WESTERN REGIONAL RESEARCH PUBLICATION

W-133

BENEFITS AND COSTS OF RESOURCES POLICIES AFFECTING PUBLIC AND PRIVATE LAND

12TH INTERIM REPORT
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INTRODUCTION

This volume contains the proceedings of the 1999 W-133 Western Regional Project Technical Meeting on "Benefits and Costs of Resource Policies Affecting Public and Private Land." Some papers from W-133 members and friends who could not attend the meeting are also included. The meeting took place February 24th - 26th at the Starr Pass Lodge in Tucson, Arizona. Approximately 50 participants attended the 1999 meeting, are listed on the following page, and came from as far away as Oslo, Norway.

The W-133 regional research project was rechartered in October, 1997. The current project objectives encourage members to address problems associated with: 1.) Benefits and Costs of Agro-environmental Policies; 2.) Benefits Transfer for Groundwater Quality Programs; 3.) Valuing Ecosystem Managment of Forests and Watersheds; and 4.) Valuing Changes in Recreational Access.

Experiment station members at most national land-grant academic institutions constitute the official W-133 project participants. North Dakota State, North Carolina State, and the University of Kentucky proposed joining the group at this year's meeting. W-133's list of academic and other "Friends" has grown, and the Universities of New Mexico and Colorado were particularly well represented at the 1999 W-133 Technical Meeting. The meeting also benefitted from the expertise and participation of scientists from many state and federal agencies including California Fish and Game, the U.S. Department of Agriculture's Economic Research and Forest Services, the U.S. Department of Interior's Fish and Wildlife Service, and the Bureau of Reclamation. In addition, a number of representatives from the nation's top environmental and resource consulting firms attended, some presenting papers at this year's meeting.

This volume is organized around the goals and objectives of the project, but organizing the papers is difficult because of overlapping themes. The last section includes papers that are very important to the methodological work done by W-133 participants, but do not exactly fit one of the objectives. -- I apologize for the lack of consistent pagination in this volume.

On A Personal Note... Any meeting or conference is successful (and fun!) only because of its participants, so I would first like to thank all the people who came and participated in 1999 - listed below. I also want to thank Jerry Fletcher for all his help at this meeting and prior to it, and John Loomis who passed on his knowledge of how to get a meeting like this to work, and who continues to have the funniest little comments to lighten the meetings up. I especially thank Paul Jakus, who helped me to organize this conference and have a lot of fun during it and afterward. Finally, I want to thank Nicki Wieseke for all her help in preparing this volume, and Billye French for administrative support on conference matters.

W. Douglass Shaw, Dept. of Applied Economics & Statistics, University of Nevada, Reno.
June, 1999

P.S. P.F. and J.C. - As far as I can tell, that darn scorpion is still dead!

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Paper Presentation: 1999 W-133 Meetings, Tucson Arizona

Implicit Prices of CRP Enrollments, Wetlands, and Soil Quality in North Dakota

By

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Abstract:

The hedonic valuation method is used to quantify the determinants of farmland rental rates in North Dakota at the county level of analysis. Rental values are specified to be a function of soil based productivity indices, the existence of wetlands and gross farm returns, and county CRP enrollments. CRP acreage was endogenized and resulted from CRP payment levels and the extent of cumulative CRP enrollments. Crop returns, soil productivity and CRP enrollments exerted significant positive influences on farmland rental rates. Increased wetland acres negatively affected county rental rates. Specific research methodologies for conducting such hedonic analyses at township and farm levels of analysis are proposed and discussed.

Introduction:

The determinants of farmland rental values in North Dakota are not fully understood which makes it difficult to evaluate and plan agricultural policy and rural development efforts. For example, it remains uncertain as to why farmland rental values are stable or increasing in many parts of the State while net farm income is decreasing. Similarly, it is not known how factors such as soil based productivity indices, levels of Conservation Reserve Program (CRP) enrollments, the existence of wetlands, farm profits, and farm ownership patterns interact with each other and influence farmland rental values throughout the State.

The hedonic valuation method (HVM) is a commonly used technique to quantify the determinants of farmland rental values with farmland rental rates or values being the dependent variable and a variety of farm characteristics including infrastructure, location, and soil productivity indices being the independent variables. However such hedonic studies of farmland rental rates have to date been limited due to their inability to incorporate information concerning CRP enrollments and the existence of wetlands, and other site specific information.

This present research will be focussed on two issues: The ability to estimate the determinants of cropland rental values at the county level of analysis, and the impact of soil productivity, agricultural market conditions, CRP enrollments and the extent of wetlands in a region on cropland rental values. This information should help to provide farmers and government officials with a better understanding of the impacts of various agricultural policy and rural development

A Review of the Literature:

Past Hedonic Studies Focused on Farmland Values:

The hedonic valuation method (HVM), commonly referred to as the hedonic or price attribute methods, uses the relationship between the prices and characteristics of a market good to estimate the value of particular characteristics associated with that good. One of the earliest applications of the HVM to value land as a differentiated factor of production was conducted by Palmquist (1989).

Past hedonic studies associated with farmland valuation have focused either on soil characteristics (Ervin and Mills 1986; Gardner and Barrows 1985; Miranowski and Hammes 1984), soil conservation, drainage technologies, or institutional factors (King and Sinden 1988; and Palmquist and Danielson 1989), and urbanization (Chicoine 1981; Pardew et al., 1986 and Shonkwiler and Reynolds, 1986.). These studies conclude high levels of farmland productivity positively affect values and that high levels of erosion, wetland acres and poorly drained soils negatively influence farmland values, that urbanization pressure positively influences farmland values, and that farm size has an ambiguous or mixed effect on farmland values.

Past Hedonic Studies Focused on Farmland and JAS Data:

A series of recently published papers by agricultural economists focused on the potential of using site specific June Agricultural Survey (JAS) land value and farm characteristics data for: hedonic analyses of farmland values (Roka and Palmquist, 1997), capitalizing government payments into farmland values (Barnard et al., 1997), and separating the effects of environmental characteristics from the other determinants of farmland values (Boisvert et al., 1997).

The focus of the Roka and Palmquist (1997) research was to examine the use of land value data from the JAS to estimate hedonic price functions in a five-state region of the cornbelt. To supplement soil attributes, crop yield and other bio-physical data missing from the JAS survey, the authors attempted to integrate (spatially reference) National Resource Inventory (NRI) data to the JAS data but his was not very successful as NRI and JAS sampling frames and scales of analysis did not coincide. Therefore, the authors used county level corn yields averaged over three agricultural censuses as a proxy measure of productivity and the percentage of land designated as 'prime farmland. The study concluded that land values from the JAS were reliable and could be potentially useful for estimating hedonic models of farmlands, especially at the national level. However, at the same time it was realized that the lack of site specific information from JAS data, specifically soil quality and productivity data, greatly diminished the ability to estimate the determinants of farmland prices.

The research project which focussed on the effect of government program payments on farmland values (Barnard et al., 1997) also used JAS land value data as the dependent variable while the explanatory variables used (farm size, soil productivity, government payments, etc) came from county level data sources. The measure of soil productivity was the 'soil relative productivity indices at the county level calculated by the NRCS. These county level explanatory variables were not found to be sufficiently detailed or site specific which limited the predictive strength of the estimated models. However this research did show that there exists a wide spatial variability (nationally) in the percentage of direct government payments that are capitalized into cropland values.

This limitation associated with hedonic studies of farmland rental rates is related to the difficulty of integrating highly site specific farmland rental rate data with GIS based land use, soil productivity, wetland and CRP related data. As noted by Reynolds (1997), none of the previously mentioned hedonic-farmland studies that used JAS or USDA Area Data were able to effectively integrate such GIS based data into their study designs. Recently available GIS data relevant to hedonic studies of farmland values include: GIS soil databases (STATSGO and SSURGO) distributed by the NRCS, the National Wetland Inventory (NWI) from the USFWS, and CRP enrollment data being compiled digitally in some States by a variety of different agencies (these issues are discussed in greater detail in the next section of the proposal).

Finally one of the AJAE papers focused on the effects of environmental characteristics from the other determinants of farmland values. This approach regressed farmland rental values (from direct farmer surveys) against field level (site specific) data describing the characteristics of the parcels, land productivity, operator characteristics and environmental variables (presence of a conservation plan, leaching and runoff measures and environmental vulnerability indices). The results showed that the value of agricultural land can be directly related to levels of farmland productivity, spatial orientation, and environmental vulnerability. However the authors explicitly stated that additional empirical testing across other regions of the country with different soil, productivity indices, cropping patterns and environmental vulnerability measures was needed.

In summary the past research on the determinants of farmland values have been limited by the lack of detailed and/or site specific data describing farmland characteristics. In order to fully evaluate the factors effecting farmland values in an entire state or in a region, more complex and larger databases are needed. It is highly unlikely that such improvements in data detail and scope will be possible without the use of GIS technologies and better access to JAS or NASS farmland valuation data.

Farmland Rental Values in ND

Farmland across the country is subject to low levels of turnover (less than 3% of farmland is sold annually). Therefore, the most commonly reported and used estimates of land value are often based on farmer's self-assessments of their land values. Such self-assessed farmland value data has been collected annually since 1994 by the USDA's June Agricultural Survey (JAS). In ND the JAS is administered by the NDASS and is based on 420 randomly selected land segments (tracts) of approximately 1.5 square miles each that involve surveys with approximately 1500 individual farm owners.

Rental values for farmland in ND are also collected annually (in January) in a separate survey by the NDASS. Such rental data may be very indicative of conditions in the agricultural economy since a large percentage of farmland in ND is rented (for example, up to 1/3 of all farmland in the Red River Valley is rented). This data, which is reported at the county level of analysis, is based on survey reports with 850 hayland farmers, 1390 pastureland farmers, and 2036 cropland farmers. Considerable range in rental rates occur between counties throughout the state as well as within individual counties.

Farmland values in ND are often classified and analyzed by 3 land use types: non-irrigated cropland, non-irrigated pasture land, and non-irrigated hayland. Further stratification results in a regional level of analysis, with the common regions being the North Red River Valley (NRRV), South Red River Valley (SRRV), Southeast Central

(SEC), Northeast Central (NEC), Northwest Central (NWC), Southwest Central (SWC), Southwest (SW), and Northwest (NW).

Preliminary analyses have shown that nominal ND rental rates and land values are currently at a 10 year high, in spite of decreasing farm incomes in most of the State. However, the real rents have remained flat for the last 10 years. Although there are no known studies to quantify and explain why farm rental values have been increasing, it has been hypothesized that the following factors may be influencing farmland rental values: high levels of CRP enrollments that tend to put a floor under rental values, low interest rates that make land ownership a competitive investment alternative, and a stable and diversified economy across the state. (Swenson, 1998)

An econometric analysis of representative farms in ND which treated cropland rental values as a moving average of annual return to land multiplied by the long-run capitalization rate, plus taxes on has estimated that in the next 10 years cropland values for medium sized farms will decrease 0.8% while cash rents will fall by 13.6%. For farms expected to experience average profits, cropland values will increase by 13.6% while cash rents will fall by 4.8% (Koo, Taylor and Duncan, 1998).

There remains considerable debate and uncertainty as to why farmland rental values are stable and/or increasing in many parts of the State while net farm income is decreasing. Similarly, it is not known with any certainty whether and how factors such as conservation reserve program (CRP) enrollments, soil and climatic conditions, distance to markets, commodity prices and net profit for particular crops, and farm ownership patterns interact with each other and influence farmland rental values. This paper represents an initial effort to understand the factors influencing rental rates.

The Demand for Rental Land

Consider the profit maximizing farmer seeking rental land to expand his farming operation in a particular year. For simplicity, we consider an appropriately aggregated production process to represent the farmer's problem as maximization a single output problem. The farmer's restricted problem can be framed:

$$\textit{Maximize} \qquad p f(\mathbf{x},\mathbf{z};\mathbf{a}) \ - \ \mathbf{w}'\mathbf{x} \ - \mathbf{r}'\mathbf{z}$$

Where **x** is a vector of variable inputs, excluding land, **z(h)** is a vector of available rental parcels differentiated by characteristics such as soil productivity, proportion of land unusable due to flooding, roads, shelterbelts, etc., and **a** is a vector of factors quasi-fixed in the relevant planning period. The vectors **w** and **r** are corresponding prices for the factors, p is output price, and f(·) is the production function.

Finding first order conditions and application of the implicit function theorem results in explicit demand functions for rental land of various characteristics, $\mathbf{z}=\mathbf{z}^*(\mathrm{p},\mathbf{w},\mathbf{r};\mathbf{a})$. For given levels of the arguments of the demand function for $\mathbf{z}^*$, quantities of land demanded of specific characteristics can be derived.

The supply of rental land results from individual landowner's decision to personally farm the land or to make the land available on the rental market. . Ignoring for this iteration of the model the decision to rent or to farm the land himself, the landowner may still decide to enroll a portion of his land in the Conservation Reserve Program (CRP). Acres under CRP designation cannot be farmed while under contract, and hence reduce the supply of land available for either cash or share rent. The effects of placing land under CRP will affect the rental land market in a fashion similar to other land characteristics that reduce the supply of productive rental land.

Enrolment in CRP by the owner of the land is also presumed to result from solution of an optimization problem. The owner who has land to rent faces the following constrained revenue maximization problem:

Maximize $r(z_f)z_f + cz_c$

Subject to $z_f + z_c = Z$

For simplicity, we assume that the landowner has land of a single quality and the land is differentiated solely on the basis of his decision to enroll some or all of the acres in CRP. Farmable acres are z_f and z_c are acres of the farmer's total parcel Z enrolled in CRP. Price c is the per acre CRP specified in the farmer's contract with the Farm Service Agency, the USDA agency administering the CRP program. We presume that the per acre rental rate received by the landowner is influenced by the total number of farmable acres available, $r(z_f)$.

First order conditions for the landowner's problem framed as a Lagrangian are:

$$\frac{\partial L}{\partial z_f} = r_f(z_f)z_f + r(z_f) - \lambda = 0$$

$$\frac{\partial L}{\partial z_c} = c - \lambda = 0$$

$$\frac{\partial L}{\partial \lambda} = Z - z_f - z_c = 0$$

Subscripts on $r(z_f)$ indicated first partials of the rental function with respect to farmable acres z_f. The implicit function theorem allows rewriting of the first order conditions in terms of the single parameter c (ignoring Z),

$$r_f\left(z_f^*(c)\right) z_f^*(c) + r\left(z_f^*(c)\right) - \lambda(c) \equiv 0$$
$$c \;-\lambda\,(c) \equiv 0$$
$$Z - z_f^*(c) - z_c^*(c) \;\equiv\; 0$$

Comparative statics relationships can be found by taking the second derivatives of the first order conditions:

$$\begin{bmatrix} z_f\, r_{ff} + 2r_f & 0 & -1 \\ 0 & 0 & -1 \\ -1 & -1 & 0 \end{bmatrix} \begin{bmatrix} \partial z_f/\partial c \\ \partial z_c/\partial c \\ \partial \lambda/\partial c \end{bmatrix} = \begin{bmatrix} 0 \\ -1 \\ 0 \end{bmatrix}$$

By the second order conditions for constrained maximization, the coefficient matrix must be positive definite, implying $z_f r_{ff} + 2r_f < 0$. Since

$$\left(z_f\, r_{ff} + 2r_f\right)\partial z_f/\partial c \;-\; \partial\lambda/\partial c = 0 \text{ , and}$$
$$\partial\lambda/\partial c = 1,$$

this results in the unambiguous comparative static result $\partial z_f/\partial c < 0$. Given the constraint of the problem, we can further expect $\partial z_c^*/\partial c > 0$ for the revenue maximizing landowner. As payments to land enrolled in CRP increase, acreage enrolled will increase. As acres in CRP increase, the supply of rental acres available for farming decreases. As supply decreases, we would expect equilibrium prices for rental land to increase.

Combining the renter's demand for rental land and the availability of parcels containing different proportions of CRP acreage based on the owner's decisions results in an inverse demand function for rental land of different categories, $\mathbf{z} = \mathbf{z}^*(\mathrm{p},\mathbf{w},\mathbf{r};\mathbf{a})$. However, lack of a suitable number of observations on land in each of the different characteristics available in North Dakota results in an aggregate demand model for rental land, where factors differentiating land characteristics are treated as explanatory variables in determining rental value, $z = z(p, \mathbf{w}, r, \mathbf{h}; \mathbf{a})$, where **h** is a set of land characteristics, including extent of participation in CRP for the aggregate land parcels considered.

We make various simplifications in determination of the land rental model. First, aggregate output prices are not available for each of the 53 North Dakota counties. We use instead the county level gross crop receipts per acre RCPAYA, lagged one year, to represent expected returns from farming (i.e., *p*). Land characteristics considered in the set **h** are the average soil productivity index for the county, SOILPRO and county acres in wetlands, WET. Finally, landowner decisions to place land in CRP each year are represented by DELCRP.

The resulting hedonic model is thus:

$$(1) \qquad RENT_{it} = f(RCPAYA(-1)_{it}, SOILPRO_{it}, WET_{it}, DELCRP_{it})$$

Observations are available for each county i over the time period 1989-1997. The explanatory variable DELCRP is itself a function of landowner decisions to enroll additional acres in CRP based on the revenue maximization model discussed above.

Consequently, estimation of a hedonic model based on (1) was conducted using two-stage least squares. Additional instruments included were CRP per acre payments (i.e., *c*), lagged one year, as well as the percentage of land within the county already enrolled in CRP. The latter term was added to represent county acres remaining eligible for CRP designation. All prices and returns were converted to 1997 dollars.

Summary statistics for the variable levels are listed in Table 1 and 2SLS results are in Table 2.

Table 1. Variable summary statistics

	Mean Levels	Standard Deviation
Rent	32.065	11.808
RCPAYA(-1)	87.647	53.262
SOILPRO	46.774	17.058
WET	45,493	40,379
DELCRP	1,297	4,648

Preliminary model results show significance for all variables chosen as well as correct signs as expected from the behavioral models posited above. Approximately 70.72% of the variation in RENT was explained by the model. First stage estimation of the endogenous variable DELCRP resulted in the expected positive (and significant) influence of CRP payments per acre. Mean county rental values are positively affected by agricultural market conditions, as expressed in the previous year's crop returns per acre for the county. Coefficients on site characteristics were also of the expected sign.

Better soil productivity within a county positively affected rental rates. Acres of wetland within the county have a depressing effect on land rental values.

Table 2. Two-stage least squares estimation results of county level mean rental values:

	Estimated Coefficient	Standard Error	t-statistic
Constant	6.18200	1.0253	6.0295
RCPAYA(-1)	0.14529	0.01250	11.628
SOILPRO	0.27432	0.03802	7.2148
WET	-0.324E-4	0.9E-5	-3.6020
DELCRP	0.00138	0.00016	8.8171

Of special interest to the objectives of the research is the strong positive influence exerted by CRP additions on land rental rates within the 53 North Dakota counties. A very loose interpretation of the results might indicate that, given the average increase in CRP land enrolled each year across the state (1,297 acres as seen in table 1), land rental rates within the "average" county are increased by $1.80 per acre. This is an approximate 5.6% increase in land rental values attributable to enrollments in CRP within the state of North Dakota.

Future Research

Model results at the county level provided fairly good explanation of cropland rental values. However, land characteristics may be very different within a single county, wetland acreage may be concentrated within smaller areas within a county, and crop incomes may vary around a county based on climate differences and other environmental factors. Consequently, the next step in analyzing factors affecting farmland rental rates will assemble data at the township level in order to further isolate the effects of environmental factors on rental rates.

Land characteristic data is available at the township level of analysis (in 5 individual ND counties). Rental values at the township level are potentially available from two sources. One option is to obtain rental data from surveying farm operators. Another option to obtain farmland rental values at the township or even more disaggregated levels of analysis would be to use farmland rental value data collected annually by the June Agricultural Survey (JAS) that is administered by NDASS. JAS data also includes various farm production information collected at the tract level of analysis (resolution of approximately 1.5 square miles). However, a preliminary request to access JAS through the NDASS has been rejected due to confidentiality concerns. Further requests are currently being investigated.

In order to identify the range of soil productivity across counties or townships (which may be important for areas with highly heterogeneous soils) it will be necessary to perform analyses on spatially referenced NRCS soils databases. Two such databases have already been obtained: the first is the STATSGO soils database at the 1:250,000 scale of analysis for the entire State, and at the more detailed and site specific 1:20,000 scale (the SSURGO database) which is presently available for only 5 counties in the State. Soil productivity indices will be estimated using the same criteria as the 1960 NDSU soil productivity classifications (based on soil types and characteristics and climatic factors) and aggregated by counties (n=52) and townships (n=180).

In order to estimate cropland acres planted at the township level it will be necessary to utilize a satellite imagery database of cropland types compiled for ND in 1997 by the NDASS.

In order to determine wetland acreages at the township level of analysis it will be necessary to utilize the NWI database. Wetlands can impose impediments to planting and harvesting nearby fields and may also be associated with poorly drained soils and are therefore expected to decrease cropland rental rates.

In order to determine the percentage of CRP lands and corresponding CRP rental payments occurring at the township levels of analysis, it will be necessary to access CRP contract files contained in county FSA offices. Permission to this data has been granted from the State FSA office.

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Water Contamination From Agricultural Chemicals:

Welfare Measures for Chemigation Producers

By P. Joan Poor
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Nebraska producers have been applying agricultural chemicals to their crop land through irrigation systems since the 1960s. Beginning in 1987, environmental regulatory requirements were implemented to prevent water contamination associated with this practice. The purpose of this paper is to use the contingent valuation method to estimate the value chemigation producers place on a reduction in ground and surface water contamination from agricultural chemicals, and to estimate producer surplus associated with a change in environmental quality resulting from the adoption of a new chemigation technology. The analysis shows that a sample of Nebraska certified chemigation applicators indicated positive willingness to pay estimates to reduce water pollution from agricultural chemicals applied through irrigation systems.

Introduction

Negative externalities that arise from the use of agricultural chemicals including groundwater and surface water contamination, have significant environmental as well as human health consequences. Over the past 40 years the increased use of agricultural chemicals has contributed to increases in crop yields and lower production costs, resulting in increased profits. However, the potential hazards associated with the use of these chemicals on the environmental and human health have raised concerns (Fernandez-Cornejo, Jans and Smith, 1998). These concerns have resulted in the increase of environmentally sustainable agricultural production systems such as integrated pest management techniques and more environmentally sound technologies such as organic pesticides. Concurrently, government intervention through regulatory actions to test and licence pesticides has occurred in an effort to ensure that the chemicals that are used are safe for the environment. More efficient application of agricultural chemicals is another method producers can use to reduce not only the risk to their own health but also minimize environmental impacts such as water contamination. One technology that enables producers to optimize (or minimize) chemical usage, reducing costs and environmental impacts, is known as chemigation. Chemigation is a term used for the process of applying agricultural chemicals to field crops through pivot irrigation systems. As long as the chemigation system is equipped with appropriate safety features to guard against water contamination and the applicator is properly trained to operate the system, this technology promotes more efficient use of agricultural chemicals.

Recent empirical studies regarding the adoption of sustainable agricultural technologies often consider whether a producer chooses to adopt these technologies given a set of socio-economic explanatory variables (Mullen et al. 1997; D'Souza et al. 1991; Caswell and Zilberman 1985; Feder and Slade 1984; and Harper et al. 1990). The environmental benefits associated with adoption are often considered separately using non-market valuation techniques. The contingent valuation method has been used to estimate compensating variation measures of producer willingness to pay for an improvement in environmental quality associated with the adoption of safer chemicals (Lohr, Park and Higley 1996; Owens, Swenton, vanRavenswaay 1997 and 1998). In addition, a utility based hedonic analysis of herbicide expenditures was used by Beach and Carlson (1993) to investigate farmer safety and water quality issues associated with chemical prices. This research furthers those efforts and uses the contingent valuation method to ask Nebraska producers who already chemigate within a regulatory setting that requires water contamination prevention devices and training, if they would adopt a new chemigation technology that would further improve environmental quality. By considering only chemigating producers, one can better determine if they are motivated to chemigate solely for profit maximization reasons, or whether their decisions also consider environmental consequences. As such, the purpose of this paper is to use the contingent valuation method to estimate the value chemigating producers place on a reduction in ground and surface water contamination from agricultural chemicals, and to estimate producer surplus associated with a change in environmental quality resulting from the adoption of a new chemigation technology.

Conceptual Framework

First consider a profit maximizing producer who chooses to apply agricultural chemicals to crop land through a pivot irrigation system. Associated with the practice of chemigation is a high probability of both surface and groundwater contamination, through surface spillage and/or back flow through the irrigation well (a negative externality or market failure which has no impact on the profit maximizing producer's production decisions). Regulations requiring producers who practice chemigation to be trained and certified, and to install safety equipment as indicated by a valid permit, act to internalize the water contamination costs. In the presence of such regulations, the practice of chemigation is considered environmentally sustainable. Considering chemigation technology to be a quasi-fixed input or factor of production is reasonable because after the initial investment in the chemigation equipment, the costs associated with certification and well permits are only required if the producer chooses to produce a positive output using this technology. It is also possible that the producer chooses to comply with the regulatory requirements but does not necessarily chemigate every year. The following dual or indirect, restricted profit function appropriately describes the profit maximizing producer:

(1) $$\pi^R = f(P,W;Z^0) \equiv \max_{y,x_i} \{P \bullet y - w \bullet x_i \;\; / \;\; (y,x_i) feasible\}$$

Where P is the price of the optimal level of a single output y, and w is a vector of prices for the optimal levels of variable inputs x_i and Z^0 the quantity of a quasi fixed or environmental input, or the current level of chemigation technology used by the producer. Equation (1), the indirect restricted profit function, satisfies the following properties:

(1) π^R is convex and continuous in prices, P and w;

(2) π^R is positive or equal to zero. (Non-decreasing in P and non-increasing in w);

(3) π^R is linearly homogeneous in prices P and w for a given Z;

(Cornes 1992; Johansson 1993; Chambers 1988; Kumbhakar and Bhattacharyya 1992; and Tsuneki 1987). Given that the indirect restricted profit function is continuous and differentiable with respect to the prices P and w, Hotelling's Lemma can be used to derive output supply and variable input demand functions as follows:

(2) $$\frac{\partial \pi^R}{\partial P} = y \qquad \text{and} \qquad \nabla_w \pi^R = -x_i$$

Now assume that π^R is also continuously differentiable with respect to Z, such that:

(3) $$\frac{\partial \pi^R}{\partial Z} = \phi$$

where Φ is the shadow price (SP_Z in Figure 1) of the quasi fixed input or the profit maximizing producer's willingness to pay for the use of an extra marginal unit of Z (Tsuneki 1987; and Wear and Newman 1991). This definition can be extended further as per Johansson (1993), in that the producer's willingness to pay represents a change in producer surplus associated with a change in the environmental input or environmental technology, as follows:

(4)

$$\phi = WTP = \pi^R(P,w,Z^1) - \pi^R(P,w,Z^0)$$
$$= \int_{Z^0}^{Z^1} \pi_Z^R dZ$$

Figure 1 illustrates the producer surplus measure where Z is defined as an environmental technology input. The following section provides an empirical application of how data from a contingent valuation survey can be used to estimate producer surplus as a change in a restricted profit function.

Empirical Application: Producer Surplus Estimate for Chemigation Producers in Nebraska

Nebraska is known for its intensely cropped and irrigated farmland. As a result, both ground water and surface water in some parts of the State have become contaminated with agricultural chemicals(USGS 1996). Agricultural chemicals have been applied to Nebraska crop land (as in many other states), through irrigation systems since the 1960s. This chemigation process, has been widespread in Nebraska since the mid 1970s. Chemigation requires unique equipment, as well as applicator training to operate such systems. This process can be advantageous to producers because it provides a cost effective method of applying agricultural chemicals in a relatively uniform and flexible manner. The major draw back is that in the absence of specialized safety equipment and training, chemical back flow into groundwater through irrigation wells can occur and may lead to significant groundwater contamination. Also, lack of training may contribute to accidental discharges of chemicals to the soil surface. Given the presence of this potential water contamination (or negative externality), Nebraska implemented legislation effective January 1, 1987, which placed numerous requirements on producers who choose to apply chemicals through their irrigation systems. These requirements include installation of safety equipment, annual site permits, mandatory equipment inspections, applicator training, testing and certification, accident reporting and penalties for noncompliance (Nebraska Cooperative Extension Service 1985; and Eisenhauer and Buttermore 1991).

In Nebraska those producers using chemigation can be characterized as profit maximizers in the presence of environmental regulations which attempt to internalize the environmental quality costs associated with chemigation. In order to estimate producer surplus or willingness to pay using the restricted indirect profit function framework described above, a hypothetical contingent valuation (CV) scenario was developed and administered to a random sample of certified chemigation applicators in Nebraska. In the spring of 1998, as part of a larger chemigation technology assessment project, 1000 questionnaires containing this hypothetical CV scenario were mailed to individuals trained and certified by the State of Nebraska to apply agricultural chemicals through permitted pivot irrigation systems. Nebraska Department of Environmental Quality personnel who administer the chemigation certification and permitting program, indicated that only about 85% of those people listed as certified chemigators were actual producers, where the remaining survey recipients, who were not able to complete the CV questionnaire, were either Natural Resource Conservation District employees or agricultural chemical sales representatives. Therefore the number of actual chemigating producers who were sent the chemigation questionnaire was assumed to be 850. Taking this calculation into consideration, the survey response rate adjusted for undeliverable and unusable questionnaires was 19%, yielding 152 usable completed questionnaires. No second mailing of the questionnaire occurred because this was part of a larger technology assessment study which did not require additional responses. Although this is considered a very low response rate, it provides a sufficient data set for the given empirical application using the restricted indirect profit function framework.

The CV hypothetical scenario asked producers what they would be willing to pay for a new chemigation technology that maintained current production levels, but reduced water contamination by 50% compared to current practices. The scenario followed a double bounded, dichotomous choice framework where four bid sets were used, described in Table 1. Based on the respondent's answer to the first willing to pay question, they were asked a follow up question in an effort to establish upper and lower bounds on their true willingness to pay for the hypothetical change in water quality. The survey respondents were reminded that this technology does not exist and that their responses in no way reflect whether it will ever be developed. The bid values were presented as an additional dollar cost per year, per irrigation well. The distribution of survey responses is presented in Table 2. Consistent with expectations, as the initial bid value increases, the proportion of respondents who answered YES to the initial bid value declined from 84% for an initial bid of $1 to 55% for an initial bid of $100. Similarly the proportion of respondents who answered NO to the initial bid increased from 17% to 45%, corresponding to the initial bids of $1 and $100, respectively. These results indicate that the survey respondents appear to have seriously considered the bid values when responding to the valuation question.

The econometric model presented in equation (4) represents respondent willingness to pay. Where Z^1 represents the new hypothetical chemigation technology and Z^0 represents the currently used chemigation technology. Willingness to pay or Φ_Z , can then be directly estimated from the CV survey data using Cameron's (1988) approach. We define Φ_Z as follows:

$$(5) \qquad Max\phi_{zi} = x_i'\beta + \varepsilon_i$$

where x_i is a vector of attributes unique to respondent i's production situation and ξ_i is a random error term. It is assumed that willingness to pay is positive and that the underlying willingness to pay commutative distribution $G_\xi(\bullet)$ has a Weibull distribution. There are four response probabilities (P^{YY}, P^{YN}, P^{NN}, P^{NY}) associated with each of the possible responses, (YES-YES, YES-NO, NO-NO, NO-YES), where the initial bid is B and the follow up bid is either B^U (upper bid) or B^L (lower bid) depending on the initial response (Hanemann and Kanninen, 1996). These probabilities yield the following log-likelihood function specification:

$$(6) \qquad \ln L = \sum_{i=1}^{S}[I^i_{YY}\ln(P_i^{YY}) + I^i_{YN}\ln(P_i^{YN}) + I^i_{NY}\ln(P_i^{NY}) + I^i_{NN}\ln(P_i^{NN})]$$

where S is the sample size and the term I_{xy} denotes an indicator function equal to 1 when the two responses are xy, and zero otherwise (Hanemann and Kanninen, 1996). Assuming willingness to pay is a non-negative random variable, the relationship can be defined from equation (5) as:

$$(7) \qquad \ln(WTP_i) = x_i'\beta + \varepsilon_i \quad i = 1.....S$$

The regression results from the chemigation producer survey, estimate a mean willingness to pay of \$143.23 and a median of \$45.59. The regression model was also run including a vector of explanatory variables in an effort to determine those factors which significantly influence a respondent's willingness to pay or the producer surplus welfare estimate. Explanatory variables include the size of the respondents farming operation in terms of irrigated acreage, and years of farming experience as indicated by their age. Variables related to chemigation include equipment costs, years of chemigation experience and the number of permitted irrigation wells they operate. A variable was also included in an effort to determine if the respondent considered whether they were located in an area known for high agricultural chemical contamination, when answering the hypothetical willingness to pay questions. The regression equation was:

(8)

$$\phi_Z = f(\underset{(-)}{TOTAL}, \underset{(-)}{REALCOST}, \underset{(+)}{YRSCHEM}, \underset{(-)}{OLD}, \underset{(+)}{NITRATE}, \underset{(+)}{WELLS})$$

Where TOTAL is the total irrigated acres per respondent; REALCOST is the deflated chemigation equipment cost in \$1991 (US Government, 1998); YRSCHEM are the number of years the respondent has chemigated; OLD is a dummy variable which equals one if the respondent is greater than 50 years old; NITRATE is a dummy variable equal to one if the respondent was located in a Natural Resource District known for high nitrate groundwater contamination (Exner and Spalding, 1990) and WELLS is the number of permitted chemigation wells operated by the respondent. These explanatory variables and their corresponding sample means are presented in Table 3.

Expectations regarding the coefficient signs for the explanatory variables are the bracketed terms directly below the variable name in equation (9). It was expected that total irrigated acreage, deflated chemigation equipment costs and the older the respondent, the less likely they are willing to pay for water quality improvements by investing in new chemigation equipment.

Also, the larger the farming operation, and higher the equipment costs, the less likely the producer will be to invest money in environmental quality improvements. The number of years of experience chemigating is expected to be positively related to willingness to pay for water quality improvements via a new chemigation technology compared to those respondents with less experience. Similarly those producers with more permitted chemigation wells are assumed to be more environmentally concerned and thus would be willing to pay more for the new hypothetical technology than those with fewer permitted wills. Finally the NITRATE variable is expected to be positively related to willingness to pay, in that respondents located in areas where groundwater contamination from agricultural chemicals is already a concern, will be likely to pay more to improve water quality than those respondents located in areas where groundwater contamination is less of a problem.

The regression results from equation (9) are presented in Table 4. The coefficient signs were consistent with expectations and all variables except REALCOST and OLD were significant at the 10% level. The results of this empirical application show that Nebraska producers who engage in chemigation on average, are concerned with water contamination from agricultural chemicals in that they indicated a positive willingness to pay to reduce associated water contamination. This conclusion is based on a change in a dual restricted profit function approach to estimate willingness to pay or a producer surplus welfare measure. The results also indicate that producers with more chemigation experience, as well as those located in regions where groundwater contamination from agricultural chemicals is a reported concern, are willing to pay more to reduce contamination than chemigation producers not possessing such attributes. Thus according to these results, profit maximizing chemigation producers do not only use their pivot irrigation systems to apply chemicals in order to maximize profits, but they also are concerned with the potential environmental degradation that may result from this practice.

Conclusions

This paper presents the conceptual framework using a dual restricted profit function, to derive producer surplus welfare estimates associated with a change in environmental quality. An empirical application is presented whereby the contingent valuation method is used to estimate producer surplus of chemigating producers in Nebraska, associated with a reduction in groundwater contamination from agricultural chemicals. The results indicate that producers who are already regulated by an environmental policy are still willing on average, to commit additional money to improving water quality particularly where the water source may be contaminated from agricultural chemicals. Chemigating producers in Nebraska do appear to be concerned with water quality and their impact on it, as well as maintaining viable and profitable agricultural production units. Further research of interest would be to extend the use of contingent valuation analysis in an indirect restricted profit function context, to additional agricultural producers, as one way of understanding producer profit motives verses their concern for the environment and natural resources.

Table 1: Contingent Valuation Scenario Bid Sets

Initial Bid	Upper Bid	Lower Bid
$1.00	$2.00	$0.25
$5.00	$10.00	$2.50
$20.00	$40.00	$10.00
$100.00	$200.00	$50.00

Table 2: Contingent Valuation Scenario Initial Bid and Associated Response Proportions

Initial Bid	%YY	%YN	%NN	%NY	Total Number of Responses
$1	78 (28)	6 (2)	14 (5)	3 (1)	36
$5	67 (24)	14 (5)	6 (2)	14 (5)	36
$20	41 (2)	33 (16)	20 (10)	6 (3)	49
$100	32 (10)	23 (7)	26 (8)	19 (6)	31
Brackets indicate actual number of respondents per category.					

Table 3: Variable Names and Descriptions

Variable Name	Mean	Variable Description
TOTAL	639.14	Total irrigated acres per respondent.
REALCOST	1656.63	Deflated chemigation equipment costs in $1991 (US Government, 1998)
YRSCHEM	9.92	Number of years the respondent has chemigated.
OLD	0.36	Dummy where age is greater than 50 years OLD=1, else OLD=0.
NITRATE	0.82	Dummy equal to 1 if the respondent was located in a Natural Resource District known for high nitrate groundwater contamination (Exner and Spalding, 1990)
WELLS	4.41	Number of permitted chemigation wells operated by the respondent.

Table 4: Equation (9) Regression Results

Variable	Coefficient Estimate	Standard Error
INTERCEPT	2.9095**	1.2069
TOTAL	-0.0024**	0.0013
REALCOST	-0.0002	0.0003
YRSCHEM	0.1178**	0.0603
OLD	-0.3022	0.7136
NITRATE	1.6362*	0.9533
WELLS	0.3612*	0.2106
Scale	1.7919	0.3001
pseudo R-squared	0.52	
Log Likelihood	-88.2684	
* and ** indicate significant levels at 10% and 5%, respectively.		

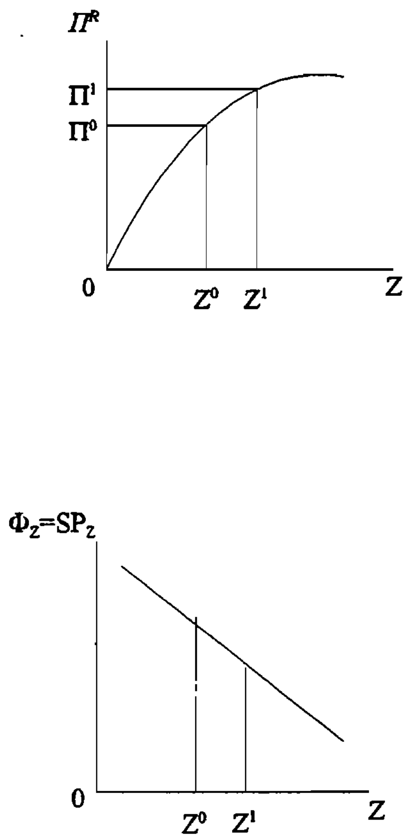

Figure 1:Producer Surplus Diagrams: Given a Change in the Environmental Technology Z.

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Welfare Losses Due to Livestock Grazing on Public Lands:
Some Evidence from the Hoover Wilderness

by

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Abstract:

Backcountry hikers' willingness to pay for removing grazing from trails in the Hoover Wilderness is analyzed by linking a multinomial logit model of trip allocation with a Dirichlet distribution so that seasonal trips can be properly aggregated. Seasonal welfare measures are derived from an incomplete demand specification. Results show that hikers' welfare losses do not everywhere exceed agency revenues and producers' surpluses. Prioritization of activities is indicated on a trail by trail basis.

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Welfare Losses Due to Grazing on Public Lands

...in furtherance of the National Environmental Policy Act of 1969, as amended (42 U.S.C. 4321 et seq.), in order to avoid to the extent possible the long and short term adverse impacts associated with the destruction or modification of wetlands..., it is hereby ordered as follows:
Each agency shall provide leadership and shall take action to minimize the destruction, loss or degradation of wetlands, and to preserve and enhance the natural and beneficial values of wetlands in carrying out the agency's responsibilities... (Ex. Ord. No. 11990, May 24, 1977, 42 F.R. 26961)

Federal agencies are required under the National Environmental Policy Act to consider the economic impacts of their management decisions. Economic impacts are broadly construed to include the non-market values assigned to recreational activities on public lands. Under the above referenced Executive Order these agencies additionally must manage their operations to preserve and enhance the beneficial values of wetlands. Again there is the implication that these values need not be market based. Conflicts can arise when public lands serve multiple types of users—particularly when some use is economic and some recreational. Clearly benefits accrue to both types of users, but the difficulty is quantifying these benefits. We illustrate an approach to benefit estimation using as our example the Hoover Wilderness of eastern California. Here both grazing and back country hiking activities occur on certain trails in the wilderness; and both some hiking and some grazing take place in riparian areas.

While multiple use of public lands has been the philosophical approach used by public land managers its application has proven difficult. One reason for this is that choosing how much of each use should be allowed is usually based upon economic criteria. Unfortunately, that presupposes enough information about prices and quantities is known so as to allow management using traditional economic principles. In the case of public lands many of the alternative uses are non-market uses. As a result, managers are hampered in their efforts.

A pressing public issue in the United States is the competition between grazing and other uses for public lands. While the price of grazing permits is an administrative decision, the value of the public lands in other uses is a non-market issue. One of the competing recreational uses is backcountry hiking. Backcountry hiking is an especially interesting competing use because the conflict is so direct. The issue is that people are viewing cattle or sheep and sharing the ecosystems with these animals.

Examining the relationship between grazing and backcountry hiking is facilitated by the fact that both grazing and backcountry hiking activities are permitted in the Hoover Wilderness Area. When the Hoover Wilderness was created grazing rights were grandfathered into the enabling legislative act. While hiking had been going on for some time, in this area the designation as wilderness brought with it an administrative structure that now accounted for hiking as well as grazing. This analysis utilizes data from this wilderness to estimate the willingness-to-pay by backcountry hikers to reduce grazing and to provide estimates of the value of several ecosystems and other trail characteristics.

Non-Market Valuation Methods

Recreation demand modeling is an important element of natural resource planning. Recreational trip data constitute the primary source of information for revealed preference methods. Recreation visitation data are, however, subject to the fact that each respondent will report a discrete number of trips to a site. Yet a single, independent recreational site rarely exists. The proper evaluation of policy changes may require a systems approach if several sites are impacted simultaneously. Or if similar recreational experiences can be obtained at places near a single recreation site of interest, there may be a high degree of substitutability among such sites. Although most travel cost studies to date have assumed independence in order to estimate demand, researchers recognize the probable important interdependencies of demands for sites due to the pioneering work of Burt and Brewer. Subsequent studies by Cicchetti et al.

and Sellar et al. have provided additional evidence to justify a systems approach. Unfortunately, travel cost analyses of household (or individual) demands for multiple recreational goods typically have not accounted for the discrete, non-negative integer characterization of trip data. Or in the case where the count nature of the data was accommodated, restrictions consistent with rational behavior were not imposed (Ozuna and Gomez). The published exceptions are recent studies by Englin et al. and Shonkwiler which employ an incomplete demand specification for non-negative integer data. Yet these papers do not fully develop the proper use of the incomplete demand model when valuing environmental goods.

A conventional recreation site choice model is the multinomial logit model of McFadden. McFadden's multinomial logit model possesses useful properties for analyzing the site allocation problem because visitation data are discrete and the model can be easily used to estimate exact per-trip welfare measures for site quality changes (we ignore the additional and tangential issue of allowance for income effects here). This model, while quite popular because of its attractive features in dealing with multiple sites, limits consideration of seasonal welfare changes due to the fact that the multinomial logit's site-specific demands are estimated conditional on total demand for all sites. Many recreation modelers have raised the point that consumer's surplus measures should come from some aggregate or unconditional demand function rather than from the site-specific conditional demands, because the former allows total seasonal consumption to change in response to site quality and price changes and the latter does not.

Intuitively, when one only has per-trip welfare measures, some assumption must be made about whether and how these can be added together to arrive at a welfare measure that can be interpreted as an annual (seasonal) maximum willingness to pay (WTP) to bring about some change. One line of research has sought to link the RUM with an aggregate demand quantity (Bockstael et al., Feather et al., Parsons and Kealy, Hausman et al.). Substantial attention has been devoted to determining the appropriate aggregate price to use in the aggregate demand equation when site-specific demands have been modeled using the multinomial logit specification. However, recent work by Shaw and Shonkwiler, Smith, and Smith and Von Haefen suggests the aggregate price indexes being proposed do not provide a utility theoretic link between the RUM and the aggregate demand equation.

The foregoing discussion leads to recognition of the fact that the data necessary to specify a random utility model are typically detailed enough to provide information on site-specific demands. In this situation the data are rich enough to allow calculation of a travel cost model to each individual recreation site, and it seems logical that this information should be exploited when developing models for multiple sites. The necessary techniques to accomplish such modeling consist of a demand system that allows calculation of unconditional welfare measures and a proper econometric technique to accommodate the discrete nature of the demand quantities.

This study attempts to synthesize the elements necessary to appropriately treat multiple site travel cost models of recreation demand when the decision variables are measured as trip counts. A multivariate count data probability model is shown to provide a link between conventional logit models of trip allocation and count data models of trip demand. Because this model generates conditional demands with exponential form, a proper incomplete demand structure (LaFrance and Hanemann) will be imposed to insure that exact welfare analysis can be performed. In general it is easier and less demanding of the data to develop quasi-indirect utility functions as opposed to a complete preference function. Such an approach adopts the general framework of Hausman's approach to derive the incomplete preference relationships and provide expressions for the welfare effects of variations in non-market goods.

Incomplete Demand Systems

Specification of a system of demand equations naturally leads to the implications of consumer choice theory for assessing the structure imposed. As LaFrance has pointed out, three practical approaches can be considered for the demand system specification. First, broad aggregates of all goods available to the consumer can be used to reflect all choices in the consumption set. Second, separability can be imposed so that conditional demand equations involving a subset of commodities can be estimated. Third, an incomplete system of demand equations can be specified. Obviously, the first approach is unsatisfactory because interest is focused on individual commodities. The second approach suffers from i) uncertainty as to the true nature of separability, ii) not identifying the overall utility function but only a subutility function, and iii) the interdependence between quantities demanded and group expenditure. This latter condition is exacerbated when many households have zero demands and consequently zero groupwise expenditure. Thus, substantial simultaneous equations bias would likely be encountered.

The incomplete demand system specification is an attractive alternative only if the preference structure it identifies is consistent with rational models of consumer behavior. Incomplete demand models can be related to an underlying utility maximization subject to a linear budget constraint and can be used to conduct proper welfare analysis (LaFrance and Hanemann, 1989). The incomplete demand structures that are consistent with such maximizing behavior were first catalogued in LaFrance and Hanemann (1984) for some common functional forms of demand equations. In the case of linear expected demands, the restrictions required for integrability are zero (or essentially zero) income effects and a symmetric negative definite cross price matrix. Burt and Brewer as well as Seller et al. imposed cross equation symmetry of the price coefficients. Hence both studies imposed restrictions generally consistent with those suggested by a linear incomplete demand system. However, because both studies modeled discrete household demand data with linear models, their welfare calculations were compromised by their assumption that demands were continuously distributed.

As mentioned, demand models which are based on an optimization hypothesis and which are applied to a subset of goods typically assume preferences are separable—thus allowing the analysis to focus on demand models for the goods of interest apart from other goods. The budget allocated to this group of separable goods is assumed known and the system yields only partial welfare measures. This can be contrasted with the key assumption of an incomplete demand system: prices outside the set of goods of interest do not vary. If this maintained hypothesis is reasonable, then unconditional welfare measures can be computed from a properly specified incomplete demand system. Given that prices of other goods are constant, the utility maximization problem under a linear budget constraint yields a system of incomplete demands which satisfy Slutsky symmetry and provide exact welfare measures for price changes of the goods of interest.

The functional form assumed for modeling the relationship between expected demands and conditioning variables will dictate the restrictions necessary to assure that the incomplete demand system satisfies proper integrability conditions. Fortunately, LaFrance and Hanemann(1984) have considered a number of functional forms and have detailed the restrictions consistent with integrability. In the empirical example which follows, their Log I specification is adopted. Consequently this particular functional form will be used to illustrate the incomplete demand system approach.

Assume that site-specific expected demands for j=1, 2, ..., J sites take the form

$$E(y_j) = \alpha_j \exp\left(\sum_{k=1}^{J} \beta_{jk} p_k + \theta_j I\right) = \gamma_j \tag{1}$$

where p_k represents the price of the k^{th} (k=1, 2, ..., J) site, I denotes household income and the observational index has been suppressed. One set of restrictions consistent with an incomplete demand system of this form is (LaFrance and Hanemann, 1984): $\alpha_j > 0$ and $\beta_{jj} < 0\ \forall\, j$, $\beta_{jk} = 0\ \forall\, j \neq k$, and $\theta_j = \theta\ \ \forall\, j$. These restrictions result in this Log I incomplete demand system having J free own-price parameters and one income coefficient. Therefore there are (1 + ½ J)*(J-1) price and income parameter restrictions implied by this functional form if it is to be consistent with the optimizing behavior underlying the incomplete demands. Although the restrictions imposed on this incomplete demand system appear severe, the requirement of zero Marshallian cross price effects is largely unavoidable when adopting a model of expected demand that yields non-negative predicted demands. In contrast, linear specification of expected demand with symmetric cross price coefficients and no income effects would result in a properly specified incomplete system—but at the cost of ignoring the discrete nature of the observed demand data and possibly predicting negative expected demand. Clearly this is a trade off that the analyst needs to consider.

Individual-specific factors can enter the incomplete demand model and still satisfy the integrability restriction that $\alpha_j > 0$ by recognizing that we can specify $\alpha_j = \exp(a_j)$ where a_j is itself a function of conditioning variables which may correspond to an individual or household. Note that the Log I specification may be restricted to reproduce the basic form of the standard conditional multinomial logit model which does not admit different own price coefficients, income, or other individual-specific shifters. This is easily accomplished by requiring that: $\beta_{jj} = \beta$ and $\theta_j = 0 \ \forall j$. These additional restrictions result in the model

$$E(y_j) = \alpha_j \exp(\beta p_j) = \gamma_j \tag{2}$$

These restrictions imply a quasi-indirect utility function and expenditure function associated with this demand system which are (LaFrance and Hanemann,1984) respectively

$$v(p, I) = I - \beta^{-1} \sum_{j=1}^{J} \alpha_j \exp(\beta p_j) \tag{3}$$

$$e(p, u) = u + \beta^{-1} \sum_{j=1}^{J} \alpha_j \exp(\beta p_j) \tag{4}$$

Now these expressions can be used to estimate the welfare effects of changes in prices and, under certain circumstances, changes in environmental goods. Of course this leads to consideration of the comparison of these welfare measures to those obtained from the logit model. To illustrate this, assume some or all of the α_j include an environmental amenity which when increased yields a new level $\alpha_j^* \geq \alpha_j$. The change in consumer's surplus under the incomplete demand specification is

$$S_1 = \beta^{-1}\left(\sum_{j=1}^{J} \alpha_j \exp(\beta p_j) - \sum_{j=1}^{J} \alpha_j^* \exp(\beta p_j)\right) = \beta^{-1}\left(\sum_{j=1}^{J} E(y_j) - E(y_j^*)\right) = \beta^{-1}\left(\sum \gamma_j - \sum \gamma_j^*\right)$$

The logit model may be parameterized so that

$$E(\pi_j) = \frac{\alpha_j \exp(\beta p_j)}{\sum_{j=1}^{J} \alpha_j \exp(\beta p_j)} = \frac{\gamma_j}{\sum_{j=1}^{J} \gamma_j} \tag{6}$$

with the $E(\pi_j^*)$ defined analogously. This formulation leads to the per-trip surplus measure

$$S_t = \beta^{-1}\left(\ln \sum_{j=1}^{J} \alpha_j \exp(\beta p_j) - \ln \sum_{j=1}^{J} \alpha_j^* \exp(\beta p_j)\right) = \beta^{-1}\left(\ln \sum \gamma_j - \ln \sum \gamma_j^*\right)$$

Two choices exist for scaling up the per-trip surplus measure S_t. They are i)multiply S_t by total expected trips before the amenity change or ii)multiply by total trips after the amenity change. Define these measures as $S_0 = S_t \sum_{j=1}^{J} E(y_j) = \sum \gamma_j$ and $S_2 = S_t \sum_{j=1}^{J} E(y_j^*)) = \sum \gamma_j^*$. Note that the γ_j have been scaled such that the expected value of their sum equals the sum of the y_j.

Proposition: Given that $\gamma_j > 0$ and $\gamma_j^* \geq \gamma_j \ \ \forall j$ then $S_0 \leq S_1 \leq S_2$

Proof: $\beta^{-1} \sum \gamma_j (\ln \sum \gamma_j - \ln \sum \gamma_j^*) \leq \beta^{-1} (\sum \gamma_j - \sum \gamma_j^*) \leq \beta^{-1} \sum \gamma_j^* (\ln \sum \gamma_j - \ln \sum \gamma_j^*)$

define $V = \sum \gamma_j^* / \sum \gamma_j \geq 1$ then by multiplying both sides by the positive quantity $-\beta/\Sigma\gamma_j$ yields $\ln V \leq V - 1 \leq V \ln V$ which holds for all $V \geq 1$ (Jeffrey, p.132).

Thus scaling up the per-trip consumers surplus measure from the random utility model by expected demand either before or after the amenity change provides bounds to the surplus measure obtained from a certain restricted incomplete demand system. Of course these results may be applied to the valuation of nonmarket goods only if the welfare effects of amenity changes can be completely recovered from the site specific demands (LaFrance, 1994). This notion if further developed by Ebert who shows that if the marginal willingness to pay functions for the environmental goods can be inferred from the specification of the incomplete demand system then unambiguous welfare measures can be determined for these environmental goods.

