A Comparison of Direct and Indirect Methods for Estimating Environmental Benefits

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Two classes of methods are currently available to estimate consumers' valuations of improvements in environmental resources—direct and indirect. This paper reports the results of a detailed comparison of the estimated recreational benefits associated with water quality improvements using one member of each class. The findings indicate that while the estimates are quite comparable, analyst judgment played a very important role in the development of both methods.

Key Words: benefit estimation, contingent valuation, travel cost, water quality.

Tight budgets, with their associated need for evaluating projects involving environmental resources, have made benefit estimation a major preoccupation of environmental economists. These benefits generally have been measured using one of two approaches—direct and indirect methods. Indirect methods rely on the behavior of households in related markets to reveal their valuations of the nonmarketed goods. Direct methods use surveys to ask individuals' valuations for hypothetical changes in such resources. While both have been used in an array of valuation tasks, most economists prefer the indirect methods because they are based on observed market behavior. Comparisons (see Brookshire et al.) between these classes have been intended to enhance the acceptability of the direct methods. Moreover, when differences were found between the respective approaches' estimates, the analysts generally have concluded that the direct (or contingent valuation, CV) method's estimates have substantial variability and are subject to design limitations (see Cummings, Brookshire, Schulze).

This paper suggests that such comparisons may provide rather limited information on the performance of either approach because the indirect methods can also have substantial variability in their estimates. We present evidence on the extent of sensitivity of consumer surplus estimates to modeling decisions that must be made to estimate the value of water quality changes with travel cost recreation demand models. These results are contrasted with those from a contingent valuation study designed to provide comparable information. Our comparison also has advantages over past studies because it is possible to develop estimates from each approach at the individual level and to use these paired estimates to evaluate the relationship between the direct and indirect methods.

Overall, we find that neither approach is free of the influence of judgment. While this conclusion is not surprising, it does suggest that explicitly recognizing the role of judgment in questionnaire design and data analysis for CV studies, and in model specification and estimation for the indirect studies, is crucial to the use of each and to the evaluation of comparisons.

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The Design of the Comparative Evaluation

Our objective is to develop a comparison of direct and indirect methods that also allows the role of judgment in the development of each approach's estimates to be described. We consider estimates derived from travel cost demand and contingent valuation methods for valuing the recreation benefits from water quality improvements.

Indirect Method: The Travel Cost Models

The travel cost model is one of the most widely used frameworks for estimating the features of a recreation demand function. It is based on the premise that even though most public recreation sites have zero (or nominal) entry fees, recreationists nonetheless pay an implicit "price" for a site's services when they visit it. This implicit price includes vehicle-related and time costs of the trip. The diversity in visitors' origins provides the information to estimate the demand function for a site's services.

Using the household production framework, the travel cost model is a derived demand for a site's services because these services contribute to the production of a household's recreation activities. A site's characteristics influence the productivity of its services. Therefore, it is reasonable to expect that the demand functions will be related to site characteristics. This view is consistent with Freeman's proposal for measuring the benefits from water quality improvements. That is, water quality contributes to the productivity of a site's services in certain activities, such as fishing and swimming. While the data to implement this suggestion are difficult to find for a single site, by using the demands for several sites that provide comparable recreation opportunities but have different characteristics, it is possible to implement Freeman's proposal.

Based on this logic, we developed three travel cost models that incorporate the effects of water quality. The first two models rely on the estimated demand functions for twenty-two water-based recreation sites to describe a general relationship between demand and site characteristics. The third model pools individuals' visits to sites in the case study area, the Monongahela River valley, assuming that they conform to a single demand function that differs only by the observed differences in water quality at the sites.

The Generalized Travel Cost Models

The first two travel cost models use a generalized approach to describe recreation sites' demand functions. This approach is illustrated below in equations (1) and (2). The model is implemented in two steps. In the first, we estimate separate travel cost demand models [i.e., equation (1)] for the twenty-two sites in our sample. Next we use these demand estimates in a generalized least squares (GLS) estimator that regresses each demand function's parameters on sites' characteristics [i.e., equation (2)], using the parameters' estimated variances to construct the relevant weighting matrix.[1] The estimator used for the first-stage site demand models is the only difference between the first two models. The first generation version used ordinary least squares (OLS), while the second used a maximum likelihood (ML) estimator that was designed to take account of truncation and censoring problems discussed below.

