statistical functions derived as meta summaries or response surfaces do not necessarily impose the preference consistency.[6] They are simply a different type of "reduced form model." Of course, one could consider using the data from meta analyses to estimate some of the parameters that underlie preferences in our proposed calibrated transfer.

For cases where different benefits concepts (e.g., option price versus consumer surplus) are being measured, it is possible, in principle, to calibrate more parameters of preferences (or to take

[5]See Smith and Kaoru (1990) or Smith and Huang (1995) as examples.

[6]Examples of these summaries include Smith and Osborne (1996), Walsh, Johnson, and McKean (1990), Boyle and Bergstrom (1992), and Van den Bergh et al. (1997).

account of more sources of heterogeneity among individuals). This is another important direction for future research.

4.3 EVALUATING BENEFIT TRANSFERS

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

Because benefit measures are never observed, their estimates are unlikely to be evaluated in a context that will be fully satisfactory. That is, there is no "true benefit estimate" that could be found to serve as a measuring stick for the transferred estimates. Thus, to close this discussion of preference calibration as a transfer method, the approach proposed in this study may possess a unique advantage for evaluating benefits transfer (especially in cases where the preference specification used in a calibrated transfer was not selected to be restrictive). That is, given a numerical characterization of the quasi indirect preference function, it is possible to consider estimating observable "quantities" at the same time as the benefits are measured. The quantity demanded of a linked good is one such observable implication of the analysis underlying the benefit estimation. Our recreation example in Section 3.1 illustrates this advantage because computing the number of trips (along with the approximate budget share implied for recreation) allowed us to discriminate between the two possible roots solving the calibration of the model.

It is also possible to use the calibrated function to estimate implied expenditure shares, price and income elasticities, and other "indexes" that may well be easier to gauge for plausibility than an estimate of the consumer surplus for an unobserved quality change. These types of estimates are not available with other transfer methods because they are not consistently linked to preferences. Large discrepancies between the predictions for the linked private good or the elasticities and what is judged to be plausible could be used to re-calibrate the missing parameters so that their correspondence with plausible or standard levels of elasticities and linked good is enhanced.[7] Alternatively, they could signal the potential for errors.

Clearly, what has been proposed here was done in the context of simple specifications to illustrate the logic of a different strategy for conducting benefit transfers. More complex functional forms are possible and numerical calibration analogous to what is used with numerical computable general equilibrium models is also possible. However, the desirability of pursuing such larger-scale efforts depends on the success of experimentation with smaller applications of the method and comparisons with current practice. These will be useful next steps in evaluating the calibration approach and should precede attempts for more ambitious numerical calibrations.

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

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