Econometric Approach

Let y_{nj} denote the number of trips from the n^{th} (n=1, 2, …, N) origin to the j_{th} (j=1, 2, …, J) individual site. Let $Y_n = \sum_{j=1}^{J} y_{nj}$ denote aggregate trips to the wilderness area from the n^{th} origin. Now suppressing the origin index, if the $y_1, y_2, \ldots, y_J$ are independently distributed as

Poisson: $y_j \sim Po(\mu_j)$, then:

i) Y is distributed $Po(\mu = \Sigma\mu_j)$

ii) $P(Y_1 = y_1, Y_2 = y2, \ldots, Y_J = y_J | Y) = \prod_{j=1}^{J} \mu_j^{y_j} e^{-\mu_j} (y_j!)^{-1} \Big/ \mu^Y e^{-\mu} (Y!)^{-1}$

$$= \frac{Y!}{y_1! y_2! \ldots y_J!} \left(\frac{\mu_1}{\mu}\right)^{y_1} \left(\frac{\mu_2}{\mu}\right)^{y_2} \ldots \left(\frac{\mu_J}{\mu}\right)^{y_J}$$

$$= \frac{Y!}{y_1! y_2! \ldots y_J!} \pi_1^{y_1} \pi_2^{y_2} \ldots \pi_J^{y_J} = Mn(\pi_1, \pi_2, \ldots, \pi_J | Y)$$

where $Mn(\bullet|Y)$ denotes the multinomial distribution.

iii) Conversely, the independent, non-negative, integer valued variables $y_1, y_2, \ldots, y_J$ have Poisson distributions if and only if the conditional distribution of these variables for the fixed sum $\sum_{j=1}^{J} y_j = Y$ is a multinomial distribution (Johnson et al. p.65).

It is obvious that the unconditional distribution is

$Mn(\pi_1, \pi_2, \ldots, \pi_J | Y) \bullet P(Y) = P(Y_1 = y_1, Y_2 = y2, \ldots, Y_J = y_J | Y) \bullet P(Y)$

$$= \prod_{j=1}^{J} \mu_j^{y_j} e^{-\mu_j} (y_j!)^{-1} = \prod_{j=1}^{J} Po(\mu_j)$$

This result was suggested by Terza and Wilson. Yet they and others have failed to recognize that if P(Y) is not specified to be $Po(\mu)$ as in (i) above, then there can be no claim that the conditional distribution is indeed multinomial. For example if the distribution of Y is $Nb(\mu,\theta)$, i.e. negative binomial, such that $V(Y)=\mu(1+\mu\theta)$ and the joint conditional distribution for the $y_1, y_2, \ldots, y_J$ is taken to be $Mn(\pi_1, \pi_2, \ldots, \pi_J | Y)$, then the unconditional results that $E(y_j) = \mu_j$, $V(y_j) = \mu_j(1+\mu_j\theta)$, and $Cov(y_i, y_j) = \theta\mu_i\mu_j$ can be obtained. This will be termed the multinomial-negative binomial model (Mn-Nb). However, we have shown that the marginal distributions of the counts should be Poisson distributed under the multinomial model. Yet if the sum of the y_j, Y, is specified to be $Nb(\mu,\theta)$ then the marginal distributions of its components, the y_j, are consequently Nb.

To derive a conditional distribution of the y_j consider that they are independently distributed with probability generating function

$\text{pgf}_j = (1+\rho-\rho t)^{-\gamma_j}$ then $\sum y_j$ has $\text{pgf} = (1+\rho-\rho t)^{\Sigma\gamma_j}$. The marginal probability mass function is $P(Y_j = y_j) = \dfrac{\Gamma(\gamma_j + y_j)}{\Gamma(\gamma_j)\Gamma(y_j+1)} q^{y_j}(1-q)^{\gamma_j}$ where $q = \rho/(1+\rho)$.

Thus $y_j \sim Nb(\gamma_j, \rho)$ and $E(Y_j) = \gamma_j\rho$ and $V(Y_j) = \gamma_j\rho(1+\rho)$. The joint conditional distribution $P(Y_1 = y_1, Y_2 = y2, \ldots, Y_J = y_J | Y)$ is

$$\prod_{j=1}^{J} \frac{\Gamma(\gamma_j + y_j)}{\Gamma(\gamma_j)\Gamma(y_j+1)} q^{y_j}(1-q)^{\gamma_j} \Bigg/ \frac{\Gamma(\Sigma\gamma_j + \Sigma y_j)}{\Gamma(\Sigma\gamma_j)\Gamma(\Sigma y_j + 1)} q^{\Sigma y_j}(1-q)^{\Sigma\gamma_j}$$ or simply

$\dfrac{Y!\Gamma(\Sigma\gamma_j)}{\Gamma(Y+\Sigma\gamma_j)} \prod_{j=1}^{J} \dfrac{\Gamma(\gamma_j + y_j)}{\Gamma(\gamma_j)\Gamma(y_j+1)}$. Termed the compound multinomial (Mosimann) or the fixed effects negative binomial (Hausman, Hall, and Griliches) or multinomial Dirichlet (MnD). Mosimann derives this distribution by assuming the multinomial probabilities $Mn(\pi_1, \pi_2, \ldots, \pi_J | Y)$ have Dirichlet distribution and notes that

$$E(\pi_j) = \gamma_j/\Sigma\gamma_j \text{ and } Cov(\pi_i \pi_j) = \gamma_i\gamma_j\left(\frac{1}{\Sigma\gamma_j + (\Sigma\gamma_j)^2} - \frac{1}{(\Sigma\gamma_j)^2}\right) < 0.$$

Woodland has recognized the ability of the Dirichlet distribution to limit shares to the unit simplex and gives several compelling arguments why the shares would likely be negatively correlated. Morey et al. have extended this discussion to the case where shares lie on the boundaries of the unit simplex and correctly noted that the Dirichlet cannot be applied to data where zero shares are observed. Although Morey et al. concluded that no multivariate density functions exist which have positive density over the entire unit simplex, boundaries included, and which are restricted to the unit simplex, the multivariate multinomial Dirichlet may properly be used in the boundary case problem because the multinomial parameters do not have a degenerate distribution in this situation.

The multinomial Dirichlet, MnD(γ_1, γ_2, ...γ_J|Y), is a conditional distribution. Consider the unconditional distribution that results when Y ~ Nb($\Sigma\gamma_j$, θ)

$$\frac{\Gamma(Y+\theta^{-1})\Gamma(\Sigma\gamma_j)}{\Gamma(\theta^{-1})\Gamma(Y+\Sigma\gamma_j)}\left(\frac{\theta^{-1}}{\Sigma\gamma_j+\theta^{-1}}\right)^{\theta^{-1}}\left(\frac{\Sigma\gamma_j}{\Sigma\gamma_j+\theta^{-1}}\right)^{Y}\prod_{j=1}^{J}\frac{\Gamma(\gamma_j+y_j)}{\Gamma(\gamma_j)\Gamma(y_j+1)}.$$

Finally for additional flexibility, consider modeling the E(Y) = $\delta\Sigma\gamma_j = \mu$, that is the hyperparameters are scaled by δ. This gives the multinomial Dirichlet-negative binomial, MnD-Nb, distribution

$$\frac{\Gamma(Y+\theta^{-1})\Gamma(\Sigma\gamma_j)}{\Gamma(\theta^{-1})\Gamma(Y+\Sigma\gamma_j)}\left(\frac{\theta^{-1}}{\mu+\theta^{-1}}\right)^{\theta^{-1}}\left(\frac{\mu}{\mu+\theta^{-1}}\right)^{Y}\prod_{j=1}^{J}\frac{\Gamma(\gamma_j+y_j)}{\Gamma(\gamma_j)\Gamma(y_j+1)}.$$

This distribution of both the allocation of trips and the sum of the trips across alternatives can be compared to the aforementioned multinomial-negative binomial model which has a scaled from as well. As seen in Table 1., the MnD-Nb has additional flexibility to model the variance within and covariance between equations due to the fact that the scale parameter enters these equations in a more complicated fashion. Note that for certain parametric combinations, the

MnD-Nb can allow for negative covariances across equations whereas the Mn-Nb restricts these to be everywhere positive.

Table 1. Some Moments of the MnD-Nb and the Mn-Nb with scale parameter

	MnD-Nb	Mn-Nb
$E(Y_j)$	$\delta\gamma_j$	$\delta\mu_j$
$V(Y_j)$	$\delta\gamma_j[1 + \delta(1+\theta)\omega(1+\gamma_j) - \delta\gamma_j]$	$\delta\mu_j[1 + \theta\delta\mu_j]$
$Cov(Y_iY_j)$	$\delta^2\gamma_i\gamma_j[(1+\theta)\omega - 1]$	$\theta\delta^2\mu_i\mu_j$
where $\omega = \Sigma\gamma_j/(1 + \Sigma\gamma_j)$		

DATA

The study area is the Hoover Wilderness area. The Hoover Wilderness area is located on the east side of the Sierra Nevada Mountain range, close to the California-Nevada state borders. The primary wilderness recreation taking place in Hoover is backcountry hiking. One of the requirements for backcountry hiking is that a backcountry hiking permit be filled out. This analysis is based on permits for 1990, 1991, and 1992.

A total of 7,661 complete permits were submitted during these three years. Of these, 7,136 were for backcountry hiking, the activity under study here. The permits included the entry point of the hiking party and the originating zip code of the party. Using these pieces of information travel distances were calculated using both computer programs and US Forest Service maps. A total of 598 residential zip code origins in Nevada and California were used in this analysis in order to more reasonably infer that the main purpose of the trip to the wilderness area was for recreation there. This resulted in a sample of 5113 permitted trips to the 14 trails.

Trail characteristics were developed from US Forest Service geographic information system information (GIS) and US Forest Service and US Geological Survey maps. The maps primarily provided information about campgrounds in the area of the trailhead, grazing allotments and trail elevation. Vegetative characteristics were obtained from the timber inventory GIS. The ecosystems found in the Hoover Wilderness include Ponderosa/Jeffrey pine, mixed pine, riparian/meadow, and rocky alpine areas. These data were merged together by digitizing the trail maps and then laying the trail map layer onto the vegetative characteristics GIS layers. This allowed us to accurately calculate the number of acres of each ecosystem that were on each trail. Grazing allotments were then added to the data base by using a US Forest Service grazing allotment map in conjunction with historical grazing figures.

Since the analysis is based on permit data there is no individual travel cost information. (Hellerstein has discussed the rationale for using aggregate trip data.) Following Englin and Mendelsohn (1991) who also worked with permit data like these, travel costs were calculated at $0.25 per mile. While this is arbitrary, the welfare estimates can easily be converted using other numbers.

Results

The multinomial Dirichlet negative binomial model was estimated using a maximum likelihood routine programmed in GAUSS. Results are reported in Table 2. Note that the likelihood estimates are from a so-called penalized model. This likelihood includes a term to insure that the estimated aggregate average number of trips to the Hoover Wilderness closely matches that of the observed average (if this is not the case, subsequent welfare calculations will not be able to reflect the average visitation rates of the sample). This factor is necessitated by the consequence that the negative binomial model, while a member of the linear exponential family for fixed and known θ (Gourieroux et al.), will not necessarily reproduce the average count when θ is estimated simultaneously with the conditional mean. In the empirical model, it is seen that this penalty function only slightly decreases the likelihood from the unconstrained specification. Further, the impact of this penalty on the calculated robust (as per White) standard errors is investigated by also obtaining bootstrap standard errors. Table 2 indicates that both sets of standard errors correspond closely. In the one case where they differ substantially (the Mixed Pine variable), the calculated standard error suggests a more liberal confidence interval.

The multinomial Dirichlet negative binomial was compared to the corresponding multinomial negative binomial model with the identical number of parameters. This latter model's log likelihood value was –1351.92 at convergence. The models differ only in the distributional assumption underlying the conditional distribution of site specific trips and thus are non-nested. Vuong's test of the superiority of the multinomial Dirichlet versus the multinomial specification yielded at test statistic of 3.04 which is distributed as standard normal under the null of no difference between the models. Thus we conclude with greater than 99% confidence that the multinomial Dirichlet better represents the data generating process.

Most of the ecosystems are positively valued as are high trails and campgrounds near the trailhead. Both sheep and cattle grazing have a negative impact on the utility of a backcountry hiking trip. Because the unit of observation is the residential zip code, the logarithms of the populations of these zip codes entered the model and were assigned parameters which could vary by destination. The rationale for the inclusion of the populations centered on the idea that more metropolitan origins likely focused their trips on the more well known trails. The coefficients on the Pn(j) variables show a diverse pattern of preferences for trails based on population of the zip code origin and generally support the notion that those from more populated areas have the propensity to visit the better known trails.

A variety of grazing scenarios could be examined using this model. We chose to examine the impacts of grazing bans on a trail by trail basis looking at sheep and cattle both individually and together. The reason for analyzing the impacts on a trail by trail basis is that the impacts of grazing depend in part on what other characteristics are on the trail. It's not only how many animals but where they are grazed. Table 3a provides these results. The first two columns of Table 3a show the current level of grazing by trail. Trails not listed in the table currently do not allow grazing. Cattle grazing is limited to Burt Canyon, Molybdenite Creek and Buckeye Creek. A total of 1354.2 AUMs (animal unit months) per year were allowed in the early 1990's. Sheep are grazed on Burt Canyon (in addition to the cattle), Leavitt Meadows, Poore Lake, Emma Lake, and Tamarack Lake. A total of 4153.5 AUMs per year of sheep have grazed in the wilderness over the last three years. It should be noted that while a cattle AUM is usually about one animal a sheep AUM consists of *five* head of sheep. So the *total number of sheep* in the wilderness could approach 20,000 head depending on the number of days that animals are grazed.

As the third column of Table 3a shows clearly the willingness-to-pay by hikers to remove cattle from the wilderness varies widely by trail. Burt Canyon shows a loss of $7,316 for all hikers visiting the Hoover Wilderness Area. Cattle grazing at Molybdenite Creek, with same number of cattle AUMs, results in losses of over $14,542. The total losses from all cattle grazing is estimated to be about $30,000. Sheep pose a more extreme picture. Leavitt Meadows is currently grazed by 1189 AUMs of sheep each year (the number of animals present at any given time would depend on the number of months sheep are grazed). The total losses from Leavitt Meadows are almost $124,000 per year. The reason for this substantial loss, and probably the large number of sheep, is that Leavitt Meadows contains a 100 acre riparian/meadow. As will be shown below riparian areas are highly valued by hikers. Removing sheep from Leavitt Meadows results is a large increase in the value of Leavitt Meadows to hikers. Comparatively speaking, the other losses are small.

A final observation about the Burt Canyon trail is useful. The cattle and sheep estimates presented above were for removing one kind of grazing but leaving the other. The final column shows the value of removing both kinds of grazing simultaneously. As you can see the value is about $42,000. This is sharply higher than the combined individual cattle and sheep estimates. This result has a straightforward interpretation however. Given that 780 AUMs of sheep are still there, removing the cattle is only worth $10,975. The *marginal* effect of removing cattle alone is small. The same argument applies to sheep. If, however, all grazing is curtailed at this site, then the sum of the two effects dominates the welfare change since now there is a complete absence of grazing on the trail.

In order to attach any policy significance to the welfare measures associated with removing grazing at a subset of the trails in the Hoover Wilderness we require benchmarks against which to compare these values. Table 3 recognizes that removing grazing can generate direct economic losses to permit holders and government agencies. While the loss in agency revenue can be easily calculated, welfare losses of permit holders require special treatment. A recent paper by Lambert and Shonkwiler has estimated the surplus under the derived demand curves associated with grazing permits over the time period analyzed. Their methods implicitly account for the non-fee costs incurred by permit holders since these costs are typically substantial relative to the grazing fee. By comparing values between Tables 3a and 3b, it is seen that at just two sites do welfare losses of hikers much exceed the revenues of the agency and the surpluses of the livestock grazers. And of those two, only grazing at Leavitt Meadows results in statistically significant net welfare losses to hikers (i.e. the 95% confidence interval for hiker welfare losses does not include the estimated losses in agency revenues and producer surplus). This result is a consequence of the ecosystem components that comprise each of the trails.

The model can then be used to the value of the ecosystems. Like the grazing, the value of the ecosystems will depend on what other characteristics are on the trail. For ecosystem valuation, the value of the ecosystem across pertinent trails is calculated rather than the total value of the ecosystem on a given trail. The values are estimated by increasing the quantity of each ecosystem on trails where that ecosystem is present by one acre and calculating the change in aggregate willingness-to-pay. Table 4 shows the results. The surplus/acre measure represents an average (across trails) marginal value of a one acre increase in the ecosystem since as many acres are added as there are trails possessing that ecosystem. These results sharply illustrate the value of riperian or meadowland to back country hikers.

Table 4. Per-Season Surplus for One Acre Increases in Existing Ecosystems

	Jeffrey Pine	Riperian	Mixed Pine	Rocky Alpine
Total Surplus	$75.04	$869.90	$ -7.16	$60.78
Acres Added	3	5	5	12
Surplus/Acre	$25.01	$173.98	$ -1.43	$ 5.06

CONCLUSION

One of the issues facing public land managers is the prioritization of those activities which may simultaneously compete for the same public areas. A pressing issue today is the appropriate level of grazing on public lands, especially those that have alternative uses. This analysis has examined i) the willingness-to-pay by backcountry hikers in the Hoover Wilderness Area to remove grazing from hiking trails and ii) the value of some Sierra ecosystems to back country hikers. The results indicate that damages to hikers varies considerably from trail to trail in the wilderness. The differences are primarily driven by the other characteristics at the trail. High country grazing by either sheep or cattle causes much lower damages than competition in riparian areas. On the Leavitt Meadows trail losses from sheep grazing are estimated to be about $124,000 annually. This is the direct result of the high value that hikers place on the 100 acre Leavitt Meadow. Welfare losses due to sheep grazing in other areas, while certainly constituting statistically significant damages, are at least an order of magnitude smaller. The increase in hiking activity is generally modest except for the change forecasted for Leavitt Meadows.

At least several limitations to this study are important to note. One is that the model cannot identify the intra-seasonal timing and patterns of hiking activities at the various trails which may result from the physical presence of livestock grazing in the Hoover Wilderness. Secondly, because of the nature of the data available, the calculation of the travel cost variable is crude and welfare effects can not be ascribed to the individual or household level. And while Leavitt Meadows does possess a wetland area which is apparently highly valued by backcountry hikers, whether the Forest Service has the ability to shift grazing from this area as per Ex. Ord. No. 11990 is as of this time an unresolved issue. Finally the use of the incomplete demand system to value changes in site attributes is proper as long as the marginal willingness to pay for quality changes can be completely recovered from the incomplete system.

Table 2. Multinomial Dirichlet-Negative Binomial Model. Log likelihood: -1244.98[a]

Variable	Coefficient	Std.Error	"t-value"	Bootstrap Std.Error[b]
Travel cost	-0.0183	0.0016	-11.3865	0.0018
Jeffrey/Pond. Pine(100 ac.)	0.2653	0.0577	4.5953	0.0565
Cattle AUMs $(100)^{1/2}$	-0.8846	0.1766	-5.0083	0.1921
Sheep AUMs $(100)^{1/2}$	-0.5892	0.0756	-7.7886	0.0588
Riperian /Meadow(100 ac.)	2.0737	0.5422	3.8243	0.3559
Mixed Pine (100 ac.)	-0.0152	0.0163	-0.9316	0.0082
Rocky Alpine (100 ac.)	0.0672	0.0150	4.4648	0.0142
Highest Elev. (100 ft.)	0.0047	0.0026	1.7968	0.0025
Campground (yes=1)	0.2910	0.1907	1.5260	0.2192
Log of Scale(δ)	-1.0333	0.1641	-6.2982	0.1543
Variance(θ)	0.6318	0.0439	14.4029	0.0434
Pn1	0.6662	0.1399	4.7620	0.1245
Pn2	0.1020	0.1355	0.7524	0.1291
Pn3	0.1407	0.0964	1.4604	0.0973
Pn4	0.3399	0.0790	4.3007	0.0811
Pn5	-0.8110	0.1136	-7.1419	0.1256
Pn6	0.2998	0.0682	4.3937	0.0705
Pn7	0.2957	0.0697	4.2403	0.0731
Pn8	-0.3215	0.0868	-3.7057	0.0863
Pn9	-0.4241	0.1004	-4.2238	0.0993
Pn10	0.6216	0.0663	9.3749	0.0671
Pn11	0.3856	0.0793	4.8604	0.0842
Pn12	-0.4187	0.1983	-2.1118	0.2191
Pn13	-0.6127	0.1926	-3.1810	0.2203
Pn14	-0.4257	0.1212	-3.5133	0.1202

[a]Penalized estimator. Unpenalized log likelihood: -1243.86
[b]Based on 400 samples

Table 3a. Total Per-Season Welfare Gains for Hikers: MnD-Nb **Incomplete Demand System**

Trail Name	Current Cattle AUMs	Current Sheep AUMs	Remove Cattle	Remove Sheep	Remove Both Cattle and Sheep
Burt Canyon	**545.6**	**780.0**	**$7,316** (3097-2328)[b]	**$4,437** (2423-6847)	**$42,354**[a] (19703-147059)
Molybdenite Cr.	**545.6**		**14,542** (6511-45828)		**14,542**
Buckeye Creek	**263.0**		**8,306** (3999-20339)		**8,306**
Leavitt Meadows		**1189.0**		**123,862** (72658-200274)	**123,862**
Poore Lake		**780.0**		**353** (157-824)	**353**
Emma Lake		**780.0**		**360** (148-701)	**360**
Tamarack Lake		**624.5**		**1,055** (609-1750)	**1,055**
Wilderness Totals	**1354.2**	**4153.5**	**$30,164**[a] (14042-90380)	**$130,067**[a] (77727-209895)	**$190,832**[a] (115053-356910)

[a]Value reflects multiple amenity changes
[b]Bootstrap 95% confidence interval based on 200 samples

Table 3b. Revenues and Surplus Measures Accruing to Non-recreationists Per-Season

Trail Name	Current Cattle AUMs	Current Sheep AUMs	Agency Revenue[a]	Surplus of Producers[b]	Total
Burt Canyon	545.6	780.0	$2,651	$19,957	$ 22,608
Molybdenite Cr.	545.6		1,091	15,277	16,368
Buckeye Creek	263.0		526	7,364	7,890
Leavitt Meadows		1189.0	2,378	7,134	9,512
Poore Lake		780.0	1,560	4,680	6,240
Emma Lake		780.0	1,560	4,680	6,240
Tamarack Lake		624.5	1,249	3,747	4,996
Wilderness Totals	1354.2	4153.5	$11,015	$62,839	$73,854

[a]Computed at $2 per AUM
[b]Computed at $28 per AUM for cattle and $6 per AUM for sheep (Lambert and Shonkwiler)

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Improving the Practice of Benefit Transfer: A Preference Calibration Approach

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I. 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 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 is therefore one of the requirements faced by the U.S. Environmental Protection Agency (EPA). 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, the 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 policy 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

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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).

theory.

A. 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, in order 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 these 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.

B. How Is Benefit Transfer Typically Applied?

Most benefit transfer methods utilize 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 is 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, and 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 et al. [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 as invalid, the implicit conditions suggested that the sites were not close substitutes.

C. 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

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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)

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 make 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 benefits 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]

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

[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.

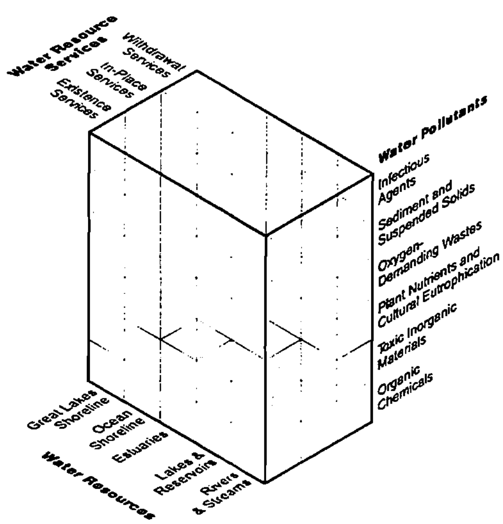
level from which the improvement occurs (i.e., each additional unit of improvement may contribute less and to individual well-being than the previous unit). Nor do they specify how the increase in well-being associated with one use of the improved resource (e.g., recreational use of a lake) may depend 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 illustrates the problems that are 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 and by asking to isolate which sources—point, non-point, storm water overflows, or municipal treatment facilities were responsible for the reductions. Finally, the cube 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

[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, etc.) or a watershed may well imply improvements in "downstream" resources as well as changes over time.

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").

Figure 1. Illustration of the Potential Scope of CWA Benefits



Often, as noted earlier, the focus of a primary research study that is a candidate for use in a benefit transfer will be on one subset or component of the cube described in Figure 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.

D. What Is an Alternative Approach to Benefits Transfer?

This paper 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:

$$U = U(C_1, C_2; Q, \alpha) \tag{1}$$

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

$$V = V(m, P, Q; \alpha) \tag{2}$$

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

WTP for a change in environmental quality from Q_0 to Q_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.

$$V(m, P, Q_0; \alpha) = V(m - WTP, Q_1; \alpha) \tag{3}$$

Assumptions about the functional form of utility should then allow the analyst to express WTP as a function of the change in environmental quality $(Q_1 - Q_0)$, income, prices, and α.

$$WTP = f(Q_1, Q_0)\, m, P; \alpha) \tag{4}$$

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.

E. 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 they are designed to take explicit account of the assumptions regarding individuals' preferences and the constraints that they face. As such it also 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 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, such 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

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

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

We begin, in section 2, 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 are designed to demonstrate how information from a hedonic study, a travel cost study and a contingent valuation 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."

II. 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 involved price or quantity changes for marketed commodities, components of market exchanges were observed and the primary focus of policy analysis was 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 there are direct uses for the affected environmental resources, 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 there has been considerable effort 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 be making 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 use of the available estimates in 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 B illustrates how one can improve the consistency of transfers using as a case study for fishing benefits in the Willamette Basin. This is compared with the logic implied by the Hicks-Harberger approximation.

The third 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, then it may be necessary to develop transfers that incorporate these restrictions. As proposed in section I, the most direct way to meet this objective is to use available information to calibrate an indirect utility function.

Section III 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.

A. 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. (5).

$$CS_1 = \frac{1}{2}\left(P_0 + P_1\right)\left(q_1 - q_0\right) \tag{5}$$

As Diewert [1992] recently explained, first order approximations to compensating (willingness to pay) and equivalent variation (willingness to accept) measures of the consumer surplus changes can also be expressed in similar terms. Eq. (6) provides the compensating variation (CS_2) and (7) the equivalent variation (CS_3) approximations.

$$CS_2 = P_1\left(q_1 - q_0\right) \tag{6}$$

$$CS_3 = P_0\left(q_1 - q_0\right) \tag{7}$$

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.[8] 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 non-market 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 quantity changes for the uses linked to an environmental resource that are permitted because of the policy change for the typical user
2. estimate the number of typical users before and after the policy change
3. transfer a per "unit" consumer surplus measure, with the unit measure comparable to the index used in step (1)
4. Combine estimates in steps (1) through (3) for each year considered in the analysis and compute the discounted aggregate benefit measures.

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 willingness to pay measure. Eq. (8) translates the steps to an equation.

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

$$CS_P = \frac{CS_T}{\Delta d_T}\left(d_1 \bullet N_1 - d_0 \bullet N_0\right) \tag{8}$$

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

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

The connection between Eq. (8) and (6) 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.[9] 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 theoretically consistent would be the marginal value of the quality change provided that change is measured in the same

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[9]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 1 below 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, etc.)

effective units as d.[10] With this amendment Eq. (8) 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. It is to the steps required to evaluate this issue that we now turn with an example.

[10]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.

Table 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 et al. (1991)
			$86.50 per trip	Oregon Rowe et al. (1985)
Trout (days)	100,218	1,002,182	$31.80 per day	Oregon and Washington McCollum et al. (1990)
			$21.38 per day	Oregon Brown and Hay (1987)
Warmwater (days)	24,207	242,069	$30.47 per day	U.S. Walsh, Johnson, and McKean (1992)
			$16.22 per day	U.S. Bergstrom and Cordell (1991)
Direct Water Contact Recreation				
Swimming	0	1,001,859	$19-$30 per day	Not given

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

B. 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 improvements attributed to the Clean Water Act (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, etc.) In this use the pre-CWA use was assumed to be zero due to bans on swimming prior to 1972. Table 1 summarizes a few of the selected estimates reported in that earlier analysis. Figure 2 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.

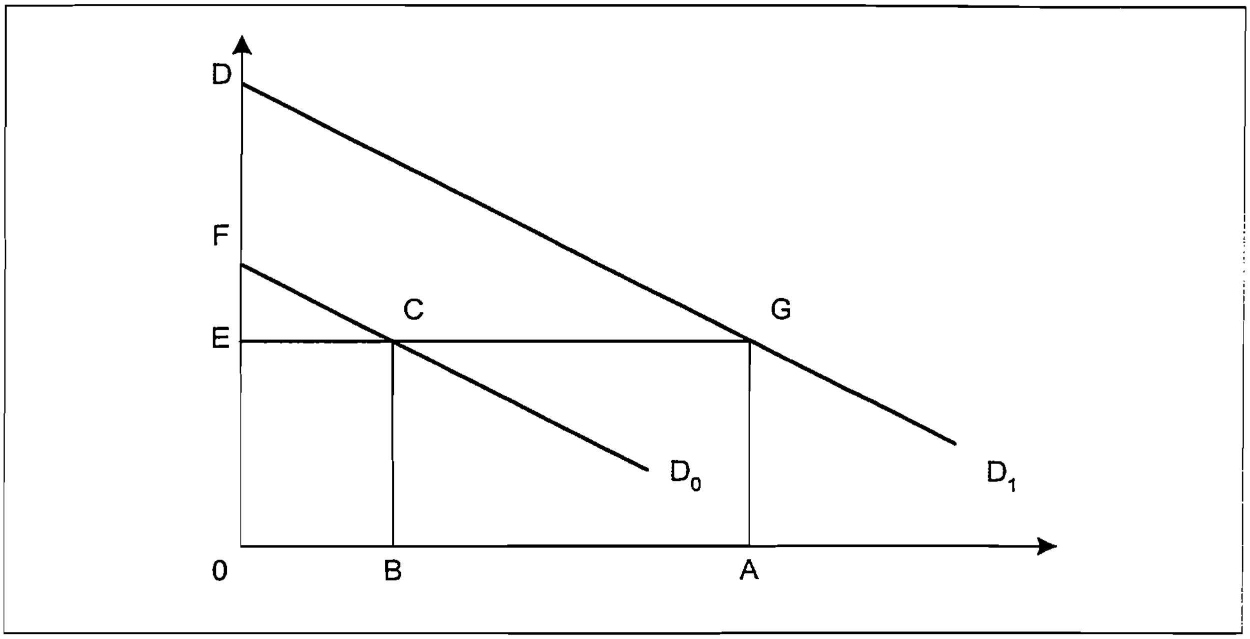
The benefit transfer logic interprets estimates of DEG/OA, consumer surplus per trip for the desired fishing experience, as the equivalent of a marginal value (or virtual price.) The benefit measure given in Table 1 is then:

$$\left(\frac{\mathrm{DEG}}{\mathrm{OA}}\right) \bullet \mathrm{BA} \tag{9}$$

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 I suggested, based on the Kirschhoff, Colby, and LaFrance 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. (9). We can use geometry and the results from Table 1 to illustrate the extent of the mistake.

Figure 2. Illustration of Logic of Willamette Analysis



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. assume 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 $OB = \alpha \bullet OA$.

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:

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

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

Simplifying the expression for DFCG, we have Eq. (10) expressing the desired benefit measure:

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

The expression given in Eq. (9) for the usual benefit transfer method can be expressed in terms of DE, OA, and α as:

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

This geometry implies we have a relationship between the "correct" benefit measure and the simple approximation. Taking the ratio of Eq. (10) to (11) we see that the correct measure is $(1 + \alpha)$ 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. (9) is correct because the quantity measure is assumed to be zero with the pre-CWA water quality conditions.[11]

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 quality improvement causes a parallel shift in the demand function. If we hypothesize that quality reduces the effective price (a 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

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

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

and the desired measure is DEG.

to non-market valuation. It can be readily generalized to contingent valuation or hedonic models.

C. 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.[12]

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

[12]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.

any role "ability to pay" has in limiting monetary measures of the value of quality change.[13]

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. Equation (12) uses this augmentation form in describing the direct utility function for a representative individual. This Cobb-Douglas specification assumes the individuals well-being is related to recreation (C_1) and all other goods (C_2) as in Eq. (12).

$$U = (A(W)C_1)^{\alpha} C_2^{1-\alpha} \tag{12}$$

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 and the analysis of the Willamette River is somewhat different from what is implied by the utility specification in Eq. (12). 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

[13]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.

expenditure function that correspond to constrained utility maximizing behavior. Equation (13) describes the indirect utility function and (14) the expenditure function.

$$V = \left(\frac{P}{A(W)}\right)^{-\alpha} m \cdot a \tag{13}$$

where P = the relative price of C_1 to C_2, with the later normalized to unity

m = income

a = constant scaling factor

$$m = \left(\frac{P}{A(W)}\right)^{\alpha} \frac{V}{a} \tag{14}$$

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 equation (15):

$$\frac{\partial m}{\partial P} = C_1^* = \left(\frac{\alpha}{a}\right) P^{\alpha - 1} \bullet (A(W))^{-\alpha} \bullet V \tag{15}$$

In logarithmic form this suggests Hicksian demand for C_1 shifts in as W increases. This is seen in equation (16)[14]

$$\ln C_1^* = \ln\left(\frac{\alpha \bullet V}{a}\right) + (\alpha - 1)\ln P - \alpha \bullet \ln(A(W)) \tag{16}$$

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

Although this seems to contradict Figure 2, to interpret what it means consider the behavior described by the compensated demands associated with this utility function: amount of effective services of C_1 the consumer realizes is larger for each P. Re-arrange equation (15):

$$\begin{aligned} \text{at } W_0\text{:}\quad A(W_0)^{\alpha} \bullet C_1 &= \left(\frac{\alpha}{a}\right) P^{\alpha-1} \bullet V \\ \text{at } W_1\text{:}\quad A(W_1)^{\alpha} \bullet C_1 &= \left(\frac{\alpha}{a}\right) P^{\alpha-1} \bullet V \end{aligned} \tag{17}$$

If $A(W_1) > A(W_0)$ and nothing else changes, then we see that to maintain a the 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 to Eq. (13) we derive the Marshallian demand in Eq. (18) and the Marshallian consumer surplus for trips to a recreation site with W* water quality in Eq. (19).[15]

[15]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.

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.

$$C_1 = \alpha \bullet \frac{m}{P} \tag{18}$$

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 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 there are no other differences in recreation sites but the water quality. 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 as 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 often analysts 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 often the concept is approximated by describing how a set of consumers' choices would change with a quality change.

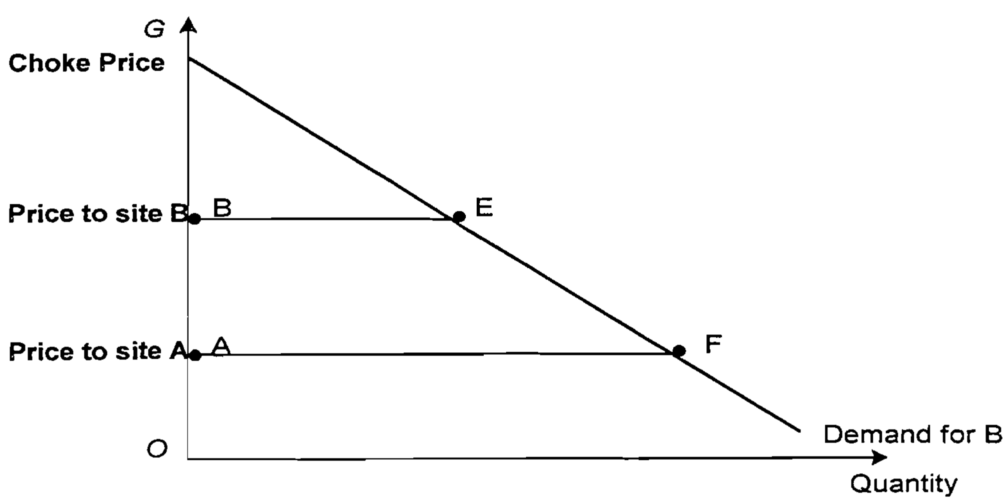
Suppose we have two lakes; A has water quality level sufficient to allow boating but not swimming, B has a water quality level that permits swimming (and therefore also with boating). In the absence of other differences (such as congestion at each site) a recreationist who wants to do both would likely use site B and not A. When we observe them using site A, it is usually the case that 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. This means 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 and thus the gain is measured as a price change along B's demand function, as in Figure 3.

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 they are perceived by users 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 chance OB to OA as in equation (19).

Figure 3. Quality Treated as a Price Change



$$MCS = \int_{P_{OA}}^{P_{OB}} \frac{\alpha m}{P} dP = \alpha m \left[\ln P_{OB} - \ln P_{OA}\right] \tag{19}$$

At this stage the logic implicit in equations (13) or (14) 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 serves to reduce the effective price, because P/A(W) enters the indirect utility function (13). This reasoning suggests 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:

$$\frac{P_{OB}}{P_{OA}} = \frac{\overline{P}/A(W_B)}{\overline{P}/A(W_S)} \tag{20}$$

or

$$\ln P_{OB} - \ln P_{OA} = \ln(A(W_S)) - \ln(A(W_B)) \tag{21}$$

Substituting into Eq. (19), we get Eq. (22)—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. This can be derived by comparing (22) with the Hicksian measure for the same quality change.

$$MCS = \alpha \bullet m\left[\ln(A(W_S)) - \ln(A(W_B))\right] \tag{22}$$

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:

$$MCS \approx \alpha \bullet m \left[\frac{A(W_S) - A(W_B)}{A(W_B)}\right] \tag{23}$$

The Hicksian measure of the willingness to pay for a quality improvement from W_B to W_S is given in Eq. (24).

$$HCS = \left(\left(\frac{P}{A(W_B)}\right)^{\alpha} - \left(\frac{P}{A(W_S)}\right)^{\alpha}\right) \frac{V}{a} \tag{24}$$

Expressing HCS per unit demanded (with the Hicksian demand function, C_1^H) we have Eq. (25) for Hicksian surplus per trip.

$$\frac{HCS}{C_1^H} = \frac{P}{\alpha}\left[\frac{(A(W_S))^{\alpha} - (A(W_B))^{\alpha}}{(A(W_B))^{\alpha}}\right] \tag{25}$$

Comparing equation (25) with (23) 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.[16]

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.[17]

The MCS approximation per unit of C_1 is given in equation (26).

$$\frac{MCS}{C_1} = P\left[\frac{A(W_S) - A(W_B)}{A(W_B)}\right] \tag{26}$$

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. α is the fraction of the consumer's budget spent on C_1 (recreation). Table 2 provides a few examples for one potential change in water quality where there is a 20% 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.

[16]Characteristics of the Willig condition (identified in footnote 5) 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 attributed to the quality per unit C_1 quality increase changes with income, i.e., how does (MCS/C_1) change with m.

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

Table 2. Illustration of Approximation Errors Due to Simple Transfer

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

D. 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 B 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 for 8-9 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.

III. 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, it 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 willingness to pay 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 II 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 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 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 willingness to pay 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 willingness to pay 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 was 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 that need to be imposed on the model needed to specify an indirect utility function and to use it to infer the value of a policy alternative.

To illustrate the process of what we have labeled calibrated benefit transfers, two numerical examples are developed 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's [1984] estimates of the value of water quality changes along with estimates in a recreation demand model by Englin et al. [1997].[18] The second uses a study by Michael et al. [1997] evaluating the effect of water quality on lake front 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 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.

A. 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 II suggested, the selection of a specification for water quality in consumer preference has important implications. So while would like to keep

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[18]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).

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 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 Equation (27) below

$$V = \left[\left(P - h\left(d_1\right)\right)^{-\alpha} m\right]^b \tag{27}$$

Because our example combines a recreation travel cost demand-based measure with the early Mitchell-Carson [1989] contingent valuation 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 there are several possible studies that 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.[19] 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 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 willingness to pay.

To calibrate the preferences defined by Equation (27) 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 Equation (28) using (27):

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

$$C_1 = -\frac{V_P}{V_m} = \frac{\alpha m}{(P - h(d))} \tag{28}$$

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.[20] This is given in Equation (29):

$$MCS = \alpha m \int_{P_0}^{P_c} \frac{1}{(P - h(d))} dP = \alpha m \ln(P - h(d)) \Big]_{P_0}^{P_c} \tag{29}$$

When we evaluate the integral, the result is Equation (30)

$$MCS = \alpha m \left[\ln\left(P_C - h(d)\right) - \ln\left(P_0 - h(d)\right)\right] \tag{30}$$

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 Equation (31) below:

$$\frac{\partial MCS}{\partial d} = \alpha m \left[-\frac{h'(d)}{\left(P_C - h(d)\right)} + \frac{h'(d)}{\left(P_0 - h(d)\right)}\right] \tag{31}$$

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

[20]Setting $C_1 = 0$ in (28) and solving for P does not yield a finite choke price because C_1 approaches zero as P assumes arbitrarily large values. For current purposes we assume there is some large finite choke price.

choke price times (-h'(d)) and the second is the demand at P_0 multiplied times h'(d). The definition of the choke price (even if it cannot be expressed in closed form) implies the first of the terms on the right side of Equation (31) 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 trips is exactly h'(d) as in (32)

$$\frac{\dfrac{\dfrac{\partial MCS}{\partial d}}{\alpha m}}{(P_0 - h(d))} = \frac{\dfrac{\partial MCS}{\partial d}}{X_1} = h'(d) \tag{32}$$

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 (32). 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., $\beta d^{\beta-1} \approx \beta\, [1 \div (\beta-1) \log (d)]$), we can express equation (32) as a quadratic as in equation (33) and solve for the roots:

$$\log(d) \bullet \beta^2 + (1 - \log(d))\beta - \hat{a} = O \tag{33}$$

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

Each of the roots is a potential solution. We discriminate between the two roots for B based on their economic properties. This task is completed by solving for α from the expression for the WTP in Eq. (34) below, using each of the roots derived from (33) and then evaluating the predicted demand and the estimates of α. The later 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. 27) is then given in Equation (34):

$$WTP = m - \left(\frac{P - h(d_F)}{P - h(d_B)} \right)^{\alpha} m \tag{34}$$

Eq. (34) 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 (33) 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 Equation (34) for α. This result is given in (35):

$$\hat{\alpha} = \frac{\ln\left(\dfrac{m-\hat{WTP}}{m}\right)}{\ln\left(\dfrac{P-\hat{h}(d_F)}{P-\hat{h}(d_B)}\right)} \tag{35}$$

Computations using Eq. (33) and (35) identify a sufficient number of the parameters for the indirect utility function in equation (27).[21] The calculations for (35) use Mitchell and Carson estimates for improving water quality from boatable to fishable conditions— \$163 (in 1983 dollars) and \$249.41 (in 1995 dollars) and income was \$32,659 (in 1995 dollars).[22]

This process yields two parameter estimates (one for each of the roots of Equation (33) as given in Table 3:

[21]While we cannot recover an estimate of b in Equation (27) with this information, this parameter did not enter the WTP function (i.e., equation (34)) and therefore an inability to isolate it with this information does not preclude our calculation of WTP or demand for new sites.

[22]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.

Table 3. 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 trips cost was $100 (including the time costs of travel).

The selection of an economically plausible root for (33) is clearest using $\hat{\alpha}$ and $\hat{X}_1$. Negative predicted trips are clearly implausable, as is a large (in absolute magnitude) value for are $\hat{\alpha}$. 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. This 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 based 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 4 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 Equation (32)) 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 times 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 4.

Table 4. 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 sound trips cost was $100 (including the time costs of travel).

B. 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 sets of information available. The first of these is a measure of the willingness to pay for a policy or plan to improve surface water quality. Once again we use Mitchell-Carson [1984]. However for this application the physical measure of water quality used to characterize their results will be different, the size of the change in water quality will differ as well. In addition, we assume that an estimate of the marginal willingness to pay for housing attribute is available using information in Michael et al. [1997].

In considering use of information from a hedonic property value model the approach must be different 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 willingness to pay evaluated at a point. The ability to estimate this marginal willingness to pay at this point does not necessarily imply it is possible to identify the full marginal willingness to pay 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.[23]

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).[24] 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.

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[23]Feenstra [1995] is a notable alternative case. In his case, however, a specific form of preference heterogeneity is assumed in order to allow the demand behavior to be represented by the utility of a representative consumer.

[24]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 form of the CES function used by Quigley is itself specialized. In the case of several attributes it is given in Eq. (36). Note that in this case we have assumed that all other prices are constant across individuals. We also maintain that the housing choice is the only way to 'select' a water quality level.

$$V = \sum_{i=1}^{K} (\theta_i \cdot A_i)^b + (m - P(A_1,\ldots\ldots A_K))^b \tag{36}$$

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
θ_i, b	=	parameters

The first order condition with respect to A_1 yields Eq. (37):

$$\frac{\partial V}{\partial A_1} = 0 = b \cdot (\theta_1 \cdot A_1)^{b-1} - \left(\frac{\partial P}{\partial A_1}\right) \cdot b \cdot (m - P(A_1,\ldots\ldots A_K))^{b-1} \tag{37}$$

With re-arrangement of (37) we can show 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]; Huang and Smith [1995]) has proposed using simply the marginal value for benefit transfers. This practice is possible because the use of the $\partial P/\partial A_1 . \Delta A_1$ 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. (38) below indicates the slope of the hedonic price function offers a point estimate of a composite of the parameters in the indirect utility function.

$$\frac{\partial P}{\partial A_1} = \theta_1^b \cdot \left(\frac{A_1}{m - P(\cdot)}\right)^{b-1} \tag{38}$$

Recognizing the role of θ_1 and b in Eq. (38) is the first step in recovering the parameters necessary to consistently transfer the value nonmarginal changes in water quality,[25] measured as changes in A_1 in this hedonic price model.

The second estimate assumed to be available from a contingent valuation study is a measure of willingness to pay for improving water quality, as described in the Mitchell-Carson contingent valuation study. We summarized the key elements in their question earlier. Eq. (39) defines the willingness to pay for their proposed plan to improve water quality from A_1 to $A_1 + \Delta$

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

using the preference function defined in Equation (36).[26]

$$(m - WTP)^b + \sum_{i=2}^{K} (\theta_i \bullet A_i)^b + (\theta_1 \bullet (A_1 + \Delta))^b = m^b + \sum_{i=1}^{K} (\theta_i \bullet A_i)^b \tag{39}$$

Eq. (39) 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. (40) is the Hicksian willingness to pay for the improvement in water quality, the benefits measure that we seek.

$$WTP = m - (m^b + (\theta_1 \bullet A_1)^b - (\theta_1 \bullet (A_1 + \Delta))^b)^{1/b} \tag{40}$$

We can use Eq. (38) to eliminate θ_1 from (40) 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.

[26]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]).

(40) 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 non-linear 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]) and thus we have a plausible range of values for it as well.

Overall, then Eq. (38) and (40) are two non-linear 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 does not require the resulting estimates to be consistent with the individual's available income or other constraints. Since 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 willingness to pay estimate using Eq (40) 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 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 since simplifying assumptions were made to allow the use of readily available information.

From the Michael et al. [1997] 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 baseline level of water quality measured = 2.96 meters measured using secchi disk

The Mitchell and Carson CV study provides:

WTP = \$242 annually for water quality improvements.

Because the Mitchell and Carson asked respondents for their willingness to pay for national water quality improvements, measured on the water quality ladder from boating to swimming levels, the following three adjustments were made. First, we multiply the \$242 figure with a fraction 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 need 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 contingent valuation study. Recall we resolved this question for the travel cost - 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.[27] Clearly, this step of the process was somewhat *ad hoc* being constrained by the information at hand. However, it can be easily modified and does not impinge on the calibration logic. It is discussed in because the process of establishing the consistency between the physical units involved in different benefits measures is important. Thus, boating and swimming level water quality are calculated as 2.96 and 5.66 meters respectively. Third, the WTP must 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 are 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 for policy purposes would update these to the years relevant for the policy.

With these adjustment, we can solve Eq. (38) and (40) for θ_1 and b. Because of the non-linearity 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

[27]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 equals10 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.

information needed to calculate WTP for alternative water quality changes using Eq. (39) because together they characterize the indirect utility function from which the WTP measure is derived. Table 5 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 5. 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	$(\partial P/\partial A_1) \cdot (\Delta)$
$\Delta A = 1$	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 underscores the message that simple approximations can generate biased estimates of the value of non-marginal 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. (39)) to explicitly address quality changes, given income and price information.

IV. Next Steps

All approaches to non-market valuation can be interpreted as providing information that offers a partial measure of consumer preferences.[28] This conclusion follows from their definitions. 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) is 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 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

[28] 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.

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 non-marketed 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 or choice occasion indirect utility functions (for 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 makes 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 non-market 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, i.e., Marshallian consumer surplus, Hicksian willingness to pay, or marginal hedonic price? Moreover, the baseline and new levels of water quality must be defined in unit measures that are consistent with those that are assumed to enter the specified utility function.

The examples developed in sections II and III 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 differences in the situation associated with each benefit transfer. A number of questions need to be considered in evaluating whether further extensions to this approach are warranted. In the balance of this chapter, three will be introduced and discussed briefly. This discussion is not intended to be complete. Rather it highlights some of the next questions that should be considered in the process of developing practices to provide a system of stand-alone calibration procedures. Such a system would allow the analyst to readily calibrate a consistent preference function. The issues for further research should evaluate the importance of

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

A. Preference Specification

At least three issues need to be considered in preference specification. First, will the focus of analysis be a small number of priced commodities and one or more non-market 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 good are considered with environment quality) has dominated the literature. In this situation it seems reasonable to argue that it will be easier to develop quasi-indirect utility functions that are consistent with the empirical estimates rather than begin 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.[29]

Recently Ebert [1998] has offered a general summary of the issues associated with non-market valuation. He deliberately adopts a system approach to describing the tasks posed in non-market 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 willingness to pay functions for the non-marketed 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 associated with generalizing the Hausman logic for the task of valuing environmental quality changes.[30] Of course, it also significantly increases the informational requirements imposed on the modeling

[29]Hanemann [1984] proposed this strategy for econometric modeling 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 contingent valuation models (see Eom and Smith [1994] and more recently Nikletschek and León [1996]).

[30]See Bockstael and McConnell [1993] and Larson [1991; 92] for a discussion of the difficulties in recovering Hicksian measures of WTP for quality changes using Marshallian demand functions.

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 6 reproduces a table from 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.

Table 6. 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	$\left(\frac{q'}{c}+\frac{b}{c^2}\right)-\exp[c(p'-p^0)]\left(\frac{q^0}{c}+\frac{b}{c^2}\right)$	$\frac{1}{c}\ln\left[1+\frac{c}{b}(q^0-q')\right]$	$m-\left[\frac{1-c}{(1+b)m^c}(p'q'-p^0q^0)+m^{1-c}\right]^{\frac{1}{1-c}}$
Equivalent variation	$\exp[c(p^0-p')]\left(\frac{q'}{c}+\frac{b}{c^2}\right)-\left(\frac{q^0}{c}+\frac{b}{c^2}\right)$	$-\frac{1}{c}\ln\left[1+\frac{c}{b}(q'-q^0)\right]$	$\left[\frac{1-c}{(1+b)m^c}(p^0q^0-p'q')+m^{1-c}\right]^{\frac{1}{1-c}}-m$
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 $\tilde{p} = p|\lim_{p\to\tilde{p}} q = 0$. 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. This table is taken from Bockstael, Hanemann, and Strand (1984).

These applications 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.

The primary advantage of this approach is 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 the Cobb-Douglas example used in section II assumes that 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 III we adopt a CES).[31] An important and unresolved issue for calibrated transfers is that only one of these studies (Espinosa and Smith [1995]) has considered the role of non-marketed goods in the preference specification that permit 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 will complicate the calibration of the full economy to a baseline set of conditions.

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

B. 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 (28) to include recreation component into 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 the preferences of individuals.

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, et al. [forthcoming] as an example). Developing multiple estimates of the same benefit concept was the strategy used in meta analyses of past benefit studies.[32] It is important to note that statistical functions derived as meta summaries or response surfaces do not necessarily impose the preference consistency.[33] They are simply a different type of "reduced form model."

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[32]See Smith and Kaoru [1990] or Smith and Huang [1995] as examples.

[33]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].

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

C. 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 this also seems unlikely to offer many practical insights for evaluating transfer practices.

Because benefit measures are never observed, estimates of them 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 evaluation of 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 III illustrated this advantage because computation of 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 such that their correspondence with plausible or standard levels of elasticities and linked good is enhanced.[34] 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 doing benefit transfers. More complex functional forms are possible and numerical calibration analogous to what is used with numerical CGE models is also possible. However, the desirability of pursuing such larger scale efforts depends

[34]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.

on the success of experimentation with smaller applications of the method and comparisons with current practice. It would be relatively easy to consider an exercise where recent benefit transfers were "redone" using the calibrated preference logic and compared with the approach used in the policy analysis. This would seem to offer a next step in evaluating the usefulness of the logic and should precede attempts for more ambitious numerical calibrations.

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Land Use Diversity and Urban Watersheds: The Case of New Haven County

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Land Use Diversity and Urban Watersheds: The Case of New Haven County

ABSTRACT : This paper presents the results of a hedonic property value analysis for an urban watershed in New Haven County, Connecticut. We use spatially referenced housing and land use data to capture the effect of environmental variables around the house location. We calculate and incorporate data on open space, land use diversity and other environmental variables to capture spatial variation in environmental quality around each house location. We are ultimately interested in determining whether variables that are reflective of spatial diversity do a better job of describing human preferences for housing choice than broad categories of rural versus urban areas.

Using a rich data set of over 4000 houses we study these effects within a watershed which includes areas of high environmental quality and low environmental quality as well as varying patterns of socio-economic conditions. Our results suggest that, in addition to structural characteristics, variables describing neighborhood socio-economic characteristics and variables describing land use and environmental quality are influential in determining human values. We also find that the scale at which we measure these spatially defined environmental variables is important.