$$V_{ij} = f_i(\bar{P}^s_{ij}, P_{ij}, Y_j, \bar{SE}_j | B_i) + e_{ij} \tag{1}$$

where V_{ij} is a measure of quantity demanded, the visits to the ith site during a predefined period (usually a season) by the jth person; P_{ij}, the implicit price of a trip to the ith site, including both vehicle costs associated with the roundtrip and the time costs of travel, for the jth individual; $\bar{P}^s_{ij}$, a vector of the implicit prices of the substitute sites for the ith site available to the jth individual; Y_j, individual j's income; $\bar{SE}_j$, a vector of socioeconomic characteristics hypothesized to be determinants of the site's demand; B_i, a vector of the demand parameters for each of the determinants of the site's demand; and e_{ij}, stochastic error.

$$\hat{B}_{ki} = g_k(\bar{C}_i | d_k) + U_{ki} \tag{2}$$

where $\hat{B}_{ki}$ is kth estimated demand parameter derived from equation (1); C_i, a vector of characteristics for the ith site; d_k, a parameter vector for each of the determinants of site demand; and U_{ki}, stochastic error.

The data for both of these models were taken from the 1977 Federal Estate Survey.

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[1] Vaughan and Russell (1982a, b) used a one-step approach to implement the model. However, the two-step allows for more testing of the sensitivity of the parameter estimates for both components of the generalized model to different specifications.

Sample sizes ranged from 35 to 202 for each of 22 Army Corps of Engineers sites. We obtained information on site characteristics from the Corps of Engineers' Recreation Resource Management System and the U.S. Geological Survey's National Water Data Exchange. To estimate the implicit price, we used round trip distance and 8¢ per mile to derive the vehicle-related 1977 travel costs and the reported travel time together with an estimated wage for the time costs of travel. The predicted wages were estimated for each individual using hedonic wage models based on the 1978 Current Population Survey (Smith 1983). Our approach assumes the opportunity cost of time is equal to the wage rate. In a separate analysis (see Smith, Desvousges, and McGivney 1983b) we found that the conventional assumption in recreation demand models (i.e., assuming opportunity cost was a fraction of the wage) was not superior to use of the full wage rate.

Data limitations constrained our analysis in two ways. First, the data did not include information on several variables in a travel cost demand function, most important among these were P_{ij}^{s}, the prices of substitute sites.[2] Second, the data were obtained in on-site surveys which truncated our available measure of site usage at one visit (because individuals had to be present to be interviewed). In addition, the survey coding procedures censored visits at the level of six visits in a season.

Because truncation and censoring can lead to biased parameter estimates using OLS, we refined our estimates of the model. First, for the OLS version of the model, we "screened" the site demand function estimates using Olsen's diagnostic index. This index gauges whether the truncation and censoring of the dependent variable would be likely to lead to severe bias in the estimates. After this screening we confined the sample of sites for GLS estimation to those cases identified to have less severe impacts.

Our second version of the generalized travel cost used a maximum likelihood estimator for these first-stage demand models. Assuming normality for the error structure, with truncation at zero and censoring at k, the likelihood function is given in equation (3).

$$(3)\quad L(\beta, \sigma^2, y) = \underset{i \in S_1}{\pi}\left[\frac{\frac{1}{\sigma}\,\phi\,[(y_i - \beta X_i)/\sigma]}{1 - \Phi(-\beta X_i/\sigma)}\right] \cdot \underset{i \in S_2}{\pi}\left[\frac{1 - \Phi[(k - \beta X_i)/\sigma]}{[1 - \Phi(-\beta X_i/\sigma)]}\right]$$

where y_i is measure of site usage for ith individual; β, parameter vector $(1 \times K)$; X_i, vector of independent variables for ith individual $(K \times 1)$; σ^2, variance in error associated with site demand function; S_1, set of observations with $0 \leq y_i < k$; S_2, set of observations with $y_i \leq k$; $\phi(.)$, density function for standard normal variate; and $\Phi(.)$, distribution function for standard normal variate.

For both versions of the generalized model we selected a semilog specification for equation (1) following earlier support in Smith (1975); Ziemer, Musser, and Hill; and Vaughan, Russell, and Hazilla. While several standard socioeconomic determinants were considered, none of the resulting estimates was uniformly superior to that using the simplest model for all sites. This model specified the logarithm of visits as a function of the implicit price and household income.

$$(4)\quad \ln V_{ij} = b_{1i} + b_{2i}P_{ij} + b_{3i}Y_j + e_{ij}.$$

Although we considered several water quality measures—including the Resources for the Future (RFF) and National Sanitation Foundation indexes—dissolved oxygen (DO) (the mean and variance over the summer months of 1977) provided the clearest association with the estimated demand parameters. This finding is consistent with earlier judgments (Feenberg and Mills; Davidson, Adams, and Seneca) as to the usefulness of this parameter as a measure of water quality.