INTRODUCTION

Over the past decade, various papers applying hedonic property analysis to environmental valuation issues have suggested that the location or proximity of a house with respect to environmental features in the landscape is of some importance in determining its market value. These studies provide evidence that the market price of a house reflects the level of some environmental good home owners are aware of and are willing to pay for. Various papers in the literature suggest that variables describing land use and environmental quality are influential in determining human values. The question of scale and pattern in land use is a less studied area, with relatively few studies published on these elements of assessing environmental preferences (Bockstael, 1996; Geoghegan, *et al.*, 1997). We suggest that these questions are important for understanding the impact of land use planning regulations on housing preferences.

The present paper therefore studies the effect of land use variables, such as open space, commercial areas and forest land on house prices and people's willingness to pay for these features in relation to the land use around their houses. We ask whether scale matters – do people consider land use features at varying distances from their house and do these factors affect property values? We also ask a similar question with regard to the spatial distribution of various types of land uses, i.e. is there a preference for homogenous or chaotic land use planning? We then compare these results with the information we receive from using the more traditional urban/rural categorization of land use in the hedonic model estimation. The use of sophisticated spatial variables is relatively new in the hedonic property value literature. We find that variables representing urban watershed health and integrity, including land use and open space, significantly affect consumer choices of location and willingness to pay for housing.

The hedonic price technique is based on isolating the contribution of various factors to the market price of a good, through the use of econometric techniques. Hence it may be used to estimate the value of a public good, such as environmental quality, by using market prices for private goods, such as houses. We apply the hedonic property model in the New Haven watershed system to measure the direct use of environmental quality. This watershed is composed of three rivers – the Quinnipiac, Mill and West rivers – which together drain an area of 600 km^2 and converge in New Haven harbor on Long Island Sound. The watershed supports a population of 610,000 people (see Figure 1) and covers a range of rural, suburban and urban levels of development. This therefore provides an opportunity for us to study linkages between ecological and economic systems, including variations in physical characteristics within the watershed. In order to effectively apply the hedonic price technique and to accurately represent the environmental conditions within the watershed we use spatial techniques and geographically referenced maps of land use. The spatial distribution of land uses within the watershed is not uniform. We therefore incorporate into our database a number of land use variables intended to capture spatial variation in environmental factors.

THE HEDONIC PROPERTY MODEL

The hedonic property value method is a revealed preference technique that utilizes actual market transactions in housing real estate. The idea is that when home buyers select a house, they are purchasing more than just the physical structure and the plot of land. They are also purchasing the site specific attributes of the neighborhood where the house is located. These site specific attributes include environmental quality, safety, demography, and the quality of local government services such as schools. Therefore, the prices paid for homes should reflect the capitalized value of environmental quality to the homeowner.

A basic assumption of the hedonic property value model is that the study area can be treated as a single market for housing and that this housing market is in equilibrium. In addition it is assumed that individuals have information on housing choices and are mobile enough to choose a house anywhere in the market area (Freeman 1993; Palmquist 1991). These assumptions imply that individuals choose housing based on utility maximization, given the prices of alternative housing choices, and that the prices just clear the market. While sometimes criticized as restrictive we feel these assumptions are not unrealistic for the relatively small area that is the subject of this study.

Houses are differentiated from each other in a variety of dimensions including structural characteristics such as the material the house is constructed from, accessibility to highways, neighborhood characteristics such as average income, racial composition and natural environmental characteristics such as land use and water quality. It is therefore necessary to control for structural housing characteristics and neighborhood characteristics if we are interested in understanding the role of land use and environmental quality on consumer preferences and willingness to pay. As is well established in the literature, we can use the hedonic price equation to estimate the equilibrium price schedule for the environmental variables we are interested in studying.[1] This function relates the price of a house, h, to its structural and environmental characteristics and may be represented by the following function:

$$P_h = f_h(S_{h1},\ldots,S_{hj},N_{hk},\ldots,N_{hk},Z_{h1},\ldots Z_{hm}) \qquad \textbf{for all } h \tag{1}$$

where S = a vector of structural characteristics

N = a vector of neighborhood characteristics

Z = a vector of environmental characteristics

A utility maximizing consumer is therefore assumed to maximize the following utility function:

$$\textbf{Max } U = U(S,N,Z,X) \tag{2}$$

Subject to a budget constraint:

$$Y = P_x X + P_h(S,N,Z) \tag{3}$$

where X is a composite commodity or numeraire consumed by the individual, S,N,Z are as defined earlier; Y refers to household income and P_x is the price vector of the commodity X, where we assume $P_x = 1$. Assuming utility maximizing behavior and an interior solution to this utility maximization problem, and assuming preferences are weakly separable in housing and its

[1]The theory underlying hedonic models was first developed by Griliches (1971) and Rosen (1974).

characteristics, we expect that the individual will set marginal willingness to pay for a housing characteristic equal to the marginal implicit price for the characteristic:

$$\frac{\partial U}{\partial Z_i} = \frac{\partial P_h}{\partial Z_i} \tag{4}$$

The hedonic price function (1) is therefore an implicit price relationship that gives the price of a house as a function of its various characteristics and the partial derivative of the hedonic price function with respect to any characteristic defined in (1) gives us the marginal implicit price of that characteristic. That is:

$$\frac{\partial P_h}{\partial Z_j} = P_{h Z_j}(S_{h1},...,S_{hj}, N_{hk},..., N_{hk}, Z_{h1},...Z_{hm}) \tag{5}$$

Since the price schedule represents a locus of the equilibrium marginal willingness to pay of all households, it cannot be interpreted as representing either the demand or the supply of characteristics. However, if the hedonic price function can be determined, then the individual's marginal willingness to pay for a characteristic may be estimated from the slope of the function with respect to the characteristic. The functional form for the hedonic equation is not determined theoretically and need not be linear since it is determined by the interaction of both supply and demand within the housing market. The hedonic equation must therefore be determined empirically.

A number of studies now exist which make use of hedonic property models to examine the effects of environmental disamenities and amenities. These studies include those highlighting the impact of variations in site-specific factors such as local climate (Haurin, 1980), air pollution (Harrison and Rubinfeld, 1978, Palmquist, 1982, Murdoch and Thayer, 1988) water quality (Brashares, 1985, Feenberg and Mills, 1980) and other amenities (Rosen, 1979; Roback, 1982). Various empirical studies also include the effects of crime, recreational opportunities, and population demographics (Berger and Blomquist 1992; Potepan 1996). Some studies have also included measures of school quality when explaining house price variations (e.g. . Li and Brown, 1980 and Pogodzinski and Sass, 1991.

CAPTURING ENVIRONMENTAL VARIATION IN THE HEDONIC MODEL

Models that address environmental externalities which characterize land use have a strong spatial component. The value of a parcel of residential land is affected by the pattern of surrounding land uses, not just the specific features of point locations (Bockstael, 1996, Geoghegan *et al.*, 1997). Hedonic models have generally utilized access and distance variables to represent these spatial components or uni-dimensional spatial variables such as neighborhood socioeconomic census data. Bockstael and Bell, 1997 suggest that the nature of the surrounding landscape will affect house values. Geoghegan *et al.*, 1997 point out that the problems with these traditional approaches are that "locational characteristics are more likely characterized by a gradient than by discrete levels that change abruptly," (i.e., census tract boundaries) and that it may not be just neighborhood effects causing the externalities, but patterns. Geoghegan *et al.* attempt to account for these patterns by including diversity and fragmentation indices which measure land use and pattern. Leggett and Bockstael, 1998 examine the impacts of the percentage of area in various types of land use in determining house values in coastal areas. In this paper we draw on the findings of these recent applications but we also compare the use of ecological indices to traditionally defined categories of urban and rural areas. Determining the extent of the differences between these measures could have important policy implications.

In this paper, we describe the nature of the landscape surrounding each house by using a set of variables which describe landscape pattern. In particular, we utilize a data set for one watershed and county. We geo-code the houses as points using their exact latitude/longitude data and are interested in showing that the value of aggregate measures of land use and landscape pattern, which affect the ecosystem's ability to provide certain types of habitat and support natural processes, are reflected in human perceptions of their environment and the value they indirectly associate with their natural surroundings. We introduce a set of land use related variables to understand the importance of using appropriate explanatory variables as well as investigate the importance of scale. We are ultimately interested in determining whether variables that are reflective of spatial diversity do a better job of describing human preferences for housing choice than traditional variables. In addition, we address potential spatial auto-correlation problems within our data set.

An explanation of the types of landscape variables we use is required before we proceed.

Aggregate variables

These categories are based on assessing the majority land use around each house using land use data. The land use categories are then aggregated into broad categories of rural, urban and semi-urban where:

Rural: open space, forest, water, fields and agriculture land use

Urban: impervious surfaces, high density residential & commercial, roof, pavement, and major roads.

Sub-urban: medium and low density residential land use.

Mosaic variables:

Diversity, richness, evenness, dominance measures are some ways of determining the relative numbers of types, sizes or shapes of land use patches present in a landscape mosaic (Forman, 1995). By analyzing the heterogeneity of a landscape, ecologists attempt to address the question of whether the abundance of patches in a landscape is ecologically important. Equally, the location of patches with reference to each other is an important area of research (see Forman, 1995, Turner, 1989).

Similarly, it is suggested that heterogeneity in land use/land cover and spatial patterns and features of the landscape may be important for property values. We investigate the importance of the following landscape features:

Diversity: This variable, used by Geoghegan *et al* (1997), measures whether an area is dominated by a few or many land uses and is defined as: $H = -\sum_i P_i \ln P_i$. The index measures the proportion of land in the number of identified land use types within the watershed.

Richness: Relative richness is an alternative diversity measure where R = $(s/s_{max}) \times 100$. This measure looks at the relative richness of land uses in an area (s) in terms of the total number of land use types (s_{max}) found within the watershed. Therefore it differs from the diversity index in that it is not a measure of area but a ratio of land use types relative to the maximum possible land use types found in the watershed.[2]

In addition, the percentage of open space around each house within a 1 mile and ¼ mile radius is included as an additional variable.

Spatial pattern:

The location of a house is, as the joke goes, the first, second and third most important criteriá in purchasing a house. Location in relation to work, roads, schools, shops, open space, water bodies etc., can be relatively easily incorporated into our study because of geographically referenced data. We examine the following features: distance to open space, distance to lakes, distance to streams, distance to ocean, distance to parks, distance to highways.

THE DATA SET

The data set includes over 4,000 houses sold in New Haven County between 1995 and 1997 (see Figure 2).[3] This data comes from actual house sale prices obtained from real estate multiple listings that are compiled by local real estate boards. The multiple listings also include detailed information about house characteristics (e.g. lot size, number of rooms, type of heating, etc.). This property information is combined with demographic, land use and socioeconomic information obtained from the 1990 U.S. Census. The data set is unique for two reasons: First, this is a small urban watershed varying on a gradient of both population density and environmental quality. Second the local economy was relatively stagnant during this time period which will allow us to isolate the effects of environmental variables on housing price without

[2] Using a land use map and an indicated analysis radius we determined the proportion of land in each of the 5 broad land use categories established. High density land use includes: impervious surfaces, high density residential & commercial, roof, pavement, and major roads. Medium density land use is medium density residential. Forest land use includes deciduous forest and coniferous forest. Water land use includes deep water, shallow water, non-forested wetland, forested wetland, low coastal marsh, and high coastal marsh. Fields and agriculture land use includes: turf & grass, soil/grass & hay, grass & hay & pasture, soil/corn, grass/tobacco, barren land, and bare soil.

[3] The real estate market in Connecticut during this time period was stagnant or falling in some places. We thus assume these to be real prices for the time period used.

introducing the bias caused by a rapidly changing economy with the associated large swings in population.[4]

To estimate a model that can discover the environmental values held by home buyers, it is critical to be able to relate the location of each home to the attributes of its surrounding environment. The geocoding process was performed for each of the 13 towns in the New Haven Watershed. These 13 towns are: Berlin, Bethany, Cheshire, Hamden, Meriden, New Haven, North Haven, Plainville, Prospect, Southington, Wallingford, West Haven, and Woodbridge. Data on land use, roads, municipal and private open space, state owned open space, and Census block groups are also incorporated into the data set. Since the watershed is demarcated based on hydrological criteria, and does not necessarily conform to economic activity, it is important to select land use and land cover features on the outer edge of the watershed map to allow spatial statistics to be properly calculated for homes near the outer edges of the New Haven watershed.

METHODOLOGY

In order to compare traditional land use measures with those utilizing indices found in the ecological literature, we examine two different models. The first model incorporates traditional measures of land use representative primarily of housing/population density. The second model incorporates variables that represent both scales and patterns of land use at varying distances from the houses.

One set of variables relates to determining whether the majority land use around a house can be classified as urban. suburban or rural. These land use categories are determined by assessing the majority land use within a 1 mile and 1/4th mile radius around each house. We then assign dummy variables to place the land within our three categories of urban, rural and sub-urban as defined earlier.

[4] A concern associated with the hedonic property value model is that if there are market forces moving consistently in one direction or if environmental quality variables are rapidly changing, bias may be introduced into the model (Freeman 1993). The stagnant economy should help us reduce any potential bias in our model.

A second set of variables concerns those explanatory variables which describe the pattern of the landscape surrounding each house. We determine the percentage of area around each house that is considered open space[5]. A summary of land use within the New Haven watershed is shown in Table 1. To describe distribution and diversity of land use within the watershed we calculate a diversity index defined earlier as a measure of how diverse land use is within a certain area (Turner, 1989; Goegehan *et al.*, 1997). The value of this index depends both on the diversity of land use and the evenness with which these land uses are distributed within the specified area. The more land use categories there are and the more even their distribution, the greater the diversity[6]. We define "land use chaos" as reflecting a higher diversity of land use, but note that an increase in the diversity index may occur with increased evenness in the distribution of land use. Figure 3 illustrates the diversity index at a ¼ mile radius. We also calculate a richness index which should be able to explain the additional effect of local variety in land use, relative to that found within the entire watershed. So, for example, if the watershed has 5 types of land use and only one is found in the vicinity of your house, you have a low relative richness of land use around your house. If an area has high diversity it may not have a high relative richness if it does not include a majority of the representative land use types found within the watershed. In general, the higher the value of these indices, the higher the number of uses within the area. Conversely, a low value suggests a single land use or relatively few land uses.

[5] All grid maps for this project have a cell size of 100 feet and use the same road map for extent. Open space was defined to include the following land use categories: turf/grass, soil/grass/hay, grass/hay/pasture, soil/corn, soil/tobacco, grass/tobacco, deciduous forest, coniferous forest, deep water, shallow water, non-forested wetland, bare soil, low coastal marsh, and high coastal marsh. Non-open space includes: impervious surfaces, high density residential and commercial, medium density residential, roof, pavement, barren land, and major roads.

[6] The Shannon index, as traditionally used as a biodiversity measure, is sensitive to changes in the abundance of rare species (i.e., a Type 1 index). In the context of this study, the value of this index is likely to be higher if rare or very abundant land use types are lower in an area because the distribution of land use types is more even. An increase in the index value can occur despite a decrease in the abundance of rare land uses. Similarly a decrease in the index value can occur with reduced evenness in the distribution of land use types.
The Simpson index, which is sensitive to changes in the abundance of the most common species (dominance), may be an alternative index for us to try. It is calculated as: $D = \sum p_i^2$ where p_i = proportion of land in category i. So as D increases (or $1\text{-}D$ decreases), diversity decreases. If there are changes in the abundance of the most common land use within the given area this index will be more affected than if there are changes in the rare land uses within the area.

We have no prior expectations on the sign of the coefficient on the diversity variable or of the richness variable. In order to use these variables in the context of development levels within the watershed, we suggest the relative richness of an area weighted by the population density in the area would be a good indicator of the level of development of that area. We therefore multiply richness by population density and use this variable to examine the differences between densely populated areas with high relative richness in land use/land cover and sparsely populated areas with low relative richness in land use/ land cover.

We also expect that social and demographic neighborhood characteristics could affect housing prices and our measures of neighborhood characteristics include variables such as percentage of white households, crime rate per 1000 people and average income. We considered various measures of school quality such as test scores, attendance rate, dropout rate, etc. In addition, to account for the variation created by differences in property taxes, we include the town mill rate which is representative of the property tax for each community. Other explanatory variables in the house value equation include those suggested by various empirical studies of urban housing demand such as distance of a property to large cities such as New Haven and Hartford in this case. These were however found to be consistently insignificant and were omitted from the final model. We include a variable for average time taken to travel to work (WORKTIME) derived from the aggregate time to work reported by households in the census data. Variable names and definitions for the variables used in the formal analysis are presented in Table 2.

RESULTS AND DISCUSSION

Keeping in mind that the hedonic price function is determined empirically and that the functional specification of the price function will have a significant effect on the estimates of the coefficients, we considered some common functional forms (linear and double log forms) of which the semi-logarithmic form provided the best fit, yielding the following hedonic model:

$$\ln(VALUE) = \alpha_0 + \alpha_1 S + a_2 N + a_3 Z + \varepsilon \qquad (6)$$

The dependent variable, *ln(VALUE)*, is the natural logarithm of the house value. A number of structural and neighborhood variables were included to control for additional factors that determine house prices.

While we found considerable heteroscedasticity in the linear and double-log models, White's χ^2 test fails to reject the null hypothesis of homoscedasticity for the log-linear model. We are however concerned that we may have a problem of spatial autocorrelation due to omitted variables which may be spatially autocorrelated. Although a map of the residuals suggests no significant spatial pattern, we test for possible spatial interaction between our observations by proposing the following spatial error model:

$$Y = X_1 B_1 + X_2 B_2 + \ldots + \varepsilon \qquad (7)$$
$$where \quad \varepsilon = W\varepsilon + \mu \;\; and \;\; \mu \sim N(0, \sigma^2)$$

W is a spatial weights matrix where W_{ij} is a normalized measure of the associated between the i^{th} and the j^{th} residuals[7]. We hypothesize that $\varepsilon_i = f(\varepsilon_j \ldots \varepsilon_k)$ where $\varepsilon_j \ldots \varepsilon_k$ are residuals of observations within a one mile radius of ε_i. We define W_{ij} as the average of the OLS residuals within this 1 mile radius. We then test our hypothesis that there may be spatial autocorrelation by running a new OLS regression with the average error term, W_{ij}, as an additional regressor. We find that the new variable has no explanatory power in our regression and does not affect the stability of the model results. Given our large data set, we suggest that this is an adequate measure of spatial autocorrelation and reject our hypothesis that there is perceptible spatial autocorrelation due to omitted spatially correlated variables within our study area.

Table 4 presents the selected results of both model 1 and model 2. Complete model results are presented in the appendix. The first column of Table 4 presents the results from model 1 which uses simple dummy variables for urban and rural areas. We define urban areas as areas where

[7] Traditional spatial weights matrices generally provide a means for comparing information on the proximity of observations in terms of their location, with information on some other variable which measures the location (Odland,1988: Anselin, 1988). We believe that unless houses are located next to each other, they are unlikely to have an impact on neighbouring house values. We are therefore interested in examining the effect of some other

the majority land use within a 1/4th and 1 mile radius is high density residential or commercial development and rural areas as areas where the majority land use is forest cover and wetlands. Medium density residential areas are defined as medium level or suburban development. We use two dummy variables, URBAN1 and RURAL1 to capture the effect of land use on house prices. The coefficients on URBAN1 is positive and somewhat significant while the coefficient on RURAL1 and RURAL4 were found to be consistently insignificant. While providing some information on the effect of housing density on property value, this classification tells us relatively little about the type of landscape or development levels preferred by house buyers.

The second model in which we include the landscape pattern variables and omit the broad rural/urban categories, includes the variable DEVELOP as an indicator of the level of development around the house. Using the second model where the spatial patterns are more explicit we see that the spatial distributions as well as the types of land use present have fairly substantial effects on property values. The results (also shown in tables 4 and the appendix) that follow support our hypothesis that both environmental conditions and population density are important in determining the level of development in an area.

The results suggest that 77% of the variation can be explained by both models. Most of the structural characteristics of the houses in the sample were found to be significant with interior space (SQFT) indicating that there are decreasing returns from the physical area of the house. The percentage of open space within ¼ mile also exhibits decreasing returns. Demographic and neighborhood effects are also significant. Average education (EDULEV) is used as a proxy for community income and social status and the coefficient on this variable is found to be significant and positive. Crime, as expected, has a negative effect on property prices as does higher population density. We also find that houses sold in the winter have a somewhat lower selling price than houses sold during the remaining parts of the year. We find that travel time to the nearest highway has a significant and positively signed coefficient. This effect we believe is reflective of a preference to live in less noisy and more suburban areas. The selling price is negatively correlated with distance to the ocean and distance to lakes as expected.

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spatially correlated variable we may have omitted in our model specification and which may be therefore reflected in the distribution of the residual terms.

Percentage of open space around a house and the diversity of land use are both found to be significant variables in determining property values. An increase in the percent of open space within a ¼ mile radius of a home increases the value of the property. Interestingly we note that the coefficient on the diversity index at a ¼ mile radius is negatively signed, indicating that people prefer to live in places with more homogenous land use in the immediate vicinity of their houses. However, as noted earlier, this index does not reflect the type of land use and therefore houses in commercial areas and near forested areas are both likely to have higher property values due to low diversity index values. The sign on this coefficient therefore establishes that there is a tendency for property prices to be higher in areas with a single land use. This is very likely influenced by zoning regulations and this effect would need to be more fully incorporated into the model in order to understand the effect of the diversity index more clearly.[8] These results are of particular interest to planning policies given that there is a higher value associated with certain types of land use and indeed with particular patterns in land use as shown by the diversity index.

The coefficient on the variable DEVELOP is found to be positive and significant. This suggests that houses in areas with high population density and high relative richness in land use fetch a higher selling price whereas houses in areas with low population density and low relative richness in land use have a lower selling price. If we translate this to urban and rural categories, based on population density, this result suggests that houses in urban areas with higher land use richness have a higher selling price than houses in urban areas with lower land use richness. This makes intuitive sense since, in urban, populated areas, there may be a preference for different amenities such as parks, shopping areas etc. Similarly, in rural areas where population density is lower, a low relative richness in land use/land cover results in lower selling prices for houses.

The interaction term implies that the elasticity of population density and richness will vary. We calculate the elasticity effect of the richness variable and estimate that the richness elasticity ranges from –0.0008 in areas of low population to 0.0045 in high population areas. This suggests that although house values are relatively inelastic with respect to the relative richness in land use/land cover, areas of high population density have a higher elasticity. As populations increase, the value of the house becomes more elastic with respect to land use/land cover.

CONCLUSIONS

This paper studies the value of environmental variables such as open space and land use diversity to choices made by human beings within a watershed context. In this it adds to the growing evidence that spatial patterns are influential in determining human preferences for their living spaces. We have also suggested an alternative test for spatial autocorrelation where we test whether our regression residuals are spatially correlated and whether these explain any variation in the model. We find that this is a simple test to use with a large data set such as ours where there is no basis for using the proximity of houses as a weighting matrix.

This paper has used a rich data set to show that variations in neighbourhood variables and land use pattern can have an effect on house values. In particular, we have contrived to show that the use of simple dummy variables to differentiate between rural and urban land use categories give us ambiguous and uninteresting results. On the other hand, the use of variables which attempt to capture some of the spatial characteristics of land use and land cover together with population density support the hypothesis that both scale and pattern are important in hedonic property analysis.

Acknowledgements: We would like to thank George Silva for his research assistance on the data set and Jackie Geoghegan, George Parsons and Robert Mendelsohn for useful comments on this paper.

[8] The diversity indices were calculated for both a ¼ mile and a 1 mile radius around each house. The ¼ mile radius was chosen to be representative of immediate walking distance or visual distance from a house. This latter variable was dropped due to insignificance.

The New Haven Watershed

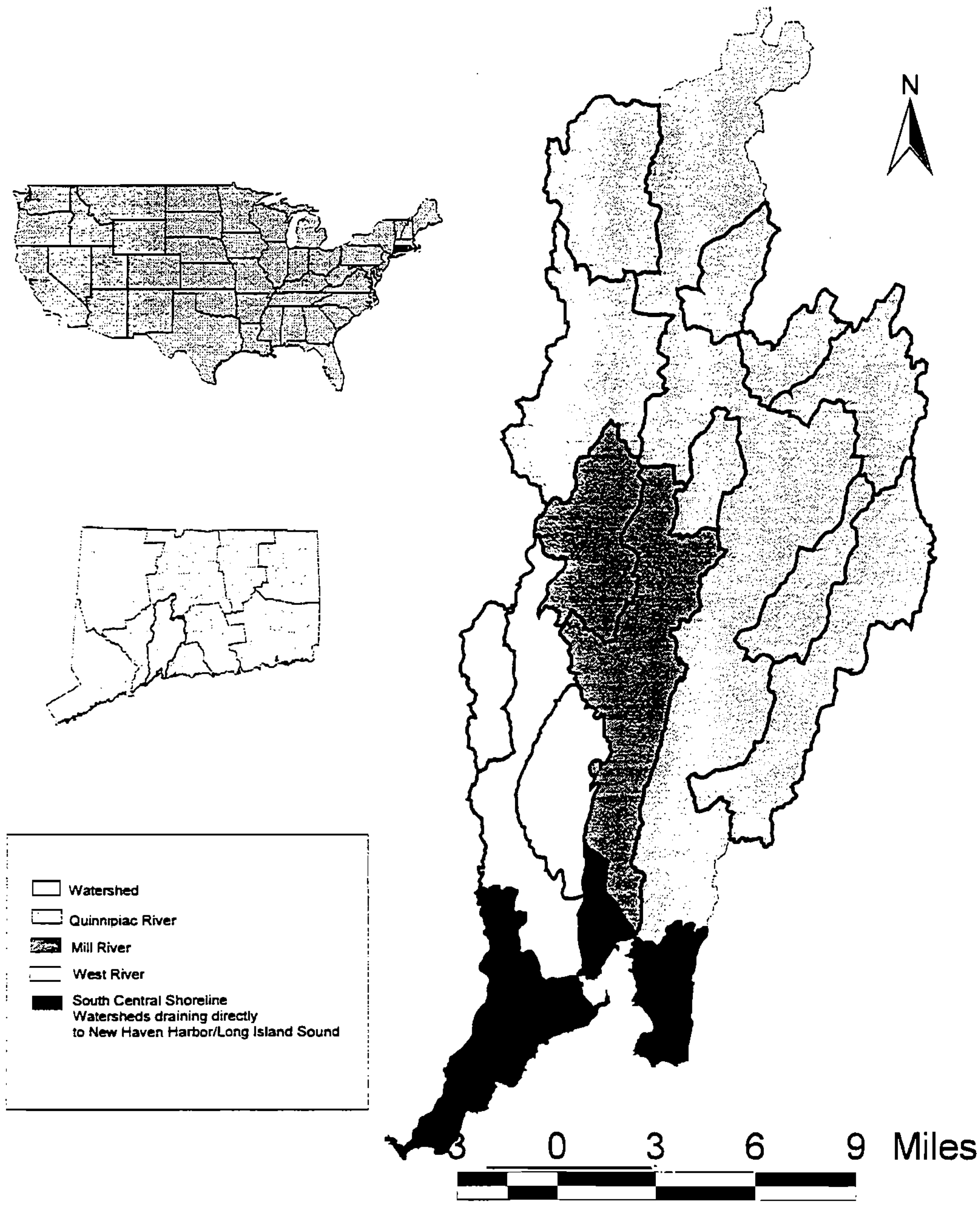

Figure 2

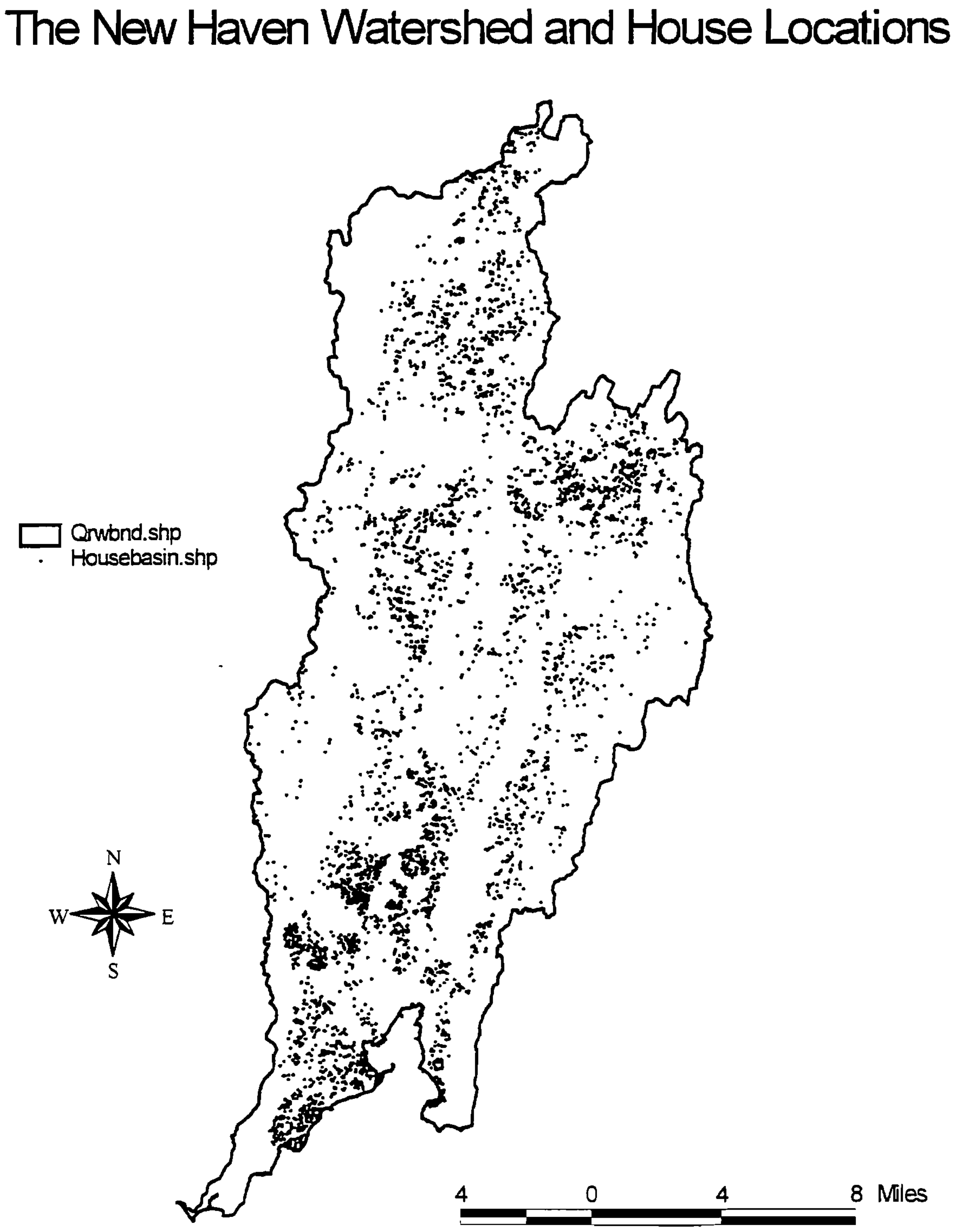
The New Haven Watershed and House Locations
Qrwbnd.shp
Housebasin.shp
N
W
E
S
4
0
4
8 Miles

Figure 3

Land Use Diversity at 1/4th mile

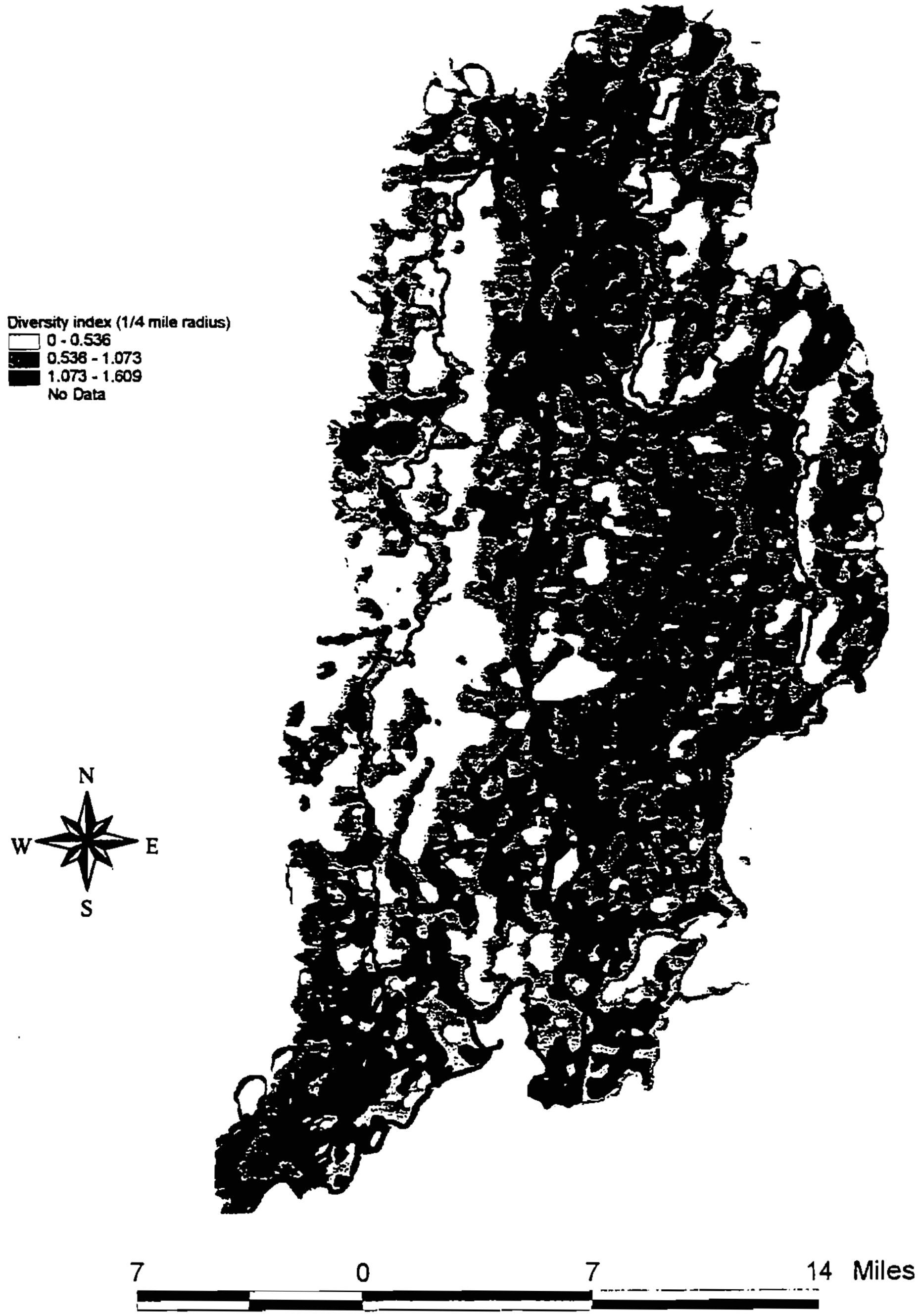

Table 1

Land Use	Land Use Code	Total Area (square feet)	Percentage of Area
Other	0	3,906,527.04	0.05%
Impervious Surface	1	291,959,641.54	4.00%
High Density-Residential/Commercial	2	390,811,852.46	5.35%
Medium Density Residential	3	1,566,848,786.62	21.45%
Surface – Roof	4	11,526,519.52	0.16%
Pavement/Road	5	2,383,238.65	0.03%
Turf/Grass	6	192,659,244.90	2.64%
Soil/Grass/Hay	7	259,141,178.26	3.55%
Grass/Hay Pasture	8	513,471,343.29	7.03%
Soil/Corn	9	27,659,796.81	0.38%
Grass/Corn	10	21,031,831.73	0.29%
Deciduous Forest	13	2,965,612,148.58	40.59%
Conifer Forest	14	92,309,041.32	1.26%
Deep Water	15	97,353,371.43	1.33%
Shallow Water	16	109,912,119.56	1.50%
Non-forest Wetland	17	3,164,307.18	0.04%
Forest Wetland	18	135,782,013.31	1.86%
Barren Land	19	185,268,873.09	2.54%
Bare Soil	20	145,496,886.99	1.99%
High Coast Marsh	22	62,213,104.30	0.85%
Major Road	25	227,096,298.97	3.11%
	Total	7,305,608,125.52	100.00%

Table 2

VARIABLE NAMES AND DEFINITIONS

Variable	Definition
SPRICE	Selling price of house
WINTER	Dummy variable for transactions occurring in winter months
ACRES	Lot size in acres
LEVELLOT	Dummy variable for a level lot
VIEW	Dummy for view (realtor determination)
SQFT	Number of square feet in the house
BATHS	Number of bathrooms
NGARAGE	Number of cars garage can hold
FIRE	Number of fireplaces
POOL	Dummy variable for the presence of a pool
DECKS	Dummy variable for the presence of a deck
CAIR	Dummy variable for the presence of central air conditioning
PUBWATER	Dummy variable for connection to a public water supply
ATTICP	Dummy variable for the presence of an attic
BNONE	Dummy variable for the lack of a basement
FINBASE	Dummy variable for finished basement
BRICK, CEDAR, CLAP, SHAKE, SHING, STONE, STUCCO, VINYL, WOODEXT	Dummy variables for the exterior construction material. Omitted is other exteriors not listed.
CAPE, COL, RAISE, RANCH, SPLIT, CONT, BUNG, VICT, TUDO, ALUM, EXTASBSES	Dummy variables for house style. Omitted is other house styles not listed.
PETRO	Dummy variable for the use of oil or gas for heating
AGE	Age of the house in years
LAKWTR	Distance to lake
EQUAL_MI	Equalized mill rate
COLLEGE	Percent of students continuing to college education at local high school
CRIME	Crime rate per 1000 people at town level (1994)
PWHITE	Percent of population in the block group that is white
WORKTIME	Average travel time to work in minutes for block group
EDULEV	Average number of years of education (adults over 25) for block group
POPDENSE	Population density for the block group (people/ha)
TCHIGH	Relative distance to nearest highway (weighted by type of road)
DOCEAN	Distance in feet from Long Island Sound
DIVERS1.4	Diversity index for a 1 and ¼ mile radius
POPEN4	Percent of landscape in open space
URBAN/RURAL	Dummy variables for majority land use determined to be either urban, suburban or rural
DEVELOP	Level of development determined by the relative richness of land use and population density

Table 3

DESCRIPTIVE STATISTICS OF SOME VARIABLES

VARIABLE	Minimum	Maximum	Mean
HOUSE VALUE (in US dollars, 1995-1997 selling prices)	$10,509	$729,416	$127,681
ACRES (area of the lot in acres)	0.03	80.7 2	0.6 2
SQFT (area of the house in sq.ft.)	300	14,000	1633.6
BATHS (number of bathrooms)	1	6.3	1.58
NGARAGE (number of garages)	0	8	1.23
EIRE (number of fireplaces)	0	8	0.71
EQUAL_MI (equalized mill rate)	12.25	30.28	21.35
COLLEGE (% of students continuing on to college)	70	86	76.63
CRIME (crime rate per 1000 people)	13.01	130.92	51.88
PWHITE (% of white population)	0	1	0.91
WORKTIME (average time to work in minutes)	9.17	25.22	19.24
EDULEV(average number of years of education (adults over 25))	10.12	17.52	13.3
POPDENSE (persons per hectare)	98.36	28,073.64	3,552.8
TCHIGH (weighted distance to highway in feet)	0.00	190,505.00	12,514.00
OCEAN (distance to Long Island Sound in feet)	100.00	143,252.00	53,003.28
DIVERS4 (diversity index for a ¼ mile radius around each house)	0.013	1.543	1.026
POPEN4 (% of open space within ¼ mile radius of each house)	0.00	100	36.19

Table 4

SELECTED REGRESSION RESULTS

VARIABLE	Model 1 (log-linear, with urban/rural classifications)	Model 2 (log-linear, with spatially explicit variables)
WINTER	-0.19052E-01	-0.19152E-01
	(-2.533)*	(-2.554)*
ACRES	0.23126E-01	0.22512E-01
	(5.708)*8	(5.562)**
ACRES2	-0.31502E-03	-0.30696E-03
	(-4.716)**	(-4.602)**
SQFT	0.36801E-03	0.36916E-03
	(25.294)**	(25.426)**
SQFT2	-0.29954E-07	-0.29970E-07
	(-16.452)**	(-16.498)**
LAKWTR	-	-0.46010E-05
		(-2.128)*
EQUAL_MI	-0.20317E-01	-0.20295E-01
	(-10.388)**	(-10.338)**
COLLEGE	0.29694E-02	0.34216E-02
	(2.767)**	(3.181)**
CRIME	-0.80597E-03	-0.79071E-03
	(-3.240)**	(-3.176)**
PWHITE	0.58581	0.58108
	(16.448)**	(16.301)**
WORKTIME	-0.45805E-02	-0.51622E-02
	(-2.703)**	(-2.995)**
EDULEV	0.10547	0.10421
	(24.084)**	(23.845)**
PDENSE	-0.15095E-12	-0.55421E-12
	(-2.712)**	(-3.276)**
DOCEAN	-0.61457E-06	-0.49279E-06
	(-3.562)**	(-2.907)**
URBAN1	0.28438E-01	-
	(1.568)	
RURAL1	-0.27118E-02	
	(-0.310)	-
DIVERSITY4	-	-0.76098E-01
		(-3.033)**
POPEN4	0.23338	0.44648
	(3.775)**	(5.374)**
POPN4^2	-0.20390	-0.41791
	(-3.139)**	(-4.995)**
RICH14	-	-0.71371E-03
		(-2.078)*
DEVELOP	-	0.59072E-14
		(2.473)*
CONSTANT	9.2333	9.3438
	(82.710)**	(79.943)**
R^2	0.77	0.78
F	413.08	393.83
OBSERVATIONS	4326	4326

t-statistics in parenthesis; * and ** denote significance levels at the 0.1 and 0.01 levels respectively.

APPENDIX

Complete Model Results

VARIABLE	Model 1 (log-linear, with urban/rural classifications)	Model 2 (log-linear, with spatially explicit variables)
WINTER	-0.19052E-01	-0.19152E-01
	(-2.533)*	(-2.554)*
ACRES	0.23126E-01	0.22512E-01
	(5.708)**	(5.562)**
$ACRES^2$	-0.31502E-03	-0.30696E-03
	(-4.716)**	(-4.602)*8
LEVELLOT	0.19697E-01	0.19521E-01
	(2.539)*	(2.520)*
VIEW	0.32624E-01	0.33771E-01
	(1.756)	(1.821)
SQFT	0.36801E-03	0.36916E-03
	(25.294)**	(25.426)**
$SQFT^2$	-0.29954E-07	-0.29970E-07
	(-16.452)**	(-16.498)**
BATHS	0.42591E-01	0.42506E-01
	(5.578)*	(5.579)**
LEVELS	0.56602E-01	0.56282E-01
	(6.698)**	(6.675)**
FIRE	0.62334E-01	0.61777E-01
	(9.959)**	(9.890)**
POOL	0.44885E-01	0.44524E-01
	(4.160)**	(4.137)**
DECKS	0.44399E-01	0.45640E-01
	(5.136)**	(5.291)**
CAIR	0.59056E-01	0.59133E-01
	(6.472)**	(6.496)**
PUBWATER	-0.63203E-02	-0.53575E-02
	(-0.521)	(-0.439)
ATTICP	0.81358E-01	0.82968E-01
	(8.084)**	(8.259)**
BNONE	-0.12208	-0.12503
	(-5.168)**	(-5.302)**
FINBASE	0.23021E-01	0.21325E-01
	(2.973)**	(2.753)**
FWOOD	0.27164E-01	0.26302E-01
	(3.661)**	(3.552)**
RANCH	0.75399E-01	0.73478E-01
	(6.466)**	(6.309)**
LAKWTR	-0.28311E-05	-0.46010E-05
	(-1.335)	(-2.128)*
VINYL	0.35120E-01	0.34468E-01
	(4.306)**	(4.234)**
AGE	-0.25112E-02	-0.24801E-02
	(-17.135)**	(-16.896)**
EQUAL_MI	-0.20317E-01	-0.20295E-01
	(-10.388)**	(-10.338)**
COLLEGE	0.29694E-02	0.34216E-02
	(2.767)**	(3.181)**
CRIME	-0.80597E-03	-0.79071E-03

	(-3.240)**	(-3.176)**
PWHITE	0.58581	0.58108
	(16.448)**	(16.301)**
WORKTIME	-0.45805E-02	-0.51622E-02
	(-2.703)**	(-2.995)**
EDULEV	0.10547	0.10421
	(24.084)**	(23.845)**
PDENSE	-0.15095E-12	-0.55421E-12
	(-2.712)**	(-3.276)**
DOCEAN	-0.61457E-06	-0.49279E-06
	(-3.562)**	(-2.907)**
TCHIGH	0.60932E-06	0.32566E-06
	(2.003)*	(1.046)
URBAN1	0.28438E-01	
	(1.568)	
RURAL1	-0.27118E-02	
	(-0.310)	
DIV1		-0.17540E-01
		(-0.660)
DIV4		-0.76098E-01
		(-3.033)**
POPN4	0.23338	0.44648
	(3.775)**	(5.374)**
POPN42	-0.20390	-0.41791
	(-3.139)**	(-4.995)**
NGARAGE	0.40991E-01	0.41299E-01
	(8.043)**	(8.120)**
RICH14		-0.71371E-03
		(-2.078)*
DEVELOP		0.59072E-14
		(2.473)*
CONSTANT	9.2333	9.3438
	(82.710)**	(79.943)**
R^2	0.77	0.77
F	255.82	260.83
OBSERVATIONS	4326	4326

t-statistics in parenthesis; * and ** denote significance levels at the 0.1 and 0.01 levels respectively.

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Exploring and Expanding the Landscape Values Terrain

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Exploring and Expanding the Land Amenity Values Terrain

Introduction

Land has many, multifaceted values to people. Throughout history, the values provided to people by land and its relative scarcity has resulted in minor and major competitions for the possession, use, and management of land. The struggle for survival leads to competition for land as an input for producing the basic necessities of life such as food, shelter and clothing. Conflicts over land to provide basic necessities for survival have occurred over and over again in past civilizations. These type of conflicts over land can be observed today in many underdeveloped parts of the world.

In most developed parts of the world, market forces are doing an adequate job of allocating land to the production of life's necessities in a nonviolent manner. In agricultural land markets for example, market generated prices for food and fiber products play a pivotal role in determining how much land will be allocated to, say, corn and cotton production. Similarly, the demand for and supply of land for housing are captured in market prices and transactions which, in the absence of some type(s) of government restrictions, determine how much land in a region will be allocated to housing. Since VonThunen, economists have been quite familiar with and accepting of the market-driven allocation of land to different uses according to its "highest valued use."

If market prices and transactions capture and reflect all relevant values of land, allocation of land to its "highest valued use" via market forces should be automatic assuming all other necessary assumptions hold. There are good theoretical reasons to believe that land markets adequately reflect the value of land as an input into the production of economic commodities in the nature of private goods such as food, fibre, timber, and mineral products. Land, however, provides additional services beyond the role as inputs in commercial production processes which are more in the nature of public goods. Certain types of land amenity values fall into this category. Economic theory suggests that the value of public good land services will not be adequately reflected in land markets. Other types of land amenity values will likely escape land markets altogether because of their incommensurable or intangible nature.

Economists typically do not venture outside of the commensurable land value terrain into the incommensurable or intangible land value terrain. Yet, as Crosson (1985) argued over 10 years ago, resolution of competing demands and interests in the use and allocation of land requires that the full scope of land values be taken into consideration. This theme is being repeated often in contemporary times, particularly in developed countries such as the U.S. where land use conflicts increasingly are centered on the amenity values of land, rather than value of land as an input into commercial economic production.

In the next section, some background discussion on why we as a society seem to care so much about land values is provided. Following this discussion, a framework for empirical assessment of these values focusing on amenity values is presented. Application of these values in the rural policy arena is then discussed. Some concluding thoughts are provided in the final section.

Land or Landscape Values: Why Do We Care?

The struggle over defining the values of land to individuals and society in nothing new in America. For example, Thomas Jefferson believed strongly that close ties with the land provided "character building" values that would result in the type of independent, moral, and productive citizens needed to support a growing, free democracy. In another corner was Alexander Hamilton who did not completely share these sentiments with Jefferson. Hamilton viewed land values more narrowly as he focused on the use of land as an input into the economic engine needed to drive the new country. These differing perspectives on the value and importance of land to individuals and society, as well as differences of opinion of other issues, led Jefferson and Hamilton to advocate different and perhaps competing visions for the development of America (Crosson, 1985; Hite and Dillman, 1981).

The Hamiltonian view of land values apparently was more in line with a young, developing America. Throughout most of American history to date, including the history of economic thought, the predominant view of land has been as an input into commercial productive process just like any other input (Crosson, 1985). In the world of economics, this traditional view of land values has been passed along from one generation of economists to the next primarily through the neoclassical production function. Using the neoclassical production function, one of the first lessons introductory economics instructors teach their charges is that land along with labor and capital provide the big three inputs for production of commercially valuable economic commodities. Values of land other than its marginal productivity in the production of food and fibre products, widgets, or some other commercial commodity are typically not discussed in microeconomic theory courses.

The traditional view of land as a commercial input has contributed to a historical emphasis in rural areas and rural/agricultural policy on "productivism." Productivism as applied to land implies a commitment to intense, industrial, and expansionist use of land supported by public policy primarily to increase productivity and commercial output. In the rural U.S. at least up to the mid 20th century, productivism has been the guiding force behind land use and policy (Lowe et al, 1993; Reed and Gill, 1997). The results of productivism can be seen throughout the rural landscape - agricultural operations producing an abundance of food and fibre products, private industrial forests where trees for lumber products are grown with utmost technical precision, and large dams and reservoirs built to provide electricity to fuel all manners of industrial production and output.

Productivism has served America well, helping to transform it in a relatively short historical time period from a struggling, developing nation to a highly developed "super power" with unprecedented standards of living. In a developing nation, necessities of life such as food, shelter, and clothing are often critically scarce. As a result, there is high individual and societal demand for increased outputs of these necessities with a corresponding high derived demand for the use of land as a commercial input. In this context, productivism naturally emerges as a means for organizing and utilizing land and other inputs to deal with the scarcity of life's necessities.

As a nation develops, two fundamental changes occur that put pressure on the established emphasis on productivistic uses of land in rural areas. First, productivism helps to mitigate the scarcity problem related to life's necessities to the point that it is no longer a major national concern. In the United States and many European nations, for example, an equally troubling concern for individuals and society are the large surpluses of food and fibre produced by the agricultural industry. Second, as a nation develops, demand for noncommercial land values such as amenity values tend to increase at a relatively greater rate than demand for more food and fibre production. The reason from economic theory for these different demand changes is that because they are in the nature of luxuries, the income elasticity of demand for noncommercial land values is greater than the income elasticity of demand for necessities such as food and fibre (Bromley and Hodge, 1990).

Economists, rural sociologists, geographers and other observers of changes in rural development patterns agree that many rural areas are moving into a postproductivism era. This postproductivism era is characterized by more diverse economic activities and attitudes with respect to the use and allocation of land. A key characteristic of a rural area experiencing postproductivism is a migration into the area of new residents attracted by landscape amenities, and increased visitation by nonresidents seeking recreational and leisure opportunities supported by landscape amenities. In addition to "on-site" benefits of landscape amenities to rural area residents and visitors, postproductivism in a rural area also receives support from "off-site" beneficiaries of landscape amenities. These off-site beneficiaries include people in urban, suburban, and exurban areas who enjoy cleaner air and water supported by the countryside (Flynn and Marsden, 1995; Lowe et al, 1993; Reed and Gill, 1997; Troughton, 1996).

Many people in rural areas of the United States remain geared toward productivism and represent the traditional "stakeholders" in rural policy. Accordingly, many local, state and federal institutions involved in rural development and policy continue to lean towards productivism. The new stakeholders in rural development and policy are the residents and nonresidents of rural areas who are more in the postproductivism camp. Institutions that serve the interests of the postproductivism stakeholders are not well-developed. The presence of these different sets of stakeholders and the lack of institutions set up to handle and mediate the interests of both groups sets the stage for land value and property rights conflicts in need of resolution (Bromley and Hodge, 1990; Reed and Gill, 1997).

Land or Landscape Values: What are They?

When two groups do not see eye-to-eye on an issue, a first step towards an acceptable solution is for each group to have a better understanding of what is important or valuable to the other group. What are the various types of land or landscape values that may be of importance to productivists, postproductivists, or both? To address this question, the full scope of land values is discussed in this section with an emphasis on amenity values.

The National Agricultural Lands Study published in 1979 was indicative of a growing national interest and concern in the loss of agricultural land and associated values of this land. The following quote from the National Agricultural Lands Study suggests the broad scope of agricultural land values of concern:

> "As prime farmland disappears, food is not our only loss. The quality of our lives is diminished. There are garish signs and glaring storefronts where leaves once caught the rain and filtered the sunlight. There is asphalt where fields and woods once beckoned and refreshed the spirit. There is the loss, also, of farm family life, and the values that spring from living close to the land" (Fields, NALS, 1979).

How many different types of values can you pick out from the above quote? Several fundamental types of land values are embodied in this quote which have been discussed in more detail by a number of authors in articles published since the late 1970s, particularly in the early 1980s, but continuing to today.

An early influential paper on the subject of agricultural land preservation and values was written by Bruce Gardner and published in the *American Journal of Agricultural Economics* in 1977. Gardner (1977) delineated four major types of values provided jointly by agricultural land:

1. local and national food production; 2. provision of local jobs in the agricultural sector; 3. better and more organized development of urban and rural land; and 4. environmental amenities. Crosson (1985) provides further elaboration on agricultural land values in an article appropriately entitled, "Agricultural Land: A Question of Values." In this article, he first discusses the market values of agricultural land starting with the tremendous market value of food and fibre products produced on agricultural land. He also highlights the considerable employment benefits provided by jobs in the agricultural sector. Both Gardner (1977) and Crosson (1985) argue that private land markets adequately allocate agricultural land to the production of food and fibre products and the associated support of jobs in the agricultural sector.

Crosson (1985) also points out the market value of agricultural land for development purposes. In the absence of development subsidies, agricultural land is converted to residential, commercial, and industrial uses whenever the market value of the land in nonagricultural uses is higher than the market value of the land in agricultural uses. In the spirit of VonThunen, both Gardner (1977) and Crosson (1985) argue that private land markets adequately value and reallocate agricultural land from agricultural to nonagricultural uses.

Gardner's (1977) fourth category of agricultural land values, environmental amenities, was defined broadly as open space and other general amenities of agricultural land of an environmental and public good nature. Crosson (1985) included visual amenities provided by open space as a type of intangible value of agricultural land. Other intangible values of agricultural land according to Crosson (1985) include wildlife habitat, "character building" values gleaned from rural life, and the value of a "sense of community" promoted by farming life. Gardner (1977) and Crosson (1985) agree that unlike market values derived from food and fibre production, employment, and development, the externality nature of the above amenity-type values means that private land markets are not likely to adequately allocate land to the support of these values.

A number of studies conducted mostly in the 1980s attempted to quantify the amenity-type values of agricultural land. Amenity-type values were defined somewhat differently by each study. Halstead (1984) referred generally to the "nonmarket value" of agricultural land including wildlife habitat, scenic vistas, and recreation. Bergstrom et al (1985) defined environmental amenities associated with agricultural land to include scenic value and the environmental qualities of agricultural land which generate nostalgic value. Nostalgic value is related to the virtues people since Jefferson ascribe to living close to the land. Beasley et al. (1986) defined amenity values to include scenic values and historical values of agricultural land. Bowker and Didychuk (1994) refer to the "external benefits" of agricultural land which they define to include open space, scenic vistas, wildlife and traditional country life.

Rosenberger and Walsh (1997) define three categories of amenity-type values which they classify as nonmarket values of agricultural land; open space values, environmental amenities, and cultural heritage. Open space values include visual, recreational and therapeutic benefits. They define environmental amenities to include watershed protection, soil conservation, plant and animal habitat, and the biological diversity supported by these amenities. Cultural heritage value is defined as the value of agricultural land as part of the unique cultural or natural heritage or history of an area (Rosenberger and Walsh, 1997).

Landscape amenities have also been a topic of considerable interest in the rural development, land planning, and environmental planning fields. Duffy-Deno (1997) and Reed and

Gill (1997), for example, discuss the role of landscape amenities such and scenic beauty and open-space in attracting new residents and recreational visitors to rural areas. As with the economic valuation studies mentioned in the previous paragraph, a major concern in the land use and environmental planning literature is with the value of undeveloped land on the urban-rural fringe (e.g., greenbelts). Amenity-type values identified in the rural development and planning literature include recreational values, open space, scenic beauty, symbolic values, environmental quality, and the shaping and containment of urban sprawl (Correll et al., 1978; Duffy-Deno, 1997; Lee and Fujita, 1997; Lee and Linneman, 1998; Reed and Gill, 1997).

Philosophers, working primarily in the area of environmental ethics, have identified and discussed values associated with nature then can be applied to land and landscapes. A general dichotomy of land values suggested by philosophers are *instrumental values* and *intrinsic values* (Ferre, 1988). Instrumental values of land are derived from the active or passive use of land to support or generate services which are useful or valuable to people, plants, animals, and ecological systems as a whole. Intrinsic values of land are the values of land which are independent of active or passive use by some other entity.

The source of intrinsic values is a rather deep ethical, philosophical, and theological question. Some schools of ethical/philosophical/theological thought identify the source of intrinsic values of land as the land itself. For example, Aldo Leopold's "land ethic" outlined and discussed in his book, *The Sand County Almanac,* suggests that land values include the values of land elements such as plants and animals to themselves. Leopold's "land ethic" is an important foundation for modern schools of ethical thought which hold to the inherent value of the biotic and abiotic elements of land and landscapes to themselves including biocentrism and ecocentrism

(Oelschlaeger, 1991). Consistent with biocentrism, the *biocentric intrinsic value* of land refers here to the value of living land elements to themselves. Consistent with ecocentrism, *ecocentric intrinsic land value* refers here to the value of living and nonliving land elements to themselves and to the land or landscape ecosystem as a whole.

Other schools of ethical/philosophical/theological thought identify the source of intrinsic values as God or other spiritual beings or entities. Judaism, Christianity, and Islam teach that God created the land and everything else in the universe. This creation has an inherent value to the creator which is apparent in common Judeo-Christian land and nature scriptures found in the Old Testament of the Bible such as Psalm 104. Consistent with the idea of the inherent value of land an nature derived from its creator, *theistic intrinsic land value* refers here to the value of living and nonliving land elements to God independent of active or passive use by anyone or anything else on earth.

Buddism, Hinduism, and parts of Native American spiritualism teach that various types of spiritual beings inhabit the land and its elements. The presence of these spiritual beings provide value to land that is not dependent on active or passive use by anyone or anything else on earth. Thus arise, for example, "Sacred Groves" which are preserved by Hindus for the benefit of the spiritual beings that are believed to inhabit the grove of trees. The inherent value of living and nonliving land elements derived from a multitude of spiritual beings or entities is referred to here as *pantheistic intrinsic land values.*

Unlike intrinsic values, instrumental values of a particular land element are derived from the active or passive use of this element to generate, produce, or support some good or service useful and valuable to people or some other living or nonliving land element. Instrumental values can be divided up into noneconomic and economic instrumental values. Noneconomic instrumental values associated with land include *biocentric instrumental land value* and *ecocentric instrumental land value.* Biocentric instrumental land value is the value of land elements to plants and animals. For example, different types of soil have instrumental value to particular plants as source of nutrients needed for life. Ecocentric instrumental land value is the value of land elements to all living and nonliving components of land ecosystems. For example, ecocentric instrumental land value would include the value of soil as a foundation for surface rock formations. The argument in philosophy and ecology circles is that the function of soil within the land ecosystem as a foundation for rock formations has value outside and independent of human activities which give rise to economic values.

Biocentric and ecocentric instrumental land values focus on the noneconomic instrumental values of land elements to biotic and abiotic components of land ecosystems. Noneconomic instrumental land values also include certain types of *anthropocentric instrumental land values.* Anthropocentric instrumental land values are derived from the active or passive use of land elements to generate, produce, or support goods or services of value to people. Philosophers such as Holmes Rolston and economists including Crosson, and Hite and Dillman identify a number of landscape values that can be classified as noneconomic anthropocentric instrumental land values. These include at least particular types of aesthetic values, historical values, cultural values, security and stability values, mental health values, physical health values, and spiritual health or religious values (Crosson, 1985; Hite and Dillman, 1981; Rolston, 1985).

Economists and philosophers have also identified a host of economic anthropocentric instrumental land values. Because economic values are always dependent on human preferences, the terms "economic" and "anthropocentric" in the label "anthropocentric instrumental land value" are redundant. We can therefore shorten the label somewhat to economic instrumental land values. Economic instrumental land values include a number of *active use values* including material consumption value, recreational use value, on-site scenic appreciation value, and commensurable mental, physical, and spiritual health values involving on-site use. Economic instrumental land values also include a number of *passive use values* including commensurable existence values, historical values, cultural values, job satisfaction values, security and stability values, off-site aesthetic appreciation values, off-site recreation and leisure values, and mental, physical, and spiritual health involving off-site passive use (Crosson, 1985; Hite and Dillman, 1981; Kline and Wichelns, 1996; Rolston, 1985; Rosenberger and Walsh, 1997). The various values of land or landscapes from an interdisciplinary perspective are summarized in Figure 1.

Land or Landscape Function and Value Linkages

How can we organize all of the various notions of land and landscape values to facilitate policy analysis and decisions? To accomplish this organization, it is useful to think of a particular land area or landscape as an asset with various *functions* as illustrated in Figure 2. In the productivism tradition, the focus of rural land and landscape values in rural policy analysis and decisions has been on the use of rural land and landscape elements as commercial inputs. The use of rural land and landscape elements as commercial inputs includes using soil and water resources as inputs into the production of commercial agriculture, forestry, mineral, and manufactured products. For the most part, rural development policy in the United States has historically emphasized the goal of maximizing the use of rural land and landscape elements as commercial inputs (Bromley and Hodge, 1990; Reed and Gill, 1997).

Land and landscapes also function as "places" which support what philosophers and sociologists refer to as "values of place" (Norton). For residents, land or landscapes function as place to live and work. In a rural area, these residents include "long-time" residents who work locally in traditional jobs in the agricultural, natural resource extraction, and manufacturing sectors and "new" residents who work in local or nonlocal nontraditional jobs in the recreation and tourism, "high technology," business service sectors, or are retired and living off of transfer payments from pension funds, retirement accounts, and other nonlocal sources of income.

Land and landscapes in rural areas also provide a place to visit. In many rural areas of the country, recreation and tourism catering to nonresident visitors is a booming business. Most of this recreation and tourism is nature-based - e.g., hunting and fishing, camping, hiking, boating, lake and river swimming, water skiing, off-road touring, snow skiing, and snowmobiling. Agricultural-based tourism such as visiting "dude ranches" has been an established business activity in many parts of the country and is taking hold in other areas of the country.

Another broad function of land and landscapes especially in rural areas in the provision of "space." Space here is defined from a human interaction perspective, as in the phrase "you're in my space." Specifically, space refers here to the physical distance between people as they engage in various life activities (e.g., work, play) and the interrelated frequency of interaction between people as they engage in these activities. One of the apparent reasons people enjoy visiting and living in rural areas is that rural land and landscapes provide them with more space.

The provision of flora and fauna habitat is often identified as an important function of land and landscapes by philosophers, ethicist, economists, ecologists, biologists, and other social and physical scientists. In recent years, the preservation of rural land and landscapes as habitat for endangered plant and animal species has been a contentious rural policy issue. Heated debate between and among residents and nonresidents of the Pacific Northwest over the preservation of "old growth" forest landscapes to provide habitat for the endangered spotted owl is a familiar and obvious example.

Another function of land and landscapes is provision of unique physical terrain. Physical terrain includes mountains, rolling hills, gorges, valleys, plains, marshes, and beaches. Use and management of physical terrain features may also be a controversial area of rural policy at certain times and regions in the United States. Clashes may arise, for instance, between and among residents and nonresidents of rural areas over the preservation and management of unique physical terrain features. Debates in both the eastern and western United States over mining practices (e.g., strip mining) that temporarily or permanently alter the physical terrain and appearance of a rural landscapes are cases in point.

A major function of land and landscapes is provision of a natural water supply system. With respect to water quantity, land and landscapes provide both a flow and stock of surface and subsurface water supplies through watershed run-off into rivers and lakes, and the seepage of surface water into subsurface aquifers. With respect to water quality, land and landscape elements (e.g., plants, soil) help to filter out chemicals in surface and subsurface water supplies which are potentially harmful to human, plant, and animal health. The function of rural land and landscapes as a natural water supply system is an especially important issue from a rural and urban development policy perspective.

The land or landscape functions shown in Figure 2 support the various land or landscape values discussed previously and listed in Figure 1. The function of land and landscapes of providing commercial inputs primarily supports the value people derive from consuming commercial goods, or material consumption value. The function of land and landscapes as a place to work also supports material consumption value as well as job satisfaction value, and security and stability value. The function of land and landscapes as a place to live supports job satisfaction value, security and stability value, cultural value, historical value, recreation and leisure use value, aesthetic appreciation value, and mental, physical and spiritual health values. The function of land and landscapes as a place to visit supports cultural value, historical value, recreation and leisure use value, aesthetic appreciation value, and mental, physical and spiritual health values.

The function of rural land and landscapes of providing "space," more specifically "open space," support recreation and leisure use value, aesthetic appreciation value, existence values, intrinsic values, biocentric and ecocentric instrumental values, and mental, physical and spiritual health values. The functions of providing flora and fauna habitat, unique physical terrain, and a natural water supply system arguably have an important role in supporting all of the values shown in Figure 2.

Commodity and Amenity Values

In rural policy and development, "commodities," "commodity interests," and "commodity values" are frequently used terms. In these cases, "commodity" takes on a more specific meaning than the use of the term in economic theory to refer to goods and services in general. When government agencies such as the U.S.D.A. and land-grant university administrators talk about commodity values, they are referring primarily to values associated with the production and consumption of "private good" commercial products using land as a commercial input including food and fiber products, timber products, and mineral products. In Figure 2, commodity values would include material consumption value, and some portions of job satisfaction value and security and stability value.

Which of the land values shown in Figure 2 can be classified as amenity values of land? Land amenity values of are defined here to be the direct benefits people receive from the sights, sounds, smells, and presence of the land or landscape around them. A key part of this definition is that amenity benefits are derived directly from the land or landscape and not from consumption of commercial products produced using the land as a commercial input. As a commercial input, land provides indirect benefits to people through the consumption of final commercial products such as food, fiber, and timber products. Thus, in Figure 2, material consumption value would not generally fit the definition of land amenity value.

Another key component of the above definition of land amenity values is that these values accrue directly to people - e.g., they are anthropocentric values. Thus, intrinsic values, biocentric instrumental values, and ecocentric instrumental values as defined previously would not generally be classified as land amenity values. All of the other values shown in Figure 1 expect for material consumption value, intrinsic values, biocentric instrumental values, and ecocentric instrumental values have significant land amenity value components. In the land value literature, cultural value, historical value, recreation use value, aesthetic appreciation value, existence value, job satisfaction value, security and stability value, mental health value, physical health value, and spiritual health value have been discussed in the context of land amenity values (Beasley et al., 1986; Bergstrom, et al., 1985; Bowker and Didychuk, 1994; Crosson, 1985; Halstead et al., 1992; Hite and Dillman, 1981; Kline and Wichelns, 1996; Rolston, 1985; Rosenberger and Walsh, 1997).

Some amenity value components are captured in market prices and transactions while other components are not. For example, amenity values associated with recreation use may be captured at least partially in prices landowners charge people to lease their land for consumption and(or) nonconsumptive recreational activities. Also, amenity values associated with aesthetic appreciation may be captured at least partially in the price of rural land sold for residential purposes.

Many land amenity values, however, are in the nature of nonmarket values meaning that they are not reflected in market transactions and prices. The nonmarket nature of certain land values results from characteristics of nonrivalry and(or) nonexclusiveness. The extent of nonrivalness in the consumption of land values is dependent on congestion levels. Figure 3 classifies the land values shown in Figure 2 according to the degree of rivalness and exlusiveness under conditions of low human congestion. In this case, the bulk of land amenity values fall into the "nonrival, nonexlusive" cell and the "nonrival, exclusive" cell. Primarily because of the nonexlusive characteristic, values in the "nonrival, nonexclusive" cell are in the nature of nonmarket values. An example is the value people derive from viewing open landscapes from public, uncongested highways. Values or goods in the "nonrival, nonexlusive" cell are known commonly as pure public values or goods.

Because of they can be made exclusive, the values in the "nonrival, exclusive" cell can potentially be "privatized" and captured in market trade and prices. For example, at least sections of a large farm, woodland area, or ranch can be closed-off from public access or view. The aesthetic appreciation values derived from viewing these private areas becomes a type of private good or value. Specifically, as long as human congestion is low, values in the "nonrival, exlusive" cell may be classified as uncongested private goods. The benefits provided by uncongested private goods may be capitalized into the market value of the land sold for residential and(or) recreation and tourism purposes.

Figure 4 classifies the land values shown in Figure 2 according to the degree of rivalness and exclusiveness under high human congestion. High human congestion occurs, for example, when more and more people travel public highways or move into the countryside to enjoy open-access land amenities such as aesthetic appreciation values. Eventually, congestion sets in causing values in the "nonrival, nonexlusive" cell to move into "rival, nonexclusive" or the "rival, exclusive" cell. In the "rival, nonexclusive" cell, values are still available on a nonexclusive basis, but because of congestion people can no longer enjoy land amenity values on a nonrival basis. For example, public lands such as National Forests are open for many types of recreational activities on essentially a nonexclusionary basis (expect for obtaining necessary nonrationed license). However, in many parts of the United States, National Forests are so congested during certain times of the year that recreational use on these public land becomes a rival activity. Conflicts between hikers, mountain bikers, horseback riders, and off-road vehicle touring who often attempt to share the same trails or roads is a case in point.

In the same way, at a certain congestion level, the values in the "nonrival, exlusive" cell that were formally available on a nonrival basis will become rival. At this point, many of the values in the "nonrival, exlusive" cell will move into the "rival, exclusive" cell. The "rival, exclusive" cell contains pure private goods or values. An example in the land amenity area are private, exlusive qual or pheasant hunting preserves in the South. The quantity and quality of qual or pheasant on these preserves available for hunting are carefully regulated. Access by a limited number of hunters is strictly enforced and is very expensive.

Figure 5 lists potential techniques for valuing different types of public and private goods. These techniques can be applied to measure land amenity values categorized by degree of rivalness and exclusivness. Land amenity values in the nature of pure private goods can be valued using traditional market price valuation techniques. The economic impacts of expenditures associated with these values can be measured using economic impact analysis techniques such as input-output analysis. The noneconomic social effects of these values can be assessed using various types of social effects or impact assessment. Land amenity values in the nature of uncongested private goods can potentially also be valued using market price valuation techniques. Economic impact analysis and social effects assessment can be used to assess economic and social impacts associated with uncongested private goods and values.

Because of the lack of market prices, nonmarket valuation techniques must be employed to measure the economic value of land amenity values in the pure public good cell. The travel cost method may potentially be used, for example, to quantify public nonconsumptive recreation use values derived from countryside landscapes. It may be possible to use the hedonic price method to quanity nonrival, nonexclusive aesthetic appreciation values which are capitalized into the value of land sold for residential and(or) recreation and tourism purposes. The contingent valuation method can potentially be used to quantify the commensurable portions of all land amenity values in the nature of pure public goods. To the extent enjoyment of pure public good land amenity values involves actual expenditures, the economic impacts of these expenditures can be measured using economic impact analysis. Noneconomic social benefits derived from enjoying pure public good land amenity values can be assessed using social effects or impact assessment (Bartik, 1988; Cheshire and Sheppard, 1995; Correll et al., 1978; Garrod and Willis, 1992; Lee and Fujita, 1997a; Lee and Fujita, 1997b; Lee and Linneman, 1998; Rosenberger and Walsh, 1997; Young and Allen, 1986).

Congested public goods in the “rival, nonexclusive” cell will not generally have market prices. Land amenity values in the nature of congested public goods must therefore be measured using nonmarket valuation techniques. For example, the travel cost method may be used to measure the economic value of nonexlusive recreational use in a congested National Forest. If congested public good land amenity values are capitalized into the value of rural land, the hedonic price method can potentially be used to quantify these land amenity values. The contingent valuation method can potentially be used to measure all congested public good land amenity values. As with pure public good values, if actual expenditures are associated with congested public good land amenity values, the economic impacts of these expenditures can be measured using economic impact analysis. Noneconomic social effects can be measured using social effects or impact assessement techniques (Bartik, 1988; Cheshire and Sheppard, 1995; Correll et al., 1978; Garrod and Willis, 1992; Lee and Fujita, 1997a; Lee and Fujita, 1997b; Lee and Linneman, 1998; Rosenberger and Walsh, 1997; Young and Allen, 1986).

Aggregate Values by Landscape Type

The aggregate land value for a particular landscape is the sum of the different land values shown in Figure 2 for that landscape. The magnitude of aggregate land value and the portion of aggregate land value represented by different types of amenity and nonamenity values will vary across landscapes. Consider first an urbanized landscape characterized by high human development and congestion. In this landscape, as shown in Figure 6, aggregate land value is dominated by pure private good values and congested public good values. Land amenity values in the form of pure public goods and uncongested private goods are relatively sparse in this landscape.

Aggregate land value in the suburban landscape (Figure 6) is also dominated by pure private goods and congested public good values. Although more of the pure private goods values may be in form of private amenity values, most of the pure private good value is made up of material consumption value. Congested public good values include, for instance, the use of congested public parks and other open areas for recreation. These areas will not likely be as congested as similar areas in the urbanized landscape, but are congested nonetheless. As shown in Figure 6, land amenity values in the form of uncongested private goods and pure public goods are still relatively low on average. Suburbs on the rural fringe will have higher levels of land amenity values in the form of pure public goods and uncongested private goods as compared to suburbs on the urban fringe.

On the other extreme from an urbanized landscape, consider a frontier/natural landscape with relatively little human development. In this type of landscape, as shown in Figure 7, land amenity values in the form of pure public goods will be relatively abundant. Because of the lack of human development, pure private good values, amenity or otherwise, will be sparse. Land amenity values in the form of uncongested private goods will also be relatively abundant. There will be few congested public goods in this type of landscape.

Figure 7 depicts the mix of public and private goods in a traditional agrarian economy landscape. In this landscape, human development is evident mainly through the presence of farming and natural-resource extraction operations such as fishing, timber harvesting, and mining. Because land and other natural resources are still relatively abundant in relation to human use and congestion, land amenity values in the form of pure public goods and uncongested private goods are relatively abundant. The higher levels of commercial economic activity and human activity lead to higher levels of pure private goods and congested public goods.

A relatively new type of landscape emerging in the United States is the *exurban* landscape. The exurban landscape is an agrarian economy landscape or a frontier/natural landscape experiencing an influx of new residents from urban areas who have skipped over the suburbs to move to an area where they can enjoy the relative abundance of land amenity values in the form of pure public goods and uncongested private goods while continuing to work in jobs closely related to their urban careers. In fact, many of these people may continue to physically commute or "telecommute" to jobs headquartered in urban areas. Some may start new careers in their new rural landscape home, but in nontraditional areas such as the recreation and leisure industry, arts and crafts industry, cottage industries, or the high tech industry. The increased economic activity spurred on by exurban residents increases the level of pure private goods and congested public goods in the landscape. However, as shown in Figure 8, land amenity values in the form of pure public goods and uncongested private goods are still relatively abundant.

Figure 9 summarizes the mix of commodity values and amenity values typically found in different landscapes. Moving from an urbanized landscape to a frontier/natural landscape, public good values and land amenity values typically rise, and private good values and material consumption values typically fall. The magnitude of aggregate land values will rise and fall across landscapes according to how the sum of the different types of material consumption values and land amenity values change across landscapes. Empirical assessment of different land values would be needed to determine how aggregate land values change across landscapes.

Moving to a larger scale, a *regional landscape* is made up of a mix of various types of landscapes as shown in Figure 10. Aggregate land value for a regional landscape is the sum of commodity values and amenity values associated with each individual type of landscape within the broader regional landscape. Comparison of aggregate land value across different regional landscapes would also require empirical assessment of commodity and amenity values .

Landscape Value Planning and Management

The preceding sections indicate that there are a broad array of commodity and amenity values provided by different types of landscapes. Two relevant policy questions are: 1) What is the desired mix of value provided by a particular landscape or landscapes? and 2) How can this desired mix be achieved and maintained? Consider first a *productivist* landscape or landscapes in which commodity values are the primary values of interest. Figure 11 shows the primary beneficiaries and land management institutional representation for this type of landscape or landscapes. Primary beneficiaries of commodity values in rural areas are commodity producers, commodity consumers, landowners, and long-time residents. These beneficiaries have traditionally had strong representation in local government planning boards, local elected officials, state agricultural agencies, state resource management agencies (e.g., state forestry commission), federal agricultural agencies (e.g., ASCS), federal resource management agencies (e.g., U.S.D.A. Forest Service, Bureau of Land Management), and private commodity NGOs (e.g., Farm Bureau, other commodity associations).

When a landscape or landscapes move into postproductivism, there is demand for both landscape amenity and commodity values. The demand for amenity values, in particular, adds new residents and nonresident visitors to the list of primary beneficiaries of landscape values. The demand for amenity values also results in government agencies and NGOs who primarily represent amenity value interests becoming involved in landscape management in a rural area. Government agencies include local recreation and tourism development boards and state recreation and tourism agencies. NGOs include conservation and environmental organizations and local chambers of commerce in some areas.

A major challenge in rural areas experiencing postproductivism development is dealing with "value conflicts" between people whose interests are primarily with commodity values, and other people whose interests are primarily with amenity values. Commodity value interests are generally well-represented in various land management institutions because of the private good nature of commodity values. There is a direct incentive for commodity producers, for example, to become involved in land management issues because their income and livelihood may depend upon it. Amenity value interests may not be as well-represented in various land management institutions because of the public good nature of these values (Bromley and Hodge, 1990; Reed and Gill, 1997). A "free-rider" problem may occur, for example, because if one person or group takes on the burden and costs of becoming involved in land management institutions to protect amenity value interests, everyone in the community who enjoys nonrival, nonexclusive amenity values will benefit from these actions.

As a result of uneven representation in established land management institutions, new institutional arrangement for resolving value conflicts between commodity value and amenity value advocates in postproductivism rural areas may need to be developed. These new institutional arrangements may include more "bottom-up" organizations including citizen advisory committees, stakeholder advisory committees, local action or interest groups, round tables, and public forums. Many of the federal land management agencies such as the U.S.D.A. have increased the use of stakeholder advisory committees, round tables, and public forums in an attempt to resolve value conflicts between people who would like to maximize the use of National Forests and Rangelands for commodity values (e.g., timber harvesting, mining, grazing) and other people who would like to maximize the use of National Forests and Rangelands for recreation and other amenity values.

Summary and Conclusions

Americans appear to have a special attachment to land that spans over a broad array of concerns and interests. These broad concerns and interests lead to a multitude of interdisciplinary values that people derive from land. Land or landscapes can be thought of as assets with a number of major functions. These functions include use as a commercial input, a natural water supply system, unique physical terrain, flora a fauna habitat, space, and a place in which to live, work and visit. The functions of land or landscapes support economic and noneconomic land or landscape values ranging from material consumption value to nonuse values including existence value and intrinsic value.

Two general categories of anthropocentric land or landscape values are commodity values and amenity values. Commodity values are derived from commercial commodities produced using land as a major input including food and fiber products, timber products, mineral products and manufactured goods. Amenity values are derived directly from the land and have large nonconsumptive or passive use components. Amenity values include recreational use value, scenic appreciation value, existence value, and certain types of cultural, historical, and health values.

The classification of land or landscape values into private or public values is an important distinction to be made for empirical valuation and land policy implementation. A major determinant of the private or public good nature of land or landscape values is their degree of exclusiveness and rivalness. *Rival and exclusive* values such as material consumption value are in the nature of pure private goods. *Nonrival and nonexclusive* values such as existence value are in the nature of pure public goods. In between classifications include *nonexclusive, rival* values and *exclusive, nonrival* values. An example of a nonexclusive, rival value is public consumptive recreation use value under conditions of low or high human congestion. An example of an exclusive, nonrival value is private historical value under conditions of low or high human congestion.

Commodity values of land or landscapes fall primarily into the rival, exclusive category. Market price valuation techniques can therefore be used to quantify these values. Under conditions of high human congestion, some amenity values may fall into the rival, exclusive category. However, because of the lack of established markets for these values, market valuation techniques may not be readily applicable to these values.

Most amenity values will fall into the exclusive, nonrival category, nonexclusive, rival category, or nonrival, nonexclusive category. Revealed or stated preference nonmarket valuation techniques must be used to quantify values associated with "pure public goods" in the nonrival, nonexclusive category. Values in the rival, nonexclusive category are typically associated with "congested public goods." Because of the nonexclusive nature of these values, revealed or stated preference nonmarket valuation techniques must be used to quantify these values. Values in the exclusive, nonrival category are typically associated with "uncongested private goods." Because markets may exist for uncongested private goods, values associated with these goods perhaps can be quantified using market valuation techniques.

The economic effects of values associated with pure private goods such as commodity values can be measured using economic impact analysis techniques. To the extent that actual market expenditures are incurred to enjoy values associated with pure public goods, congested public goods, and uncongested private goods, economic impact analysis techniques can also be used to measure the effects of these values on local and regional economies. The enjoyment of amenity values of different types often involves actual market expenditures. Thus, economic impact analysis can and has been used to measure the economic effects of amenity value expenditures on local and regional economies. Social effects assessment can be used to assess the noneconomic effects of all types of land values on individuals and communities.

The distribution of land values associated with different types of private and public goods varies across landscapes. A highly urbanized landscape typically provides a high proportion of pure private good values such as commodity values and a low proportion of pure public good values including amenity values. On the other extreme, a frontier/natural landscape provides a low proportion of pure private good values such as commodity values and a high proportion of public good values such as amenity values. Landscapes in between these extremes including suburban, agrarian economy, and exurban landscapes provide more balanced mixes of private of public good values and commodity and amenity values. The aggregate value of each type of landscape must be determined on a case by case basis through empirical valuation.

A regional landscape is made of different mixes of specific landscapes including urbanized, suburban. agrarian economy, exurban, and frontier/natural landscapes. The aggregate value of each regional landscape is a function of the land values provided by each landscape and the interaction of values between landscapes in the region (e.g., substitute and complement effects). Holistic empirical valuation studies which account for value interactions between different landscapes must be conducted to determine the aggregate value of particular regional landscape.

Within a particular landscape or in a regional landscape, people residing inside or outside of that landscape will have different preferences for the current and future mix of land values; for example, commodity vs. amenity values. Productivism, which has been a traditional focus of public policy in rural areas, focuses on commodity values. Many rural areas in American are moving into a postproductivism era which focuses on both commodity and amenity values. When a rural area moves from productivism to postproductivism, value conflicts often arise between individuals and groups whose primary interests are commodity values and individuals and groups whose primary interests are amenity values. Rural institutions for handling such conflicts may not be well-established. There is a need to explore what institutions will be most effective in rural areas for moderating and solving value conflicts between people who desire different mixes of commodity and amenity values from land and landscapes.

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Figure 1. Interdisciplinary Rural Land or Landscape Values

Support of local agricultural industry

Support of local resource extraction industry

Support of local agricultural jobs

Support of local resource extraction jobs

Job satisfaction value

Support of job security and stability

Support of community security and stability

Support of national security and stability

Provision of local food supplies

Self-sufficiency in production of food items

Dispersion of food production

Continued production of unique food products

Land input for residential development

Land input for commercial development

Land input for recreational activities

Support of local tourism industry

Provision of wildlife habitat

Provision of open space

Provision of scenic views

Support of aesthetic enjoyment

Surface water storage

Ground water recharge

Natural water filtration

Support of rural life values

Provision of character building opportunities

Support of national identity/ideals

Cultural symbolization value

Historical value

Ecological life-support

Provision of genetic diversity

Intrinsic value

Existence value

Therapeutic value

Physical health value

Religious/spiritual value

Educational value

"Natural laboratory" value

Protection of cultural heritage

Nostalgic value

Environmental amenities

Countryside amenities

Promotion of orderly development

Figure 2. Land or Landscape Functions and Values

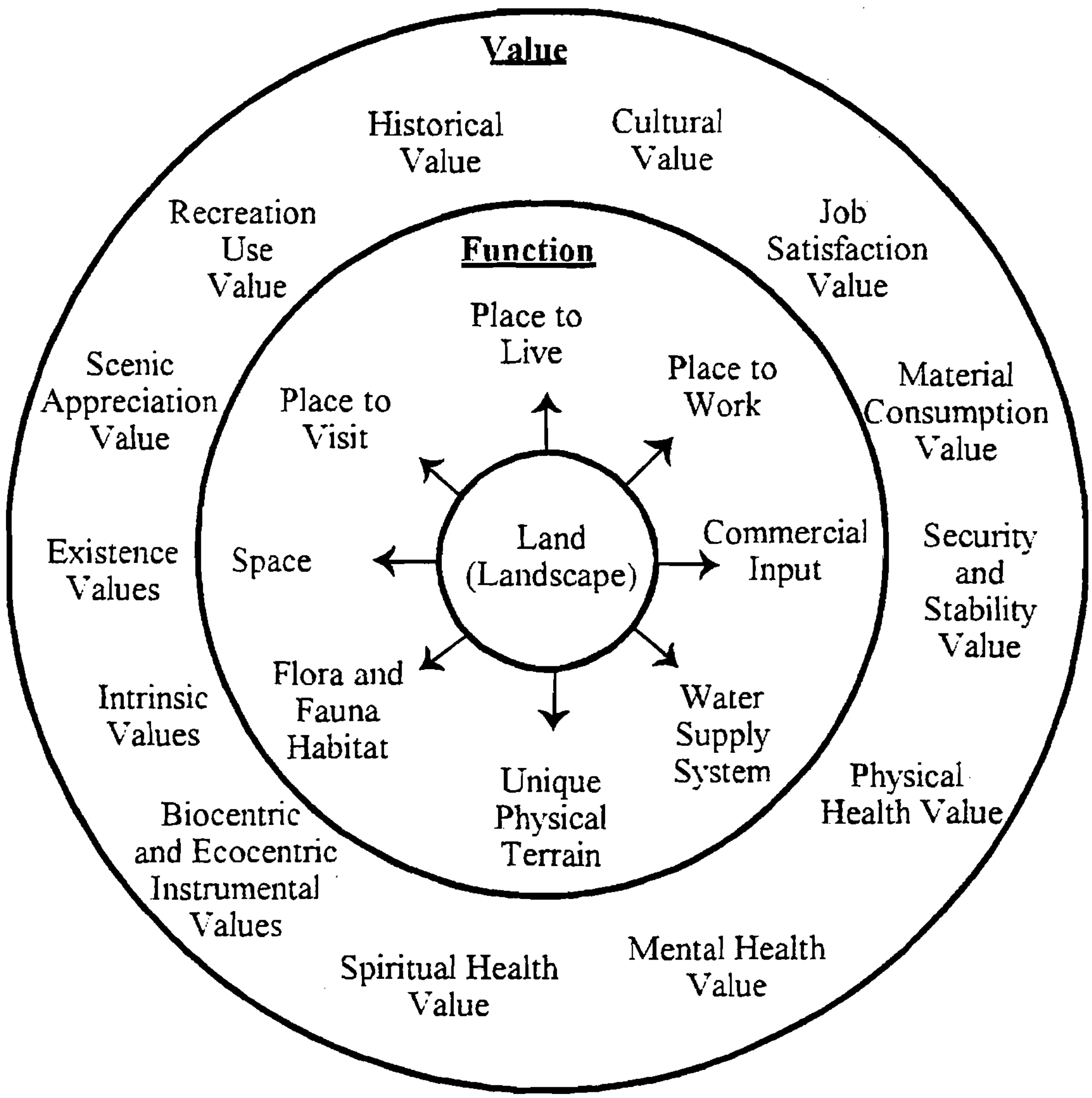


Figure 3. Landscape Value Classifications Under Low Human Congestion

	Rival	Nonrival
Exclusive	Material Consumption Private Consumption Recreation Use Individual Job Satisfaction	Private Nonconsumptive Recreation Use Private Scenic Appreciation Private Cultural Value Private Historical Value Private Security and Stability Private Physical Health Private Mental Health Private Spiritual Health
Nonexclusive	Public Consumptive Recreation Use	Public Nonconsumptive Recreation Use Public Scenic Appreciation Public Cultural Value Public Historical Value Public Security and Stability Public Physical Health Public Mental Health Public Spiritual Health Existence Value - - - - - - - - - - Biocentric Instrumental Value Ecocentric Instrumental Value Intrinsic Value

Figure 4. Landscape Value Classifications Under High Human Congestion

	Rival	Nonrival
Exclusive	Material Consumption Individual Job Satisfaction Private Consumption Recreation Use Private Nonconsumption Recreation Use Private Scenic Appreciation Private Physical Health Private Mental Health Private Spiritual Health	Private Cultural Value Private Historical Value Private Security and Stability
Nonexclusive	Public Consumptive Recreation Use Public Nonconsumption Recreation Use Public Scenic Appreciation Public Physical Health Public Mental Health Public Spiritual Health	Public Cultural Value Public Historical Value Public Security and Stability Existence Value
	Biocentric Instrumental Value Ecocentric Instrumental Value	Intrinsic Value

Figure 5. Valuation Techniques for Values Associated with Different Types of Public and Private Goods

	Rival	Nonrival
Exclusive	Pure Private Goods ("Commodities")	Uncongested Private Goods
	Market Price Valuation Techniques Economic Impact Analysis (e.g., CGE, Input-Output) Social Effects Assessment	Market Price Valuation Techniques Economic Impact Analysis (e.g., CGE, Input-Output) Social Effects Assessment
Nonexclusive	Congested Public Goods	Pure Public Goods
	Revealed-Preference Extramarket Valuation Techniques (e.g., travel cost method, hedonic price method) Stated-Preference Extramarket Valuation Techniques (e.g., contingent valuation method) Economic Impact Analysis Social Effects Assessment	Revealed-Preference Extramarket Valuation Techniques (e.g., travel cost method, hedonic price method) Stated-Preference Extramarket Valuation Techniques (e.g., contingent valuation method) Economic Impact Analysis Social Effects Assessment

Figure 6. Private and Public Good Values Associated with Urbanized and Suburban Landscapes

<u>The Urbanized Landscape</u>

Pure Private Goods	Uncongested Private Goods
Congested Public Goods	Pure Public Goods

→High relative value of individual and public amenities leads to more interest on use of remaining undeveloped land as stock and flow of amenity services.

<u>The Suburban Landscape</u>

Pure Private Goods	Uncongested Private Goods
Congested Public Goods	Pure Public Goods

→Unstable balance between commodity and amenity values - competing land value and policy interests often lead to "value struggles" on urban-suburban and rural-suburban fringes.

Figure 7. Private and Public Good Values Associated with Frontier/Natural and Agrarian Economy Landscapes.

<u>The Frontier/Natural Landscape</u>

Pure Private Goods	Uncongested Private Goods
Congested Public Goods	Pure Public Goods

→Amenity values are relatively abundant and accrue mostly to non-resident visitors. High relative value of commodity production leads to high interest on use of land as a commercial input (productivism).

<u>The Agrarian Economy Landscape</u>

Pure Private Goods	Uncongested Private Goods
Congested Public Goods	Pure Public Goods

→Economic activity is centered on agricultural and natural resource commodity production. Amenity values accrue mostly to long-time residents and occasional visitors.

Figure 8. Private and Public Good Values Associated with Exurban Landscapes

The Exurban Landscape

Pure Private Goods	Uncongested Private Goods
Congested Private Goods	Pure Public Goods

→High amenity values attract new residents and business who increase private good production in nontraditional areas including recreation and tourism, arts and crafts, cottage industries, "high tech" industries, and the business service sector via "telecommunicating."

Figure 9. Landscape Value Spectrum

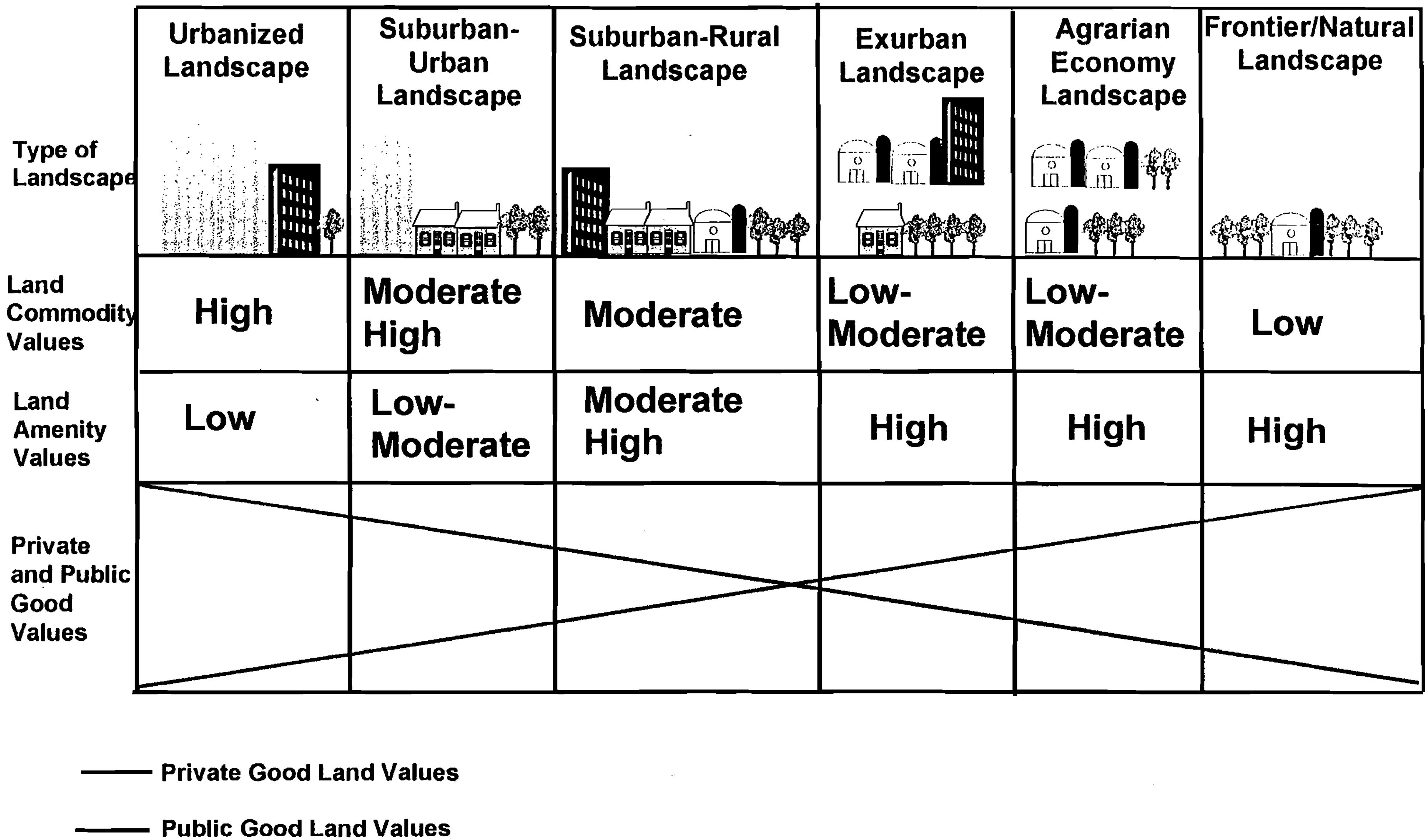
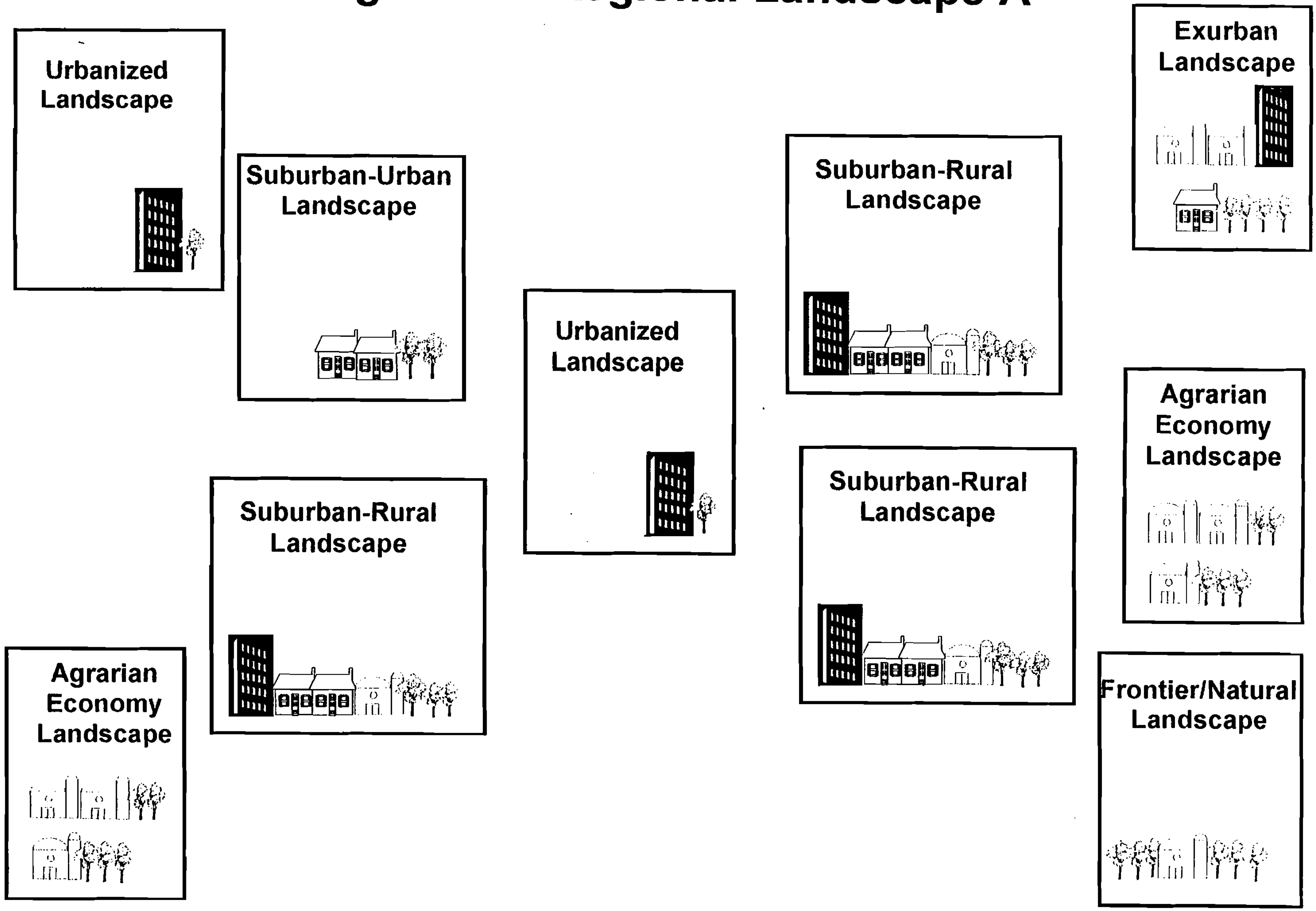


Figure 10. Regional Landscape A



Watershed Economics: Resource Valuation Issues

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Background

Many agencies and organizations that focus on natural resource and environmental management issues related to water resources now consider the watershed the appropriate level for implementation. In addition to an understanding of the environmental impacts, broadly defined, the efficient development, implementation, management, and analyses of policies and programs developed from this perspective require adequate, reliable information on the socio-economic effects that such policies and programs induce. Reliable benefit-cost analyses, for example, require that the economic consequences of all impacts of a policy or project be included.

The primary goal of these policies and programs is to affect environmental qualities and natural resource use patterns; the economic consequences of such changes are not fully reflected in standard economic measures of output, employment, and consumption. The effects of interest, commonly referred to as non-market goods and services by economists, may be significant factors in the overall economic evaluation. Recent developments in valuation methods for non-market goods can provide measures of the economic value of such consequences; information that gives valuable insights into all stages of program development, from initial conceptualization through implementation and evaluation. Such economic welfare measures can be pivotal in the development of benefit-cost analyses of alternative projects or policies.

While economic welfare measures are appropriate for the assessment of economic efficiency from a national perspective and as a guide to federal policy, there may be other concerns at the watershed level. While local communities are concerned with environmental quality, they also want to know about the anticipated impacts of any program or policy on local jobs, incomes, and economic activity as well as the distribution of such impacts among those affected. Such economic impacts affect significantly the acceptance of a specific program or policy by local groups and organizations. Economic impact analysis provides insight into the response by local communities and organizations to alternative water management alternatives that affect not only water quality and/or quantity, but the economic base of the region as well.

Introduction

The primary goal of the FETC Watershed Economics program is to provide a framework for analyzing a wide variety of programs, policies, and issues that affect water quality and quantity in terms of cost effectiveness, economic efficiency, and equity given a watershed approach to water resource management. This paper reviews the economic valuation methods available to quantify the effects of programs and polices on non-market goods and services that arise from changes in the quantity, quality, and availability of water resources. That is, the paper discusses the process of developing values and measures for the benefits side of benefit-cost analyses. An overview of current issues in non-market benefit estimation approaches and the relationship to watershed related policy development, implementation, and analysis forms the core of the paper. The paper presents an overview only. Many topics of interest will be given scant attention or left out entirely. Hopefully the information included will provide an introduction to the issues and serve as an initial guide to those who wish to explore the topics raised here in additional depth and detail. The critical overview presented by Peterson, Driver, and Gregory presents a reasonable starting point for further study. Those with more technical interest may be guided by specific references in the text that follows.

The watershed approach to water quality issues focuses public and private sector efforts on the highest priority problems within hydrologically defined geographic areas. The focus on watersheds represents the "environmental federalism" move that parallels the shift in focus from national to state to local level decision making in many public policies and programs. Such a locality based definition is a natural outcome of the attempts of regional, state, national, and international agencies to adopt more holistic approaches to assess, monitor, inventory, and manage resources. From an environmental perspective, this approach represents both: 1) a shift from dealing with single issues and point-source problems to a broader approach that adds consideration of the spatial interrelationship of non-point pollutant sources, natural ecosystems, anthropogenic forces, and both ground and surface water flows, and 2) a shift from reliance exclusively on centrally developed and mandated regulations and other command and control approaches to at least partial local control and a wide variety of alternative approaches to induce water quality improvement. Water quantity issues have focused primarily on flood control and storm damage issues, and, where appropriate, irrigation and other uses. As water use pressures have increased, water availability issues, including in-stream uses, are increasingly important.

Evaluating the economic implications of alternative programs and policies requires a broad understanding of the costs of the programs or policy impacts, the process of implementation, and economic measures of the consequences. This paper focuses on economic measures of the consequences of water quality and quantity management and use. Potential differences in value measures developed from a national perspective from those living in the watershed are discussed and potential issues addressed. The value measures to be addressed are presented and alternative approaches to defining and quantifying appropriate measures discussed. The paper also addresses the need for information on the economic impacts of the same policies and programs at the watershed level and discusses the use of such information in policy development and implementation.

Values for Watershed Management

Overarching Issues

The watershed approach to environmental management encompasses two current trends. The first, the continuing trend toward political and fiscal federalism, reflects current beliefs that the move toward central control and decision making has gone too far. The reaction to this has been to reallocate control from Washington to state and local levels as appropriate as seemed appropriate. In the environmental arena, the watershed has emerged as the consensus choice for the appropriate level for management and decision making for water quality and quantity questions. A second trend is also of potential importance to watershed economics. This is the increasing reliance on disclosure strategies as a fundamental feature of environmental management and enforcement. Disclosure strategies encompass public and/or private attempts to increase the available information on pollution to all concerned parties. Referred to at the third wave in pollution control policy (following legal regulation and market-based instruments) (Tietenberg and Wheeler), disclosure focuses on the information needs for local communities, groups, and organizations to participate effectively in the environmental management process. We will return to the effects that these trends have on the questions of valuation.

A major issue in benefit-cost analysis involves measuring the benefits of goods and services, called nonmarket goods, that are not provided by markets but are provided by nature, such as water quality, water-based recreation, and scenic beauty. Economists have generally accepted methods of valuing changes in the quality of goods sold in markets based on actual consumer purchases. For nonmarket goods, such as those generally provided by water and watersheds, economists have to rely on methods that attempt to infer willingness to pay (WTP) either from observed purchases of related market goods (called revealed-preference methods), or through survey techniques such as contingent valuation.

Issues of Nesting and Spatial Extent

Watersheds vary in size from several hundred acres for small streams to the Mississippi River basin. A watershed can be defined for any point in a stream and, by definition, watersheds of smaller size are nested within the watershed boundary defined for any downstream point. It is also possible to designate a subwatershed that includes the drainage area between two stream points. While not all inclusive, this concept of a watershed may focus attention on areas of interest and concentrate resources on the primary issues. In general, the watershed approach does not attempt to define the appropriate size or scope of the watershed for analysis but takes the definition and size as given. The definition is left as a function of the problem and area of concern. In any particular situation, the size of the watershed will be an important determining factor in the types of problems to be addressed and the methods of analysis used to determine the economic and social consequences of alternative actions.

One issue that arises is the relationship between the definition of the watershed and the spatial pattern of benefits and costs. While the watershed approach is ideal for focusing on the extent of the physical phenomena related to water issues, the socio-economic factors are not necessarily constrained by the same boundaries. For example, it has long been recognized that an individual's value for unique natural resources like Yellowstone Park and the Grand Canyon were not necessarily constrained by the distance of the individuals home from the resource nor necessarily enhanced by proximity. Recent studies indicate that the values we place on many other natural resources may be similar. Papers by Loomis (1996) and Klocek and Fletcher indicate that the relationship between value and distance is much more tenuous than originally thought.

Values – Definitions and Descriptions

The paths by which the quantity and/or quality of natural resources or environmental amenities affect an individual's welfare can be divided into three categories:

1. indirectly as a factor input into market goods or services that yield utility,
2. as an input into the household production framework of goods or services that yield utility, and
3. those that yield utility directly as an argument in an individual's utility function.

We concentrate on 2 and 3 in this paper; 1 is a topic for the paper on Cost Methods for Watershed Economics.

Economists have developed a number of categories of values that individuals might have for environmental amenities. This includes values that are not related to the use of the resource, a concept that deserves attention. What does it mean, however, to have values not related to use? Prior to discussing approaches to empirical estimation of individual values, it is appropriate to outline the concepts that must underlie such procedures.

For a starting point, we take as given the concept that the values an individual holds are determined by personal preferences. We are interested in the total value that an individual places on a change in the quality or quantity, q, of a resource; water is one example. Consider two levels of q -- an initial level, q^0, and a level after the change, q^1. Consider also the maximum level of welfare or utility the individual can obtain in the initial state, u^0, given the prices for market goods, $\mathbf{P}$, (assumed not to vary with respect to the change in q) and a predetermined level of income. We can now define the value of the change in q, called the compensating surplus

(CS). This is the difference in the money an individual must pay to obtain the utility level in the initial state (called the expenditure function) less the money that he must pay to obtain the same level of utility or welfare in the changed state. This is the amount of money that makes the individual indifferent between the two levels of the environmental resource. This difference is positive if the increase in the utility derived from the resource change allows the individual to be just as happy spending less on goods and services. The difference is negative if the change in the resource injures the individual so that additional expenditures are necessary to obtain the initial level of utility at the new level of resource quantity or quality (Freeman 1993a, 1993b). That is,

$$\mathrm{CS} = e(\mathbf{P}, q^0, u^0) - e(\mathbf{P}, q^1, u^0)$$

We make no argument that this measure of value is necessarily "reasonable" or, in all cases, capable of measurement. The cost of environmental degradation may be considered immeasurably large by some individuals. It does, however, provide an explicit definition of what we mean by value -- an economic concept that can be measured in money terms.