GLS estimates of the demand parameter—site characteristic models—were developed using both the OLS and ML estimates. Because detailed descriptions of these findings are available (see Smith, Desvousges, and McGivney 1983a; Smith and Desvousges 1985), the primary concern here is the relative performance of each version of the generalized travel cost (GTC) model in estimating water quality benefits. In developing this evaluation, we want to highlight the factors influencing our judgment as to which version appears to provide the best estimates.

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[2] Our formulation maintains that differences in each site's characteristics provide the basis for site substitution, and that with this information it would be possible to construct a homogenous index of site services. To the extent this treatment of substitute sites is inappropriate, our estimates will be biased. For a more detailed discussion see Desvousges, Smith, and McGivney.

Table 1. Generalized Least Squares Estimates Using Maximum Likelihood Site Demand Estimates

	Model		
Independent Variables[a]	Intercept	Travel Cost Parameter	Income Parameter
Intercept	$-.044$	$-.022$	$.17 \times 10^{-4}$
	(-0.024)[b]	(-0.431)	(0.657)
Shore	$.001$	$-.11 \times 10^{-4}$	$-.60 \times 10^{-7}$
	(0.782)	(-0.382)	(-1.449)
Access	$-.039$	$.27 \times 10^{-2}$	$.14 \times 10^{-6}$
	(-1.071)	(1.301)	(0.074)
Water Pool	1.461	$-.089$	$.86 \times 10^{-4}$
	(1.030)	(-1.522)	(2.731)
DO	$.020$	$-.10 \times 10^{-3}$	$-.24 \times 10^{-6}$
	(2.076)	(-0.286)	(-0.766)
VDO	-6.47×10^{-5}	1.48×10^{-7}	5.28×10^{-10}
	(-2.077)	(0.127)	(0.573)
R^2	.475	.196	.455
F	2.89	2.50	2.68

[a] The definitions for the site characteristics are: shore, total shore miles at site during the peak visitation period; access, number of multipurpose recreational and developed access areas at the site; water pool, size of the pool surface relative to total site area; DO, dissolved oxygen (percent saturation); and VDO, variance in the dissolved oxygen over the six observations during the recreational season of the survey.

[b] The number in parentheses below the estimated coefficients are the ratios of the estimated coefficient to the asymptotic standard error; they follow a normal distribution asymptotically.

First, we evaluated the GLS models for demand parameters in each version. The mean DO had statistically significant effects on both the intercept and the travel cost parameters in the first generation model, but did not with GLS models based on the ML demand estimates. As shown in table 1 in the second generation model, water quality had a significant and plausible effect for only the intercept parameter.

Second, we evaluated each version's estimates of the value of water quality improvements for the Corps of Engineers' sites. Table 2 reports these results, along with the household income and the price limits used in these calculations. Since a semilog specification for the demand function does not allow for a finite choke price, we have used the maximum observed travel cost for each site. Two consumer surplus measures are reported for each version of the GTC model—a Marshallian measure (*M*) and a Hicksian measure (*H*) derived using Hausman's quasi-expenditure function for this demand specification and the same finite choke price.[3] A change from boatable to fishable (i.e., game fish) water quality conditions was selected because this can be compared with results in the literature.

Clearly, the two versions of the GTC model imply different estimates. The first version yields per trip estimates of the value of a water quality change that are much larger than the second. Comparing these estimates with the available findings for comparable resources with approximately comparable water quality changes (for example, Vaughan and Russell 1982a, Loomis and Sorg) would suggest that because the second generation GTC falls within the range of past estimates, it offers a more plausible framework for benefit analysis.

The large differences in Marshallian and Hicksian measures of the value of a water quality change in table 2 are also interesting. It is important to recognize that conventional

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[3] For the semilog specification of the demand function, the quasi-expenditure function is

$$E(TC, U) = -\frac{1}{\alpha_2}\log\left(-\alpha_2 U - \frac{\alpha_2}{\alpha_1} e^{\alpha_0+\alpha_1 TC}\right)$$

where $\alpha_1 < 0$; U is a constant reflecting the level of utility and other variables that must be assumed constant, such as other goods' prices; and TC is the travel cost.

This approach was suggested to us by Jerry Hausman and is based on his 1981 paper on exact welfare measures. Hanemann (1980) proposed a similar approach for this demand function earlier, but the connection between the two contributions has not been recognized.

Since the generalized travel cost model specifies α_0, α_1, and α_2 to be a function of site attributes including water quality, it is possible to use this approach to develop a Hicksian measure of the value of water quality. We have defined it as the change in the value of the site with a water quality change.

Table 2. Value of Increments to Water Quality: A Comparison of First- and Second-Generation Generalized Travel Cost Models in 1977 Dollars

Site	Site No.	Average Income	Average Travel Cost	Maximum Travel Cost	Boatable to Fishable (per trip)			
					First		Second	
					M	H	M	H
Arkabutla Lake, MS	301	13,184	20.04	209.35	19.37	42.33	5.44	10.17
Lock & Dam No. 2 Arkansas River, AR	302	10,409	3.04	70.01	5.38	93.26	4.27	8.58
Belton Lake, TX	304	17,279	33.18	302.86	19.31	50.02	1.60	3.12
Benbrook Lake, TX	305	19,135	30.23	344.44	54.19	59.56	2.84	5.68
Blakey Mt. Dam, Lake Ouachita, AR	307	17,144	45.39	286.03	11.52	29.96	0.79	3.37
Canton Lake, OK	308	17,392	32.30	106.16	9.31	[a]	1.07	1.91
Cordell Hull Reservoir, TX	310	15,491	29.65	184.35	12.06	33.77	2.49	8.16
DeGray Lake, AR	311	19,235	42.04	210.48	17.23	42.16	2.16	5.97
Grapevine Lake, AR	314	19,309	38.45	307.28	18.12	[a]	0.61	1.27
Grenada Lake, MS	316	9,199	24.57	207.05	15.49	31.76	3.00	4.84
Hords Creek Lake, TX	317	16,263	39.46	304.01	25.54	28.28	0.71	1.59
Melvern Lake, KS	322	18,087	31.48	130.50	13.07	14.88	1.27	2.68
Millwood Lake, AR	323	18,630	37.62	309.24	27.81	46.96	6.00	14.98
Mississippi River Pool No. 6, MN	325	19,589	52.23	843.86	20.87	24.10	0.08	0.17
New Savannah Bluff Lock & Dam, GA	329	12,609	18.65	157.36	14.64	16.44	2.25	[a]
Ozark Lake, AR	331	12,654	58.71	457.44	19.32	51.53	1.30	6.46
Philpott Lake, VA	333	14,268	26.09	268.76	20.34	61.34	2.89	6.51
Proctor Lake, TN	337	17,510	46.08	172.41	12.77	14.51	0.15	0.29
Sam Rayburn Dam & Reservoir, TX	339	19,515	40.23	155.30	12.03	24.44	2.28	5.21
Sardis Lake, MS	340	13,141	36.08	429.20	19.84	49.44	1.41	2.75
Whitney Lake, TX	344	18,688	35.40	303.62	21.96	59.07	1.36	2.61

[a] The predicted income coefficients for these models were exceptionally small, making the calculation of the change in the compensating surplus from Hausman's quasi-expenditure function unstable.

bounds on the relationship between Marshallian and Hicksian measures of a price or quantity change do not necessarily transfer to the case of quality changes.[4]

The Simple Travel Cost Model

Our third model was based on the recreation behavior reported by the sixty-nine contingent valuation survey respondents who used one or more of the recreation sites along the Monongahela River. We used records of their use and travel costs and the variation in the mean dissolved oxygen across the thirteen sites visited along the Monongahela River to estimate a simple travel cost demand model. Travel costs were constructed using the same procedures that were described for the generalized models, but we used the consumer price index to adjust vehicle travel costs and wage rates for the date (1981) of the survey. Equation (5) provides the estimated demand model, derived by pooling visits across all thirteen sites as if they corresponded to a single site.[5] The t-ratios for the null hypothesis of no association are reported in parentheses below the estimated coefficients.

$$(5)\quad \ln V = \underset{(-3.075)}{-3.928} - \underset{(-2.846)}{.051}\,TC + \underset{(1.109)}{.00001}\,Y + \underset{(3.917)}{.058}\,DO$$

$$R^2 = .225,\ F = 6.09.$$

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[4] The nature of the relationship between Marshallian and Hicksian measure of consumers' valuations of quality changes depends on how quality enters the individual demand function. For a discussion of some of the valuation of quality, see Hanemann (1982).

[5] The sample corresponds to those user trips where responses to the contingent valuation survey questions were not protest bids or judged to be outlying observations. The basic rationale for deleting observations stems from an assumption that acceptance of the hypothetical market implies a consistent relationship between an individual's bid and income. Regression diagnostic indexes were used to judge outlying observations according to their individual effect on the measured relationship between income and the bid. See Desvousges, Smith, and Fisher for more details.