The measure of value can be divided into values associated with the use of the resource, $\mathrm{CS_U}$; values that do not depend on use, $\mathrm{CS_{NU}}$; and existence values that one can hold for a particular resource with or without use, $\mathrm{CS_E}$. Freeman (1993b) presents a technical discussion that supports this division and explains the relatively esoteric situations under which certain combinations can or cannot arise. An essential element for this discussion is that developing estimates of the various measures of value require different measurement methods and approaches.

There is an additional issue that has been raised with regard to values for environmental services – the possibility that altruism is a significant factor in local decisions related to environmental quality and natural resource management. Altruism captures social values for environmental goods that do not appear to rise from factors related to personal use but rather from factors that affect other segments of society. Papers by Shabman and Stephenson and Norton, Phipps, and Fletcher indicate that altruistic motives can be significant factors in the apparent values individuals place on various measures of q. While altruism can be conceptually encompassed in the outline of values provided, the measurement issues may differ from those commonly used. The importance of such motivations in public actions remains a point of contention and a potential focus of future research.

Valuing Natural Amenities and Environmental Quality

The economics literature encompasses literally thousands of studies that develop demand estimates and values for natural amenities and environmental qualities using both nonmarket and market data. The two primary approaches to empirical measurement are commonly referred to as direct and indirect valuation methods (Smith, Desvousges and Fisher 1986). Direct methods generate demand or willingness-to-pay (WTP) relationships through the use of surveys, experiments, or voting results. Indirect methods rely on observable data on market transactions including individual decisions and actions made in conjunction with the expenditures on market goods to infer information about the demand for and value of related public goods. Natural resource based amenities and environmental qualities are two classes of public goods of special interest to watershed economics.

Such nonmarket goods are particularly difficult to value because they often generate both use and nonuse values. For example, a lake may generate several types of use values: recreational uses such as swimming, fishing, and boating. It may also provide drinking water, another type of use value. The lake may also generate nonuse values if some members of society value the existence of the lake whether or not they ever visit or use the lake. This could be the case if the lake provides habitat for an endangered or other wild species. These nonuse values need to be included in a benefit-cost study but are extremely difficult to measure.

Indirect Methods

Indirect or revealed preference methods use observed market behavior to value nonmarket goods. This approach relies on stable relationships between the market goods and services that all of us deal with on a regular basis and the natural resource or environmental amenity based goods or service of interest. If such stable relationships exist and the values of the two types of goods are inextricably linked, the value of the non-market good can be determined by information on the consumption patterns of the market good. One important method, the travel cost method, uses the costs of travel to recreational sites as a proxy for the price of the recreational good. The travel cost method can be used to infer the use value of a nonmarket good but cannot be used to infer nonuse values. Another revealed preference method, the hedonic method, tries to link the price paid for, say, a house to an environmental good such as proximity to a lake or air quality. The hedonic method works best when the nonmarket good can only be utilized if the house is purchased, otherwise the true value of the nonmarket good is understated. Loomis and Walsh provide an excellent introduction to the basics of valuation methods and many additional references to more in-depth analyses.

Travel Cost Models

The simple travel cost model has been used for over forty years to value recreation sites. First suggested by Harold Hotelling in a letter to the US National Park Service, the principal idea is that the value of a site can be derived from observations on the number of visitors to the side combined with information on costs of visitation. The approach presumes that visitors will make repeated trips until the amount of benefit they gain from the last trip is just worth what they must pay to visit. Choosing to take a trip is similar to any market transaction except that the price of the trip includes the travel cost expended as well as the entrance fee, if any, rather than the money one pays for items at the local supermarket. Just as for any consumption item, as the amount consumed increases the value of an additional unit falls. Suppose, for example, that an individual fisherman makes multiple trips to a given site. While the first few trips may be worth substantially more than the last trip, the final trip must still be at least worth the travel cost.

If all trips were worth just the travel cost, then the site would have no value. If the site were not there, the people would still have the money they would have spent on the trips which they could allocate to other goods and services and remain as well off as with the site. The value of the site lies in a difference between the value of the trip and the cost of visiting.

The travel cost approach captures this logic in a formal model that relates visitation rates to travel costs. By observing how often people visit from different places (distances), one can infer the value of the site. Since recreation sites are a type of public good so that many people can value the site jointly, the value of the site is the sum of the individual values of all visitors.

Simple travel cost models can be implemented with relative ease. Suppose a site maintains a guest book at all times and that all visitors to the site sign the book and include their home address. This information is sufficient to estimate a simple travel cost model and develop an initial estimate of the value of the site. While such as approach is crude and fraught with oversimplifying assumptions, the point is clear – values can be developed with reasonable ease.

The simple travel cost model as outlined has been expanded in many ways in the years since its inception. The multiple site travel cost model extends the analysis to multiple sites. An initial approach is to include the travel cost (price) of alternate destinations in the travel cost demand equation. A more complex arrangement is to develop simultaneous estimates of a system of sites. While this is intuitively appealing, especially since it includes the possibility of explicitly considering the effects of alternative site characteristics, the increase in complexity and information required expands the analysis significantly. The issue of how much is enough also arises. Since the users of a site are generally a spatially diverse group, the alternative sites for two individuals who visit the site but come from opposite directions is unlikely to be the same.

The generalized travel cost method extends the simple model to encompass the value of site characteristics. Many of the recent innovations to the travel cost model concentrate on including more complex measures of site quality, environmental considerations, or other aspects of particular interest to owners, managers, and policymakers. For example, methods to assess the value of water quality changes have been a prime motivation for many innovations. As more complex questions are considered, the data required and model complexity increase. It seems clear, however, that for many of the issues that may be addressed as part of an economic analysis to guide watershed level decisions, the travel cost model provides significant insights into appropriate decisions by watershed organizations, by local governments, and by private industry. Smith and Desvousges provide an excellent example of using travel cost methods to develop measures of water quality for the Monongahela River. Fletcher, Adamowicz, and Graham-Tomasi provide an overview of some of the perplexing issues in travel cost models.

Hedonic Models

Hedonic models are based on the simple observation that many goods are not homogeneous and that people make a particular purchase choice based on the observed characteristics of individual items. While originally develop to control for quality differences among goods (Griliches), the hedonic price method has become a tool to measure the value of specific attributes. It is possible, for example, to decompose the price of housing into prices for individual characteristics such as the number of bedrooms, the number of bathrooms, the total square footage, and so on. It is a short jump to extend such analyses to external characteristics such as the local public services, schools, shopping, transportation, and so on. Within the environmental literature, this includes many aspects of environmental quality.

Early, applications of the hedonic model concentrated on goods that were identical except for a limited number of attributes. For many goods of interest, however, there is a vector of attributes that vary across individual items. The movement has been toward models of greater statistical complexity to enable the analyst to disentangle the effects of numerous factors with specific concentration on characteristics of specific interest.

The hedonic approach is not without problems, however. There are serious issues as to whether changes in characteristics can be controlled for appropriately through the statistical model. Since this is an empirical question with little theoretical guidance, the appropriate functional form is not known. An issue is the trade-off in precision in explaining the market price of the good versus good explanatory power for the effect of alternative characteristics. Many goods of particular interest for this approach, such as housing, are characterized by a large number of characteristics that vary significantly among individual offerings. As the number of explanatory variables increase, in many applications the effect of decreasing degrees of freedom set in; the ability to understand the effect of any single attribute is diluted. If significant attributes are left out and these variables are correlated with attributes of interest, the hedonic price equations for the attributes of interest may be significantly biased. Another issue deals with the non-marginal changes in characteristics. Many economic analyses, including most statistical implementations of such analyses, assume smooth relationships. If this is not the case, significant under or overestimates can result.

While many applications in the environmental literature have considered air quality due to its ubiquitous characteristic (e.g., Kim, Phipps, and Anselin), water quality, flood pressures, and similar watershed issues are of specific interest at the watersheds level. In any case, care must be taken that only the appropriate environmental characteristics are included. If water quality has been poor in a given area but the fact was not generally known, one would not expect to see direct effects on market prices. It is only when the information is known and freely available that we expect to see environmental characteristics seriously impact housing, land, or related prices.

Hedonic Travel Cost Models

The hedonic travel cost model, like the generalized travel cost model, focuses on the characteristics of sites, sites that form the set of alternatives available from a specified origin. Like the hedonic model from which it derives its name, the hedonic travel cost model assumes that choices can be explained by variations in characteristics of the site. Assuming that a site can be represented as a package of attributes or characteristics, individuals are assumed to select among sites depending on their personal preferences for characteristics and the cost (distance) of the alternatives. The set of alternatives must be sufficiently dense to support changes made on marginal differences in attributes. While appealing, applications have been relatively unsuccessful. The primary issue seems to be that the set of choice are sufficiently rich to disentangle the effects of alternative preferences over characteristics.

Random Utility Models

Random utility models of recreation demand focus on the choices individuals make. Given assumptions about how recreation choices are made, readily estimable models of recreation demands for several sites can be derived. These models admit zero consumption for some sites, allow substitution possibilities across sites, incorporate site qualities, and are consistent with utility maximization behavior. They can be estimated using data only on the last trip taken without implying that all trips are taken to the same site. From an environmental perspective, this approach provides significant insight into the issues environmental quality, flows, or similar characteristics.

The major assumption is that trips are taken independently of one another which implies a fairly restrictive separability of preferences across trips. The second assumption is that trip decisions are made one at a time rather than all at the beginning of the season as is the case with the standard travel cost model. On each choice occasion, the choice is assumed to have a stochastic element to it; this is the random utility assumption. Two interpretations are possible: that the choices of individuals are deterministic but that there are stochastic elements of these from the perspective of the researcher, or that the utility of the individuals themselves has a random component.

There are two positive things to note about this approach. First, in the context of a recreation demand model, variables to be incorporated into the utility function include travel costs and characteristics of sites that make them attractive to individuals. Thus, the attractiveness of a site relative to other sites governs the probability of a visit on a choice occasion. This ready incorporation of choice among multiple sites is a key feature of the model. Second, since one has estimated the parameters of a utility function, it is a fairly straightforward matter to compute welfare estimates for changes in site characteristics (see Hanemann 1982).

Direct Methods

As the name states, direct methods derive value estimates from direct statements of value; statements in answer to a question (contingent valuation), as reaction to a hypothetical situation (experimental economics), or in the voting booth (referenda). Survey techniques, such as contingent valuation, are used when the value of the nonmarket good cannot be inferred from observed market behavior and there are no appropriate questions of public will to be answered via the ballot box. Such is the case for certain unique ecosystems, such as the Grand Canyon, and for all nonuse values. The contingent valuation method (CVM) uses a combination of survey and bidding techniques to infer consumers' WTP for nonmarket goods.

Contingent Valuation

The contingent valuation approach uses surveys to elicit individual WTP for public goods, natural resource amenities, or environmental services. Successfully applied, such a technique yields a compensated valuation measure of consumer surplus for each individual which can be aggregated over the population of interest to yield a market demand for the public good of interest. A CVM application includes at least four components (Portney 1994).

1. A survey is designed that will as the respondents to value a hypothetical scenario which may be a policy or the outcome of a specified action,
2. The survey must contain a mechanism to capture the respondent's value with respect to the scenario defined (this can include a change in tax rate, a charge to a utility bill, etc.; elicitation methods include open ended questions, a choice from a list of values, or a yes/no response to a stated amount similar to the voting response one gives to a referendum or the decision to purchase or not purchase a good at a specified price),
3. Questions on the socio-economic characteristics of the respondents, and
4. The surveys must be administered to minimize bias.

Once the basic information is obtained, statistical techniques are used to develop appropriate measures of individual and population values. CVM studies have contributed significantly to the improvement of survey techniques and the statistical analyses of categorical and referenda data.

While the CVM has evolved greatly over the last decade (due in part for the need to estimate the environmental damages resulting from the Exxon Valdez oil spill), controversies remain. A series of recent papers in the *Journal of Economic Perspectives* published by the American Economics Association outlines the debate on CVM (Portney 1994), and presents the pros (Hanemann 1994) and cons (Diamond and Hausman 1994). The National Oceanic and Atmospheric Agency (NOAA) in the Department of Commerce put together a blue ribbon panel of Nobel Laureates and other distinguished economists to consider the issue. The developed a list of recommendations for appropriate implementation of CVM studies (Arrow et al. 1993). The research focus since that time has been to test the implications of these recommendations through empirical analyses.

The issues are significant. Much of the controversy results from attempts to apply dollar values to goods that have a strong ethical component such as preserving endangered species or protecting lands of cultural significance. The recommended practice in such cases is to attempt to quantify the benefits and costs of the market and nonmarket goods and services provided by the policy and to list and describe those elements of the policy, such as ethical issues, that cannot be quantified. Decision-makers can then develop a final decision or evaluate a benefit-cost analysis in light of such non-economic factors. For many questions of interest, direct elicitation of values is appropriate and desirable.

Many of the issues to be addressed within the watershed framework are less controversial. The primary goal is to develop measures that summarize the economic and community welfare implications of environmental change to guide decision-making. A description of the benefits to be gained from specific action plans can provide a basis for community support and galvanize local action. To the extent that benefits extend beyond the watershed boundaries, appropriate documentation can support the case for revenue sharing from state and federal sources.

The CVM is a developing method of analysis; the watershed framework can be expected to impose unanticipated restrictions and requirements on valuation methods. Thus, applications of CVM within a watershed framework can be expected to raise research issues. For example, one issue not yet resolved is the appropriate extent of the market, i.e., how does one define the appropriate population for nonmarket goods, especially goods with significant nonuse components in the total value.

Benefits Transfer

While resource valuation is an essential component of any benefit-cost analysis, studies based on the CVM or other primary analysis are expensive, particularly when they involve extensive surveys. While a benefit cost analysis of the alternatives for each watershed is needed for informed decision making, an agency or local organization could easily exhaust its budget on analysis and have nothing left for program implementation. For such cases, an alternative to primary studies is necessary. The approach is to take what has been learned of costs and values for similar situations and apply this knowledge to a specific study area. Termed benefit transfer, this is a method for extrapolating the results of valuation studies from one region or watershed to another.

Methods similar to those now considered benefit transfer have been a standard approach in recreation analyses for many years (Loomis and Walsh). Agencies rely on various measures of recreation use including the recreation day or the recreation visitor day to summarize use. Specific studies at a variety of locations provide values for each measure of use. Careful consideration of the results of such studies provides the basis for developing values for the specified measure of recreation use. The analysis results in a series of unit day values for a wide spectrum of activities. With this background, studies of value can be changed to studies of use which are more easily understood and simpler to implement. This approach provides a simple means of developing use values for existing facilities or anticipated use values for new or proposed facilities.

Current applications in benefit transfer are based on a similar set of assumptions concerning comparability among areas or regions. The approach is to develop a consistent set of measures including both socio-economic characteristics of the population and physical characteristics of the natural resource or environmental amenity for areas where primary studies were developed and for areas where the similar questions exist but primary studies have not been done. The approach is to use statistical models to apply the relationships developed in one region to another while correcting for differences in characteristics. This is an evolving area with significant promise for watershed studies. A well-planned and implemented series of primary studies should provide the basis for many groups and organizations to understand the economic implications of specific local concerns. Visitor Day Equivalency Analysis (VDEA), an approach proposed by Kealy, Rockel, and Tomasi, combines recent developments in nonmarket valuation for environmental goods with the traditional visitor day approach in an attempt to simplify value estimation for smaller environmental projects.

Summary

The watershed approach to the management of water quantity and quality issues is expected to be an increasingly important component in the implementation of environmental policy to meet societal goals in the US as we move into the 21st century. To attain the success possible, it is important that information on the benefits of alternative actions be available in appropriate form to all those involved -- watershed organizations, citizen groups, policymakers, business interests, and so on. The valuation methods discussed in this paper can provide at least some of the information necessary to fill this need. Valuation methods are an essential tool in the development of an integrated approach to the environmental management of watersheds.

A series of studies to develop baseline information on the values of environmental benefits from local management input of selected watersheds could provide valuable baseline data to guide the development of general features of watershed management. This information could also serve as the basis for appropriate benefit transfer applications to other watersheds to provide initial estimates of the potential returns to alternative action plans.

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Valuing Non-indigenous Species Control and Native Species Restoration in Lake Huron

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Abstract: Sea Lamprey are a non-indigenous aquatic nuisance specie that prey on lake trout – a key native specie in the Great Lakes. Lamprey induced mortality is partially credited with the collapse of lake trout populations in the Great Lakes in the late 1940s. Ongoing lamprey control efforts have permitted the recovery of lake trout stocks in Lake Superior and parts of Lake Michigan. However, rehabilitation of lake trout in Lake Huron has been hampered by large populations of lamprey originating from the St. Marys River. This research estimates some of the economic benefits and costs associated with several new options for controlling sea lamprey in the St. Marys River. All treatment options are shown to have positive net present value, even though only part of the economic benefits have been measured. The paper also highlights areas where further research might reduce some of the uncertainties associated with the present analysis.

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Introduction[1]

Numerous non-indigenous aquatic species pose threats to the native species in the Great Lakes (Mills et al). One of the best know invaders is the sea lamprey. Sea lamprey prey on lake trout – a key native specie in the Great Lakes. Earlier research has shown the benefits of sea lamprey control in the Great Lakes outweighs the costs (Talhelm and Bishop). We use a model of the demand for recreational fishing in Michigan to estimate some of the economic benefits associated with lamprey control in Lake Huron. Specifically, we estimate benefits that accrue to Michigan's recreational anglers as a result of lake trout recovery scenarios that are linked to lamprey management options on the St. Marys River. These benefits are then compared to the costs of the alternative management options available for the St. Marys. Since no attempt has been made to document all possible economic benefits, we refer to this as a "partial" benefit-cost analysis. Even though not all the benefits are quantified here, the results provide important evidence about the benefits that readers should bear in mind. First, assuming the scenario descriptions being valued are reasonable characterizations of the effects of lamprey treatments, the estimated benefits presented here serve as a lower bound on the total benefits since many important potential benefits have not been quantified. Second, the benefits to Michigan anglers are likely a major portion of the benefits associated with changes in lamprey control. Finally, the results suggest that all of the lamprey treatment options yield substantial economic benefits to Michigan anglers, and the portion of benefits that are estimated here exceed the program costs.

Sea Lamprey and Lake Trout. As mentioned, the sea lamprey is a non-indigenous aquatic nuisance specie in the Great Lakes.[2] The lamprey likely made its way into the Great Lakes following the 1829 construction of the Welland Canal around Niagara Falls. Sea lamprey prey on lake trout and other species of

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[2] The material in this section is drawn the St. Marys River Control Task Force report (see "SMRCS" in the references), and from the following fact sheets prepared by the Great Lakes Fishery Commission: Fact sheet 1, "The Great Lakes Fishery Commission: History, Structure, and Mandate," Fact sheet 3, "Sea Lamprey: A Great Lakes Invader," and Fact sheet 9, "International Sea Lamprey Management on the St. Marys River."

Great Lakes fish. Lamprey are credited, along with over-fishing, for the collapse of the lake trout populations in the Great Lakes. The presence of lamprey in the Great Lakes led to the creation of the Great Lakes Fishery Commission (GLFC) which is jointly funded by Canada and the United States. The GLFC oversees lamprey control in the Great Lakes.

Ongoing chemical and barrier lamprey control efforts have successfully reduced populations of lamprey in Lake Superior and most of Lake Michigan. This has allowed for the restoration of lake trout populations in Lake Superior and some more limited success in Lake Michigan. Efforts to achieve restoration of lake trout on Lake Huron and northern Lake Michigan have been hampered by the large numbers of lamprey that spawn in the St. Marys River, the channel connecting Lake Superior and Lake Huron. The sea lamprey population in northern Lake Huron is estimated to be larger than in all of the other Great Lakes combined (SMRCS). The primary means of controlling lamprey is by treating streams in the Great Lakes basin with the lampricide TFM (3-trifluoromethyl-4-nitrophenol). TFM kills larval lamprey before they can migrate to the Great Lakes. However, due to the flow volume and depth of the St. Marys River, TFM treatment would require substantial funds and would be of reduced effectiveness. This has led the GLFC to search for other potential control options for the St. Marys River. Were it not for the difficulties associated with the treatment of the St. Marys River, it is estimated that sea lamprey abundance in Lake Huron would be about 50,000 (approximately the levels of Lakes Superior and Michigan) rather than 400,000 (SMRCS). The large number of lamprey in northern Lake Huron (and Lake Michigan) coincide with vast areas of critical spawning habitat for lake trout. Increasing lake trout populations in the critical spawning areas in northern Lake Huron is crucial for achieving self sustaining stocks of lake trout – a goal laid out in the Fish-Community Objectives for Lake Huron (DesJardine et al).

Three St. Marys River Treatment Options. In this research, we examine three recently refined sea lamprey control options for the St. Marys River. The three options consist of combinations of two treatments: sterile male release and trapping (SMRT) and granular bayer applications (GB). The sterile male release and trapping program involves the trapping of lamprey, the sterilization of the males, and the release of the sterile males. Granular bayer is a chemical treatment that is effective in killing larval lamprey. Spot treatments with the bottom toxicant granular bayer do not appear to cause significant mortality in non-target organisms (SMRCS). GB is produced in a granular form so that it can sink to the river bed where the larval lamprey are located. GB is applied by helicopters to larval lamprey "hot spots" identified based on a mapping and sampling of lamprey spawning areas in the river. Uncertainty associated with the long run effectiveness of SMRT is thought to be larger than the uncertainty associated with the long run effectiveness of granular bayer. One reason for this increased uncertainty is the possibility of enhanced growth and reduced mortality of larval lamprey at lower spawning rates (a compensatory response).

In sum, the three sea lamprey treatment options considered in this analysis are as follows: The first option is ongoing sterile male release (SMRT only). The second option includes ongoing sterile male release along with applications of granular bayer every five years (SMRT + GB). The third option includes ongoing sterile male release along with a *one-time* application of granular bayer (SMRT + GB 1.x). In terms of cost, granular bayer is much more expensive than sterile male release. Applications of granular bayer cost just under 5 million dollars (US) per application. The sterile male release and trapping program costs about three hundred thousand dollars a year.

Recreation Demand Model for Michigan

A repeated-random utility travel cost model of recreational fishing in Michigan is used to estimate the economic benefits to recreational anglers in Michigan of increases in lake trout populations in Lake Huron. The travel cost method is widely used to estimate the use-values associated with recreational activities (see Freeman or Bockstael et al, 1991 for reviews of the travel cost method). Travel cost methods that are based on random utility models (RUM)[3] are well suited to estimating recreation demand when there are numerous substitute sites and can be used to value of changes in the quality of recreational fishing sites. In a repeated

[3] For general texts on the RUM, see Train or Ben-Akiva and Lerman. For early applications of the RUM to recreation site choices, see Bockstael *et al.* (1984) and Bockstael *et al.* (1989). Recent applications include Feather *et al.*, Hausman *et al.*, and Shaw and Jakas, while Jones and Sung represents an earlier application of the RUM to fishing in Michigan.

RUM such as the Michigan model,[4] the season is divided into a series of choice occasions in which anglers decide whether to take a trip, and if so, where to fish. In the Michigan model, all other fishing and non-fishing activities are reflected in the "don't go" alternative.

The data describing where and how often anglers go fishing in Michigan was collected in an extensive telephone panel survey that followed anglers during the course of the 1994-95 fishing year. The panel members were recruited from the general population of Michigan residents to ensure that the results would be representative of the general population. Computer assisted telephone interviewing was used to streamline all interviews and improve response accuracy. Additional techniques to ensure response accuracy included the use of the following: a large pilot survey, fishing logs as memory aides, bounded recall to avoid double counting of trips across panel interviews, and providing multiple opportunities to revise trip counts. To balance the need to collect timely and accurate data against the burden of the interviews, frequent anglers were called more often than infrequent anglers -- panel interview frequencies ranged from eight interviews for the most avid anglers to three interviews for the least avid anglers. The model and data used here draws on the work of previous research documented in Hoehn *et al.*

Here, the survey data is used in two stages. In the first stage, fishing location choices are modeled using the survey data for anglers who took a fishing trip to the Great Lakes and fished for trout or salmon. In the second stage, the number of Great Lakes trout and salmon fishing trips is modeled. The second stage estimates the propensity of all the anglers in the panel to participate in Great Lakes trout and salmon fishing trips, i.e., the go fishing/don't go fishing level. There are 1902 potential anglers in the panel data sample; 1080 of these took some type of fishing trips in 1994 during the April to October open-water fishing season. Of these participants, 90 individuals took Great Lake trout and salmon trips for a total of 312 trips. Of these trips, 70 are multiple day trips and 242 are single day trips. There are 9 choice occasions per month from April to October.

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[4] For applications of the repeated-RUM, see Morey *et al.*, 1991 and 1993, and Chen *et al.* Morey provides a thorough review of repeated RUM models in the context of modeling seasonal recreation demand and site choices.

The fishing sites are characterized by their travel costs and catch rates. Travel costs are defined as the sum of driving costs, lodging costs, and time costs. Driving costs are the round trip travel distance multiplied by the estimated per mile driving cost for each sample member. Time costs are defined as each individual's estimated time costs multiplied by the travel time for each trip. The individual specific time cost and driving cost regressions, as well as the lodging cost calculations are documented in detail in Appendix 1 of Hoehn et al. Each site is also described by its catch rate for the following species: salmon, lake trout, and other trout (other trout includes rainbow and brown trout). These catch rates are specific to each county and vary on a monthly basis from April to October. These catch rates are based on an analysis of the Michigan creel survey party interview data (described in the next section). The spatial and temporal variation in the catch rates reflects seasonal differences across sites in the abundance of salmon and trout.

Destination sites (fishing locations) are defined by the stretch of Great Lake shoreline within a Michigan county that offers opportunities to catch Great Lakes trout and salmon. While there are 41 Michigan counties that border the Great Lakes, not all of these provide access to trout and salmon fishing. For example, the Michigan counties bordering Lake Erie, Lake St. Clair and some of the Saginaw Bay counties are excluded because the warmer water does not provided substantive opportunities to catch Great Lakes trout and salmon.[5] Single day trips and multiple day trips to any of these sites are treated as distinct alternatives in the RUM choice sets. Feasible sites that enter anglers' choice sets for single day trips include the Great Lake counties in Michigan within 150 miles of an individuals permanent residence. For multiple day trips, all 35 sites enter the choice set of each individual.

The repeated RUM that is estimated here is specified as a nested logit with the participation level nested above the site choice level (see McFadden or Morey for details on nested logit). In the nested logit, the

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[5] The following Great Lake counties were not included in the analysis: Monroe, Wayne, Macomb, Tuscola, Bay, and Cheboygan. With the exception of Cheboygan, these are warm-water areas where trout or salmon fishing is essentially non-existent (these counties had no more than a handful of trout or salmon anglers in ten years of creel survey data). Cheboygan County was not included because no angler in the sample fished there and because in 10 years of creel survey data there were no trout or salmon observations from Cheboygan Co. However, unlike the other excluded counties, it is possible to catch trout or salmon from the waters off Cheboygan.

probability of selecting a site conditional on taking a trip is given by

$$Prob(j|go) = \pi_{j|go} = \frac{\exp(\beta X_j)}{\sum_j \exp(\beta X_j)} \tag{1}$$

where go refers to taking a trip, j refers to the possible sites, X_j is a vector of characteristics describing the sites, and β is a vector of parameters to be estimated. X_j will include site characteristics such as travel costs and catch rates. The index βX_j is referred to as the indirect utility of taking a trip to site j. The relative value of the elements of the estimated β are estimates of anglers preference for different site characteristics.

The probability that an angler chooses to take a trip on any given occasion is given by

$$Prob(go) = \pi_{go} = \frac{\exp(\theta IV + \gamma Z)}{1 + \exp(\theta IV + \gamma Z)} \tag{2}$$

where Z is a vector of angler characteristics, γ is a vector of parameters to be estimated, IV stands for inclusive value and θ is the parameter on the inclusive value. The IV is a summary index that describes the utility of the recreation site choices, and it is given by $IV = \sum_j \exp(\beta X_j)$.

The use of the inclusive value as a variable is a way of introducing potential correlation in the error terms associated with sites. If θ, the estimated parameter on IV, is less than one, then the estimates suggest that the indirect utilities associated with the alternative fishing sites are more correlated with one another than they are with the "don't go fishing" alternative (McFadden). The IV formula is also used in the calculation of the economic value (benefits or costs) associated with any changes in the site characteristics, X_j (Small and Rosen; McFadden; Morey). Even though the present model differs, the procedures for calculating economic benefits and extrapolating these to the Michigan population are the same as those described in Lupi et al, 1998 or Lupi and Hoehn, 1998.

Catch Rate Modeling

Part of the research effort was devoted to updating the Great Lake trout and salmon catch rate estimates so that the recreational demand model could be re-estimated using catch rate estimates more in line with the 1994 survey data on anglers' fishing site choices. Specifically, the Hoehn *et al* version of the recreational fishing model is based on Great Lake catch rates that were estimated by MDNR personnel using data from the mid to late 1980s. The catch rates vary by site, species, and month. However, since the angler survey data is from 1994, it is possible that those catch rates do not reflect the status for the Great Lakes fisheries in the year that anglers made their fishing site choices. Because of potentially important changes in these fisheries, we went back to the raw creel survey data and re-estimated the catch rates to include more recent years. The catch rate estimation is documented in Lupi, Hoehn, and Jester.

Negative binomial regression models were used to estimate species-specific catch-per-hour for recreational anglers fishing for trout and salmon in Michigan waters of the Great Lakes. Dependent variables were observations on catch and hours fished for angler parties interviewed in Michigan creel surveys from 1986 to 1995. The estimated models relate catch rates to independent variables for year, month, and fishing location. Interactions between months and locations are included to permit a rich array of spatial and temporal variation in estimated catch rates. Additional variables control for charter boat use, angler party size, and extent of species targeting (e.g., fishing for "salmon" versus "chinook"). Separate models are estimated for nine combinations of species and Great Lakes. The nine catch rate models range in size and include from 35 to 110 explanatory variables and from 5,000 to 50,000 observations. The estimation results indicate significant relationships between catch rates and most independent variables, including large positive effects for charter boats and targeting, positive but declining effects for increases in fishing party size, and significant spatial and temporal differences.

By utilizing the annual data, the catch rate modeling approach provides predictions of the 1994 catch rates that are specific to specie targeted, lake, site, month, and year -- even for combinations of specie, site, and month where any one year might contain few observations. The estimates of catch rates for 1994 serve as independent variables describing sites in the recreational fishing model. The complete set of estimated catch rates for all species and lakes are given in Lupi et al.

Estimated RUM Parameters

The nested-logit recreational fishing model was estimated sequentially by applying maximum likelihood techniques to the site choice and participation levels of the model. The choice probability functions used at the two stages of estimation are given above by equation (1) for the site choice level and equation (2) for the participation level. As shown in Table 1, the estimated parameters on the travel cost variables are negative. The estimated parameters on the catch rate variables are positive. Notice that the travel cost parameter for multiple day trips is lower than the travel cost parameter for single day trips, and the catch rate parameter for multiple day trips is larger than for single day trips. This means that catch rates are relatively more important and travel costs are relatively less important determinants of where anglers take multiple day trips than they are for single day trips. This suggests that any changes in catch rates will be more valuable for anglers taking a multiple day trip than for anglers taking a single day trip.

Table 1 also presents the estimated parameters on the Lake Superior and Lake Michigan constants for both single and multiple day trips. The Lake Superior and Lake Michigan constants for the single and multiple day trips are dummy variables that take the value of 1 if a site lies on the lake and a value of 0 otherwise. Including these constants in the model assures that, on average, the estimated model will predict that the share of trips the each Great Lake will match the shares in the survey data.

The third part of the table presents the participation level results. The estimated inclusive value parameter is significantly less than one indicating that the nested logit is a significant improvement of the multinomial logit formulation. Roughly speaking, the inclusive value parameter estimate implies that the Great Lakes trout and salmon fishing sites are closer substitutes for each other than they are to the "don't go" alternative. This suggests that, relative to an un-nested version of the model, the total number of Great lakes trout and salmon fishing trips will be less responsive to changes in fishing quality than will be the allocation of trips across sites. In addition to the inclusive value parameter, Table 1 also presents several other parameter estimates for variables that entered the model at the participation level. Males, older individuals, and more educated individuals are more likely to take Great lakes trout and salmon fishing trips. Conversely, individuals with more adults or more children living in their household are less likely to take great Lakes trout and salmon fishing trips (though the effect of adults is not significantly different than zero at conventional levels of significance). In addition, individuals who do not have a paying job are less likely to take Great Lakes trout and salmon fishing trips.

Re-estimation of the recreational angling demand model using the updated catch rates revealed some interesting results. Recall that the catch rate models reported in Lupi, Hoehn, and Jester provide estimates for catch rates specific to three specie groups: lake trout, salmon, and other trout. The combined catch rate variable used in the model presented in Table 1 was derived by taking the sum of these three catch rates at each site in each month. That is, the catch rate for trout and salmon at site j in time t is given by

$$CR_{j,t}^{T+S} = CR_{j,t}^{lake\ trt} + CR_{j,t}^{salmon} + CR_{j,t}^{other\ trt} \tag{4}$$

where the subscript j,t represents site j at time t, and the superscripts represent the specie groups with T+S meaning "trout and salmon." Several preliminary models were estimated using the three separate catch rate variables, one for each of the specie groups. A general finding after estimating under a variety of model specifications was that the parameters on the catch rates for individual species were fairly unstable and were often insignificant. Some specifications resulted in the lake trout parameter being insignificant and sometimes even negative, while other specifications resulted in the salmon catch rate being very low and insignificant. Interestingly, in almost all specifications examined, we could not reject the restriction that all species of Great Lakes trout and salmon had the same parameter. One difficulty that bears on this result is that the specie-specific catch rates are significantly correlated with one another which complicates attempts to identify their separate effects. A second difficulty relates to the correlation between species-specific catch rates and the lake specific constants. Another explanation may be that anglers who are targeting a specific species may not care about the catch rates of other species when they make their site choices.

Table 1: Estimated Model Parameters.

Single day trip, site choice level		
variables	**parameter**	**t-stat**
Travel cost/100	-5.70	-16.6
Catch rate	1.89	2.51
Lake Superior constant	1.04	1.36
Lake Michigan constant	1.89	4.99

Multiple day trip, site choice level		
variables	**parameter**	**t-stat**
Travel cost/100	-0.81	-5.77
Catch rate	4.60	5.19
Lake Superior constant	0.15	0.27
Lake Michigan constant	1.37	4.19
Trip constant	-5.89	-10.0

Participation Level		
variables	**parameter**	**t-stat**
Inclusive value	0.17	1.93
Participation constant	-17.2	-8.00
Male	1.56	6.57
Ln(age)	1.74	5.34
Ln(education)	1.81	3.16
Adults in hhd.	-0.11	-1.21
Children in hhd.	-0.20	-2.78
No job	-0.71	-3.47

Log likelihood values at site choice level, -510; and at participation level, -1493.

Figure 1: Steps required to link treatment options to economic value.

```
                 Step                 Step                Step               Step
                  1                    2                   3                  4

Treatment          Changes in           Changes in          Changes            Benefits
options     -->     lamprey      -->    lake trout   -->    in catch    -->    to Mi.
                   population           population          rates              anglers

     linkages between treatment                       assumed                from the
| <-- options, lamprey & lake trout  --> |<-- proportional -->|<--  economic   -->|
     from Sitar, SMRCS, & GLFC                                        model
```

The implication of the result that the catch rate variables have the same parameter is that each of the species is equally important to anglers and equally valuable. Put differently, it means that when making a Great Lakes trout and salmon fishing site choice, anglers prefer high catch rates, and there were not significant differences in this preference among trout and salmon species. This result has potentially important implications for the current analysis as well as for any future analyses of anglers preferences regarding fish-community objectives. Whether the result accurately characterizes the general population of anglers or whether it may be due to present data limitations is recommended as an area for future research.

Linking the Lamprey Treatments to Economic Values

In order to use the RUM to value changes in the fishery, we need to establish a link between the treatment options and variables that enter the RUM. While the obvious variable is catch rates, the diagram emphasizes that a complex chain of information is needed in order to evaluate the treatment options. First, the effect that treatments will have on lamprey populations needs to be established. Second, changes in lamprey populations must be linked to changes in the lake trout populations. Third, one needs to map the changes in lake trout populations into changes in lake trout catch rates. Finally, the RUM is used to estimate the use-value that accrue to anglers due to increased catch rates. Thus, the diagram illustrates one pathway in which changes in management actions result in changes in value. Anderson refers to this as marginal analysis to emphasize that we seek to identify how value changes in response to some management action.

Projections of lamprey and lake trout populations associated with the three treatment options as well as the no treatment option were derived from the models of Sitar (the first two linkages in Figure 1). That study models the relationship between lamprey populations in Lake Huron and lake trout populations. These are linked to the control options using assumptions provided by the Great Lakes Fishery Commission (SMRCS). Thus, for each treatment option as well as for the no treatment option, we have a time series of lake trout population levels for various regions of Lake Huron. The projected age 8+ lake trout population levels in the three regions are presented in Figure 2.

The third step in Figure 1 involves relating lake trout populations to the catch rates that are used in the recreational angling model. To relate changes in lake trout populations to changes in catch rates, we will assume that a proportional relationship holds for each site. Such a relationship is often used in the fisheries literature and can be written as $C/E = \alpha S$ where C represents total catch, E represents angling effort, α represents a catchability coefficient, and S represents the population or stock size (this relationship is discussed further in Lupi, Hoehn, and Jester). Thus, an X% increase in the lake trout population associated with a site will increase the lake trout portion of the catch variable for that site by X%. Referring back to equation (4), when Lake Huron lake trout populations increase by X%, only $CR^{lake\ trt}$ is increased by X%. Since only the lake trout portion of the catch rate variable in the recreational angling model is adjusted, the overall catch variable will increase by less than X%.

To complete the linkage, the regional lake trout population estimates were translated into proportional changes in regional lake trout populations by dividing by the regions lake trout population levels in 1994, the year of the behavioral survey. The absolute and proportional changes over time in the populations of mature lake trout for each region are presented in Figure 1. For each county in the recreational demand model, a time series of catch rate changes is derived by multiplying the 1994 catch rate for lake trout by the proportional change in lake trout population for the region associated with the site. This approach preserves the spatial variation in catch rates that existed in 1994.

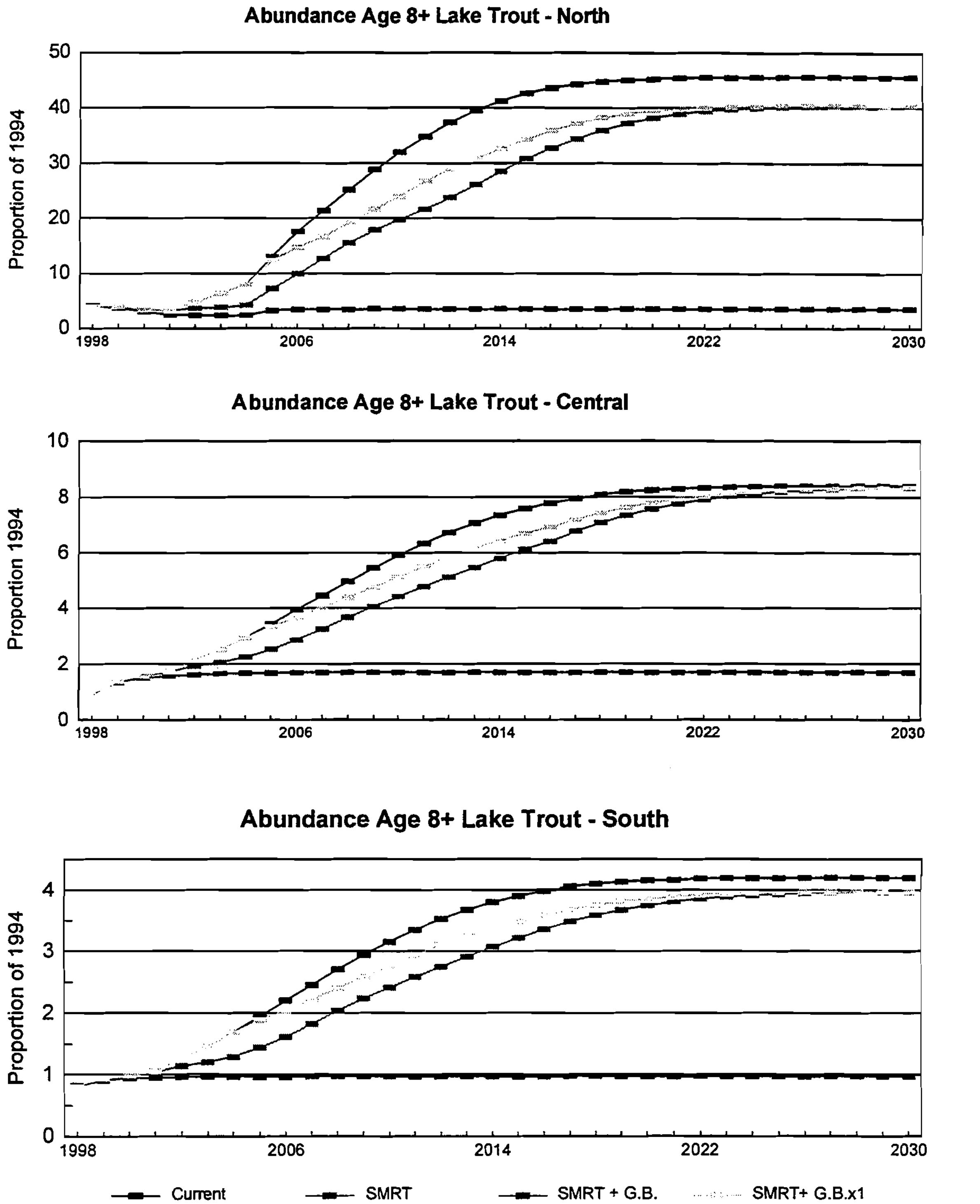

Figure 2: ***Proportional Change* in Estimated Annual Abundance of Mature Lake Trout by Regions of Lake Huron for each Treatment Option**

Valuation Results

The estimates of the economic use-values associated with each of the policy options in the year 2015 are: $2,617,000 for Option A; $4,742,000 for Option C; and $3,333,000 for Option E (see Figure 2). These are estimates of the economic use-values accruing to Michigan resident anglers, and they are denominated in 1994 US dollars. The estimates reveal that each of these options yield substantial benefits in future years.

Table 2: **Estimated recreational angling benefits for the projected lake trout populations in 2015 for each St. Marys River treatment options.**

	Option 1 SMRT only	**Option 2** SMRT + GB	**Option 3** SMRT + GB 1.x
Estimated benefits to Michigan anglers in 2015	$2.62 mil	$4.74 mil	$3.33 mil
Estimated population increase (absolute)†			
Northern region	62,000	90,000	71,000
Central region	122,000	156,000	135,000
Southern region	137,000	175,000	152,000
Lake Huron (total)	321,000	421,000	357,000
Estimated population increase (proportional)‡			
Northern region	30.8	42.6	34.3
Central region	6.1	7.6	6.7
Southern region	3.2	3.9	3.5
Lake Huron§	4.8	6.0	5.3

† Projected absolute increase in mature lake trout population for each region.
‡ Projected factor increase in mature lake trout population for each region (2015 regional population /1994 regional population).
§ Projected factor increase in mature lake trout population for all of Lake Huron (2015 lake population / 1994 lake population).

Table 2 shows the estimated annual use value that would accrue to Michigan's recreational anglers in the year 2015 if lake trout catch rates were to increase by the factors in the table. The table also shows that, as one would expect, the treatment options that yield the largest lake trout population increases have the largest benefits. The absolute changes in population are largest in the southern region and smallest in the northern region. However, since the current population level in the northern regions is so low, the proportional increases in population are much larger in the north than in the south.

Does the spatial pattern of changes in fish population matter? In the above scenarios, the proportional changes in catch rates are much larger in the northern region than in the other regions as seen in Table 2. The final row of Table 2 also presents the lakewide average proportional change in lake trout population. One might ask whether the spatial delineation of the changes in lake trout population affects the estimated economic values. It turns out that the spatial (regional) composition of the catch rate changes makes a substantial difference for the estimated economic values. If the average lakewide change in population were applied to all sites at Lake Huron, then the estimated benefits of option A would be about $8 million. The estimated value based on the lakewide average population change is much larger because the southern portions of the lake that are closer to population centers get a larger catch rate change when the lakewide average lake trout population change for the lake is used than they do when the regional changes are used. The outcome reflects the economic result that, all else equal, changes in fishing quality will be more valuable the closer they are to users. This is a reflection of the use values that are being measured by the travel cost method.

Net Present Values:

Just looking at the estimated economic benefits for the different treatment options in the year 2015 only reveals part of the picture since the costs of the policies differ, as does the timing of the costs and benefits for each policy. The get the stream of annual benefits, the regional changes in catch rates were evaluated for the years 1999 to 2030 with the populations in future years assumed to stay at the 2030 levels. Figure 3 graphs the annual stream of net benefits (benefits to Michigan anglers minus costs). Figure 3 shows that for the treatment options involving granular bayer, there are large downward spikes that reflect the large costs of the granular bayer treatments in those years. One can also see from Figure 3 that the net benefits of each policy are negative for the initial years following the initiation of each of the treatment options. Then, in later years, as the lake trout population begins to grow, net benefits become positive.

Figure 3: Annual Benefits to Michigan Anglers Minus Program Costs

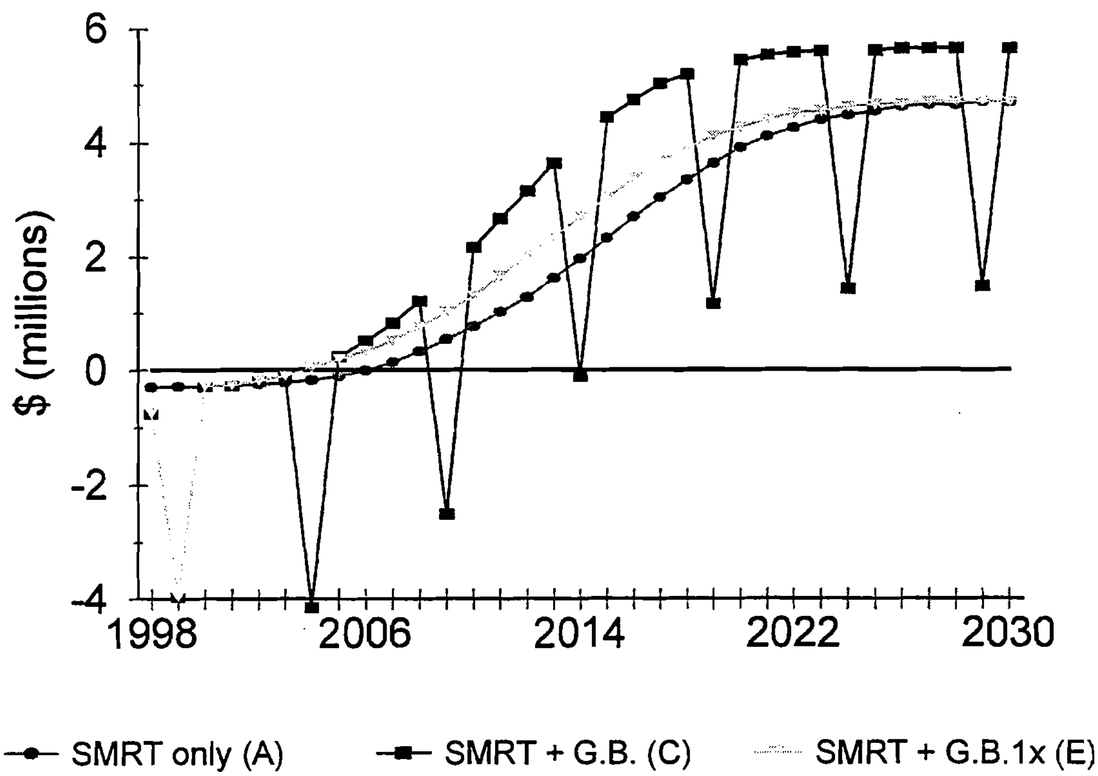

Table 3: **Net Present Value of the St. Marys River policies under alternative interest rates**
(The economic values in table are in $1,000 units)

interest rate	**Option A** (SMRT only)	**Option C** (SMRT + G.B. every 5 yrs.)	**Option E** (SMRT + G.B. only once)	**Option C-A** (which is better?)
1.0%	394,030	406,210	804,720*	12,180
2.0%	165,700	171,530	340,510*	5,820
3.0%	93,340	96,470	192,110*	3,130
4.0%	59,370	60,860	121,700*	1,480
5.0%	40,410	40,730	81,910*	320
6.0%	28,720	28,160	57,070*	(550)
7.0%	21,030	19,800	40,520*	(1,230)
8.0%	15,730	13,970	28,980*	(1,760)
9.0%	11,960	9,780	20,660*	(2,180)
10.0%	9,200	6,680	14,510*	(2,520)
12.5%	4,940*	1,840	4,820	(3,100)
15.0%	2,690*	(740)	(380)	(3,430)
25.0%	(40)*	(3,730)	(6,730)	(3,690)

† Negative numbers in parentheses (based on partial benefits estimate; measures only the use-value that accrues to Michigan resident anglers as a result of the changes in lake trout catch rates).

* Option with largest net present value (present value of Michigan angling benefits minus present value of costs).

The net present value of benefits minus cost was calculated for each option using a variety of discount rates (see Table 3). The results show that all three treatment options are estimated to have positive net present values at reasonable discount rates, even though not all of the benefits have been quantified here. In addition, treatment Option E which involves the one time granular bayer application combined with lamprey trapping and release of sterile males is best in the sense that it yields the largest net present values, (except at very high discount rates). Referring back to Table 3, the net present value results imply that the accumulated difference in benefits between options C and A are enough to offset the added GB application cost that occurs up front -- except at extremely high discount rates. Also, option C is better than option A at lower discount rates (<6%) with the converse holding at higher discount rates. The economic value of the three treatment options differs for several reasons. While option C grows fastest and leads to a larger lake trout population, it also has large recurring costs. Alternatively, option A has the lowest costs, but it also has the slowest growth in lake trout populations. The best alternative, option E, suggests the faster initial growth provided by the first treatment of granular bayer is beneficial, but continued granular bayer treatments do not yield enough additional growth to offset the large application costs.

Limitations and Future Research

It is important to bear in mind some of the caveats associated with the numbers reported in table 3. For instance, the estimated benefits used to calculate the net benefits are based only on the estimated recreational use-value accruing to Michigan recreational anglers. There are likely other economic benefits associated with the treatment options that have not been measured. Potentially important benefits that have not been measured include such things as: benefits to non-resident anglers that fish in Michigan; benefits to anglers that fish in Canadian portions of the lake; benefits due to possible increases in catch rates in northern Lake Michigan; potential reductions in stocking costs; and values that the general public might have for rehabilitation of native fish stocks. Moreover, the changes in lake trout catch rates are based on changes in the growth of age 8+ lake trout which likely over-states the growth in the population of lake trout entering the recreational fishery (about age 5+). Also, In addition, the analysis does not account for uncertainties associated with the projected lake trout growth for each scenario, nor does the analysis account for any uncertainties associated with the economic value estimates. Finally, a sensitivity analysis of the physical and economic assumptions underlying the results has not been conducted. A list of some of the key assumptions underlying the analysis follows:

- used the yearly proportional changes for the age 8+ year classes and these were applied only to lake trout (no changes in other species are assumed);
- all changes in catch rates are proportional to the 1994 values (so sites with very low baseline catch rates tend to stay low);
- the above table uses the complete stream of benefits and costs into perpetuity;
- annual benefits are only comprised of the use-value estimates from the recreational demand model where all trout and salmon species where equally desirable;
- the benefits only apply to Michigan resident recreational anglers and do not include non-use values;

- there's no accounting for savings in fish stocking costs or benefits to commercial or tribal fishers;
- the season for lake trout is held at its current level (May to early Sept);
- the recreation model values travel time at the full opportunity cost;
- any possible increases in lake trout in northern Lake Michigan (due to reduced lamprey populations in northern Lake Michigan) have not been valued;
- there is no accounting for the uncertainty associated with the economic model estimates;
- nor is there any accounting for the degree of uncertainty associated with population projections for each of the options, etc.

Research issues: Several important research issues have been raised in the course of this project. A key issue regards anglers' preferences for alternative species of trout and salmon. In the model applied here, we lacked enough data to identify potential differences in anglers' preferences for various trout and salmon species. As a consequence, the model treats all these species as equally valuable and implicitly holds the allocation of fishing effort constant across species. There are many possible research steps that might shed more light on this issue. One approach would be to incorporate more data into further refinements of the recreational angling model. The additional data might permit the modeling of anglers specie target decisions in addition to their site choices. Another possibility would be to directly question anglers about their species preferences and their preferences for alternative lake management plans. The information about anglers specie preferences could be used to augment the travel cost data, while the information on preferences for lake management plans would permit the estimation of some of the non-use values associated with native specie restoration. Finally, preferences for lake management plans could be collected the general public, as opposed to just anglers.

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THE VALUE OF WATER LEVELS IN WATER-BASED RECREATION: A POOLED REVEALED PREFERENCE/CONTINGENT BEHAVIOR MODEL

by

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THE VALUE OF WATER LEVELS IN WATER-BASED RECREATION: A POOLED REVEALED PREFERENCE/CONTINGENT BEHAVIOR MODEL

Abstract

In this paper we present estimates of the recreation value of preventing a decline in water levels at, and even the total loss of, a large western lake that is drying up. We use a Poisson version of the count data travel cost model; however, in addition to and in combination with revealed preference (RP) data, we employ contingent behavior (CB) responses to hypothetical questions on alternative water levels and number of trips. The panel data model used allows for tests of differences between results using RP and CB data. This particular pooled RP/CB approach has not to our knowledge previously been applied to examine the values of alternative water quantities in water-based recreation.

THE VALUE OF WATER LEVELS IN WATER-BASED RECREATION: A POOLED REVEALED PREFERENCE/CONTINGENT BEHAVIOR MODEL

1. INTRODUCTION

In this paper we use stated preference (SP) data on recreation trips that are responses to hypothetical water level scenarios constructed for survey respondents. We use these data in order to ascertain whether and to what extent water levels matter in the demand for trips to a lake recreation site. These SP or contingent behavior (CB) data supplement revealed preference (RP) data giving actual trips taken during a season. Oddly, though a great deal of recreational economic analysis has focused on water quality issues, far fewer valuation studies have been conducted with the focus being the importance of the quantity of water at a recreation site. Our application is to a Nevada lake, a state where virtually all surface water is of interest because it is so scarce.

One of Nevada's four terminus lakes, Walker Lake, is an important sport fishing location, but is at serious risk of becoming useless in this regard. The lake's level has declined approximately 140 feet since 1880, though very recent wet years in the region have ended a drought period and apparently slowed the decline. Upstream agricultural uses on the Walker River, which feeds the lake and has its headwaters in California, are usually blamed for the decline. Walker River water is about 140 percent allocated, with this overallocation possible because of return flows. Agriculture is also often currently blamed for the accompanying increase in total dissolved solids (TDS), though this point is debatable, as a newly developed water quality model shows that even if all TDS loading from agriculture were eliminated, TDS would still be a problem at the lake (Humberstone, 1999).

Increasing TDS levels have increased the likelihood that certain species of fish cannot survive in Walker Lake (see Thomas, 1995; Humberstone, 1999). Laboratory experiments suggest that the lake's key species of importance, the Lahontan Cutthroat Trout, cannot survive at TDS levels equal to or greater than 16,000 mg/L (Vinyard and Dickerson, 1998). Recent measurements of TDS at Walker Lake are 13,300 mg/L and it has been suggested that volumes of water at Walker Lake greater than 2.3 million acre feet must be maintained to avoid the 16,000 mg/L critical threshold and maintain the fishery (Humberstone, 1999). Even at the current TDS levels the Lahontan Cutthroat Trout currently must be stocked in order to grow to sizes of interest to sport anglers. In summary, if average annual deficit conditions continue, this will lead to a TDS level at Walker Lake that will exceed 16,000 mg/L in approximately twenty years (Humberstone, 1999).

As a possible way to halt the decline of Walker Lake as well as address other important allocation issues, several parties in Nevada and California have begun discussion of the potential for a regional or state water bank. At the practical level, however, a great deal of water would have to be somehow moved to Walker Lake to bring about a substantial change. Humberstone's forecasting model shows that a 16% reduction in all upstream current diversions is not sufficient to maintain the fishery in all future years, though it remains an issue as to what level of upstream diversions, or alternatively, what existing volume at the lake, would be sufficient.

We estimate a recreation demand model to address these issues, using the popular count data specification for the model (see for example Hellerstein, 1991; Creel and Loomis, 1992; Shonkwiler and Shaw, 1996). In combination with the actual trip (RP) data, however, we use CB data. A panel data model is used to combine these data; each respondent represents a cross sectional unit and with each respondent there is associated an observation of RP data along with multiple-scenario CB observations (in ordinary panel data applications, these would be the time series units).

Several previous studies have estimated recreational use values for water quantity changes (e.g., Creel and Loomis, 1992; Cameron, Shaw and Ragland, 1999; Cameron et al., 1996; Ward et al., 1997; Cordell and Bergstrom, 1993; Fadali and Shaw, 1997). However, the panel data approach that we use to combine RP and CB data, initially developed by Hausman et al. (1984) and subsequently applied to the analysis of recreation behavior by Englin and Cameron (1996), has not previously been used to examine the value of water quantities in recreation. To our knowledge, the use of CB data to examine the impacts of water level changes has only been performed previously by Cameron et al. (1996) and Cameron, Shaw and Ragland (1999). Cordell and Bergstrom's approach is essentially a contingent valuation study of use values for North Carolina reservoirs.

2. THE MODEL

The standard count data model is well-developed in the literature and grounded in consumer theory (Hellerstein and Mendelsohn, 1993). In contrast to multiple-site recreation demand models, the count data model can typically handle only a single site; however, it provides a useful framework for dealing with total seasonal trips and seasonal welfare measures. We use a Poisson specification to model the underlying distribution of trips because this is an appropriate way to accommodate the presence of zero values (thereby allowing the inclusion of nonparticipants) and nonnegative integer values. We apply the model to estimate demand and welfare for recreation at Nevada's Walker Lake.

Travel cost models most often only use data on actual reported trips (RP approach). However, with increasing frequency, modelers are supplementing these data with CB (e.g., Cameron, 1992; Adamowicz, Louviere and Williams, 1994; Cameron et al., 1996; Adamowicz et al., 1997). CB responses are those one would get in response to a question such as: "how many trips would you take to this lake if the water level was 20 percent higher than it was when you visited in June?" CB data of this sort asks about behavior rather than a value or willingness to pay for a good, and it may be that people find responding about a behavior related to use of the public good to be easier than valuing the public good.

There are three chief potential gains from using CB data in combination with RP data. First, CB survey questions can be constructed in order to elicit information about scenarios that lie outside of observed historical values for variables such as water levels, site amenities, and travel costs. Since RP approaches are confined to actual values of the data for such variables, extrapolation of the results to conditions outside of observed reality may be problematic. The use of CB data in combination with RP data addresses this issue.

Second, combining RP and CB data in one model allows one to test for similarities (or differences) in empirical results derived from these two different types of data. This capability is useful in certain applications.

Third, the use of panel data generally allows for higher precision in estimation for a given sample size of respondents (alternatively, sample sizes can be smaller for any given targeted level of precision). These benefits have made the panel data approach popular in other areas of empirical analysis, including the assessment of problems in economic history, the behavior of labor markets, the analysis of energy and water conservation measures, and program/project evaluation. Each of these applications involve cross sectional units (e.g., facilities, households, states, countries) that can display substantial heterogeneity. The use of panel data to study such cross sections allows for higher precision in estimation.

The starting point for the model is the demand for trips to a single recreation site:

$$TRIPS = F(C, X, Z, D) \qquad 1$$

where *TRIPS* is the quantity of recreation trips demanded, C is the cost of travel to the site, X is a vector of respondent-specific attributes, Z is a vector of site-specific attributes, and D is a (1,0) indicator variable indicating whether the data for the observation is CB (D=1) or RP (D=0) data.

As mentioned above, the Poisson regression model provides an appropriate specification given the nature of recreation site trip data. The log likelihood function for the Poisson is:

$$\text{Log}\, L(\beta) \;=\; \sum_{i=1}^{N}\left[-\lambda_i + TRIPS\beta'\mathbf{x}_i - \ln TRIPS_i!\right] \qquad 2$$

where $\lambda_i = \exp(F(C_i, X_i, Z_i, D_i)$.

While the model above yields a welfare measure that is an approximation of the exact Hicksian measure, it has a simple and attractive feature allowing calculation of the consumer's surplus per trip. Assuming γ is the coefficient on the travel cost, for a change in the travel cost to a very large (infinite) travel cost, CS per trip is simply $-1/\gamma$. Total seasonal consumer's surplus is simply the total predicted trips divided by γ. Unfortunately, when using a single-site cross-sectional model, the site characteristics cannot be used to explain the model (as they obviously do not vary for individuals in the data), and therefore there is no direct link to be made between marginal site quality changes (such as the water quantity change of interest here) and estimated welfare impacts.

We use a panel data formulation in which each cross section "i" corresponds to an individual respondent. and for each i there are multiple observations "j" that correspond to the source of data. In our analysis, we have two sources of data: one based on RP data and one based on different CB survey scenarios. The CB scenarios are described briefly in the following section.

Finally, we deal with the potential for overdispersion by reference to White's standard errors, from which inferences can be drawn even in the presence of misspecification (White, 1982). In addition, we note that the Poisson distribution yields unbiased estimates for the parameters even when the distribution is misspecified (Gourerroux, Montfort and Trognon, 1984).

3. DATA AND VARIABLES

Between November 1995 and March 1996, a mail survey questionnaire was sent to a group of recreators who visit several lakes in the region of northwest Nevada. The mail survey was implemented using most of the guidelines suggested by Dillman (1978), but the budget for this project precluded extensive efforts to obtain a return from those who failed to respond. Approximately 44 percent of the questionnaires were returned, after subtracting those surveys which were returned because of bad addresses. Following cleaning of the sample to exclude illogical responses and incomplete or unusable trip data, 679 completed surveys remained. A large proportion of the sample are anglers, and had participated in the annual Walker Lake fishing derby, typically held in the winter.

3.1 Contingent Behavior Scenarios

Economic analysis on this project took place simultaneously with beginning research in the hydrologic and other physical sciences, so the survey design could only incorporate a scant amount of existing scientific information. The main source of scientific information in 1996 was found in Thomas (1995), and records from the U.S. Geological Survey. There were six different versions of the mail survey questionnaire, and key design elements were varied to allow some flexibility. Four of these presented baseline and hypothetical scenarios at Walker Lake, while the other two dealt with other water recreation sites. Each respondent received only one version of the survey. This manuscript deals only with the surveys pertaining to Walker Lake. Each survey version depicted slightly different hypothetical scenarios, with each being a variation on possible water levels. Scenarios were presented using information in text form. Additionally, three of the versions presented to the respondent a pair of computer-enhanced photographs, one with "baseline" actual 1996 conditions and the other with enhanced "new" conditions. After being presented with a scenario which described a water level increase or decrease, respondents were asked whether they would change behavior from their actual number of reported trips for the season because of this different water level. If they stated yes, they were asked whether they would take more or fewer trips, and asked to report how many trips they would take and during which month(s) the trips would be increased or reduced. Only the three scenarios pertaining to Walker Lake water levels were retained for the dataset used in this manuscript. The three scenarios were:

- a text-only high-water scenario which described conditions (lake surface area, level of TDS, condition of sport fish, and number of usable boat ramps) at water levels approximately 20 feet higher than end-of-1996 levels of 3,946.5 feet,
- an identical highwater scenario that included computer generated photos of the higher water level at Walker Lake, and
- a low-water scenario including photos, which described conditions associated with water levels approximately 20 feet lower than 1996 conditions.

It is clear that the 20 foot increases depicted in the surveys would translate to large increases in volume at the lake (approximately 700,000 acre feet), but this number was chosen based on available physical science information that suggested increases which might prevent fishery loss (Thomas. 1995). These scenarios are perhaps both politically and practically improbable in view of current institutions, but the key point in doing such analysis relates to whether the respondents believed in the scenarios. If the respondents thought the scenarios were plausible, as may people asked to rate a currently unavailable automobile design, their responses can be tested for consistency. If respondents thought the scenarios were implausible, they were given the option of no response.

After eliminating surveys with inconsistent/missing contingent behavior or demographic data, a sample of 236 respondents remained. Each of these respondents contributed two observations to the model (reported actual 1996 trips and contingent trips under the new water level scenario), for a total of 472 observations. Of the 236 respondents, 82 completed surveys involving lower water level contingent scenarios. The remaining 154 respondents completed surveys involving a higher water level scenario. Of the 154 higher water scenario completes, 91 were the version with photos accompanying the text. Of the 236 respondents, 136 said they would not change the number of trips they would take under different hypothetical conditions. 117 of the respondents did not take any trips to Walker Lake originally and 99 of the respondents did not take any trips under the contingent scenarios. Because of the manner in which the original sample was obtained, there are potential biases, including those associated with on-site recruitment (see Shaw 1988). Therefore, we make no attempt to generalize from our sample to a larger random sample of the population of recreators.

3.2 Variables

The dependent variable is number of trips to Walker Lake. The independent variables included in the model consist of travel cost, one site-specific attribute (water level), six respondent-specific characteristics, an indicator variable CB denoting the source of the data point (RP *versus* CB), and an interaction term between CB and travel cost. The independent variables are shown in Table 1.

4. RESULTS

We first discuss the results of the pooled Poisson model (Section 4.1). Then, using those results we develop and present estimates of the value of recreation at Walker Lake, the impact of changes in water level on trips taken, and the influence of water level changes on recreation values (Section 4.2).

4.1 Model Results

Table 2 presents the results of the pooled Poisson models. We estimated the model in two ways:

1. The first specification includes the variables CB and CB*COST (unrestricted model). Inclusion of these variables allows one to test the null hypothesis that the source of data (CB versus RP) is not a statistically significant influence in the model. The results of this model specification are shown in the second column of Table 2.
2. The second specification omits the variables CB and CB*COST (restricted model). Estimation of the model without these variables allows one to determine the influence of their omission on other parameters of interest. The results of this model are shown in the third column of Table 2.

Inspection of Table 2 shows that most results are similar across the two specifications. The estimated coefficient on Walker Lake water level is positive and significant at the .01 level. This means that, *ceteris paribus*, higher water levels are associated with higher numbers of trips to the lake. The estimated coefficient on travel cost is negative, as expected, and statistically significant at the .01 level as well. The gender and age of the respondent both have the expected signs (positive) and are statistically significant. The indicator variable denoting that the respondent is retired has a negative coefficient, perhaps contrary to typical expectations, but is significant at only the .10 level. Size of household, level of respondent's education, and household income are not statistically significant.

While the estimated coefficient of CB is not statistically significant, the coefficient on CB*COST is of marginal statistical significance (at the .10 level for a two-tailed test). At first glance this suggests that the source of data (contingent behavior scenario *versus* actual revealed preference data) may have a marginal influence in the model. Comparison across columns 2 and 3, however, shows that inclusion of the two CB indicator variables has very little (in some cases no) influence on estimated coefficients for the remaining variables. The parameter most affected is that for water level, which falls from .028 to .024 (14% decrease) due to inclusion of the data source indicators. This is a relatively modest alteration.

To explore this issue further, we conducted a Wald test, which provides the appropriate hypothesis test for the influence of the source of data (CB *versus* RP). This test is preferred over a likelihood ratio test because the results do not depend on the validity of the assumed underlying (Poisson) distribution. The null hypothesis is the set of restrictions:

$$H_0: \beta_{CB} = \beta_{CB*COST} = 0 \qquad 3$$

Under H_0, the Wald test statistic W has a chi-squared distribution with two degrees of freedom (the number of restrictions). The critical value for the chi-squared distribution (n = 2, P = 0.95) is c = 5.99. For the (unrestricted) regression shown in column 2 of Table 2, W = 4.492 < 5.99 = c. Therefore, one cannot reject the null hypothesis that the set of restrictions holds. The indicator variables denoting the source of data (CB *versus* RP) are not significant factors in the model.

4.2 Estimated Values

Table 3 shows estimated values derived from the results of the models. First, we estimated average consumer surplus per trip to Walker Lake, calculated as $-1/\beta_{COST}$. For the unrestricted model, the estimate of per-trip consumer surplus equals $88/trip. The estimate of consumer surplus from the restricted model equals $120/trip.

At first glance, these per-trip values may appear somewhat high. For example, the median value for cold water recreational fishing as reported in Walsh et al. (1990) is approximately $40 per day (1997 dollars). However, it is important to remember that the values estimated by our model are in units of dollars per trip. According to the results of on-site surveys conducted at Walker Lake, the mean trip length is about 3 days (Fadali, Shaw, and Espey, 1998). Multiplying the Walsh et al. per-day value by 3 days yields $120/trip. This is quite close to the values we estimate in this manuscript, in fact equal to the estimate of consumer surplus from the restricted pooled Poisson model.

The second type of result included in Table 3 is our estimate of average annual consumer surplus (per person) from recreational visits to Walker Lake. The estimates range from $485/person/year to about $665/person/year. We do not in this manuscript develop estimates of aggregate annual consumer surplus (i.e., for the entire population of recreators at Walker Lake).

There are two chief reasons for this. First, the mail survey sample on balance is thought to exhibit over-avidity on the part of recreators (Fadali, Shaw, and Espey, 1998). Second, precise estimates of the number of persons who make visits to Walker Lake are not available.

The third row of Table 3 presents estimates of the effect of changes in Walker Lake water level on the number of trips. The results indicate that for a one-foot decline in water level, each recreator would take (on average) between 0.1 and 0.2 fewer trips per year. In the fourth row, we show the consumer surplus losses associated with this decline in trips. Each one-foot drop in water level is estimated to result in a loss on the order of $12 to $18 per person per year.

Another way of examining the effects of water quantity changes relates to changes in the volume of water at Walker Lake, and these can be linked with recent work in the physical sciences relating volume or storage to the critical TDS levels. Using a quadratic relationship between storage and the Lake's elevation, it can be estimated that a volume of 2.3 million acre feet translates to an elevation of about 3,951.33 feet. This elevation was approximately the actual end of year lake level in 1997, as determined from U.S.G.S. records. Recent hydrology modeling suggests that even this elevation and volume are not sufficient to maintain the fishery at Walker Lake.

Our hypothetical scenarios pose an increase of 20 feet in the Lake's water level, which corresponds approximately to the lake's end of year level in 1984. Actual storage at the end of that year was about 3.05 million acre feet. Thomas (1995) estimated that to maintain TDS at the July 1994 level of 13,300 mg/L would require about 33,000 acre feet more than a long term average. and that to reduce TDS from 13,300 to about 10,000 the lake-surface would need to increase by about 20 feet, corresponding to about 700,000 acre feet of water. This 20 foot increase provided the basis for the hypothetical scenario. Others have suggested that Walker Lake needs approximately 50,000 acre feet additional volume to maintain the fishery, but the exact additional volume is not yet known. If we assume that 50,000 more acre feet is adequate, this would imply a sustainable fishery volume of approximately 2.35 million acre. Again using the quadratic relationship, this would translate into about a 1.4 foot increase in the water level at Walker Lake, only slightly more than the marginal "one foot" value reported in Table 3. Using this marginal value as an indicator, the values of roughly $12 to $20 per person for this change in water level are large enough to compensate agriculture to move 50,000 acre feet of water down to the lake, provided the total number of "willing" recreators is sufficient to rent water from other sources. Fadali and Shaw (1998) suggest, using conservative estimates, that this total number is large enough.

5. CONCLUSIONS AND POSSIBILITIES FOR FURTHER RESEARCH

Some hydrologists have suggested that if the 1987 to 1994 drought at Walker Lake had existed for just another two years, the lake would have been unable to recover for future use. At best it is currently a fragile ecosystem. As stated in the introduction, it is possible that a water bank will be created for this region, though national, state and local politics will undoubtedly play the deciding role. On the positive side, water banks may in fact be most beneficial during drought periods (Loomis, 1992). Part of the success for the bank depends on whether a market exists for the water, with one possibility being the demand that recreators have for increases in water supplies at one or more recreational sites. As shown in Fadali and Shaw (1998), in principle, the demand for water from a bank exists to some extent; their results using only RP data indicate that value per acre foot on the part of recreators may be high enough to bid away agricultural water on a rental basis.

These results are supported here, where the SP data from the same study are used for the first time. An advantage over the Fadali and Shaw (1998) study is that we are able to focus more carefully on values for marginal water level changes at Walker Lake.

As in this study, economic analysis often must be performed ahead of physical science analysis because of the funding and timing of research projects, even though having the best physical science results often improves the quality of economic analysis. This suggests that it is wise to build an economic model flexible enough to incorporate better scientific data and measurements as they become available. Using the storage-elevation relationship discussed above, we can flexibly translate water level changes to volume changes, identifying the critical water level change needed. Our model then allows one to recover the value for additional water for a variety of water levels considered. We demonstrate that under our assumptions an increase in Walker Lake's level of about 1.4 feet may be sufficient to maintain the fishery, and it seems certain that a 20 foot increase would do so. A key science finding yet to come is a more precise identification of the critical volume of water for sustaining the Walker Lake fishery. If more water is needed to avoid the TDS level critical for sport fish species, our model can be used to examine that situation.

As noted above, obtaining large increases in volume at Walker Lake may not be possible given the current political climate, existing institutions, and withdrawals from the system. Future study of the Walker River Basin needs to better address the exact volume needed to avoid the 16,000 mg/L TDS level over the years to come, and the role of uncertain factors such as global climate change and the incidence of extreme precipitation events. Finally, there needs to be much more research on the willingness to sell on the part of agricultural users in the Basin, and other factors that could lead to actual development of a water bank.

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Table 1: Variables

Variable Name	Variable Definition
WATER	Water level at Walker Lake, in feet
GENDER	Indicator variable denoting respondent gender (= 1 if male; otherwise 0)
AGE	Age of the respondent in years
HOUSEHOLD	Number of persons in respondent's household
EDUCATION	Years of education of respondent
INCOME	Annual household income (including interest, dividend, and retirement income)
COST	Travel cost, including opportunity cost of time
RETIRED	Indicator variable denoting respondent is retired (= 1 if retired; otherwise 0)
CB	Indicator variable denoting whether the observation is from CB or RP data (= 1 if from CB data; otherwise 0)
CB * COST	Interaction term composed of CB * COST

Table 2: Results of Pooled Poisson Models[1]		
Variable	**With CB Indicator Variable and CB*Cost Interaction Term**	**Without CB Indicator Variable and CB*Cost Interaction Term**
Constant	-92.55*** (23.43)	-108.92*** (26.48)
WATER	0.024*** (0.006)	0.028*** (0.007)
GENDER	0.529** (0.269)	0.526** (0.267)
AGE	0.036*** (0.007)	0.036*** (0.007)
HOUSEHOLD	0.102 (0.104)	-0.101 (0.103)
EDUCATION	-0.151 (0.095)	-0.150 (0.095)
INCOME	$-6*10^{-6}$ $(4*10^{-6})$	$-6*10^{-6}$ $(4*10^{-6})$
COST	-0.011*** (0.003)	-0.008*** (0.002)
RETIRED	-0.498* (0.279)	-0.501* (0.281)
CB	-0.239 (0.400)	Not included
CB* COST	0.005* (0.003)	Not included
Log Likelihood	-2725	-2755

1 White's standard errors are shown in parentheses.
* Denotes statistical significance at the .10 level for a two-tailed test.
** Denotes statistical significance at the .05 level.
*** Denotes statistical significance at the .01 level.

Table 3: Consumer Surplus and Changes in Trips		
Estimate	**With CB Indicator Variable and CB*Cost Interaction Term**	**Without CB Indicator Variable and CB*Cost Interaction Term**
Average Per-Trip Consumer Surplus	$88	$120
Average Annual Consumer Surplus	$485	$664
d(TRIPS)/d(WATER) at Mean Predicted Value of TRIPS	0.132 trips annually per person per change in water level (feet)	0.154 trips annually per person per change in water level (feet)
Values per Trip due to Water Level Change	$11.60 annually per person per change in water level (feet)	$18.54 annually per person per change in water level (feet)

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The Effect of Fluctuating Water Levels on Reservoir Fishing

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Proposed Running Head: Water Levels and Fishing

ABSTRACT: The effect of Tennessee Valley Authority reservoir water levels on recreational fishing is evaluated using a time-series cross-section data set. The data were collected for fishing in East Tennessee during March through August in each of the years 1994-1997. The recreation demand model shows that water levels do not represent a major barrier to participation during the six month period. Water levels do, however, affect the number of trips that anglers take during the season. On average, maintaining TVA lakes at full pool for one additional summer month (i.e., until the end of August) would result in an additional ⅔ trip per angler, or an additional 100,000 reservoir fishing trips per year in the study region. The net benefit to anglers is, on average, about $3.75 per season, or approximately $562,500 in the region.

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The Effect of Fluctuating Water Levels on Reservoir Fishing

The Tennessee Valley Authority (TVA) system of dams and reservoirs is designed to provide the Tennessee Valley with flood control, navigation along the Tennessee River, power generation, and economic development in the region. Current TVA policy begins lake drawdowns on August 1 of each year to generate electricity and provide downstream flood control. The drawdown date was the result of an intensive review of reservoir operations by TVA in cooperation with local government and business representatives, as well as the general public. TVA personnel have declared the process and its outcome a model of success, and cite its applicability to other water management agencies facing controversy (Ungate 1996).[1]

But the August 1 drawdown remains controversial, especially among users of tributary lakes at the upper end of the Tennessee Valley watershed. These lakes tend to have deeper channels with shallower, high elevation coves. The drawdown leaves many coves and boat ramps at these lakes landlocked for much of the year, or with a long mud-flat eventually leading to water. An extensive number of land parcels are exposed to these mud flats, depressing property values. Recreational users, including anglers, may find access limited or precluded through the drawdown. A recent study found that delaying drawdown until October 1 on two major tributary lakes could have an economic impact to just six surrounding counties as high as $7 million as people increase lake recreation in response to higher water levels (Murray et al. 1998).

The effects of the drawdown policy are likely to differ across recreational activities. While some activities such as swimming are clearly impacted in a negative way, the effect that drawdowns have on sportfishing is in question. Some say that drawdowns help anglers because fish become concentrated in smaller pools of water, improving fishing quality. Others say that access issues are more important because dry boat ramps restrict the ability of reservoir anglers to launch boats, or that the aesthetic impact of a "bathtub ring" around the lake discourages recreational fishing.

A multi-year recreational fishing data set is used to evaluate the response of reservoir anglers to the TVA water management policy. Angler response is modeled with a combined multinomial site choice/double-hurdle (MNL-DH) count data trips model, following Shaw and Jakus (1997). The MNL-DH modeling strategy allows us to model the effect of water levels not only on site choice (which reservoir to fish), but also on the "desire" to fish in reservoirs, where water levels may represent a site-quality hurdle. We also provide estimates of the benefits to anglers under alternative water level policies.

The Advantage of a Double Hurdle Approach[2]

The MNL-DH model allows us to separate the sample of anglers into three groups. The first group is composed of reservoir users, those who actually fish in reservoirs of the TVA system. The second group is potential reservoir users, those who fish other types of water bodies but would consider fishing in reservoirs under circumstances favorable to them. These people might wish to fish in reservoirs but face a hurdle that prevents reservoir fishing, e.g., a site-quality hurdle caused by inadequate water levels which may limit access or increased the chance that a boat may strike a submerged object. The final group is composed of those anglers who would rarely, if ever, consider fishing in reservoirs (e.g., the die-hard fly-fisherman), and they never get over this "participation" hurdle. The first hurdle is fundamentally economic: if site quality improves enough the consumer will move from a corner solution (no trips) to an interior solution (non-zero trips). The second hurdle is fundamentally non-economic: the feasible set of policy relevant price/quality combinations is very unlikely to move the consumer from a corner to an interior solution.

The key advantage of the MNL-DH modeling strategy is that one can specify different data generating mechanisms for the different hurdles that define each group. The probability of observing zero trips for any observation is composed of two parts: the probability that desired consumption is zero (the participation hurdle) plus the probability that desired consumption is positive, but another hurdle prevents consumption (the site-quality hurdle). Following Shonkwiler and Shaw (1996), let D_i represent the latent decision by person i to participate in reservoir fishing, with observed trips $y_i = 0$ if $D_i \leq 0$. Let Prob($D_i = 0$) $= 1-\Phi(Z_i'\gamma)$ describe the probability of observing zero trips, where Z_i is a vector of factors influencing the participation hurdle. Z_i can include individual specific variables such as demographics, so that $\Phi(Z_i'\gamma)$, the cumulative normal distribution evaluated at $Z_i'\gamma$, is the probability that a person wishes to make a reservoir fishing trip. Additionally, let λ_i be the Poisson parameter describing the number of reservoir fishing trips, where $\lambda_i = \exp(X_i'\beta)$ where the X_i are variables that influence the trip making process. The probability that any observation i is a non-user with little or no interest in reservoir fishing is $1-\Phi(Z_i'\gamma)$, whereas the probability of a corner solution (potential user) is given by $[\Phi(Z_i'\gamma)]\times\exp(-\lambda_i)$. This second probability is simply the product of the probability of clearing the participation hurdle and the probability that desired trips is zero (perhaps because of site quality reasons). Finally, the probability that observation *i* is a reservoir user (clearing both hurdles) is given by $[\Phi(Z_i'\gamma)]\times[1-\exp(-\lambda_i)]$.

Combining a multinomial logit site choice model with a double-hurdle count data model allows us to gauge the influence of water levels not only on site choice, but also on participation in reservoir fishing. To see this, recall that in linked site choice/trips models some form of the inclusive value, the summary measuring capturing all characteristics of all sites, is passed on to the trips portion of the model. The inclusive value is part of the X_i vector, and is an argument of λ_i.[3] With site water levels appearing in the MNL site choice model, it is easy to gauge the impact of water levels on potential users by calculating the probabilities described above.

Study Area and Data Sources

The study area consists of a set of thirteen reservoirs located in a 35 county region of East Tennessee. Nearly all of the reservoirs are located adjacent to an Interstate highway and stretch along a corridor from Bristol, TN to Chattanooga, TN. The reservoirs in the northeast portion of the study area are tributary reservoirs subject to relatively large drawdowns in the fall; the most popular tributary reservoirs are Cherokee, Douglas, and Norris reservoirs. Water levels on Norris Lake, for example, range from a March 1 elevation of 995 feet, to a peak elevation of 1023 feet about June 1 (Figure 1). Douglas receives 750,000 visitor-days per year, Cherokee 950,000 visitor-days, and over 2 million visitor-days per year at Norris (Murray et al. 1998).

Recreational fishing data were collected between 1994 and 1997, a four year period. A random digit dial survey was used in each year to contact and identify people who fished in Tennessee.[4] Once identified, anglers were asked about all fishing activities during the six month time period (March 1 through August 31) immediately preceding the survey. We did not contact the same anglers each year, so we do not have a panel data set; rather, we have four different cross-sectional data sets. The final data set is composed of 977 East Tennessee anglers from whom complete trip and income data were obtained.[5] Not all anglers fished in reservoirs during the six month period; they could have fished in private ponds, trout streams, or warmwater streams.

Daily water level information for each lake were obtained from the Tennessee Valley Authority. The four year time period showed considerable variation in water levels for the tributary lakes. Figure 1 shows the daily elevations for a typical tributary lake, Norris Lake, located about 30 miles north of Knoxville, TN. The figure shows that 1996 and 1997 were relatively "normal" years as the reservoir filled during the spring, but in 1994 the lake filled very rapidly while in 1995 the lake filled very slowly. In 1994 and 1996 "full pool" was reached around May 15, whereas in 1995 and 1997 full pool was reached on roughly June 1. On August 1, TVA's policy of maintaining a full pool expires, and the agency begins unrestricted lake drawdown. Relative to the 1996 drawdown, the lake was drawn down swiftly in 1995 and 1997, while in 1994 it remained near full pool. The seasonal pattern of water levels on Norris was similar to that experienced by the other two major tributary reservoirs (Douglas and Cherokee). For contrast, Figure 2 shows the change in elevation for a typical "run-of-the-river" reservoir

(Fort Loudon, located in Knoxville) not subject to large drawdowns. None of the downstream reservoirs are subject to large drawdowns (on average, about 4 to 6 feet elevation change).[6]

Water level data augments the recreation data for each year and, by "stacking" the data for all years, the variation in water level across years can be exploited to identify the effect of water levels on fishing participation and reservoir choice. The data must capture both the "fill" rate and the "drawdown" rate for each reservoir, so the periods April 15 through May 15 (fill) and August 1 through August 31 (drawdown) were chosen. For each time period in each year the average daily water elevation was calculated. The water level characteristic for each reservoir was calculated as deviations from the 1996 "base" year. Water levels above those in 1996 were measured as positive values whereas levels that were lower than 1996 had negative values.

Empirical Results

The full sample consisted of 977 East Tennessee anglers, of whom 55.2% were reservoir users. Simple statistics from the recreational data indicate that water levels may be important to anglers. During the high pool water year in 1994, over 62% of anglers fished in reservoirs, whereas during the low pool water year of 1995 fewer than 50% of anglers fished in reservoirs (Table 1). This is evidence that water levels may be part of a significant site quality hurdle as suggested by Cameron et al. (1996). Further, the average number of trips by reservoir anglers was lowest in 1995 and highest in 1994.

Site Choice Portion. As noted above, water levels during two time periods, April 15 through May 15 and August 1 through August 31, were used to characterize the spring "fill" and late summer "drawdown" phases of this reservoir characteristic. Other reservoir characteristics included travel cost, the number of boat ramps at each reservoir, and the average catch rate (summed over all species and averaged across anglers).[7] A dummy variable indicating the presence/absence of a fish consumption advisory was the final reservoir characteristic.[8]

All variables in the site choice portion of the model had the expected sign and were statistically significant (Table 2). Rising travel costs made a site less likely to be visited (the negative sign); more boat ramps–a measure of site access–made a site more likely to be visited (a positive sign); higher catch rates made a site more likely to be visited (a positive sign); a fish consumption advisory made a site less likely to be visited (negative sign).

Focusing now on the role of water levels in site choice, a positive coefficient means that a site is less likely to be visited when water levels are below 1996 levels, whereas a negative coefficient means a site is more likely to be visited if water levels are below 1996 levels. The site choice model shows that low water levels in the late summer negatively impact site choice: relative to the 1996 water levels, low water made a site is less likely to be visited whereas higher water levels made a site more likely to be visited. The coefficient for spring water levels was negative, but was not significant at conventional significance levels.

Trip Frequency/Participation Portions. The information contained in the site-choice model is passed to the trip frequency model via the inclusive value. The inclusive value contains both economic information (the effect of travel cost) and site quality information (e.g., the effect of water levels). The sign of the inclusive value is positive, as expected, and is statistically significant. Trip frequency also increases as income increases. In the participation portion of the model, anglers who fish waters other than reservoirs were less likely to participate in reservoir fishing. College educated anglers are more likely to fish reservoirs relative to anglers with a high school education or less. Anglers who were nonwhite were less likely to fish reservoirs than white anglers. Residence in an urbanized county was statistically insignificant, indicating that participation in reservoir fishing was independent of the angler's residence.

Evaluating Baseline and Alternative Water Level Scenarios

Two alternative water level scenarios were considered in comparison to a "baseline". The baseline is the historically experienced water level in the sample. The first alternative scenario assumes that the August water levels experienced in 1996 were standard and held in the spring/summer season of each year contained in the sample period. The water levels in 1996 represent a relatively slow drawdown through the month of August. The second alternative is that advocated by local lake user groups: maintain a full pool through the end of August. The full pool scenario is reasonably close to the water level experienced in 1994, so that neither policy alternative would result in a site quality characteristic level that is outside what respondents' have already experienced.[9]

Water Levels as a Site Quality Hurdle. In the MNL-DH model the influence of water levels on participation is captured in the inclusive value index passed from the site choice portion to the hurdle portion. Following the formulas presented in the methodology section, the probability of any angler being a user (someone who fished in a reservoir), a nonuser (someone who would not fish reservoir regardless of the water level) and a potential user (someone who would fish reservoirs if water levels were high enough) were calculated (Table 3). It is also possible to calculate the expected number of reservoir fishing trips.

Under all policy scenarios the probability of being a nonuser is constant because the inclusive value index does not enter this hurdle. The mean probability of being a nonuser was constant at just over 44.7 percent. The other two probabilities–being a user or a potential user–do change as alternative policies change. Under the baseline (actual) policy, the mean probability of being a user is just under 55.3 percent, while the mean probability of being a potential user is very small (5.9×10^{-5} percent). The mean expected number of reservoir fishing trips, conditional on being a user, is 14.61, while the unconditional estimate of mean trips is 8.15.

As water levels are raised under the alternative policies, the probability of being a user rises while the probability of being a potential user falls. The change is quite small: for the change from baseline scenario to the full pool scenario the probabilities change by only 2.0×10^{-5} percent. Additional water in August does not appear to draw anglers from a corner solution (no reservoir fishing) to an interior solution (making at least one trip), suggesting that August water levels are not a major hurdle for potential reservoir anglers. This makes some intuitive sense in that lakes are at full pool from roughly late May through July 31. If water levels were a "make or break" site characteristic reservoir anglers would choose to fish during this portion of the season.

Just because water levels were not a major *hurdle*, however, does not mean that levels were unimportant. Water levels had a large impact on the estimated number of reservoir fishing trips. The mean number of fishing trips, conditional on being a reservoir user, is 14.73 for the 1996 scenario and 15.27 for the full pool scenario. Thus, the full pool proposal put forth by advocacy groups within the study area would result in, on average, an additional ⅔ trip per season per reservoir angler. Given that approximately 150,000 people in the study region fish in reservoirs, this means that maintaining full pool through August 31 would result in an additional 100,000 fishing trips.

Willingness to Pay for Alternative Policies. Willingness to pay measures were calculated for each of the alternative policy scenarios for the linked site choice/trips MNL-DH model. For the "1996" policy the seasonal WTP measure was $0.64, with a 95% confidence interval (CI) between $0.04 and $1.30. For the full pool scenario the mean seasonal WTP was $3.75 (with 95% CI $0.07 - $8.46). With 150,000 reservoir anglers in the East Tennessee study region, the aggregate benefit of a full pool policy would be approximately $562,500.

The estimated welfare change is similar to recent study of anglers in Nevada, but is still relatively small in comparison to most of the past literature. Shaw et al. (1999) studied a Nevada lake that had been drained in 1992, killing all the fish. Their model found an aggregate benefit to anglers of $100,000 to have maintained the "average" minimum pool in 1992 rather than having the lake drained (a per trip measure could not be calculated). Cordell and Bergstrom (1993) estimated the aggregate benefit for a policy holding four TVA lakes in North Carolina at full pool for one additional month as $5.1 million. The Cordell and Bergstrom estimate is about nine times as large as the estimate for the Tennessee lakes, but also includes benefits accruing to recreationists other than anglers (i.e., campers, hikers, picnickers, etc.). Fadali and Shaw (1998) estimated the benefit for keeping a volume of water sufficient to avoid fish kill at a remote Nevada lake with few substitutes. The per trip benefit was just under $30, with an aggregate benefit of $4.2 million. The range of benefits for maintaining water levels in lakes is clearly quite wide; the estimate from this study is within this range, though at the smaller end of the scale. Our estimate of aggrgegate benefits, however, may be somewhat understated because TVA does not maintain a "standard" policy for post-August 1 drawdowns. An established, predictable drawdown policy might yield surplus gains in excess of this amount if anglers, especially out-of-state anglers with little access to lake-level information, could rely upon a full pool until August 31.

Conclusions

The MNL-DH model indicates that lake water levels in the month of August are important to anglers. August water levels do not appear to act as a hurdle to participation (anglers will fish in reservoirs before the August drawdown), but water levels do affect the number of trips that anglers make. Assuming the full pool scenario advocated by local lake user groups is adopted, anglers would make an extra 100,000 trips per season. The aggregate consumer surplus of this policy is approximately $562,500.

Economic development is a primary goal of TVA, and development could be stimulated by the additional 100,000 fishing trips a full pool policy would spur. But a water management agency such as TVA is often faced with multiple and, sometimes, conflicting objectives. TVA, for example, is also responsible for providing flood control, downstream navigation, and hydroelectric power. A full pool policy can stimulate economic development, but may also engender costs associated with increased risk of flooding, increased risk of barge accidents if downstream channels are shallower, and decreased power generation. Future research would evaluate the full pool policy against costs of not meeting these additional objectives.

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Table 1. Angler Visitation to Tributary Reservoirs, by Year

Year	% of Anglers Visiting Reservoirs	Average # of Visits to All Reservoirs
1994	62.3%	16.40
1995	49.6%	12.90
1996	54.0%	15.32
1997	55.2%	14.72

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Table 2: Combined Site Choice/Trips Model for Reservoir Fishing

Variable	
Site Choice Portion	
Travel Cost	-0.04* (-9.13)
# Boat Ramps	0.03* (6.92)
Fish Consumption Advisory	-0.34* (-2.61)
Catch Rate	0.10* (3.04)
August Water Level Dev.	0.07* (2.02)
4/15 - 5/15 Water Level Dev.	-0.05 (-1.53)
Trip Frequency Portion	
Intercept	2.17* (9.01)
Inclusive Value	0.18** (1.76)
Income ($1000)	0.005* (1.99)
Participation Portion	
Intercept	1.22* (11.92)
Fish Other Waters	-1.17* (-14.03)
College Education	0.32* (2.78)
Nonwhite	-0.30** (-1.70)
Live in Urbanized County	0.09 (1.41)
# Observations	977

[a] Number in parentheses is the ratio of a coefficient to its asymptotic standard error. Standard errors determined using White's general covariance matrix.

* = significant at α=0.05
** = significant at α=0.10

Table 3: Evaluating Water Levels as a Hurdle: Mean Probabilities for the Sample

	Baseline	1996 Standard	Full Pool through 8/31
P(Nonuser) = $1 - \Phi(Z'\gamma)$	0.44700502	0.44700502	0.44700502
P(Potential User) = $\Phi(Z'\gamma) \times \exp(-\lambda)$	5.9×10^{-7}	5.5×10^{-7}	3.9×10^{-7}
P(User) = $\Phi(Z'\gamma) \times [1 - \exp(-\lambda)]$	0.55299439	0.55299443	0.55299459
E(Trips|Trips>0)	14.61	14.73	15.27
E(Trips)	8.15	8.22	8.51

Table 4: Mean WTP for Water Level Scenarios

	1996 "Standard" Drawdown	Full Pool through August 31
Site Choice/Trips Model		
Mean WTP	\$0.64 (\$0.04 - \$1.30)	\$3.75 (\$0.07 - \$8.46)
Mean Δ E(Trips|Trips>0)	0.12	0.66

[a] 95% CI calculated by Krinsky-Robb method.

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Figure 1.

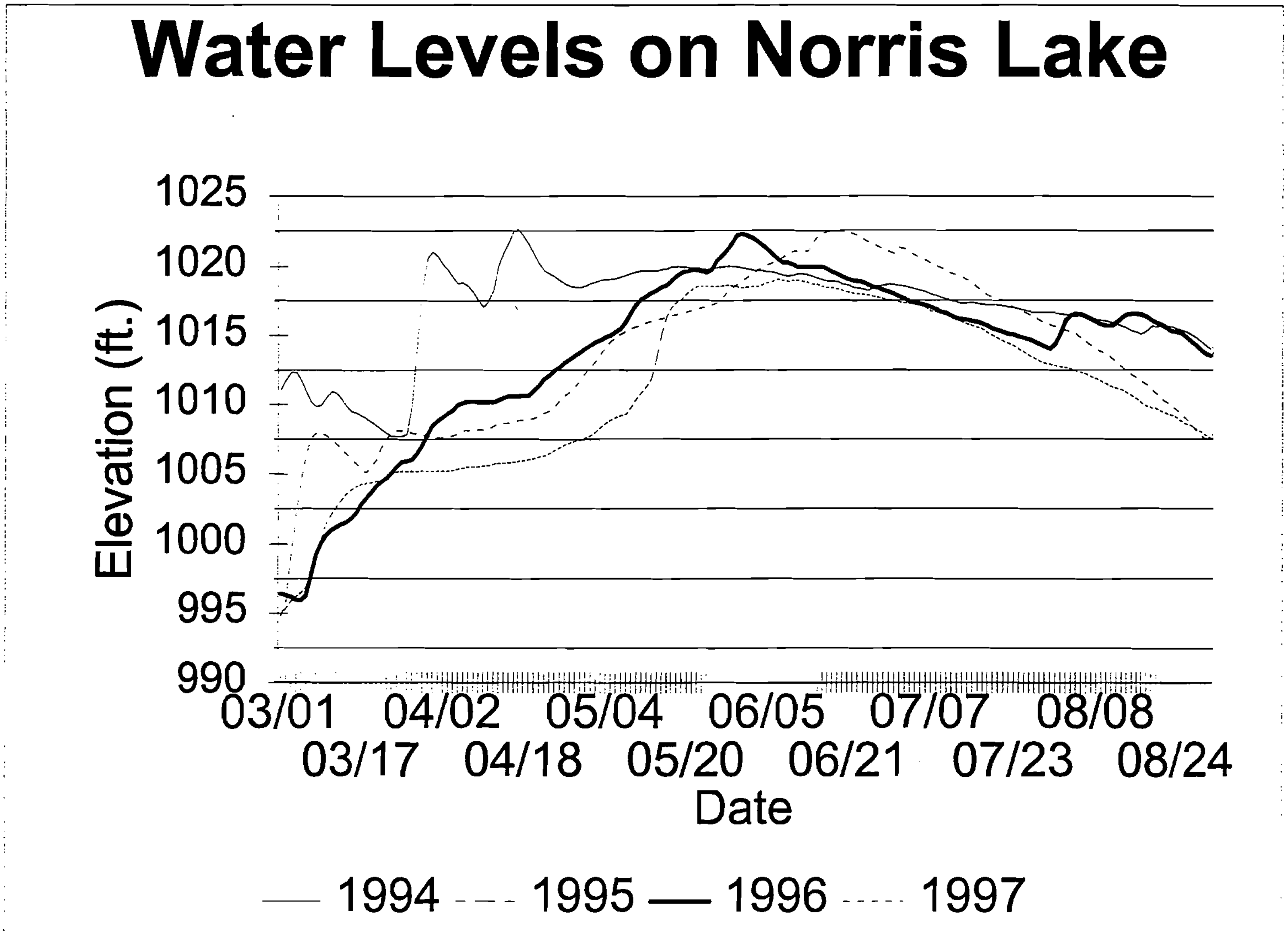

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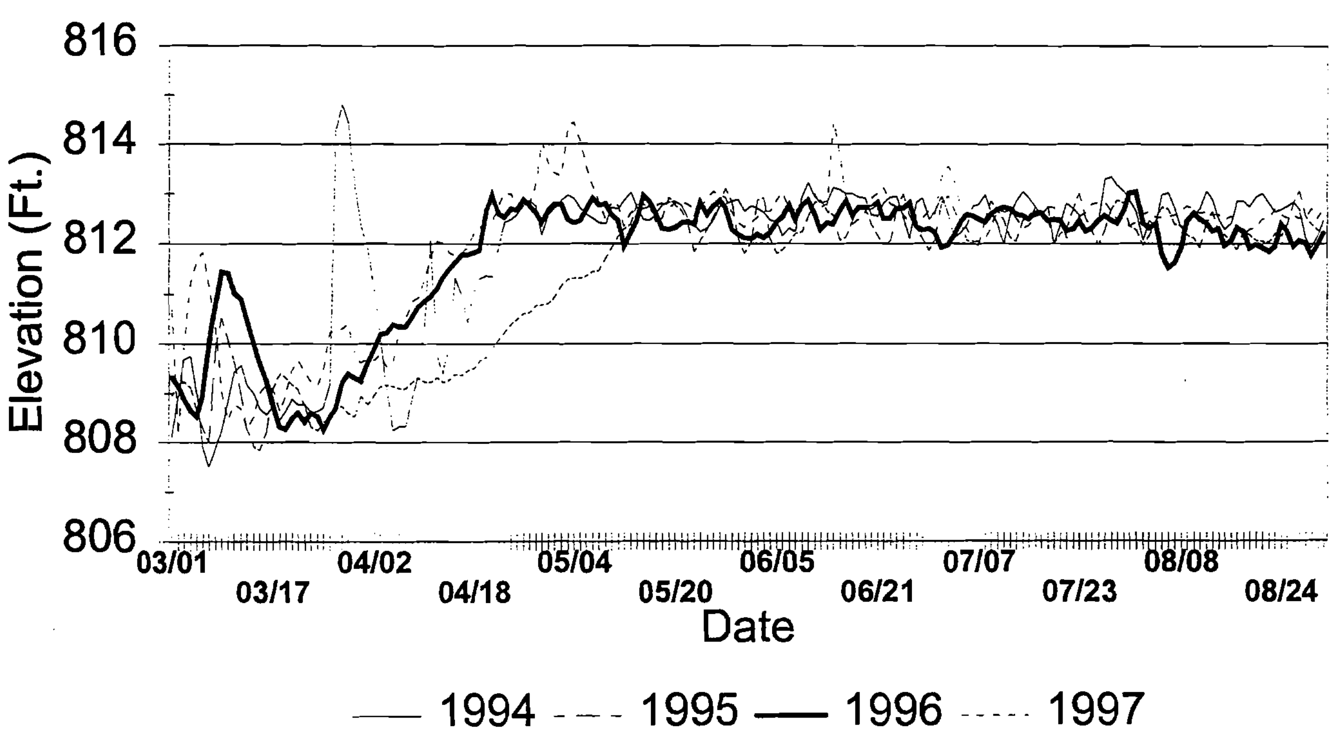
Figure 2.

Water Levels on Fort Loudon Lake



ENDNOTES

1. Controversy over reservoir drawdown policies is not new or unique to TVA, and a small economic literature addressing the issue has developed. A sampling of recent literature includes Ward (1989) who estimated a four reservoir demand system to gauge the economic losses of draining three reservoirs in New Mexico. Cordell and Bergstrom (1993) used contingent valuation methods to estimate the impact of TVA drawdown policies on four TVA lakes in North Carolina. Cameron et al. (1996) examined recreationists' actual behavior in the Columbia River Basin in the Pacific Northwest, finding that low water levels affected the decision to recreate at all, as opposed to affecting the number of times a lake was visited. Ward et al. (1997) used a CES demand system to evaluate various water level policies at New Mexico reservoirs. Fadali and Shaw (1998) looked at a remote lake in Central Nevada, using a nested logit model to estimate anglers' WTP to prevent water volume losses that would cause the fishery ecosytem system to collapse. Shaw et al. (1999) estimated angler losses of a fish kill that resulted from draining a lake in Northern Nevada.

2. This section draws heavily upon Shonkwiler and Shaw (1996), who provide a very lucid development of the double hurdle count data model, which can be extended to a mix of discrete and continuous distribution functions. Yen and Adamowicz (1994) and Haab and McConnell (1996) also present hurdle count data models.

3. The inclusive value is calculated as $IV = \ln [\Sigma_j^J \exp(W_j'\tau) + 0.577]$, where W_j is a vector of characteristics of site j, τ is the coefficient vector, and the summation is over all J sites. Whether or not the inclusive value is the appropriate index to pass from the site choice model to the trips model is the subject of current research. Shaw and Shonkwiler (1999) argue that the inclusive value is not utility theoretic and propose an alternative aggregate demand measure and an alternative price index. They do not, however, propose a quality index.

4. Details about each survey are available upon request.

5. The sample was relatively uniform in its composition between the different years: 25.3%, 26.7%, 25.4%, and 22.6% from 1994, 1995, 1996, and 1997 respectively.

6. The exception is Hiwasee Reservoir in the southwestern mountains of North Carolina, about 1 hour east of Chattanooga. While this lake is included in the study as an important potential substitute site, it receives only 1.5% of all trips made by East Tennessee reservoir anglers.

7. The catch rate measure used in this study results in the errors-in-variables problem recently highlighted by Morey and Waldman (1998). The solution proposed by these authors does solve this problem, but Train et al. (1998) demonstrate that the Morey-Waldman solution only works when there are no omitted site attributes, measurement error in other variables or

"other random events". If these problems cannot be eliminated, then the Morey-Waldman method introduces correlation between the residuals and the catch rate coefficient. In effect, the analyst trades one type of bias for another. Train et al. conclude that the standard procedure "...is consistent under weaker and more realistic assumptions...". The standard procedure is adopted for our study.

8. See Jakus et al. (1997) and Jakus, Dadakas, and Fly (1998) for other reservoir fishing models that have included fish consumption advisories as a site characteristic.

9. This problem is why Cameron et al. (1996) and Cordell and Bergstrom (1993) had to use hypothetical valuation methods to augment their actual behavior models.

ESTIMATING MULTIPLE-NESTING-STRUCTURES IN A SINGLE RANDOM UTILITY MODEL: AN APPLICATION TO FRESHWATER FISHING

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INTRODUCTION

Nesting is the most frequently employed method for overcoming the restrictive properties of the random utility model (RUM). Models without nests assume the errors across all alternatives within the choice set are independently and identically distributed (i.i.d.), which is often an unrealistic assumption. Nested models are less restrictive because they construct groups of alternatives which share similar, but unobserved characteristics. Nested models impose the i.i.d. assumption on alternatives within the same group, but across groups the error distribution can vary.

In order to utilize the advantageous statistical properties of the nests, the researcher usually chooses a single behavioral model that describes the anglers decision process. Within these single-nesting-structure models, the behavioral model is often viewed as the process that generates the similar, but unobserved characteristics shared by alternatives within a nest. However, the nesting structure can also be a purely statistical artifact of the data and not necessarily behavioral. For example, fishing sites may be grouped into rivers and lakes. Anglers might be described as first choosing whether to fish at a lake or a river based on personal characteristics. Once that decision has been made, the angler then chooses among the sites within that nest. An alternative view of the nests might be that the quality of the data for the site characteristics varies significantly between rivers and lakes, which induces different degrees of correlation among the errors for each type of site.

Alternative single-structure nested models are often viewed as mutually exclusive; the researcher must choose one nesting structure. However, Kling and Thomson (1996) have shown that the choice of nesting structure can have a significant impact on the welfare calculations. Therefore, the choice of nesting structure needs to be made carefully. The Kling and Thomson results show that the most "natural" structure, based on type of fishing trip, does not perform as well as other, more counter-intuitive models. This lends support to the view that the appropriate nests are statistical and not behavioral. It also complicates the researcher's job because it may not be possible to find the most appropriate nesting structure by relying on economic intuition.

Our paper develops a flexible method for determining the appropriate nesting structure that overcomes some of the limitations of single-nesting-structure models. It also suggests there may be a behavioral basis for designing nesting structure. Alternative nesting structures need not be mutually exclusive; different structures may apply to different groups within the sample. Rather than impose one decision process and nesting structure, we estimate a multiple-nesting-structure RUM based on a finite mixture approach (Shonkwiler and Shaw, 1997, Titterington, Smith and Makov 1985). Section 1 compares the single and multiple-nesting-structures and shows how the later significantly reduces the i.i.d. assumption. Section 2 describes the data. Section 3 describes the model results. Section 4 describes the welfare calculations. Section 5 describes future research.

1. NESTING STRUCTURES

One consequence of assuming that errors are independently and identically distributed in a conditional logit model is the independence from irrelevant alternatives (IIA) property. That is, the model prescribes the ratio of choice probabilities between two alternatives to be solely a function of the characteristics of the two alternatives. Whether or not the assumption is justified is an empirical question. In general, it is better not to impose this assumption a priori.