Using this model, the estimated Marshallian consumer surplus for each household, with the maximum travel cost as the choke price (\$22.65 in 1977 dollars), ranged from \$0 to \$8.74 (in 1982 dollars) for a water quality improvement from boatable to fishable. This is consistent with the range of values for this type of improvement in the literature and with that predicted for Corps sites using the second generation generalized travel cost model as well as with past estimates.

On the basis of these comparative appraisals, two of the three travel cost models would be judged as a plausible basis for valuing water quality changes—the second version of the generalized model and the simple travel cost model. We used both of these models in the comparison with contingent valuation results. However, each implies different assumptions in transfer of benefit estimates for a particular scenario. For example, use of the generalized travel cost model to value water quality improvements along the Monongahela requires that we assume these sites are comparable to the Corps of Engineers sites on which this model was based. Alternatively, the simple travel cost model assumes that all sites along the river have identical demands, and improvements in water quality can be described by considering the reported range of water quality experience from these sites. The details of calculating the consumer surplus increments illustrate how some of these differences influence the estimates.

As shown in figure 1, a change from WQ_1 to WQ_2 would increase consumer surplus by *ABCD* at price P_1 (assuming the choke price is finite). When the demand function holds income constant, it is the Marshallian surplus; with utility constant it is the Hicksian surplus measure. The generalized model adjusts all of the demand parameters to reflect all of the characteristics of each Monongahela site and the income of the household in estimating the value of a water quality improvement. By contrast, the simple travel cost model adjusts only the intercept for the income of each user and for the postulated water quality change.

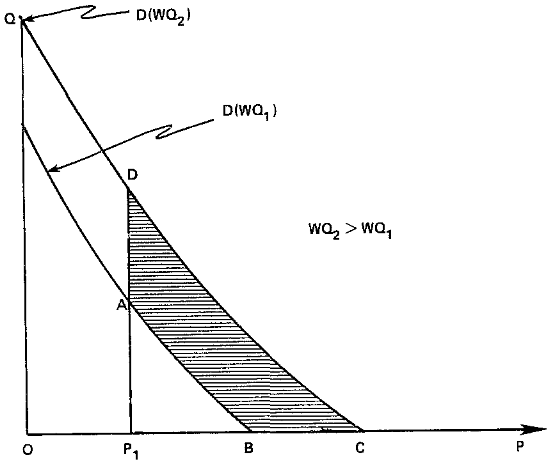

Figure 1. Consumer surplus increment for water quality change

Direct Method

The contingent valuation estimates are based on a single-stage, stratified cluster sample of households in the five southwestern Pennsylvania counties that comprise the Pennsylvania portion of the Monongahela River basin.[6] One objective of this survey was to develop estimates of use values for water quality improvements that could be compared with the estimates from the travel cost models.

After asking about recreation behavior, the questionnaire explained to each respondent the reasons why one might value water quality in terms of a framework that distinguished use, option, and existence values. The questionnaire then elicited an option price value for a water quality change.[7] That is, each respondent was asked to provide an annual bid for a specified water quality improvement based on his actual use and possible future use of the river. This formulation acknowledges that an individual's demand for the services of the sites along the Monongahela River may be uncertain. To describe water quality to respondents the interviewers used the RFF water quality ladder shown in figure 2. Beginning with the existing conditions on the river—water quality levels that are generally regarded to be consistent with recreational boating—option prices were asked for two water quality improvements to levels associated with fishing and swimming. In addition, individuals were asked to value avoiding a deteri-

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[6] Professional interviewers who had received an intensive two-day training session for the survey obtained 301 usable interviews during the late fall of 1981. This resulted in a response rate of 80.6%.

[7] The specific details associated with the contingent valuation questions are quite involved and cannot be completely summarized here. They are provided, including the visual aids, in Desvousges, Smith, and McGivney, pp. 4–9 to 4–19. The report is available from the authors.

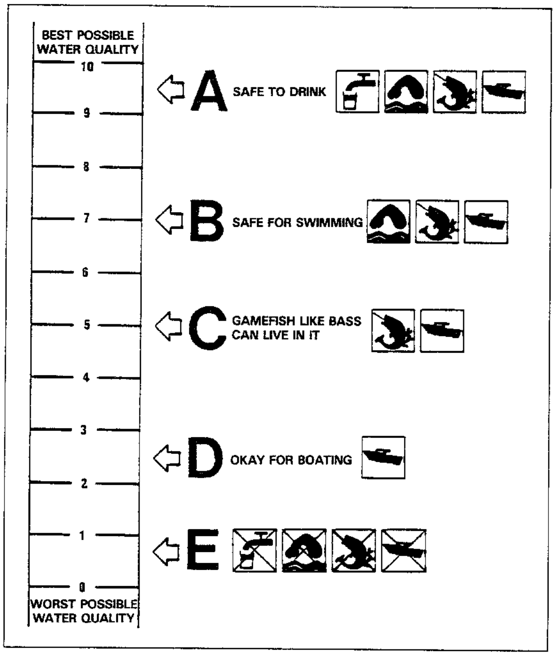

Figure 2. Water quality ladder

oration in water quality to levels that would not permit any recreational uses of the river. The payment mechanism (i.e., how their bids would be paid) was selected on the basis of work by Mitchell and Carson. It was explained as the annual increase in the prices of goods and services and taxes required to meet the pollution control costs associated with each water quality level.

Four approaches to asking for these bids were each applied to approximately equal shares of the sampled households. They included two iterative bidding forms with different starting points—$25 and $125 (see Brookshire, Ives and Schulze), a direct question (see Hammack and Brown), and a direct question using a payment card (see Mitchell and Carson). For each of these formats we asked each individual how much of his option price bid was due to actual use of the river. These estimated use values can be compared with the estimates of the consumer surplus increment for the same water quality changes from our two travel cost demand models.

Our survey respondents include both users and nonusers of sites along the Monongahela River. Because the travel cost models measure the use values of water quality improvements, we restricted the sample to users for the comparative evaluation. In the analysis of the contingent valuation responses, a further restriction was made to the sample. Protest bids and outlying observations, individuals who had not accepted the terms of the hypothetical questions, were deleted (see footnote 5). These two adjustments reduced our sample to sixty-nine observations for comparative analysis.

Comparative Performance of the Benefit Estimation Methods

Past comparative studies (i.e., Bishop and Heberlein, Brookshire et al.) have had to focus on comparisons of means for groups of individuals. In these comparisons, it was not possible to pair estimates from the same individuals and thereby control for the effects of differences in the socioeconomic variables of the groups being compared. By contrast, our design compares benefit estimates for each of the sixty-nine individuals. That is, the predictions from each of the travel cost models are matched with the contingent valuation responses for each individual. This design also allows consideration of the effects of each of the following on the comparative performance of direct and indirect methods: (*a*) the travel cost model used—simple versus generalized travel cost; (*b*) the surplus concept—Marshallian versus Hicksian; and (*c*) the question format used to elicit willingness to pay in the contingent valuation experiment.

The comparison considers the incremental consumer surplus from three water quality changes: (*a*) avoiding a decrease in water quality from current boatable conditions to levels that would preclude all recreation, (*b*) an improvement in water quality from boatable to fishable conditions, and (*c*) an improvement from boatable to swimmable conditions. In the contingent valuation the water quality ladder depicted these changes. To provide the required water quality link for the travel cost model, we used the corresponding values of DO for the improvements and assumed a complete loss of the site for the deterioration scenario.[8]

Table 3 reports the average annual values

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[8] The values for DO are given as follows: (*a*) improvement from boatable to fishable—a change from 45% to 64% saturation; (*b*) improvement from boatable to swimmable—a change from 45% to 83% saturation.

These estimates were derived from Vaughan's 1981 analysis of indexes of water quality that led to the RFF water quality ladder used in our survey.

Table 3. A Comparison of Mean Benefit Estimates in 1981 Dollars

	Water Quality Change[a]		
Approach	Loss of Area	Boatable to Game Fishing	Boatable to Swimming
Contingent valuation			
Direct question	19.71	21.18	31.18
	(17)	(17)	(17)
Payment card	19.71	30.88	51.18
	(17)	(17)	(17)
Iterative bidding ($25)	6.58	4.21	10.53
	(19)	(19)	(19)
Iterative bidding ($125)	36.25	20.13	48.75
	(16)	(16)	(16)
Travel cost methods			
Generalized travel cost			
Marshallian[b]	1113.46	209.17	463.30
	(69)	(69)	(69)
Hicksian	1323.73	509.64	1194.40
	(69)	(69)	(69)
Simple travel cost	3.53	7.16	28.86
	(69)	(69)	(69)

Note: The travel cost models' estimates were converted from 1977 to 1981 dollars using the consumer price index for December 1981, the last month of the survey.

[a] The numbers in parentheses below the means are the number of observations on which each of these estimates was based.