Single-structure nested models reduce, but do not eliminate this restriction. For example, suppose we nest fishing trips based on whether they occur at rivers or lakes and assume that there are only two sites of each type. Using Morey's (1997) basic notation for a repeated nested model[1], the choice probability ratio for the first lake site (Lake_1) vs. the first river site (River_1) is:

$$\frac{P(\text{Lake}_1) = \dfrac{L_1(L_1^{SL}+L_2^{SL})^{(1/SL)-1}}{(L_1^{SL}+L_2^{SL})^{1/SL}+(R_1^{SR}+R_2^{SR})^{(1/SR)-1}}}{P(\text{River}_1) = \dfrac{R_1(R_1^{SR}+R_2^{SR})^{(1/SR)-1}}{(L_1^{SL}+L_2^{SL})^{1/SL}+(R_1^{SR}+R_2^{SR})^{(1/SR)-1}}}$$

[1] We use Morey's notation for the similarity coefficient. Morey's SR is equivalent to McFadden's $1/(1-\sigma_R)$

where $L_i = \exp(X_i\beta)$, the exponentiated utility index for lake site i=1,2 , and $R_i = \exp(X_i\beta)$ for river sites i=1,2. and SR and SL are the similarity coefficients for alternatives within the nest. For choices across nests, this expression simplifies to:

$$\frac{P(Lake_1)}{P(River_1)} = \frac{L_1(L_1^{SL} + L_2^{SL})^{(1/SL)-1}}{R_1(R_1^{SR} + R_2^{SR})^{(1/SR)-1}}$$

Therefore, this probability ratio allows for dependence on alternatives other than $Lake_1$ and $River_1$ and the IIA assumption is not imposed. However, the probability ratio between the two lake sites is simply L_1 / L_2 and therefore imposes the IIA assumption.

A multiple-nesting-structure model further reduces reliance on the IIA assumption. The multiple-nesting-structure model employs a finite-mixture approach to estimate which nesting structure is most appropriate for each trip in the sample. This approach adds another layer to the nesting structure to estimate the probability a trip belongs within a particular nest. Figure 1 provides an overview of the approach. Suppose we have two alternative nesting structures—A, which is river vs. lake, and B, which is major vs. non-major fishing site, as defined by a popular angler resource book. Instead of forcing the researcher to choose between the two nesting structures, the multiple-nesting-structure model uses the characteristics of the individual or the trip to determine which nesting structure is best suited for that trip. The probability of choosing site j on trip i (Pij) is:

$$P_{ij} = (P_i(NSA)P_{ij}^A + (1 - P_i(NSA))P_{ij}^B$$

where,

$$P_i(NSA) = \frac{1}{1 + \exp(\lambda Z_i)}$$

$P_i(NSA)$ is the probability that a trip is best described by nesting structure A and is a function of the characteristics of the trip Z_I. Only if $P_i(NSA)$ = 1 or 0 for all cases would either of the single-structure nested models be preferred to the multiple-nesting-structure model.

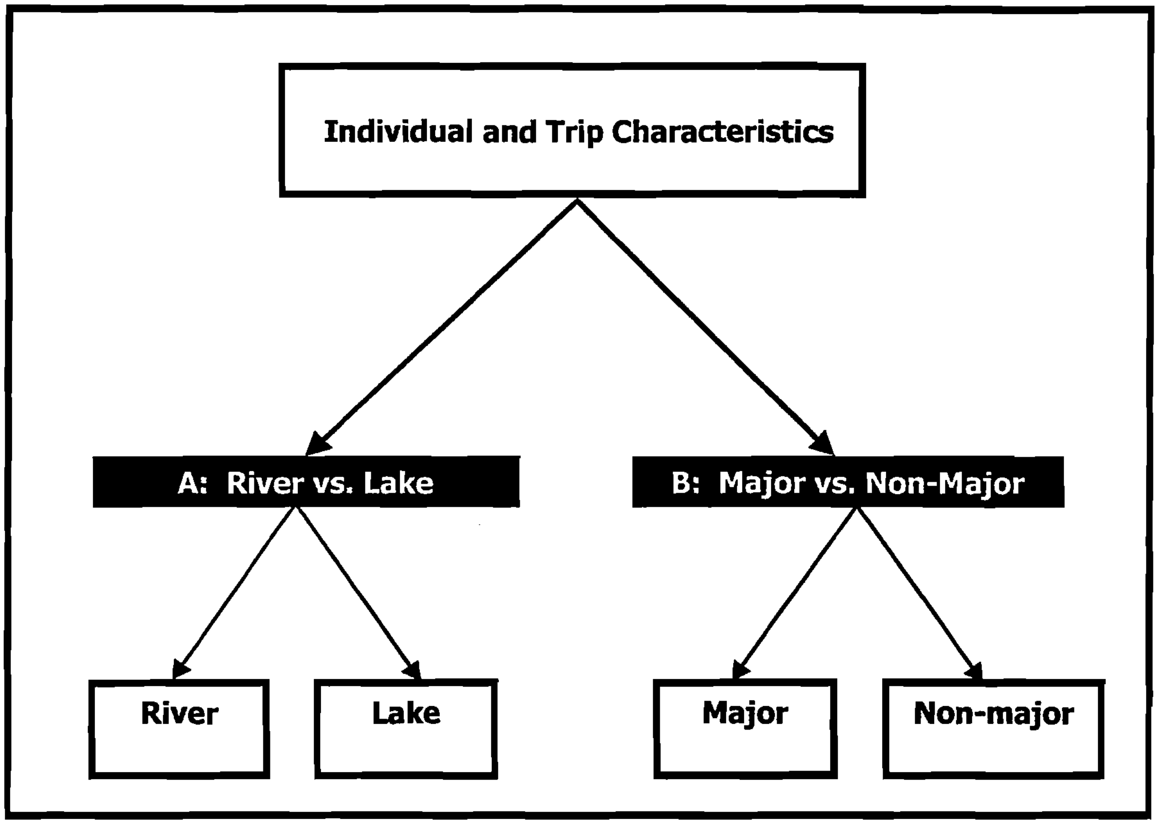

Figure 1.
Multiple-Nesting-Structure Model

The reduction in the need for the IIA assumption using this model can be seen by looking at the probability ratios. Using our original example, nesting structure A is still rivers vs. lakes, but in nesting structure B (major vs. non-major fishing water body), the nests are $Lake_1$, $River_1$ and $Lake_2$, $River_2$. The probability ratio for $Lake_1$ and $Lake_2$ no longer depends on only those two alternatives:

$$\frac{P(Lake_1)}{P(Lake_2)} = \frac{L_1[P_i(NSA)(L_1^{SL} + L_2^{SL})^{(1/SL)-1} + (1 - P_i(NSA))(L \quad + R_1)^{(1/SM)-1}]}{L_2[P_i(NSA)(L_1^{SL} + L_2^{SL})^{(1/SL)-1} + (1 - P_i(NSA))(L_2 + R_2)^{(1/SNM)-1}]}$$

In general, the IIA assumption is only necessary between alternatives that are in the same nests in both nesting structure A and B for trips that have identical $P_i(NSA)$. Overall, the multiple-nesting-model will significantly reduce the instances where IIA is assumed to be valid.[2]

The potential improvement from a multiple-nesting-structure model can also be seen when comparing hypothesis testing of multiple vs. single structure models.[3] Of course, there are many ways in which the data can be nested in a single-structure model. Typically, one would estimate several models then choose a single structure based on whether or not the similarity coefficients are correct (>1) or based on a model selection criteria such as the Pollack and Wales (1991) likelihood dominance criteria (LDC).

The LDC is used for comparing "non-nested" models. Here the term "non-nested" is unrelated to the nesting as described above. "Non-nested" models are situations in which we want to compare two competing models and one model cannot be stated as a restricted version of the other model. Also, LDC can be used when a composite model that incorporates both models cannot be designed and when the hypothesis that one of the two models performs better cannot be tested.

For example, a model with no nesting structure can be directly compared to a nested (e.g., river vs. lake) model because the no-nest model is a restricted version of the river/lake nested model. A no-nest model imposes the restriction that SL and SR are both equal to one. Therefore, it is a restricted version of the nested model, and standard hypothesis tests can be employed.

[2] A single-structure model could also be based on four nests: major lakes, non-major lakes, major rivers, and non-major rivers. We are currently exploring the relationship between this model and the multiple-nesting-structure approach.

[3] We are grateful to Kerry Smith for pointing this out.

However, when comparing two nested models, neither nesting structure is a restricted version of the other, and the standard hypothesis tests do not apply. An alternative approach would be to construct a single composite model (Davidson and McKinnon, 1993):

$$P_{ij} = (\alpha)P_{ij}^{A} + (1-\alpha)P_{ij}^{B}$$

where the two superscripts refer to the probabilities estimated under alternative nesting structures A, B. Therefore, we've created a composite model that incorporates the two alternative models. If the composite model could be estimated, the result α=1 would support accepting nesting structure A while the result α=0 would support acceptance of nesting structure B.

The LDC recognizes that it is often impossible to parameterize and estimate a composite model. Pollack and Wales describe the conditions under which the differences in the log-likelihood between two competing non-nested models are large enough to assume that one model dominates the other with adjustment for the difference in the number of parameters in the two models. Therefore, the composite model need not be estimated, but the models can be compared. This is the approach used by Kling and Thompson.

A multiple-nesting-structure model can be viewed as a composite model that obviates the need to choose among competing nested models. $P_i(NSA) = \alpha$ and only if α=1 or 0 for all cases would either of the nested models be preferred to the composite model. In other words, the two alternative single-structure nested models are restricted versions of the multiple-nesting-structure model. This additional flexibility should improve the performance of the RUM models.

2. DATA

This study uses data from a 14 month panel survey of Montana anglers from July 1992-August 1993. The respondents were recruited using random-digit dialing and 75% of anglers agreed to participate. Once recruited, the respondents were sent a trip diary every two months in which to record details of their fishing trips. The respondents then were called and asked to read back the information from their trip summaries to the interviewer. The response rates for each of the seven panels range from 61 to 78 percent. In total, 2,919 trips were reported. After removing trips that lack key information and trips lasting for more than one day, 1,473 trips remain for use in this analysis. Table 1 provides demographic information of survey respondents, and Table 2 provides key information on the trips used in this analysis.

Table 1.
Demographic Information on Survey Respondents

VARIABLE	MEAN	STANDARD DEVIATION
AGE	41	15.19
INCOME ($1992)	26,011	17,995
OWNBOAT	0.43	0.50
FEMALE	0.37	0.48

Table 2.
Trip Characteristics

VARIABLE	MEAN	STANDARD DEVIATION
TARGET TROUT	0.65	0.48
MAJOR SITE	0.63	0.48
RIVER SITE	0.51	0.50

The choice set for this model is comprised of 182 river sites and 71 lake sites. In most cases, the lake sites are defined around a single lake. River site definitions are based on Montana River Information System river reaches, the smallest segments of each river. The fishing sites are characterized using the variables listed in Table 3.

Table 3.
Description of Site Variables

VARIABLE	DESCRIPTION	MEAN	STANDARD DEVIATION
Specific to Lake Sites			
BIODUM	Dummy variable for lakes with "abundant" fish.	0.54	0.50
CGCIRC	Number of campgrounds relative to circumference of lake.	0.14	0.24
LOGAREA	Log of the surface area of the lake.	5.59	2.11
Specific to River sites			
BIOMASM	Biomass rating measure of pounds per 1,000 feet of river.	82.94	154.85
SRAMILE	Number of State Recreation Areas per mile of river reach.	0.03	0.07
LCTYMILE	Number of large cities (pop. > 10,000) within 30 miles of river reach, divided by reach length.	0.06	0.09
LOGLNGTH	Log the length of reach in miles.	2.59	0.71
AESMDUM1	Aesthetics rating for rivers.	0.20	0.40
Specific to both Lake and River sites			
MAJOR	Dummy variable for site defines as major fishing sites.	0.35	0.48
RIVER	Dummy variable for river sites	0.72	0.45
TRIPCOST	Costs of trip calculated as trips costs plus maintenance costs plus oil costs.	19.83	17.14

3. RESULTS

For this analysis, we first nest the data using two different schemes: river vs. lake and major vs. non-major.[4] The model results are reported in Tables 4 and 5. In both models, all the variables have the expected sign and are significant at the 90-percent confidence level with the exception of BIODUM in the lake nest of the river vs. lake nesting structure. The Pollack and Wales LDC (1991) points to the river vs. lake nesting structure as the better model as its log-likelihood is -4698 while the major vs. non-major nesting structure yields a log-likelihood of –4851. The river vs. lake model has just one more parameter and the improvement in the log-likelihood passes the χ^2 test at all normal levels of significance. In addition, the similarity parameters (SR and

SL) are both greater than one in the river vs. lake model, which is consistent with utility-maximization theory. For the major vs. non-major nesting structure, SNM is less than one. This condition supplies further evidence that the river vs. lake nesting structure is the more appropriate of the two.

Table 4.
Results from River vs. Lake Nesting Structure

NEST	VARIABLE	PARAMETER
R	AESMDUM	0.54**
R	LOGLNGTH	0.06**
R	SRAMILE	0.96**
R	MAJOR	0.34**
L	LOGAREA	0.16**
L	BIODUM	0.28
L	MAJOR	0.22**
Both	TRAVCOST	-0.05**
	SR	1.84**
	SL	2.06**

Mean LL = -3.189 McFadden's R^2 = 0.42

Table 5.
Results from Major vs. Non-Major Nesting Structure

NEST	VARIABLE	PARAMETER
M	BIODUM	1.12**
M	LOGAREA	0.30**
M	LAKE	-1.31**
NM	BIODUM	0.50**
NM	LOGAREA	0.28**
NM	LAKE	-0.69**
Both	TRAVCOST	-0.10**
	SM	0.81**
	SNM	1.06**

Mean LL = -3.293 McFadden's R^2 = 0.40

The results for the model utilizing the multiple-nesting-structure model are presented in Table 6. This model uses the same specification as the two separate

[4] The designation of a site being a major site comes from "The Angler's Guide to Montana" by Michael S. Sample (1984).

nested models. The similarity coefficients for all four nests are now all greater than 1 and consistent with utility maximization. The log-likelihood for the multiple-nesting-structure model is significantly higher than log-likelihoods for the two single-structure models. This suggests that it would be inappropriate to impose either structure individually on the entire data set.

The results clearly show that different nesting structures apply to different trips. The positive and significant coefficient on OWNBOAT implies that trips by people owning boats are more likely to be best modeled using the major vs. nonmajor nesting structure. The negative and significant coefficient on TARGET TROUT shows that trips taken to target trout are best modeled using the lake vs. river nesting structure. Boat owners who target trout are slightly more likely to fall into nesting structure B. Across the entire sample the average $P_i(NSA)$ is .601, which means that more trips fall into the river vs. lake nesting structure. This is consistent with the result that the single-structure river vs. lake nest works better than the major vs. non-major. Other variables such as age, income and gender were not significant in the model.

As stated previously, nests are generated to group sites believed to have similar but unobserved characteristics. The multiple-nesting-structure model allows the similarities to be in the eye of the beholder, i.e. the trip-taker. Additionally, the discovery of behavioral indicators of appropriate nesting structure provides evidence that nesting structure has an important behavioral component.

Table 6.
Results from Multiple-Nesting Structure Model

STRUCTURE	NEST	PARAMETER	ESTIMATE
		CONSTANT	-0.30
		OWNBOAT	1.71**
		TARGET TROUT	-1.61**
River vs. Lake	R	SR	1.25**
	L	SL	1.58**
Major vs. Non-Major	M	SM	1.28**
	NM	SNM	1.56**
Both	Both	TRIPCOST	-0.07**
River vs. Lake	R	AESMDUM	0.85**
	R	LOGLNGTH	0.13**
	R	SRAMILE	1.48**
	R	MAJOR	0.45**
	L	LOGAREA	0.02
	L	BIODUM	1.09**
	L	MAJOR	0.36*
Major vs. Non-Major	M	BIODUM	0.15
	M	LOGAREA	0.32**
	M	LAKE	0.94**
	NM	BIODUM	0.05
	NM	LOGAREA	0.39**
	NM	LAKE	-0.03

Mean LL = -3.122 McFadden's R^2 = 0.44

4. WELFARE CALCULATIONS

Competing models often are compared based on welfare measures. The compensating variation is computed using a simulated change in a policy-related variable such as catch rate or a particular site closure. The potential problem with this approach is that the results may be sensitive to the policy variable or the site chosen for the simulation. To provide a more comprehensive comparison of the models, we simulate the closure of each of the 253 sites and calculate change in compensating variation.

For the multiple-nesting-structure model, we simply multiply the CV from each nesting structure by the probability that the trip is in that nesting structure:

$$CV_{ij} = P_i(NSA_i)CV_{ij}^A + (1 - P_i(NSA))CV_{ij}^B$$

Table 7 summarizes the results for the three estimated models. For this set of models, there are no discernable differences between the average site values from the models. However, the range of estimates from the multiple-nesting-structure model has a larger standard deviation than the other models. This may indicate that this model provides more sensitive estimates by allowing CV estimates to differ by angler and type of trip.

Table 7.
Consumer Surplus Estimates from Site Closures

NESTING STRUCTURE	MIN	MEAN	MAX	STANDARD DEVIATION
River vs. Lake	$0.001	$0.046	$0.908	0.0804
Major vs. Non-Major	0.001	0.048	0.436	0.0604
Multiple	0.001	0.047	0.983	0.0954

5. CONCLUSION

Multiple-nesting-structure models provide a flexible method for further relaxing the restrictive assumptions of a conditional logit model. Rather than imposing a single nesting structure on data, the approach developed here allows the data to determine which structure best applies to each trip. The results suggest that multiple-nesting-structure models may outperform single-nested models, although for this application there is not a significant impact on the welfare calculations.

This analysis also suggests that there may be a behavioral basis for determining nesting structure. Montana anglers who own boats appear to group sites according to waterbody type, i.e. river or lake. In contrast, anglers who target trout are more likely to group sites according the quality of the site. The extent to which sites share similar, but unobserved, characteristics appears to be a function of the characteristics of the potential users of the site.

To further refine multiple-nesting-structure approach, our future research will focus on identification conditions. Testing this approach with other datasets will provide additional insights into this issue. Identification can be a problem with any probabilistic allocation of the sample among alternative model structures. Therefore, we also intend to compare this probabilistic approach with a deterministic approach, whereby trips are assigned to nesting structure based on responses to survey questions about their decision process. This should provide a better understanding of the behavioral basis for nesting structures.

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META-ANALYSIS OF OUTDOOR RECREATIONAL USE VALUE ESTIMATES: CONVERGENT VALIDITY TESTS

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ABSTRACT

We update the Walsh et al. (1988) literature review of outdoor recreation economic studies to the present and merge our database with MacNair's (1993) coding of the Walsh et al. (1988) review. The database we use for analysis has 163 studies providing 741 outdoor recreation use value estimates. We then perform meta-analysis on the data to develop models for use in benefit transfer. Unbalanced panel models were tested on the data, finding no significant panel effects. Several OLS models are developed coinciding with different geographic divisions of the US studies, including a national model and four census region models. Convergent validity testing was performed on each model, assessing their precision in predicting the raw average values for each recreation activity in each defined geographic zone. While the census region models have the best statistical fit to the data, they are less robust to changes in the magnitude of explanatory variables under benefit transfer scenarios than the national model. We also compare the national model's precision to a simple national average value transfer and find that for point estimates alone, the simple transfer is as accurate as using the national meta model. However, meta provides the ability to adapt the values to recreation activities and recreation settings outside the bounds of the data set.

I. Purposes of Meta-Analysis

A. Traditional Uses

Meta-analysis was originally developed to understand the influence of different methodological and study specific factors on the outcomes of the studies and provide a statistical summary and synthesis of past research. The first two meta-analyses by Walsh et al. (1989, 1992) and Smith and Karou (1990) sought to explain the variation in consumer surplus per day estimated from contingent valuation and travel cost methods. More recent applications of meta-analysis for this purpose include groundwater (Boyle, et al., 1994), air quality via the hedonic property method (Smith and Huang, 1995), endangered species (Loomis and White, 1996), visibility (Smith and Osborne, 1996), price elasticities of water (Epsey et al., 1997), health effects (Desvousges et al., 1998), and recreational fishing (Sturtevant et al., 1998). Desvousges et al. (1998) and Sturtevant et al. (1998) also investigate panel data estimators.

B. Benefit-Transfer

A more recent use of meta-analysis is to more systematically utilize the existing literature for the purpose of benefit transfer. Essentially, the meta regression equation coefficients estimated using available study sites could be used to "forecast" benefits at unstudied policy sites. Thus, rather than use an average of a few point estimates from past studies, the meta equation has at least three advantages. First, it utilizes information from a greater number of studies providing more rigorous measures of central tendency sensitive to the underlying distribution of the study values. Second, methodological differences can be controlled for when calculating a value from the meta-analysis equation. Third, by setting the independent variables in the levels specific to the policy site, the analyst is potentially accounting for differences between the original studies and the policy studies. These advantages may sum up to better measures of central tendency than averaging approaches. Thus benefit transfer using a meta-analysis equation shares some of the potential advantages of benefit function transfer espoused by Loomis (1992).

In 1998, an entire workshop on meta-analysis for the purpose of benefit transfer was held at the Tinbergen Institute in Amsterdam. Krichhoff's paper (1998) illustrates the basic approach of using an estimated meta equation to predict consumer surplus values. She then evaluated the relative accuracy of the meta-analysis derived benefit transfer as compared to the original study and a benefit function transfer. She found that multi-site benefit functions outperformed meta-analysis, but that meta-analysis outperformed single-site benefit function transfer. However, in light of the bias of her evaluation criteria toward benefit function transfer, she concludes that the use of meta-analysis for benefit transfer is still encouraging. Sturtevant et al. (1998) support this conclusion by showing that, in general, estimates from the meta-analysis are more precise than point estimate transfers.

The purpose of our paper is to contribute to the refinement and testing of meta-analysis as a benefit transfer tool. To do this, we first update the meta-analysis of Walsh et al. (1988. 1989, 1992) with additional studies and investigate the empirical importance of the panel nature of the reported study values. Second, we perform an evaluation of the relative accuracy of the meta-analysis derived estimated benefits.

II. Econometric Issues in Meta-Analysis Estimation

A. Panel Nature of Data with Multiple Estimates from Same Study

Many of the recreation studies reported multiple estimates for targeted outcomes, such as benefit estimates for a sample population, subset of the sample population, different activities, or different sites. Multiple observations from the same source may be correlated and the error processes across several of these studies may be heteroskedastic. In the presence of panel effects, the classical OLS and maximum likelihood estimators may be inefficient and their estimated parameters biased.

The classical OLS model is:

$$y_i = \mu + \beta' x_i + \varepsilon_i, \tag{1}$$

where *i* indexes each observation, y is the dependent variable (in this case, consumer surplus per person day adjusted to 1996 dollars), x is a vector of explanatory variables including methodology, site, and user characteristics, and ε is the classical error term with mean zero and variance σ^2_ε.

A generic panel model is:

$$y_{ij} = \mu_j + \beta' x_{ij} + \varepsilon_i \tag{2}$$

where *j* indexes the individual study. Accounting for the panel nature of the data when estimating a statistical model is important because of the potential unobserved correlation of a unit's multiple observations. Classical regression models are inefficient if they cannot account for this correlation of the observation unit's multiple responses, if said correlation is present. An additional twist on the panel nature of the data is that it is unbalanced, that is, there are not a uniform number of observations from each unit. Each study has at least one, but can have several value estimates.

B. Candidate Panel Models: Fixed Effect and Random Effect

The panel data effects can be modeled as either having a unit-specific constant effect or a unit-specific disturbance effect.[1] The fixed effect model treats the panel effect as a unit-specific constant effect. The group effect parameter, μ_j, in the case of the fixed effect model, takes on the form:

$$y_{ij} = \alpha_j d_{ik} + \beta' x_{ij} + \varepsilon_i, \tag{3}$$

[1] Desvousges et al. (1998) identify candidate models for meta-analysis as being an equal effect model (the classical OLS), a fixed effect model, a random effect model, and a Bayesian approach. Sturtevant et al. (1998) test a fixed effect, random effect, and a separate variances model (no common error term). We test the equal effect, fixed effect and random effect models.

where d_{ik} is a dummy variable taking on a value of one for all observations where i = k. The first term can be reduced to α_j, signifying a group effect constant for each study in our meta-analysis. The fixed effect model is simply the classical regression model with unit-specific constants.

The random effect model treats the panel effect as a unit-specific disturbance effect. The group effect parameter, μ_j, in the case of the random effect model, can be written as:

$$y_{ij} = \alpha + \beta' x_{ij} + \varepsilon_{ij} + \mu_j, \tag{4}$$

where μ_j is the unit-specific disturbance effect and has a mean zero and variance σ_μ^2. Each study has an overall variance:

$$\mathrm{var}[\varepsilon_{ij} + \mu_j] = \sigma^2 = \sigma^2_\varepsilon + \sigma^2_\mu. \tag{5}$$

The random effect model is a generalized regression model with generalized least squares being the efficient estimator.

Two test statistics aid in choosing between classical OLS, fixed effect, and random effect models: Lagrange multiplier statistic and chi-squared statistic. Breusch and Pagan's Lagrange multiplier statistic tests whether a group effect specification is significant (H_0: $\mu_j = 0$). Hausman's chi-squared statistic tests the random effect model against the fixed effect model (H_0: μ_j as a random effect; H_1: μ_j as a fixed effect).

C. Pooled vs. Disaggregated Models: Hypothesis Tests

An additional issue with this data is whether all the studies can be pooled to estimate a single national model, or whether separate geographic regional models should be estimated. Separate regional models are preferred if the regions are structurally different in either the intercept parameter(s) or slope parameters. A Chow test (F-test) can be performed on the data to determine if the data can be pooled to estimate a national model, or whether regional models should be estimated. The hypotheses are:

$$\begin{aligned} &H_0\text{: National model, } \alpha_n\text{'s} = \alpha_m\text{'s, and } \beta_n\text{'s} = \beta_m\text{'s,} \\ &H_1\text{: Regional models, at least one } \alpha_n \neq \alpha_m\text{, or at least one } \beta_n \neq \beta_m\text{,} \end{aligned} \tag{6}$$

where *n* subscripts the estimated regional model coefficients and *m* subscripts the estimated national model coefficients. That is, if at least one region-specific constant or region-specific slope parameter is different from the others, then pooling the data to estimate a national model imposes a restriction on the coefficients.

III. Testing the Performance of Meta Equations for Benefit Transfer

A. In-Sample Comparisons Involving Individual Study Values

One means to evaluate the relative accuracy of the predictions from the meta-analysis equation is to compare the predictions to the actual individual study values. While the model R^2 provides some indication of goodness of fit, our real interest is in whether the dollar magnitude of the errors would be acceptable for a benefit transfer exercise. Further, we are interested in whether the meta-analysis estimated values might be less subject to small sample errors likely to arise from simply averaging the few available studies for that recreation activity in that region.

B. In-Sample Comparisons Involving Regional Average Values

Some government agencies perform benefit transfer by relying upon a set of standardized "unit day values". The USDA Forest Service has done this since 1980 using their Resources Planning Act (RPA) values. In the last decade, these values are specific to groups of similar activities and region of the country. The U.S. Bureau of Reclamation and U.S. Army Corps of Engineers have relied upon the U.S. Water Resources Council Unit Day Values (U.S. Water Resources Council 1979, 1983) for decades.

Recently, the USDA Forest Service has investigated the possibility of using consumer surplus estimates from a meta-analysis equation to fill in the missing values in their recreation activity by region table. Thus another evaluation is to compare original study values averaged by recreation activity and region to the meta-analysis equation's estimate of these same values for cells in the table which have original study values. This may provide some indication of the relative accuracy of using the meta-analysis equation to fill in the missing values in the table.

C. Evaluation of Out of Sample Accuracy: Within Time Period and Out of Time Period

Another way to evaluate the performance of a statistical model is to compare its estimates to those from original studies that were not used to estimate the model. These observations can be from the same time period or literally out of the sample time period. In this reporting of results, we only test the performance of the meta-analysis models by means of the in-sample comparisons, providing a form of convergent validity testing.

IV. Data Sources

Since we are updating the previous literature review effort of Walsh et al. (1988), new valuation studies performed since they completed their effort were collected. Thus, we limited our search to studies from 1988 to the present. Study values for years prior to this are obtained from MacNair's (1993) database previously assembled for the USDA Forest Service. We also added studies from Walsh et al. (1988), not used by MacNair (1993).

A. Data Search and Limitations

We searched a wide range of electronic databases including the American Economic Association's Econ Lit, First Search Databases, the University of Michigan-Dissertation and Master's Abstracts, NTIS and Water Resources Abstract Index. Unpublished or "gray literature papers" were also searched using W133 Proceedings from 1987 to 1996, Carson et al.'s (1994) CVM bibliography as well as our own collections of working papers, conference papers and reprints.

We focused primarily on studies in the U.S. but included Canadian studies as well (with appropriate currency conversion). Studies in Europe or the rest of the world were not included as the recreation settings are quite different than North America.

We did not look for or emphasize fishing studies as these are subject of two previous significant literature reviews: (a) Sturtevant et al. (1996); and (b) a joint effort directed by Kevin Boyle and Industrial Economic Incorporated (Markowski et al, 1997). Our initial study coding sheet was patterned after Markowski et al.'s (1997) to maintain comparability. Thus we concentrated our effort on activities that had not been previously studied such as rock climbing, snowmobiling, mountain biking as well as activities commonly valued by agencies such as the USDA Forest Service or U.S. Bureau of Reclamation. Therefore, saltwater boating or ocean activities were not given great emphasis either.

B. Coding Procedures

A master coding sheet was developed and used to code the studies we collected, and to guide the recoding of the MacNair (1993) database. The main database of values that underlies the averages contains 126 fields, with the last field being a comment section. There are six main coding categories:

1) complete citation to the study;
2) the benefit measure (original value, deflated to 1996$, adjusted to common units);
3) the nature of the benefit measure (e.g., WTP vs WTA, mean vs median);
4) details of CVM application if CVM used;
5) details of TCM application if TCM used; and
6) study location details (e.g., whether National Forest, Park, State Park, etc.), environment type (e.g., forest, wetland), recreation activity, etc.

We also recorded the geographic region of the country for the study and whether it provided an estimate of a site-specific, state, regional or national average recreation use value. Census Regions represent the four Assessment Regions (Northeast, Southeast, Intermountain and Pacific, as well as a separate region for Alaska) for USDA Forest Service, RPA purposes. We also recorded the USFS Regions (e.g., R1=Montana and No. Idaho; R2=WY and Colorado; R3=Arizona and New Mexico; R4=Nevada, Utah and So. Idaho; R5=California; R6=Oregon and Washington; R8= Southeastern U.S.; R9=Northeastern U.S.; R10=Alaska; note R7 does not exist).

In the past, the RPA average recreation values were reported per USFS Region. However, this results in two problems: (a) very small sample sizes per activity/region cell; and (b) numerous activity/region cells with no average value (due to the lack of any original studies). To address both of these problems, the USFS Regions were aggregated into the Census Regions. In the database, individual study values are identified by both Census Region and USFS Region, so users can sort the data to compute their own values for a USFS Region if desired.

All study values were updated from their original study year (not publication year) values to

1996 using the Implicit Price Deflator. Originally there were slightly more than 170 individual studies that produced slightly more than 750 individual values. A couple of Random Utility Model estimates were on a choice occasion basis and we were unable to determine a way to convert them to a per day value using the information provided in the publication, thus dropping them from the database. Additional studies were removed from the database because they did not report enough information to convert their reported units to a per day basis. Therefore, we ended up with 163 studies providing 741 individual values. We examined these remaining studies for outliers, or per day values for an activity which were more than two standard deviations from the activity mean value. These outliers were removed from the calculations of regional average values (table 1), which are based on 701 individual estimates. Due to recoding of MacNair's (1993) values into our categories, we have studies ranging back to as early as 1967, although the bulk of the values are from the 1980's and 1990's.

Table 1 provides the average consumer surplus per day estimates for the 22 primary recreation activities defined by the USDA FS RPA. These estimates are a simple averaging of the individual study reported values with region and activity segregating them.

V. Statistical Results

All of the subsequent models were estimated using LIMDEP software. Table 2 lists and defines the variables tested across the models. Out of the 741 observations recorded, 672 had reported enough information to fully code for each of the variables listed in table 2. These 672 observations were provided from 131 separate studies. The number of estimates per study ranged from 1 to 134. If there is correlation among these multiple observations for each study, then OLS assumptions are violated. While these studies may provide estimates that relate to tests of methodology, different sites, or different activities, there may still be unobservable, yet systematic effects of the study on their estimates. Panel models can account for these unobservable systematic effects.

ii) In-sample Comparison Involving Regional Average Values Results

The rest of the Treatments (B, C, E, F, G, and H) compare predicted regional values per activity to the raw average regional values per activity. Treatments B (national model) and F (CR model) are directly comparable since they both use national mean values of the explanatory variables to predict average regional values. Table 7 provides the overall results of the assessment. Treatment B of the national model had a grand average absolute difference of 41% for all activities and Treatment F of the CR models had a grand average absolute difference of 118%. For an activity in a region, the difference estimates ranged from –80% to 234% for Treatment B, and from –2567% to 513% for Treatment F. This result makes sense since the mean values used to adapt the regression models are based on national averages for each activity, which is the level of development for the national model. More variability is introduced to the CR model transfers because the national mean values are not sensitive to regional model differences.

Treatments C (national model) and G (CR models) are directly comparable since they both use census region mean values of the explanatory variables to predict average regional values. Table 7 provides the overall results of the assessment. Treatment C of the national model had a grand average absolute difference of 60% for all activities and Treatment G had a grand average absolute difference of 122%. For an activity in a region, the difference estimates ranged from –75% to 299% for Treatment C, and from –391% to 809% for Treatment G. This result does not support previous conclusions drawn and does not meet expectations. We expected that using CR specific mean values for adapting the regressions to predict average values would be more precise than using national averages, and that these CR specific mean values for the explanatory variables would better fit the CR models. The opposite is true. The national model is more robust to perturbations in adapting the model for benefit transfer than the CR models. Because of this and the Chow test results, we decided to assess the CR models further.

Treatment E arose because we wondered if the CR models, despite better statistical fit to the data, were as volatile in predicting benefits if even more specific values were used for the explanatory variables. Therefore, in Treatment E, the CR models are adapted to benefit transfer by using the most precise mean values available for the explanatory variables. Mean values in this treatment are for studies on an activity in a given region. For example, when predicting a value for camping in CR1, we used the mean value for the independent variables from just camping studies in CR1. One disadvantage of this approach is that where there is no data, the model cannot be adapted to that region. Table 7 provides the overall results of the assessment. Treatment E of the CR models had a grand average absolute difference of 9% for all activities. For an activity in a region, the difference estimates ranged from –64% to 67%, which is significantly different than the other treatments.

Treatment H arose because we thought that maybe the CR models would be more responsive to a mixture of mean values for adapting the models for benefit transfer. We speculated that methodology would be invariant to application across the models, but that site characteristics would be somewhat unique to each region. Therefore, we used the national mean values from an activity for methodology variables and census region mean values for site characteristic variables. Table 7 provides the overall results of the assessment. Treatment H of the CR models had a grand average absolute difference of 118% for all activities. For an activity in a region, the difference estimates ranged from –227% to 990%. This result supports the conclusion that even though the CR models have a better fit for the data, they are not robust to perturbations of the explanatory variables. All treatments of the national model are significantly more precise than comparable treatments of the CR models, and are not significantly different from each other for the national and CR models. Only when the adaptation of the CR models is specific to the within group characteristics of the models does it perform better than the national model.

C. Efficiency of National Average Value Transfer

We use the same procedure to assess the simple benefit transfer of a national average value of a recreation activity to a region. That is, we calculate the percent difference between the national average value to the raw average value of an activity in a region. The grand absolute average difference for this transfer approach is 38%, which is not significantly different than the national model. For an activity in a region, the difference estimates ranged from –62% to 269%, which is also similar to differences for the national model. What is gained over the simple national average value transfer with the national model approach is that the national model approach provides the ability to adapt the model to the unique characteristics of the policy site. For example, a value for camping near a lake is needed but not all of the studies behind the regional average or national average are based on lake camping. One could then adapt the national model by turning the lake variable 'on' (setting equal to one), providing a value for camping near lakes.

VII. Conclusions

Several criteria have been suggested for selecting candidate studies for benefit transfer.[7] Desvousges et al. (1998) grouped these into three distinct categories: scientific soundness, germaneness, and richness of detail. Likewise, the quality of a meta-analysis will be dependent upon these criteria. We were strictly interested in the quantitative aspects of a meta-analysis and did not make any qualitative decisions concerning the studies we included in our analysis. However, studies which did not have sufficient richness of detail reported in them could not be included in the meta analysis since observations were missing on variables.

Meta-analysis as a benefit transfer tool provides several advantages over simple point estimate, average value, or benefit function transfers. First, it utilizes information from a greater number of studies providing more rigorous measures of central tendency sensitive to the underlying distribution of the study values. Second, methodological differences can be controlled for when calculating a value from the meta-analysis equation. Third, by setting the independent variables at the levels specific to the policy site, the analyst is potentially accounting for differences between the original studies and the policy studies.

While meta-analysis is a conceptually sound approach to benefit transfer, the quality of original research and full reporting of data and results is as necessary a component to critical meta-analysis as the statistical methods used. A meta-analysis can be no better than the data that it is

[7] See Brookshire and Neill (1992), Rosenberger (1998), and Desvousges et al. (1998) for a summary of these criteria and other issues related to benefit transfer. A special issue of *Water Resources Research* (1992) was devoted to the issue of benefit transfer.

built on. The ability of meta-analysis to capture nuances in the data – differences between sites, user populations, and/or affected activities – is dependent upon not only the quality of the original studies, but also on the sheer volume of studies conducted. One of the limitations of our meta-analysis is the lack of an adequate number of studies for certain recreation activities. Separate meta-analyses of different recreation activities, given enough observations, may provide models that are more robust to factors affecting them, and therefore an increased ability to function for benefit transfer.

Our database of outdoor recreation use value studies contains over 131 studies providing more than 700 use value estimates. We estimate several models from this data and test the convergent validity of the models' ability to accurately predict the raw value from the averaging of the available estimates. We found that while the regional models statistically fit the data better than the national model, they also have the greatest variability in predicting values than the more robust national model. This implies that in addition to sensitivity of these regional models to variability within the bounds of the data, they are probably more sensitive to 'noise' emanating from outside the bounds of the data set. We also found that the simple transferring of a national average recreation value is as precise as using the national model to predict values for transfer. However, the national model has the advantage of being controllable for factors outside of the existing database and specific to the policy site. However, we did not out-of-sample test these models or apply them under real transfer conditions.

There are different approaches to using existing information for benefit transfer when original data collection is not possible or not warranted. Our database and meta-analysis provides for each of these approaches. First, study specific values, or an average of a subset of the available estimates, can be accomplished by sorting the database on those studies deemed relevant to the issue at hand. Second, the simple average values per activity per region (table 1) can be transferred to a policy site. Third, the national average value of an activity can be transferred (table 1). Fourth, the meta-analysis predicted value for an activity in a region can be transferred to the policy site. And fifth, the meta regression equations (table 5) can be adapted to specifics of the policy site and issue to predict a recreation use value. As Desvousges et al. (1998) remind us, an important component in any benefit transfer is the involvement and judgment of the transfer analyst. While it would be nice to have a purely mechanistic approach to benefit transfer, this is not the case. Meta-analyses will probably never be a panacea for valuation needs. But it can be another important tool for analysts to add to their toolbox.

Several studies have performed convergent validity tests on benefit transfer trials (Loomis, 1992; Loomis et al., 1995; Downing and Ozuna, 1996; Kirchhoff, Colby and LaFrance, 1997; Kirchhoff, 1998). While the evidence provides some confidence in pursuing benefit transfers, with several cases producing values very similar to the 'true' values (as low as a few percentage points), in other cases the disparity between the 'true' value and the transfer value are quite large (in excess of 800%). On average, we found our national model to predict values within about 40% of the average value of the relevant studies. Individually, we found the difference between predicted values from the national model to the 'true' values to range from –80% to 299%. The regional models predicted, on average, values in excess of 100% of the 'true' values, and ranged from more than –2000% to 990%, depending on the treatment of the model.

We also tested the data for panel effects based on study and did not find any such effects. The only effect external to our model specification is an outlier effect. Therefore, a classical OLS regression was used to estimate the different models.

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Appendix Table A-1. Unbalanced Panel Data Models.

VARIABLE	OLS	FIXED EFFECTS	RANDOM EFFECTS
CONSTANT	-3.9427	α_i^b	43.299
	(24.65)[a]		(28.18)
METHOD	-32.951***	-9.4462	-23.534*
	(9.791)	(17.38)	(13.34)
DCCVM	10.720**	11.149	14.193
	(4.910)	(15.49)	(9.726)
ZONAL	-13.256	-0.49514	-2.6643
	(8.844)	(18.10)	(12.23)
INDIVID	2.4391	14.173	14.468
	(8.790)	(13.13)	(11.10)
TTIME	1.1009	-9.5802	-8.5826
	(6.185)	(9.769)	(7.954)
SUBS	-22.950***	4.4830	-11.211
	(4.987)	(11.60)	(7.371)
MAIL	8.0480**	-31.479	2.0925
	(3.335)	(36.77)	(8.275)
LINLIN	12.782*	26.022	21.847*
	(7.774)	(18.19)	(11.33)
LOGLIN	5.6491	-6.2556	-3.7053
	(7.499)	(15.19)	(10.15)
LOGLOG	5.1274	-5.2087	-5.5307
	(8.755)	(16.08)	(11.72)
VALUNIT	-10.364***	-10.847	-3.4213
	(3.492)	(15.96)	(8.278)
TREND	1.0187***	-24.437	0.30565
	(0.3311)	(18.55)	(0.6988)
SPECACT	8.3077	10.349	11.514
	(11.50)	(15.89)	(13.51)
FSADMIN	-13.085***	1.9487	-3.0723
	(4.426)	(4.953)	(4.547)
R1	18.582*	3.1767	0.2422
	(9.644	(10.20)	(9.668)
R2	4.5550	1.8732	-3.5341
	(7.754)	(8.680)	(8.207)
R3	-7.9537	0.82868	-4.9432
	(11.10)	(12.40)	(11.63)
R4	10.521	3.3670	-0.05561
	(7.805)	(8.414)	(7.993)
R5	2.8964	2.5269	-2.3637
	(11.19)	(11.21)	(10.64)
R6	-1.7217	-5.9663	-11.118
	(8.377)	(8.925)	(8.486)
R8	1.5595	-0.5165	-6.1615
	(8.963)	(9.739)	(9.222)
R9	8.0652	-2.6286	-7.0506
	(7.818)	(8.865)	(8.388)
LAKE	-14.957**	25.892**	0.8703
	(7.255)	(11.22)	(8.484)
RIVER	24.676***	40.798**	23.562**
	(6.986)	(18.62)	(10.35)
FOREST	-5.2554	-4.4470	-1.811
	(3.947)	(5.710)	(5.023)

Appendix Table A-1. Continued.

VARIABLE	OLS	FIXED EFFECTS	RANDOM EFFECTS
PUBLIC	21.858*** (8.246)	-1.7534 (18.80)	4.8764 (12.40)
DEVELOP	-3.8083 (6.171)	-2.4187 (9.433)	0.18788 (7.234)
NUMACT	0.60509** (0.2837)	-2.3539 (3.311)	-0.41916 (0.8073)
CAMP	16.720 (12.36)	9.9388 (13.83)	5.9536 (12.30)
PICNIC	9.8333 (16.00)	-7.526 (14.27)	-6.6941 (13.24)
SWIM	7.7678 (15.20)	-0.5456 (15.18)	-2.0466 (13.88)
SISEE	13.679 (11.75)	22.654 (16.79)	20.382 (14.31)
NOMTRBT	-0.21016 (13.64)	-9.7448 (14.18)	-8.679 (12.39)
MTRBOAT	5.4431 (11.44)	42.789** (17.86)	23.106 (14.39)
HIKE	4.7191 (13.23)	-21.774 (15.42)	-14.635 (13.39)
BIKE	0.5711 (17.29)	25.788 (19.69)	21.963 (17.69)
DHSKI	12.699 (16.72)	-27.130 (17.84)	-25.248 (15.73)
XSKI	0.05126 (15.29)	-19.791 (18.51)	-18.576 (16.41)
BGHUNT	14.916*** (3.961)	-20.574** (8.797)	-7.7539 (7.555)
SMHUNT	8.7073 (7.841)	-32.017*** (10.37)	-18.485** (9.124)
WATFOWL	10.701** (5.255)	-26.652*** (9.306)	-12.531 (7.997)
FISH	7.5243 (4.821)	-28.965*** (9.237)	-14.708* (7.885)
GENREC	5.9279 (12.42)	22.051 (28.95)	1.4363 (16.98)
INCOME	-0.72574 (0.7228)	0.5910 (0.5379)	0.447 (0.5283)
AGE	43.527 (86.24)	-84.099 (76.92)	-30.047 (73.98)
EDUC	70.395 (56.30)	-45.803 (42.09)	-23.054 (41.37)
POPUL	-0.05591 (0.04035)	-0.0082 (0.0324)	-0.0164 (0.0312)
BLACK	24.318 (24.66)	-7.0473 (18.50)	1.7083 (18.12)
HISPAN	72.294** (34.33)	41.579 (27.50)	43.789* (26.59)

Appendix Table A-1. Continued.

VARIABLE	OLS	FIXED EFFECTS	RANDOM EFFECTS
ADJ R2	0.25	0.68	0.03
F-STAT	5.40*** [50, 621]	8.81*** [180, 491]	na
N	672	672	672

[a] Standard errors in parentheses, except for F-stat where degrees of freedom are given in brackets.
[b] 131 individual group effects constants were estimated.
*p<0.10; **p<0.05; ***p<0.01 (all variables are p<=0.20).
Dependent variable is CS per person day.

Appendix Table A-2. Raw Average Recreation Values per Person/Day for Use in Model Assessments.

(Review: 1967 - 1998)

			CR=1		CR=2		CR=3		CR=4		CR=45		CR=5		
Act.#*	Activity	N	NO.EAST	N	SO.EAST	N	Intermtn	N	Pacific	N	Pac+AK	N	Alaska	N	National only
1	Camping	3	$30.07	6	$20.35	10	$22.42	3	$113.88	3	$113.88			1	$28.61
2	Picnicking	1	$55.22	1	$37.24	1	$32.30	2	$73.95	2	$73.95			1	$15.69
3	Swimming	5	$37.21					1	$14.95	1	$14.95			1	$20.67
4	Sightseeing	14	$34.23	17	$31.27	19	$32.42	4	$39.56	5	$39.56	1	$13.09	1	$18.83
5	Off-road driving							1	$33.64	1	$33.64			1	$19.94
6	Float boating			2	$8.40	4	$72.42	4	$21.69	4	$21.69			1	$21.61
7	Motor boating	2	$52.44			2	$68.76			1	$15.13	1	$15.13	1	$38.70
8	Hiking/Backpacking	2	$45.01	2	$109.96	3	$37.42	5	$21.88	6	$21.88	1	$12.93	1	$20.87
9	Biking	1	$34.11	1	$56.27	2	$58.89							1	$17.61
10	Downhill Ski					2	$23.23	1	$20.90	1	$20.90			1	$19.61
11	Cross Country Ski	2	$28.83			1	$11.71							1	$13.20
12	Snow mobiling					1	$36.23								
14a	Big game hunting	55	$45.22	26	$35.99	68	$45.05	11	$43.77	16	$43.77	5	$52.40	2	$104.90
14b	Small game hunting	3	$36.73			13	$25.75							2	$103.02
14c	Waterfowl hunting	23	$32.09	11	$17.70	19	$36.74	5	$24.51	6	$24.51	1	$60.08		
15	Fishing	42	$31.63	13	$27.74	39	$42.49	15	$36.97	16	$36.97	1	$39.22	4	$37.26
16	Wildlife Viewing	43	$27.06	25	$30.38	26	$34.03	13	$40.33	18	$40.33	5	$54.57		
17	Horseback riding													2	$16.08
19	Rock Climbing	2	$85.74			3	$42.04								
20	General Recreation	8	$15.21	6	$14.65	20	$33.51	12	$16.93	15	$16.93	3	$11.84		
21	Others			2	$26.21	5	$57.92								
	Total # of cases	206		112		238		77		95		18		21	

N = Number of cases

* = Act-13, Snow playing, and ACT-18, Resorts, were not considered as there was no study.

Investigating Heterogeneity of Preferences in a Repeated Logit Recreation Demand Model Using RP Data*

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Discussion paper
If a more recent version exists it can be found at
http://spot.Colorado.EDU/~morey/index.html

Abstract

Estimating a demand system under the assumption that preferences are homogeneous will lead to biased estimates of individuals' parameters and significantly different expected consumer surplus estimates if preferences are in fact heterogeneous. This paper investigates with revealed preference data several different parametric methods to incorporate heterogeneity in the context of a repeated discrete-choice logit model. The first is the classic method of assuming preference parameters to be functions of individual characteristics. Allowing parameters to vary across individuals as a function of individual characteristics results in preference heterogeneity that persists across choice occasions. Second, a random parameters method is proposed, where preference parameters have some known distribution. Random parameters logit causes the random components to be correlated across choice occasions and, in a sense, eliminates IIA. Finally, two methods are proposed to relax the assumption that the unobserved stochastic component of utility is identically distributed across individuals: the logit scale as a function of individual characteristics and randomization of the logit scale. The randomization of the scale, which is new, allows noise levels to vary across individuals without the added burden of explaining the source using covariates, or the added econometric difficulties with individual-specific scales. Results from an application to Atlantic salmon fishing suggest that imposing homogeneity leads to significantly different expected consumer surplus estimates.

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1. Introduction

A common assumption in most consumer demand models is homogeneity of preferences. That is, all individuals are forced to have the same preference parameters in the deterministic portion of utility, and the variance of the random component is assumed to be iid. These strong assumptions will lead to biased parameter estimates on the individual level and significantly different mean consumer surplus estimates if in fact preferences do vary across individuals, which is expected to be the rule rather than the exception.[1]

Several methods are proposed and applied to relax the assumptions of preference homogeneity. Each is discussed separately in the context of a repeated discrete-choice multinomial logit model (MNL) using revealed preference (RP) data,[2] and the advantages and disadvantages are considered. The classic method allows demand parameters to vary as a function of observable socioeconomic characteristics of the individual. Classic models include income-effects models and all other models that make model parameters a function of individual characteristics.

Another technique assumes that preference parameters for all individuals are drawn randomly from some known PDF, although the parameters for any specific individual are unknown. *Random paramters logit (RPL)* is appealing because it allows correlation of random disturbances across choice occasions, and because, in a sense, it eliminates independence of

[1] Fowkes and Wardman (1988) demonstrate by simulation that the mean of sample-wide parameters, if allowed to vary, may be statistically significantly different from parameters estimated assuming no taste variation, in the presence of nonlinearities.

[2] Morey and Rossmann (1999) have recently examined heterogeneity of preferences using choice experiment data. A *nested* logit model is not used as the basic model because some of the heterogeneous methods can be used instead of nesting to allow for more general substitution patterns without imposing an *a priori* nesting structure. The basic MNL form is used to conduct likelihood ratio tests; otherwise the models would not be comparable on the basis of nested tests.

irrelevant alternatives (IIA). Independent stochastic terms and IIA are two often-criticized assumptions inherent in a logit model. The *nested* logit model was originally introduced to reduce IIA assumptions and to increase the flexibility of the MNL. See, for example, Herriges and Kling (1995), Morey (1999), Morey et al. (1993), Morey et al. (1995), and Shaw and Ozog (1996). However, the nesting structure still imposes restrictions on substitution patterns, while the RPL can allow even more general substitution patterns.

New methods to allow the variances of the random components to vary across individuals or groups are also discussed. While permitting variances to differ across data collection methods is fairly common in the literature (e.g., RP versus stated preference (SP)), there exist very few studies that allow variances to differ across individuals to incorporate heterogeneity. These methods allow some individuals to have "noisier" choice patterns than others, which is empirically indiscernible from *parameter proportionality*, where the demand parameter vector only varies across individuals by a factor of proportionality. A new contribution of this paper is the introduction of a random logit scale parameter, which allows noise levels to vary across individuals without the added burden of having to explain the source of the different levels as a function of individual characteristics that may or may not be correlated with the noise, or the added econometric difficulties associated with trying to estimate individual-specific scales.

The application is a utility-theoretic repeated logit model of Atlantic salmon fishing site choice and participation. Empirical results demonstrate that accommodating heterogeneity significantly improves model performance in each case, and restricting preferences to be homogeneous often leads to significantly different consumer surplus estimates. For models that include socioeconomic characteristics to address heterogeneity, preferences vary as a function of these characteristics in plausible ways.

In Section 2, the techniques are described in detail, along with discussion of relevant existing applications. In Section 3, the methods are applied to the Atlantic salmon recreation demand model, and extensions are proposed throughout. In Section 4, conclusions are presented.

2. Techniques to accommodate preference heterogeneity

Heterogeneity of preferences can be addressed either through the vector of demand parameters (denoted β_i for individual *i*) or by assumptions about the distribution of the stochastic component of utility (or by using multiple methods simultaneously addressing both components). The first two methods mentioned in the previous section, which allow β_i to vary across individuals either as a function of individual characteristics or randomly based on some distribution, take the former approach.

Other techniques pursue the latter by letting error variances differ across individuals, which may reflect different levels of coherence in decision-making or interest in the activity or the included variables. Allowing the variance of the disturbance term to differ across individuals results in the same likelihood function as allowing the preference parameters to vary across individuals up to a factor of proportionality, because the logit scale parameter and the vector of preference parameters are confounded in estimation.