[b] These are calculated with a finite choke price of the maximum travel cost for the Monongahela sites.

per household (in 1981 dollars) for the estimated benefits associated with each of the water quality changes. The consumer price index was used to adjust the travel cost estimates based on the generalized travel cost model from 1977 (the year of the Federal Estate Survey) to 1981 dollars. The contingent valuation estimates are reported according to the question format used to elicit the valuation responses. Marshallian and Hicksian surplus measures are reported for the generalized travel cost model. Marshallian and Hicksian measures are identical for the simple travel cost model.

Two observations can be drawn from table 3. The contingent valuation estimates appear sensitive to the question format used, with the ratio of the largest estimated mean for a question type to the smallest about six to one. Even more striking, however, is the range of estimated consumer surplus increments across the travel cost models. It more than encompasses the contingent valuation estimates, with very high estimates from the generalized travel cost model for the Monongahela sites. Although the generalized model provided reasonable estimates for the Corps of Engineers sites, its poor performance for the Monongahela sites has a straightforward explanation. This is found in table 4's description of the site characteristics for all Corps of Engineers sites in comparison to the thirteen Monongahela sites.[9] These data indicate that the Corps of Engineers sites were substantially larger on average than Monongahela sites. They also had a larger number of access areas than the Monongahela areas. Finally, the size of the water area relative to the overall site area was smaller for the Corps sites. Thus, prediction of the demand functions for Monongahela sites based on this framework is a projection substantially outside the range for site characteristics that provided the basis for the GTC model. Accordingly, it is not surprising to observe the wide discrepancy in benefit estimates. In fact, the generalized model appears to overestimate the absolute magnitude of the intercept and the travel cost parameters. Predicted values for the intercept with Monongahela sites range from 1.77 to 2.24 and the slope from $-.083$ to $-.112$.[10] The latter is substantially larger in absolute magnitude than our estimated demand function, i.e., equation (5), based on the survey respondents' visita-

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[9] Water quality is included in this comparison. However, this is not a factor in explaining these results because the water quality levels were specified in the scenarios used in both cases independently of the actual values reported for the sites. These scenarios were specified independent of the actual conditions to present a comparable set of water quality changes and thereby provide a basis for gauging the effects of other site attributes on their valuation.

[10] The slope parameter for the travel cost variable must be converted to reflect 1981 dollars to be comparable to the results in equation (4). These results are in converted terms.

Table 4. Comparison of Site Characteristics: Corps versus Monongahela Sites

	Shore		Access		Water Pool	
Type of Site	Mean	Range	Mean	Range	Mean	Range
Corps sites	164.3	11–690	16.1	0–39	.67	.01–.94
Monongahela sites	2.9	1–12	2.5	1–7	.94	.67–.99

Note: The definitions for the site characteristics are: shore, total shore miles at site during the peak visitation period; access, number of multipurpose recreational and developed access areas at the site; water pool, size of the pool surface relative to total site area.

tion patterns. Indeed, the travel cost slope predictions are also larger in absolute magnitude than the predictions for the Corps sites which ranged from −.0106 to −.0607.

Our comparison of contingent valuation and travel cost findings can also be made using the individual data rather than the means. Table 5 reports the findings of an adaptation of an approach used in evaluating the predictive performance of econometric models (see Theil). To compare for each individual's benefit estimates, we regressed the contingent valuation bids on each of the travel cost benefit estimates for the corresponding water quality change. If the two methods yield comparable estimates, then we would expect that the intercept would not be significantly different from zero and the slope not different from unity. These models have been augmented with qualitative variables (i.e., 0,1 variables) for the question format used in the contingent valuation experiment in order to provide a basis for appraising its effect on the relationship between the approaches.

These results largely confirm the evaluation based on the means benefit estimates from each method. The GTC model's estimates (both Marshallian and Hicksian surpluses) are inconsistent with the contingent valuation bids across survey respondents. The null hypothesis that the slope parameter of these models was unity (in equivalent dollars) was rejected. Most of the intercepts were not significantly different from zero. Thus, the generalized travel cost model and contingent valuation estimates appear different. Moreover, the estimates of the mean benefits for water quality changes would almost certainly indicate that the travel cost estimates were implausible. This is not the case for contingent valuation results.

By contrast, the same two tests comparing

Table 5. Regression Comparisons of Contingent Valuation and Travel Cost Benefit Estimates

	Water Quality Change								
	Loss of Area			Boatable to Game Fishing			Boatable to Swimming		
Independent Variables	GTC-M[a]	GTC-H	STC	GTC-M	GTC-H	STC	GTC-M	GTC-H	STC
Intercept	37.750	38.160	25.032	5.428	9.732	−18.193	14.548	21.753	−25.907
	(3.888)[b]	(3.947)	(1.779)	(0.341)	(0.625)	(−0.810)	(0.552)	(0.845)	(−0.714)
Travel cost benefit estimate	.002	.001	6.822	.098	0.25	8.411	.072	.017	3.542
	(0.423)	(0.274)	(1.299)	(1.891)	(1.555)	(2.030)	(1.829)	(1.469)	(2.148)
	−370.388[c]	−672.762	1.004	−28.147	−96.796	1.656	−37.313	−131.282	1.208
Qualitative Variables									
Payment card	−31.731	−31.748	−33.577	65.688	65.928	61.177	104.303	104.617	96.767
	(−2.115)	(−2.114)	(−2.556)	(2.766)	(2.751)	(2.576)	(2.667)	(2.651)	(2.488)
Direct question	−23.881	−23.585	−27.144	12.320	14.353	8.878	8.459	11.374	2.141
	(−1.804)	(−1.784)	(−2.035)	(0.584)	(0.678)	(0.417)	(0.244)	(0.325)	(0.062)
Iterative bid ($25)	−25.969	−25.764	−27.982	−13.421	−12.256	−15.022	−23.711	−22.226	−25.043
	(−1.721)	(−1.704)	(−1.869)	(−0.561)	(−0.509)	(−0.629)	(−0.617)	(−0.574)	(−0.658)
R^2	.088	.086	.109	.180	.166	.187	.174	.159	.189
F	1.49	1.46	1.90	3.41	3.08	3.57	3.31	2.97	3.68

[a] The abbreviations for methods are: GTC-M, generalized travel cost—Marshallian surplus; GTC-H, generalized travel cost—Hicksian surplus; and STC, simple travel cost—Marshallian surplus.

[b] The numbers in parentheses below the estimated coefficients are *t*-ratios for the null hypothesis of no-association.

[c] These statistics are *t*-ratios for the hypothesis equivalent to unity for the slope coefficient for the travel cost benefit estimater after adjustment is made for the fact that the travel cost estimate with the generalized travel cost model was measured in 1977 dollars and the contingent valuation bids in 1981 dollars.

the simple travel cost results with the contingent valuation bids indicate a much closer correspondence. From table 3, the simple model's estimates are generally less than the contingent valuation results and fall within the range spanned by the CV estimates across question formats for two of the three water quality changes. The two approaches' estimates are quite comparable.

Question format does affect the relationship between the contingent valuation and travel cost results. The iterative bidding with a $125 starting point was the omitted category in forming the qualitative variables. The *t*-ratios test the null hypothesis of difference between the other question modes and this category.

These general conclusions are remarkably close to those we reported in our earlier comparison (Desvousges, Smith, and McGivney) based on the first generation GTC model. One might ask how this can be, given the results of the closer examination of the characteristics of the Monongahela sites and the conclusion that they were quite different from the Corps' sites on which both versions of the GTC model have been based. The explanation is straightforward. The decision to use the first generation model in that initial comparison was based on a plausibility check using the respondents from the contingent valuation survey who were identified as users of Monongahela sites. This check compared the resulting estimates with other studies of the recreational value of water quality improvements before using the model in the comparative analysis. The first generation GTC model implied a smaller weight to the site characteristics that were most different between Monongahela and Corps sites and thus did not exhibit the same poor performance as the second version of the model.

Implications

Brookshire et al. concluded their comparative analysis noting that their findings provided evidence "towards the validity of survey methods as a means of determining the value of public goods" (p. 176). More recently Cummings, Brookshire, and Schulze have drawn a somewhat more cautious conclusion by defining the reference operating conditions under which one might expect the survey methods to perform comparably to the indirect (market) methods. In past comparisons of direct (CV) and indirect benefit estimation methods, a lack of correspondence has led analysts to question, or at least examine more closely, the survey approach to estimating benefits.

Our findings, however, suggest that a lack of correspondence can be traced to indirect methods as well. This finding is consistent with the literature on specification searches and the implications for statistical inference (e.g., Leamer, Judge and Bock, and Wallace). That is, judgment is an inevitable component of any empirical model of an economic process. It is also a part of the design and implementation of contingent valuation surveys. In the past economists have felt more comfortable with judgments applied in indirect methods than those involved in survey research. Our findings suggest that an understanding of the limitations in both types of methods is essential to interpreting the results of comparative analyses. These results do not imply benefit estimation is infeasible for practical purposes. Rather, they suggest that it is not a mechanical process. Judgment, combined with sensitivity analysis and plausibility checks, are likely to be more important to the quality of a resulting set of benefit estimates than strict reliance on methods based exclusively on observable behavior.

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