2.1 *A repeated multinomial logit model of recreation demand with homogeneous preferences*

Consider a logit model of recreation demand. On each of T choice occasions, the individual chooses from J alternatives the alternative that provides the greatest utility. The utility individual i receives on choice-occasion t if he chooses alternative j is:

$$U_{jti} = V_{ji} + \epsilon_{jti}, \; j = 1, 2, \ldots, J. \tag{1}$$

Assume the term V_{ji} is deterministic from both the individual's and the researcher's perspective. It is a linear function of β_i and a vector of explanatory variables x_{ji} associated with angler i and alternative j that are time-invariant, so $V_{ji} = \beta_i' x_{ji}$.[3] The ϵ's vary from period to period and across individuals in a way the researcher cannot observe. Assume ϵ_{jti} is independently drawn from a univariate Extreme Value Distribution with the cumulative distribution function:

$$F(\epsilon_i) = \exp[-e^{-s_i(\epsilon_i)}] \tag{2}$$

where s_i is a positive scale parameter (Morey, 1999). This distribution has $E(\epsilon_i) = (0.57721/s_i)$ and $\sigma^2_{\epsilon i} = \dfrac{\pi^2}{6s_i^2}$. The probability that individual i will choose alternative j on choice-occasion t is:

$$Prob_{jti} = \frac{e^{s_i V_{ji}}}{\sum_{k=1}^{J} e^{s_i V_{ki}}}. \tag{3}$$

Given these assumptions, the observed trips (y_{ji}) have a multinomial distribution, and the log of the likelihood function for the N recreators is:

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[3] The model can easily be generalized to allow time-variant explanatory variables, leading to a more complicated likelihood function than that presented here.

$$\ell = \sum_{i=1}^{N} \sum_{j=1}^{J} y_{ji} \times \ln(Prob_{ji}) \qquad \textbf{(4)}$$

Homogeneity of preferences is defined as $\beta_i = \beta$ and $s_i = s \; \forall \; i$. Preference homogeneity implies that the random components are independent and identically distributed, a restrictive assumption which means that the error variances across anglers are assumed to be the same, and also that there is no correlation in random components across choice occasions for a given angler. Homogeneity of preferences also means that choices for a given individual are no more correlated than choices across individuals.

Without loss of generalization, let $s = 1$, the usual assumption in logit models. In Section 3.4, s will be allowed to vary across anglers, introducing heterogeneity in the variance of the stochastic component. It is clear from Equation 3 that allowing to s's to vary across individuals is empirically equivalent to allowing the β's vary up to a factor of proportionality, although the underlying theoretical assumptions are quite different.

2.2 Interacting preference parameters with individual characteristics

This and the following section relax the assumption that $\beta_i = \beta \; \forall \; i$, while maintaining the assumption that $s_i = 1 \; \forall \; i$. The utility angler i receives from alternative j during choice-occasion t is therefore:

$$U_{jti} = \beta_i' x_{ji} + \epsilon_{jti}, \; j = 1,2,\ldots,J. \qquad \textbf{(5)}$$

The random component ϵ_{jti} is iid.

The classic and perhaps most straightforward way to allow preferences to vary across individuals is to interact individual socioeconomic characteristics, such as age, gender, or income, with model parameters (Adamowicz et al., 1998). Pollack and Wales (1992) summarize methods of using demand parameters that depend on demographic variables, based on their earlier work and the work of others. Two applications of this technique are Morey (1981) and Morey et al. (1995). The first is a choice-share model of skiing in Colorado, in which individual demand parameters for ski area characteristics are assumed to be functions of skier attributes. The second is a repeated nested logit model of recreational trout fishing in southwestern Montana, where model parameters are interacted with resident status to allow nonresident anglers to have different preferences from residents. In the latter case, forcing nonresidents to have the same preferences would significantly lower economic values for environmental improvements.

Any model that admits income effects also allows for systematic heterogeneity among individuals as a function of their incomes, and there are a multitude of examples. Morey (1999), McFadden (1996), and Herriges and Kling (1997) discuss the theoretical underpinnings of income effects in logit models, and some empirical examples include Morey et al. (1993 and 1998) and Buchanan et al. (1998). Models with income effects are not investigated here. Also, a new literature is emerging that includes latent constructs and psychometric measures based on individual attitudes and perceptions in addition to demographic factors in discrete choice models. McFadden (1986) initiated work in this area to develop market forecasts. See Boxall and Adamowicz (1998) for an application to explain wilderness park choice, and also Ben-Akiva et al. (1997).

Note that allowing β_i to vary across individuals as a function of individual characteristics results in preference heterogeneity that persists across choice occasions. For example, suppose that the catch parameter in a fishing model is increasing in skill level. An individual with high skill will have a higher than average catch parameter in each of the choice occasions. As a result, the deterministic component of utility, V_{ij}, is correlated across choice occasions.

One reason that an application of this well-known method is included in this paper is because it is useful to compare the assumptions and results to other heterogeneous models, and because a complete treatment of heterogeneous preferences is incomplete without it. The main advantage of this technique is to allow β_i to vary across individuals in a systematic way as a function of individual characteristics. The researcher can predict how different types of individuals are affected by different policies, and consequently reach conclusions about distributional impacts.[4] The primary drawback is that β_i may not in fact vary as a function of observable individual characteristics, and model results are expected to be sensitive to the way in which the parameters and data are allowed to interact.

2.3 Random parameters logit

Another way to incorporate heterogeneity through β is to assume that one or more parameters in the vector is drawn from a known distribution, although the unique values of the parameters for a given individual in the sample cannot be known. RPL is a special case of *mixed logit* because the probability of observing an individual's sequence of choices is a mixture of logits with a prespecified mixing distribution (Revelt and Train, 1997).

[4] Benefits can vary widely as a function of individual type. See, for example, Morey et al. (1998).

Two recreational site choice examples using RPL with revealed preference data are a partial demand system of fishing site choice in Montana (Train, 1998) and a complete demand system of participation and site choice in the Wisconsin Great Lakes region (Phaneuf et al., 1998). Our random parameters model, unlike Phaneuf et al., addresses preferences for unobserved characteristics. Both of these studies find that randomizing parameters significantly improves model fit and significantly affects consumer surplus estimates for changes in environmental quality. RPL has also been applied to choice experiments (conjoint analysis) to model demand for a wide array of commodities and environmental amenities, including alternative-fuel vehicles (Brownstone and Train, 1996); appliance efficiency (Revelt and Train, 1997); forest loss along the Colorado Front Range resulting from global climate change (Layton and Brown, 1998); and the level of preservation of marble monuments in Washington, DC (Morey and Rossmann, 1999).

RPL addresses heterogeneity across the population without having to confront the sources of individual heterogeneity, which is both its strength and weakness. As noted by Adamowicz et al. (1998), RPL provides more flexibility in estimating mean utility levels but little interpretability in terms of distributional impacts associated with heterogeneity.

Like interaction, the RPL model specification assumes the β_i's vary across anglers rather than being restricted to be the same as assumed in Section 2.1. The coefficient vector for each individual is expressed as the sum of two components, the population mean vector (b) and an individual vector of deviations (υ_i): $\beta_i = b + \upsilon_i$. By assuming that υ_i is constant over choice-occasions for each individual, the unobserved components of utility become correlated. By allowing for preference heterogeneity in this fashion, the restriction of independence associated with the nonrandom logit model is removed (Phaneuf et al., 1998). Train (1998) expects such persistence in the unobserved factors that affect utility over time and over sites.

If we knew each angler's preferences, the β_i's, the probability of observing angler i's choices over the season would be:

$$P_i = \prod_{j=1}^{J} \left[\frac{e^{\beta_i' x_{ji}}}{\sum_{k=1}^{J} e^{\beta_i' x_{ki}}} \right]^{y_{ji}}. \tag{6}$$

However, the individual deviation υ_i is unobservable. Only the PDF $f(\beta)$ is assumed to be known, so the joint probability of observing angler i's choices conditioned on υ is the integral of Equation 3 over β:

$$P_i = \int_{-\infty}^{\infty} \prod_{j=1}^{J} \left[\frac{e^{\beta' x_{ji}}}{\sum_{k=1}^{J} e^{\beta' x_{ki}}} \right]^{y_{ji}} f(\beta|\theta) d\beta \tag{7}$$

where θ represents the parameters of the distribution of β. V_{ji} is no longer deterministic, but is now a random variable. Analytical evaluation of this integral is generally not possible, but advances in computer simulations allow for easy approximation based on a large number of random draws, R, from $f(\beta)$ using a pseudo-random number generator:[5]

$$SP_i = \frac{1}{R} \sum_{r=1}^{R} \prod_{j=1}^{J} \left[\frac{e^{\beta_r' x_{ji}}}{\sum_{k=1}^{J} e^{\beta_r' x_{ki}}} \right]^{y_{ji}}, \tag{8}$$

[5] If only a small number of elements of β are randomized, other techniques to evaluate the integral such as Gaussian quadrature may be used to increase speed and accuracy (Abramowitz and Stegun, 1965). See Stern (1998) for a discussion of simulated ML and its advantages.

where β_r is a single draw from $f(\beta)$, and SP_i is the simulated probability of observing the individual's choices. The simulated log-likelihood function for the RPL is therefore:

$$s\ell = \sum_{i=1}^{N} \ln(SP_i) \tag{9}$$

Nonrandom logit has the property that the ratio of probabilities of visiting two sites is unaffected by the inclusion of or change in a third site. RPL does not have this property due to the correlation in unobserved utility across choice occasions and alternatives resulting from persistent preference heterogeneity incorporated randomly, rather than deterministically as in the previous section. [6] The absence of this property is clear by examining the ratio of probabilities of visiting sites j and j' on the t-th choice occasion (conditioned on the PDF of β, but unconditional on alternatives chosen on other choice occasions):

$$\frac{\displaystyle\int_{-\infty}^{\infty} \frac{e^{\beta' x_{ji}}}{\sum_{k=1}^{J} e^{\beta' x_{ki}}} f(\beta|\theta) d\beta}{\displaystyle\int_{-\infty}^{\infty} \frac{e^{\beta' x_{j'i}}}{\sum_{k=1}^{J} e^{\beta' x_{ki}}} f(\beta|\theta) d\beta} \tag{10}$$

In a nonrandom logit model, the denominators cancel out, leaving the ratio as a function only of the characteristics of j and j'. In the RPL, the denominators do not cancel out of the integrands because they also contain υ.

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[6] Hausman and Wise (1978) were the first to incorporate correlation across choice occasions in the context of a random probit model. Heterogeneity of preferences was addressed by allowing correlation among the random components of utility across alternatives. At the time, simulation methods were not available, so they were limited to three alternatives because their probit model integrates over utility differences between alternatives rather than parameter distributions as in the RPL.

If IIA means that the ratio of probabilities of visiting two sites is unaffected by changes in a third site, then RPL eliminates IIA. This interpretation is shared by Brownstone and Train (1996), Train (1997), Revelt and Train (1997), and Phaneuf et al. (1998). However, even with random parameters, how an individual ranks any two alternatives is independent of the existence or characteristics of any other alternative. Remember that β_i is not a random variable from the individual's perspective. In this sense, how the individual ranks alternatives is independent of irrelevant alternatives.

2.4 Heterogeneity of the stochastic component

The interaction and RPL methods address heterogeneous preferences by allowing β in the conditional indirect utility functions to vary across the population. Another strategy is to allow for heterogeneity in the stochastic components, the ϵ's. Although it is assumed that all individuals have the same β's, and therefore *expected* behavior of two individuals with the same characteristics would be identical, the assumption that each individual's ϵ's are drawn from the same distribution is relaxed. The assumption that the ϵ's are independent across choice occasions is retained, but different individuals can have different error variances ($\sigma_{\epsilon i}^2$). As a result, different individuals with the same characteristics are allowed to have different levels of noise in their decision-making (for example, see Johnson and Desvouges (1997)).

As discussed in Section 2.1, it is typical to assume that all individuals have stochastic components drawn from the same distribution. Under this assumption, all of the individual scale parameters, the s_i's in Equation 3, are the same and usually normalized to one without loss of generality. To allow for heterogeneity in the stochastic component, this restriction is relaxed, and individual- or group-specific s's are estimated separately, or s can be randomized as in the RPL, the latter being a completely new method proposed in this paper. One scale must be normalized (to one or some other value) to achieve identification in the model. As emphasized above, allowing s to be heterogeneous is empirically indistinguishable from parameter proportionality (Louviere, 1996); that is, all β_i's are scaled up or down proportionately across individuals, as shown in Equation 3. In that sense, the methods in this section are more restrictive than either RPL or interaction. While heterogeneous scales require parameters to vary only up to a factor of proportionality across individuals, the other methods allow more general variation.[7] Note that s_i is inversely proportional to $\sigma_{\epsilon i}^2$. Therefore, an individual with a small (large) amount of noise in the decision process will have a relatively large (small) s_i, and the model will predict the individual's choices relatively well (poorly).

Several studies allow for differing levels of noise in different data sets or resulting from different data-generating processes, rather than to admit unobserved heterogeneity across individuals.[8] Incorporating heterogeneity of preferences through s is a much different exercise with a groundbreaking goal, and which also presents new challenges. For example, when merging k datasets, only k - 1 scale parameters need to be estimated, where k is some small integer; preference heterogeneity may require that a different s_i be estimated for every individual, or subsets of individuals where grouping is nonrandom and based on logic or some expectation.

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[7] Note that when there is only one slope parameter, allowing s or β to vary is equivalent because s and β cannot be separated in estimation.

[8] For example, Swait and Louviere (1993) propose a test for multinomial logit parameter comparisons using identical utility specifications but different data sources. Louviere and Swait (1997) propose a nonparameteric approach for estimating scale parameters when different data sets are aggregated. Swait et al. (1994) use scaling to explain differences in the magnitudes of unexplained variance between SP and RP data from the same respondents in a model of freighter shipping choice under the assumption that the SP data reflects tradeoffs more robustly and therefore contains less noise. Ben-Akiva and Morikawa (1990) also examine the differences between RP and SP data-generating processes using scales.

Alternatively, the new random scale method is an appealing way to circumvent the problems associated with estimating a huge number of individual-specific s's. First, it may be difficult or impossible to estimate a different $\hat{s}_i$ and its standard error for each individual in the sample. RP data sets may have many corner solutions and limited variability across the data, and attempting to estimate individual-specific parameters may be asking too much. A finite ML estimator of s, a prerequisite for consistency, may not exist for those who make purely random choices, or for those whose choices are completely explained by β, because the likelihood function may be continuously increasing as $s_i \to 0$ or $s_i \to \infty$ (Morey and Rossmann, 1999). Second, even if individual scales could be estimated for everybody, they would provide no information on why a given individual's error variance is high or low. Using a random scale parameter in a similar way as the random preference parameters in the RPL allows for heterogeneity across individuals in the variance of the stochastic term, but it requires estimating only enough parameters to characterize the distribution of the scales (e.g., two for the lognormal distribution) rather than n - 1 different individual-specific scale parameters. Further, the random scale does not require estimation of the scale parameter as a function of individual covariates, the specification of which may lead to specification bias.

2.5 *Individual-specific preference parameters*

It is possible in theory to estimate individual-specific models, in which no parameters are shared, if the quantity of data is sufficient and the data exhibit enough variation. Usually the data do not allow identification or estimation of all of the parameters at the individual level. Successful estimation of individual-specific models is most likely using choice-experiment data, with many observations per individual that exceed the number of parameters to estimate.[9] Individual-specific recreation demand models using RP data would be difficult to estimate because of the typical lack of variation in choices and a small number of trips taken by many individuals, which could be avoided by the careful design of a choice-experiment survey (for discussion, see Morey and Rossmann (1999)).

A consistent estimator of slope parameters in models with some individual-specific parameters and some shared parameters does not exist. Chamberlain (1984) demonstrates that a unique feature of the logit formulation is the ability to estimate individual fixed-effect constants without introducing inconsistency in the other shared parameters. This result does not extend to slope parameters, so interaction and RPL are the best alternatives to individual-specific β_i's to admit heterogeneity in V_{ji}.

3. Repeated logit models of Atlantic salmon fishing participation and site choice that allow heterogeneity

The empirical application is a repeated nested logit recreation demand model of Atlantic salmon fishing participation and site choice. The model is utility-theoretic and complete. Each of the techniques is applied to the model, and expected compensating variations are estimated for changes in site characteristics. The basic model assuming homogeneous preferences is developed

[9] For examples, see Johnson and Desvousges (1997); Beggs, Cardell, and Hausman (1981); and Morey and Rossmann (1999). Estimating a large number of individual-specific coefficients is always a daunting task. For example, about one-fourth of the individuals are nonconvergent in the first study, and about half have undetermined coefficients in the second study.

and estimated in Section 3.1. The techniques to add heterogeneous preferences are discussed and applied in Sections 3.2 through 3.4, along with comparisons to the basic model.

3.1 Model 1: A logit model of Atlantic salmon fishing with homogeneous preferences

During a fishing season, an Atlantic salmon angler has a finite number of choice occasions, assumed to be 100,[10] to allocate to nine alternatives, including five salmon river groups in Maine (Penobscot, Machias group, Dennys, Kennebec group, and Saco), three salmon river groups in Canada (Nova Scotia rivers, New Brunswick rivers, and Quebec rivers), and a nonparticipation alternative that allows substitution in and out of fishing.

The data used to fit the logit model are from a sample of 145 Maine anglers who held Atlantic salmon fishing licenses in 1988 and were active at these sites. The data set includes complete trip records on the number of visits each angler took to each of the eight Atlantic salmon fishing areas (y_{ji}). The average angler took about 20 trips to these sites and 14 to the Penobscot River in Maine alone. The data also include exogenous expected catch rates, angler incomes, and fishing costs, the p_{ji}'s, which vary widely across anglers and sites. Trip costs comprise transportation costs, on-site costs such as guides and lodging, and the opportunity cost of time, including fishing, travel, and additional on-site time (e.g., waiting time, overnight time). Finally, the data set includes socioeconomic characteristics for each angler, including age, years of fishing experience, and whether the angler belongs to a Penobscot fishing club. This particular data set was first used to estimate a nested logit model of participation and site choice with income effects in Morey et al. (1992).

[10] Over 97% of sample anglers took 100 or fewer trips during the season.

The deterministic portion of angler i's conditional indirect utility function for fishing at site j, V_{ji}, is a function of a constant (α_0), a dummy (D_j) that equals one if the site is in Canada, the budget per choice occasion (B_i), the trip cost to visit site j (p_{ji}), and the site-specific expected catch rate: $V_{ji} = \alpha_0 + \alpha_{0C}D_j + \beta_0(B_i - p_{ji}) + \alpha_1(1 - D_j)(catch_j) + \alpha_1(\alpha_{1C}D_j)(catch_j), j = 1, 2, ..., 8$. The expected catch rates at the Canadian sites are considerably higher than at the Maine sites. To account for this difference, the catch parameter is a step function constructed by multiplying the catch parameter (α_1) by a catch-scale parameter (α_{1C}) if the site is in Canada. The price parameter, β_0, is interpreted as the marginal utility of money. The conditional indirect for nonparticipation, V_{9i}, is a function of a constant, the budget per-choice occasion spent on the numeraire if fishing is not chosen, and socioeconomic characteristics of the angler: $V_{9i} = \alpha_{09} + \beta_0(B_i) + \alpha_2 age_i + \alpha_3 yrs_i + \alpha_4 club_i$, where *age* is the angler's age, *yrs* is years of fishing experience, and *club* equals one if the angler is a member of the Penobscot fishing club.[11] This is a no-income-effects model; the choice-occasion probabilities are not a function of the budget.

The ML algorithm (version 4.0.18) in Gauss (Aptech Systems, 1996) was used to find the estimates of the parameters that maximize the likelihood of observing the sample trip records, given exogenous trip costs, expected catch rates, and angler characteristics. The parameters are all significant and are reported in Table 1. The estimated parameters indicate that site visitation increases in expected catch and is a decreasing function of trip cost. The catch step function shows that increases in catch are more important when catch is lower (the Maine sites) than when

[11] The three constants α_0, α_{0C}, α_{09} were included to account for the effects of any unobserved variables in the participation decision and the choice of region. The model was identified by setting α_0 equal to zero. Note that because the conditional indirect for nonparticipation is a function of angler characteristics, Model 1 does allow preferences to be heterogeneous in the classic sense to some degree in terms of the participation decision of how much to fish, although the model is called "homogeneous". Modeling participation as a function of demographic variables is common in the recreation demand literature, so this does not represent a new contribution.

Table 1
Parameter Estimates[1]

Parameters	Model 1 Homogeneity	Model 2 Interaction	Model 3 RPL	Model 4 Group scales	Model 5 Random Scale
Fishing parameters[2]					
α_{0C}	4.409 (7.95)	5.124 (13.88)	2.971 (1.87)[6]	3.969 (7.46)	3.993 (7.48)
β_0	-0.0157 (-43.00)	-0.0154 (-42.61)	-0.0175 (-38.34)	-0.0144 (-29.99)	-0.0147 (-30.32)
α_1	15.834 (15.00)	32.363 (9.46)	39.220 (8.21)	14.808 (13.95)	12.466 (11.57)
α_{1C}	0.0620 (4.65)	0.0377 (5.87)	0.133 (1.45)	0.0630 (4.68)	0.0743 (4.51)
Particip. parameters[3]					
α_{09}	2.296 (17.93)	3.726 (11.89)	4.517 (6.03)	2.304 (18.47)	2.322 (18.10)
α_2	0.0200 (12.08)	-0.0121 (-1.87)	-0.0240 (-1.49)	0.0167 (8.94)	0.0218 (11.68)
α_3	-0.121 (-18.09)	-0.152 (-6.04)	-0.0577 (-1.09)	-0.100 (-12.10)	-0.134 (-15.60)
α_4	-0.647 (-13.52)	0.386 (2.14)	-0.404 (-1.01)	-0.868 (-12.80)	-1.0025 (-13.80)
Hetero. parameters					
γ_1 for age		-0.373 (-5.22)	-0.613 (-5.41)		
γ_2 for yrs		-0.437 (-1.55)	1.210 (2.76)		
γ_3 for club		12.314 (6.16)	6.158 (2.14)		
σ_{0C}			6.653 (9.50)[6]		
σ_{09}			1.995 (14.84)[6]		
group scales[4]				1.0 (fixed), 1.270 (20.05), 0.945 (28.33), 1.236 (18.70), 1.087 (28.56), 1.211 (21.53), 1.034 (28.82), 1.117 (25.14)	
σ_{ls}[5]					0.642 (16.15)
Lik. ratio stat. [d.o.f.]	NA	67.96 [3]	4054.20 [5]	47.30 [7]	1956.95 [1]

1 Asymptotic t-statistics in parentheses.
2 Fishing : $V_{ji} = \alpha_0 + \alpha_{0C}Dj + \beta_0(B_i - p_{ji}) + \alpha_1(1 - D_j)(catch_j) + \alpha_1(\alpha_{1C}Dj)(catch_j), j = 1, 2, ..., 8$; α_0 fixed at zero for identification in all models.
3 Participation: $V_{9i} = \alpha_{09} + \beta_0(B_i) + \alpha_2 age_i + \alpha_3 yrs_i + \alpha_4 club_i$.
4 Group 1: young, inexperienced, no club; Group 2: young, inexperienced, club; Group 3: young, experienced, no club; Group 4: young, experienced, club; Group 5: old, inexperienced, no club; Group 6: old, inexperienced, club; Group 7: old, experienced, no club; Group 8: old, experienced, club.
5 σ_{ls} is the standard deviation of $\ln(s)$ in Model 5 where s is the random scale. The mean of $\ln(s)$ is fixed at zero for identification.
6 α_{0C} (α_{09}) is the mean and σ_{0C} (σ_{09}) is the standard deviation of the normal distributions of the Canadian (nonparticipation) contant in Model 3.

catch is high (the Canadian sites). Socioeconomic characteristics such as age, years of fishing experience, and club membership are all important in the participation decision of how often to fish. Older anglers tend to fish less, and those with more years of experience or belonging to a fishing club tend to fish more.

The Penobscot River in Maine is a very popular fishing site with relatively high catch for a Maine site. Expected compensating variations, E(*CV*)s, are estimated for three environmental changes at the Penobscot for all models: doubling the catch rate, halving the catch rate, and elimination of the site entirely. Both improvement and deterioration experiments are conducted because million-dollar fish stocking policies to improve the catch rate and dam projects for hydroelectric power (which would lower the catch rate) are relevant to the Penobscot. The E(*CV*) per choice-occasion for angler i for a logit model with no income effects is simply calculated as:

$$E(CV_i) = (1/\beta_0) \times [V_i^0 - V_i^1], \quad \textbf{(11)}$$

where $V_i = \ln(\sum_{j=1}^{9} \exp(V_{ji}))$, the expected utility per choice occasion, and the superscripts denote conditions before and after the change (Morey, 1999). The total seasonal E(*CV*) is the choice-occasion E(*CV*) multiplied by 100. Seasonal E(*CV*)s and confidence intervals for the mean E(*CV*)s simulated using 500 pseudo-random draws based on the estimated covariance matrix of the parameters are presented for Model 1 in Table 2. The mean seasonal E(*CV*) for doubling the catch, for example, is $2,270, which is consistent with the very avid, serious nature of these recreational anglers who pay high trip costs to go fishing (often in the hundreds or thousands of dollars), and receive high benefits from the activity.

Table 2 Expected Seasonal Compensating Variations for Three Penobscot Scenarios[1]					
E(*CV*)s	**Model 1 Homogeneity**	**Model 2 Interaction**	**Model 3 RPL**	**Model 4 Group Scales**	**Model 5 Random Scale**
Double catch					
mean	\$2270	\$2978[2]	\$3514[3]	\$2300	\$3140
conf. interval	\$1707 to \$2833	\$2345 to \$3301	\$2720 to \$4308	\$1606 to \$2994	\$1976 to \$4304
median	\$2081	\$1489	\$2104	\$2102	\$2748
minimum	\$4	-\$68	-\$26	\$6	\$1
maximum	\$6859	\$17305	\$20561	\$7421	\$10115
Halve catch					
mean	-\$470	-\$521	-\$743[3]	-\$475	-\$164[3]
conf. interval	-\$537 to -\$403	-\$590 to -\$452	-\$908 to -\$578	-\$557 to -\$393	-\$213 to -\$115
median	-\$346	-\$220	-\$391	-\$361	-\$92
minimum	-\$1960	-\$3711	-\$4931	-\$2030	-\$1446
maximum	\$0	\$36	\$13	-\$1	\$0
Eliminate site					
mean	-\$908	-\$939	-\$1312[3]	-\$911	-\$582[3]
conf. interval	-\$965 to -\$851	-\$996 to -\$882	-\$1579 to -\$1045	-\$989 to -\$833	-\$653 to -\$511
median	-\$638	-\$627	-\$900	-\$681	-\$230
minimum	-\$4129	-\$5074	-\$6704	-\$3882	-\$1789
maximum	-\$1	-\$1	-\$1	-\$2	-\$1

1 Confidence intervals for the mean E(*CV*)s were simulated using the estimated covariance matrix.
2 Statistically significantly different from the Model 1 mean at a 10% level of significance.
3 Statistically significantly different from the Model 1 mean at a 5% level of significance.

3.2 Model 2: A logit model with the catch parameter as a function of angler characteristics

In Model 1, only the participation decision is a function of angler characteristics. In this section, the model is generalized by making the site-choice decision also a function of angler characteristics. The catch parameter is interacted with age, years of experience, and the club dummy. The variables are interacted linearly, so the conditional indirects for the fishing alternatives become: $V_{ji} = \alpha_0 + \alpha_{0C}D_j + \beta_0(B_j - p_{ji}) + \alpha_1(1 - D_j)(catch_j) + \alpha_1(\alpha_{1C}D_j)(catch_j) + \gamma_1(1 - D_j)(age_i)(catch_j) + \gamma_2(1 - D_j)(yrs_i)(catch_j) + \gamma_3(1 - D_j)(club_i)(catch_j) + \gamma_1(\alpha_{1C}D_j)(age_i)(catch_j) + \gamma_2(\alpha_{1C}D_j)(yrs_i)(catch_j) + \gamma_3(\alpha_{1C}D_j)(club_i)(catch_j), j = 1, 2, ..., 8$.

Parameter estimates reported in Table 1 indicate members of a fishing club are more concerned with catch, and older anglers are less concerned. Perhaps fishing club members are more interested in the sporting aspect of fishing, and older anglers are more interested in fishing for the pure enjoyment of the activity, regardless of what they catch. Years of fishing experience is not a significant determinant of the catch parameter in Model 2. Two individuals in the sample who are older and are not club members have negative catch parameters, and thus negative E(*CV*)s for catch improvements. Negative catch parameters may simply be an artifact of the model specification. Alternatively, some individuals may in fact have negative catch parameters if omitted variables are correlated with catch. For example, a larger catch rate is expected to be correlated with higher visitation and possibly more congestion at a given site. Perhaps these anglers have a strong preference for isolated sites with few visitors, even if it means catching fewer fish.

A likelihood ratio test shows that Model 2 is statistically superior to Model 1; incorporating heterogeneity matters. Further, although older anglers who are not club members have E(*CV*)s for catch improvements in the interactive model that are lower than in Model 1, the mean E(*CV*)s for the sample as a whole, reported in Table 2, are somewhat larger in absolute value from Model 2 for all Penobscot scenarios, although the medians are smaller. Club members tend to have the highest E(*CV*)s. Perhaps the most important finding is that the range of E(*CV*)s over the sample is much larger in Model 2. Incorporating heterogeneity by making the catch parameter a function of socioeconomic characteristics not only allows the researcher to determine which groups are most affected by environmental changes, but also allows a much wider range of behavior of and estimated impacts on different types of anglers.

3.3 Model 3: A RPL model with interaction

Model 3 is a RPL model that is an extension of Model 2, and therefore uses two heterogeneous methods; heterogeneity of the catch parameter is again accomplished using the same interactions as in the previous section. The constants α_{0C} and α_{09} (the Canadian and nonparticipation constants, respectively) are natural candidates to be random parameters because they represent the effects of all other relevant variables on the participation decision and regional choice (α_0 for Maine is still fixed for identification of the model). It is likely different anglers respond to unobserved variables differently, but the way specific individuals respond is by definition unobservable. Therefore, it is assumed that these constants vary across anglers randomly. They are drawn from normal distributions with means α_{0C} and α_{09} and standard deviations σ_{0C} and σ_{09}, which are estimated in the model. A normal distribution was used for both constants because there are no restrictions on the signs, and because the proportion of possible values decreases for value ranges farther from the mean.[12]

Note that Model 3 allows for systematic variation in the catch parameter because it is reasonably explained in terms of socioeconomic characteristics. It may be more informative for the researcher to be able to explain why parameters differ across individuals in a systematic fashion where possible, as in Model 2 (and again in Model 3) where interaction of the catch parameter with angler characteristics is statistically significant. Therefore, only the constants are randomized in Model 3 with unobserved variables.

A comment is warranted about the choice of the number of repetitions, *R*, an issue that is not examined extensively in the literature. The simulator is unbiased with only one draw of β.

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[12] Distributional assumptions are simply approximations of the true distributions, which are unknown. The normal and lognormal are typically used, the latter to impose restrictions on a parameter's sign. Train (1998) uses a lognormal distribution for the parameter on fish stock to constrain it to be positive (i.e., all anglers are assumed to gain utility from catching fish), but Phaneuf et al. (1998) use a normal distribution. Train (1998) also allows the price parameter to be random and lognormally distributed. Layton and Brown (1998), however, warn of undesirable effects on the distribution of the E(*CV*)s as a result (because the price parameter is in the denominator of the *CV* formula), and hold the price parameter fixed. Phaneuf et al. (1998) also hold the marginal utility of money fixed. RPL results may be sensitive to the distributional assumptions. For example, in the Atlantic salmon model, both normal and lognormal distributions for the catch parameter were investigated in preliminary analyses. E(*CV*)s were found to be highly sensitive to the standard deviation of the catch parameter, especially in the lognormal case. The long right tail combined with the nonlinearity of the E(*CV*) calculation led to unrealistically enormous economic values.

However, increasing the number of repetitions increases the accuracy of the simulator and reduces *simulation noise* (Layton and Brown, 1998).

The number of draws should be large enough so that the model parameters and E(CV)s are insensitive to different random number draws. A total of 2,500 draws was used in the integration simulators in this paper. With this large number, most model parameters did not vary at two or three significant digits. Perhaps more importantly, mean E(CV)s changed by less than 1%, whereas with only 100 draws they changed by more than 10%. Note that this number of draws is considerably larger than the numbers reported in other studies, which range from 250 to 1,000, although one can expect the appropriate number to vary with the study.[13]

Model 3 is statistically superior to both Models 1 and 2 on the basis of likelihood ratio tests, and in addition, σ_{0C} and σ_{09} have highly significant asymptotic t statistics. The parameter estimates for the RPL are reported in Table 1.[14] The values of α_{0C} and σ_{0C} are 2.97 and 6.65, and the values of α_{09} and σ_{09} are 4.52 and 2.00. The ratios of the standard deviation to the mean are 2.24 and 0.44, which match well with the ratios for random parameters in other studies valuing environmental improvements. The range over 20 parameters in 3 studies is 0.40 to 14.29, with a mean of 2.28 and a median of 1.43 (Train (1998), Phaneuf et al. (1998), and Layton and Brown (1998)).

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[13] Brownstone and Train (1996) examine the sensitivity of average probabilities, the log-likelihood function, and parameter gradients to different numbers of draws and different sets of random numbers (i.e., different values for the random number generator seed), but hold the estimated parameters constant as they conduct the tests (i.e., a new model is not estimated for every value of the seed).

[14] In estimation of RPL models, it was found that scaling the parameters was critical to obtain convergence. Parameters should be scaled so that the diagonal elements of the Hessian are roughly the same order of magnitude. Note that the RPL software developed by Kenneth Train was not used to estimate Model 3. In this study, site characteristics do not vary over choice occasions, so the coding of the likelihood function was simpler than that coded by Train. Programs are available from the first author upon request.

For a RPL model, the E(CV) per choice occasion for angler i is obtained by simulating integration of Equation 11 over the PDF of β:

$$E(CV_i) = \frac{1}{R} \sum_{r=1}^{R} (1/\beta_0) \times [V_{ri}^0 - V_{ri}^1], \tag{12}$$

Because seasonal E(CV)s are additive and each component can be integrated separately, the seasonal E(CV) can be computed as the simulated E(CV) per choice occasion multiplied by the number of choice occasions. The mean and median seasonal E(CV)s from the RPL in Table 2 are statistically significantly higher for all scenarios than for either Models 1 or 2, indicating that randomization has a significant impact on economic values. The ranges on E(CV)s are also wider.

Removal of IIA assumptions in terms of probabilities is a desirable property of RPL. Making the regional constants random removes IIA between Maine and Canada, but not within a region. As an example, the percent changes in the probabilities of visiting each of the eight fishing sites on the t-th choice occasion (independent of what occurs on other choice occasions) when the catch rate at the Penobscot doubles are computed. The RPL and nonrandom Model 1 are compared using a representative angler.[15]

The nonrandom model predicts the angler's probabilities of visitation to all other sites on the t-th choice occasion when the Penobscot catch doubles will decrease by the same amount (60%), as a result of IIA. The probability of visitation to the Penobscot will increase by 155%. The particular randomization of the constants in Model 3 results in a feature similar to that of a nested logit model. When the Penobscot catch doubles, the probability of visiting any of the other

[15] For this example, the angler is 63 years old, has fished for 11 years, is a member of the Penobscot fishing club, faces low trip costs to the Penobscot of $37, and took 4 trips to the Penobscot.

Maine sites falls by 76%, and the probabilities for the Canadian sites all fall by 33%. IIA is retained within the two regions, and the higher level of substitution among Maine sites is a reasonable result. The probability of visiting the Penobscot increases by 88% when the catch doubles. To eliminate IIA assumptions entirely, different random α's could be estimated for each alternative, rather than for each region.

3.4 Models 4 and 5: heterogeneity in the stochastic component

Three approaches were investigated to allow heterogeneity in the random component of utility: 1) individual-specific scales; 2) group-specific scales; 3) and a random scale parameter. Compared to Model 1, all three resulted in significant reductions in the likelihood function and different monetary values. The results from two of those models, Model 4 (group scales) and Model 5 (a random scale) are presented below.

A model attempting to estimate individual-specific s_i's was run, but it only converged when a restrictive upper bound was placed on the scales, suggesting that the likelihood is monotonically increasing in the scale for certain individuals. The Hessian for the full model would not invert, although inversion was obtained separately for the β's. When individuals were examined on a case-by-case basis, it was discovered that about 60% had undetermined scales. Johnson and Desvousges (1997) also estimate a model with individual-specific scales using choice experiment data and report difficulties with convergence, although they do not report the proportion of individuals for whom the model did not converge. These findings are not surprising; as noted above, the ML estimator may not even exist for some individuals. Below, two alternatives to the individual-specific method are discussed, both of which have desirable features.

Although Johnson and Desvousges (1997) state that individual-specific scales can indicate whether groups of respondents make random or repetitive choices, or are having trouble with the survey design in the case of choice experiments, the individual scales themselves contain no information about the type of person fitting these categories. An alternative that does allow the researcher to reach conclusions about how scale varies across types of individuals is the use of different scale parameters for different types of groups.[16] This specification is also much easier to estimate because it reduces the number of scales to be estimated considerably.

Model 4 examines whether scales vary significantly based on age, experience, and club status. The Atlantic salmon anglers were divided into eight groups on the basis of the mean values of age and years of experience (47 and 6.5, respectively) and club status. Each angler was assigned a corresponding group-specific scale, and one scale was normalized to one to achieve identification.[17] The estimated parameters are reported in Table 1, and a likelihood ratio test shows that Model 4 is statistically superior to Model 1. For models with s's that vary, E(CV)s are a function of the scales:[18]

$$E(CV_i) = [1/(s_i\beta_0)] \times [\ln(\sum_{j=1}^{9} \exp(s_i V_{ji}^{0})) - \ln(\sum_{j=1}^{9} \exp(s_i V_{ji}^{1}))], \qquad \textbf{(13)}$$

Again, the mean E(CV)s are higher for Model 4, but only slightly as compared to Model 1. They are not statistically significantly different from the Model 1 means.

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[16] Note that an alternative to group scales would be to estimate scales as a nonnegative function of individual characteristics (see, for example, Cameron and Englin (1997) and Morey and Rossmann (1999)).

[17] The scale was fixed for the younger, inexperienced anglers who are not members of a club. They are the most numerous and took approximately the average number of trips for the sample. As a result, they had a large influence on the likelihood function of Model 1, the source of the starting values for Model 4.

[18] If there is only one alternative in each state of the world for the proposal being evaluated, the s's drop out of the formula for E(CV), although the estimation of β is still affected by the presence of heterogeneous scales in the likelihood function.

The club members as a group have the smallest random component variance, which suggests that they are very careful and systematic about fishing decisions. This is consistent with membership of a fishing organization. Of club members, younger anglers have smaller random components, but of the nonclub group, older anglers have smaller random components. The group scales range from 0.94 to 1.27.

Model 4 is estimated under the assumption that β does not vary across anglers, only $\sigma_{\epsilon i}^2$ varies (or similarly, that parameter proportionality holds). Section 2.5 discussed why individual-specific models cannot usually be estimated for revealed preference data, but group-specific models, where all parameters are allowed to vary across groups, may be identified if there are multiple individuals in each group with adequate variability in choices and with each facing a large number of choice occasions. Group models can be used to test the hypothesis of parameter proportionality (which is empirically indistinguishable from testing that preference parameters are the same across individuals, but only variances of the stochastic term differ) by adding up the log-likelihoods across the group models and comparing to a model with group-specific scales (Swait and Louviere, 1993). Parameter proportionality could not be tested here.[19]

Louviere (1996) notes that parameter proportionality is retained consistently across different types of data sets in numerous studies. If that finding extends to types of individuals, then using group scales is an appropriate way to address heterogeneity. Even in cases where parameter proportionality is statistically rejected, Louviere suggests that modeling only error variability will account for most of the heterogeneity. However, the significance of Model 3 suggests that parameter proportionality may not hold. Because s_i and β_i are confounded in a MNL

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[19] Of eight group-specific models using the group definitions listed above, inversion was obtained for only two, primarily because of the small number of anglers in some groups and the small number of trips taken to the Canadian sites (only 34 across the sample, and zero in several groups).

model, if parameter proportionality is rejected, it is not possible to discern whether: 1) both parameters and scales vary, versus 2) just the parameters vary on the basis of individual-specific estimates of s_i and β_i (Swait and Louviere, 1993). However, as noted by Morey and Rossmann (1999), the model would be identified if s_i and β_i were estimated as functions of individual characteristics, or if both s and a subset of the elements of β were randomized.

Model 5 takes a different approach. While it is assumed that the s's vary across people, it is also assumed that they vary unsystematically from the researcher's perspective. Using a similar procedure to Model 3, s is assumed to be a random scale parameter with some distribution. The lognormal distribution is chosen to restrict $s_i > 0 \; \forall \; i$. To obtain identification, the median scale is fixed at one (by setting the mean of $\ln(s) = 0$). Anglers whose choices are rational and appropriately sensitive to characteristics of alternatives will have scale parameters close to the normalized value of one (Johnson and Desvousges, 1997). Those making random choices or those who do not care about the alternatives or their characteristics will have smaller scales; those for whom the model predicts extremely well, or who make repetitive choices, will have larger scales.

Again, 2,500 draws were used to minimize simulation noise. Given the lognormal distribution, the following formulas can be used to determine the mean and standard deviation of the random scale: $E(s) = exp(\sigma_{\ell s}^2/2)$; and $\sigma_s = exp(\sigma_{\ell s}^2/2) \times \surd\,[exp(\sigma_{\ell s}^2) - 1]$, where $\sigma_{\ell s}$ is the estimated standard deviation of $\ln(s)$. The mean s is 1.23, and the standard deviation of s is 1.99. Model 5 is statistically superior to Model 1. E(CV)s were simulated, and again the mean (and median) E(CV)s for doubling the catch rate are larger in absolute value than Model 1. For the site-deterioration scenarios, the means and medians are significantly smaller.

4. Conclusions

Several methods to incorporate heterogeneous preferences have been proposed to generalize the restrictions inherent in assuming homogeneity. These methods address four broad category types of heterogeneity: 1) systematic heterogeneity in the deterministic component of utility; 2) random heterogeneity in the (formerly) deterministic component; 3) systematic heterogeneity in the stochastic component; and 4) random heterogeneity in the stochastic component. While each of these types is dealt with individually in this paper, multiple types could be dealt with at once to reduce model restrictiveness and to allow for a much richer treatment of heterogeneity. One could envision an even more general model that combines the interaction and random parameters of Model 3 with the random scale of Model 5.

An important empirical finding in this paper is that restricting preferences to be homogeneous tends to result in significantly different mean expected consumer surplus and has important implications for the range of expected values as well. In some cases, mean E(*CV*)s for models incorporating heterogeneity were not statistically different from the means from Model 1 that does not have the features. However, in all cases allowing heterogeneity significantly improved model fit, which alone justifies use of the methods. Also, heterogeneity results in larger ranges in the E(*CV*)s across the sample, which is an implication of the model allowing for a wider range in individual behavior.

The systematic heterogeneity methods should be used where possible to allow the researcher to reach conclusions about subgroups of the population, which may be relevant for environmental policy targeting different types of recreationists. Systematic heterogeneity allows the researcher to assess the distributional impacts of policies. However, the random logit scale parameter provides the researcher a way to allow for variation in the distribution of the random component across individuals without the potential biases associated with estimating the scale as a function of covariates, or the difficulties associated with individual-specific scales.

Final model selection can depend on a mix of economic theory and intuition combined with empirical comparisons. In developing a model with heterogeneous preferences, it is important to consider the types of individuals in the sampling frame. How they differ in terms of geographic proximity, socioeconomic variables such as income and education, and avidity in terms of dependent variables such as number of recreational trips, and how responses differ to attitudinal questions, may provide insight on whether (and which) preference parameters should be expected to vary much across individuals, and whether those variations can be observed. These same factors, plus written and verbal comments perhaps, might be used to assess the level of coherence in decision-making and interest in the activity, and therefore could be used to decide whether iid assumptions about the random components are reasonable. As heterogeneity features are added to the basic model, their relative importance and impact can be evaluated not only on the basis of the likelihood function, but other factors including predictive power, and the robustness of parameters and other model results such as consumer surplus.

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Are Revealed Preference Measures of Quality Change Benefits Statistically Significant?

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ABSTRACT

Environmental economists typically invoke weak complementarity in order to measure use values of quality changes from revealed preference information such as recreation demand functions. But they do not typically associate standard errors with the estimates. The assessment of standard errors for quality change welfare measures is more complicated than for the price change case, in part because a structural assumption must be employed to explain how the constant of integration back varies with the quality change parameter of interest. Even when the widely-invoked weak complementarity assumption is used to provide this structure, the underlying quasi-preferences are unknown for many demand functions. Numerical approximations can be used, but require changes in multiple variables to maintain the structural hypothesis, which can inflate standard errors.

We explore these and other issues involved with assessing standard errors of quality change measures. Not surprisingly, we find that when the underlying quasi-preferences are unknown, the variance of compensating variation is substantially larger than when they are known. What is surprising is how large the standard errors of quality change welfare measures are relative to their means, even when the underlying quasi-preferences are known. Demand models with statistically-significant parameters frequently yield compensating variations for quality change that include zero in their 95% confidence intervals. The covariances between demand parameters play a strong role in the magnitude of the standard error of compensating variation.

Are Revealed Preference Measures

of Environmental Quality Benefits Statistically Significant?

Two main avenues are available to researchers interested in measuring the value of changes in non-market amenities such as environmental quality: the stated preference approach (exemplified by contingent valuation, contingent ranking, and other direct questioning methods) and the revealed preference approach, the best-known example of which is the travel cost method of recreation demand. Each results in a willingness-to-pay function, typically based on compensating variation or surplus, whether estimated directly (as with stated preference data) or inferred from an auxiliary relationship (such as demand functions, in the case of revealed preference).

While the topic of precision of welfare measures generally has received some attention (e.g., Adamowicz, Fletcher, and Graham-Tomasi; Kling and Sexton; Kling, 1991, 1992), much, though not all, of this work has centered on price changes and access values for recreational resources. Relatively less work has focused on the precision of quality-change welfare measures, particularly those derived from revealed preference work. An exception is Kling (1988a,b), whose focus primarily is comparing alternative estimation strategies for recreation demand based on their relative error in estimating a known true welfare change measure, but who also calculates root mean squared errors that suggest the empirical quality change measures are often statistically insignificant.

The reason why insights about quality change welfare measures might differ from those obtained from evaluating price changes is that the integrability problem generally poses an additional challenge for welfare measurement. For quality changes, it is well known that unique welfare measures cannot be obtained from revealed preference alone without imposing some additional structure on preferences (e.g., LaFrance and Hanemann). The structure typically used is *weak complementarity* between one or more market goods and the quality characteristic. First proposed by Mäler and further articulated by Bradford and Hildebrandt, Willig, and Bockstael and McConnell (1983; 1993), weak complementarity allows the researcher to recover the value of an environmental quality change by focusing on how the market demands for the weak complements to quality change as quality changes. It corresponds to the familiar graph of the area between two (Hicksian) demand curves that shift with a quality variable.[1]

The problem this poses for welfare measurement with quality changes is that if the underlying preferences for quality are not known, as is typically the case when one estimates a demand system dependent on the quality characteristic, implementation of the welfare measurement procedure under weak complementarity requires a three-step procedure, outlined in Mäler, involving changes in price as well as the change in quality. (This is described further below. Because it would typically be implemented using numerical methods such as those outlined in Vartia or Porter-Hudak and Hayes, we refer to this as the *numerical* approach.) One would expect, intuitively, that this would inflate the standard errors of the welfare measure, relative to the case where the underlying quasi-expenditure function is known and can be evaluated directly for a change in quality alone (the *analytic* approach).

The operational question we explore is whether, in either case, the point estimates of welfare change for quality changes are sufficiently precise to be of any use in policy analysis. One dimension of the problem, clearly, is the relative increase in standard error of the welfare measure when underlying quasi-preferences are unknown, compared to when they are known. This issue is probably most relevant to revealed preference studies, where demand functions are estimated but the corresponding weakly complementary quasi-expenditure function may not be known.[2]

The other dimension we explore is the role that correlation between willingness to pay function parameters plays in determining the standard error of the resulting compensating variation. This issue is common to both revealed and stated preference methods, because each ultimately derives willingness to pay as a function of correlated random variables, namely the estimated parameters of the statistical model. This issue, somewhat to our surprise, is quite important to the precision of the resulting standard errors.

We consider these questions within the framework of a linear demand model, because this is one commonly-used functional form for which the underlying weakly-complementary quasi-expenditure function is known. This facilitates the comparison of the numerical and analytic approaches, because each can be performed on the same model and the resulting standard errors compared.

The Welfare Measurement Framework

The issue of concern is measuring the consumer's valuation of a change in exogenous quality, represented by the variable z, from an initial level z_0 to a subsequent level z_1. There are n market goods denoted by $\mathbf{x}=(x_1,...,x_n)'$ with corresponding prices $\mathbf{p}=(p_1,...,p_n)'$, and the consumer is presumed to choose market goods in a way that minimizes the cost of utility, represented by the dual problem

$$\min_{\mathbf{x}} \mathbf{p'x} \quad \text{s.t.} \quad u^0 = u(\mathbf{x},z). \tag{1}$$

The solution to (1) is the Hicksian demands $x^h(\mathbf{p},z,u)$, which when substituted into (1) yields the minimum expenditure function $e(\mathbf{p},z,u)$. The corresponding primal problem has the form

$$\max_{\mathbf{x}} u(\mathbf{x},z) \text{ s.t. } m \geq \mathbf{p'x} \tag{2}$$

where m is the consumer's exogenous budget constraint. The solution to this problem yields Marshallian demand functions $x(\mathbf{p},z,m)$ which are estimated empirically.

In practice, we typically work with incomplete demand systems, which do not identify all the structure of $e(\mathbf{p},z,u)$. Suppose that a b-good (with b<n) incomplete demand system

$$x_i = x_i(\mathbf{p}^b,\mathbf{p}^{-b},z,m), \qquad i=1, \dots ,b$$

is estimated, with $\mathbf{p}^b \equiv [p_1,...,p_b]'$ the vector of prices included in the estimated demand system and $\mathbf{p}^{-b} \equiv [p_{b+1},...,p_n]'$ the prices of other goods outside the empirical demand system. The empirical demand system integrates back to a quasi-expenditure function $\tilde{e}(\mathbf{p}^b,z,\theta(\mathbf{p}^{-b},z,u))$ which can be used for exact welfare measurement with respect to any of the prices in $\mathbf{p}^b$, conditional on $\mathbf{p}^{-b}$, but not for z without further structure on $\theta(\cdot)$ (LaFrance and Hanemann).

As noted above, weak complementarity of z with $\mathbf{x}^b \equiv [x_1,...,x_b]'$ is the typical assumption made about the structure of preferences that is sufficient to identify the curvature of $\theta(\cdot)$ with z and, therefore, the way that z enters the quasi-expenditure function $\tilde{e}(\mathbf{p}^b,z,\theta(\mathbf{p}^{-b},z,u))$. Mäler showed how one could use this assumption in a three-step process (raising price to the choke level given original quality level; changing quality while simultaneously adjusting choke price to keep "use" at zero; and reducing price from the new choke level to its original level) to measure the compensating variation associated with a change in quality for arbitrary demand functions.[3] Larson showed how one can analytically recover the weakly complementary quasi-expenditure function corresponding to a linear single-equation demand function.

The Demand Specification and Implied Quasi-Preferences

If we write the Marshallian demand function for a good x of interest as

$$x = \alpha + \beta p + \gamma z + \delta m, \tag{3}$$

the quasi-expenditure function obtained from integrating back from (3) is (e.g., Hausman)

$$\tilde{e}(p,q,\vartheta(z,u)) = \vartheta(z,u) \cdot e^{\delta p} - (1/\delta)[\alpha+\beta p+\gamma z+\beta/\delta], \tag{4}$$

where ϑ(z,u) is a constant of integration that may depend on all other parameters of the problem besides the variable of integration p, including the quality variable and other prices (which are not made explicit in this model).

As is well known, equation (4) does not provide a basis for unique welfare measurement for quality (z) changes without further structure being imposed. What is typically invoked is weak complementarity between the public good or quality attribute whose value is of interest, and a set of related market goods whose demand can be observed (e.g., Mäler). Weak complementary is essentially an assumption that there is no "passive-use" value. In the present model, the assumption is of weak complementarity between x and z. When this is imposed as part of the process of integrating back to recover quasi-preferences, the resulting quasi-expenditure function is

$$e(\mathrm{p,q},u) = u\mathrm{e}^{(\delta/\beta)(\gamma z+\beta p)} - (1/\delta)[\alpha+\beta\mathrm{p}+\gamma\mathrm{z}+\beta/\delta], \tag{5}$$

where $u < 0$ is the utility index, independent of both z and p (Larson). The corresponding weakly complementary indirect utility function is

$$v(\mathrm{p,q,m}) = [\mathrm{m}+(1/\delta)(\alpha+\beta\mathrm{p}+\gamma\mathrm{z}+\beta/\delta)]\mathrm{e}^{-(\delta/\beta)(\gamma z+\beta p)}.$$

The compensating variation for a change in quality from z_0 to z_1 is

$$\mathrm{CV} = e(\mathrm{p}_0,\mathrm{z}_0,\mathrm{u}) - e(\mathrm{p}_0,\mathrm{z}_1,\mathrm{u})$$

$$= \mathrm{M} - u\mathrm{e}^{(\delta/\beta)(\gamma z_1+\beta p)} + (1/\delta)[\alpha+\beta\mathrm{p}+\gamma\mathrm{z}_1+\beta/\delta]. \tag{6}$$

Approximating the Standard Error of CV when Quasi-Preferences are Known

In the *analytic* approach, the compensating variation is obtained directly from substituting the two levels of quality, z_0 and z_1, directly into the quasi-expenditure function, as in (6). The resulting CV is a random variable because it is a nonlinear function of a set of correlated random variables. Given the variance-covariance matrix Ω_ϕ for the vector of estimated parameters $\hat{\phi} \equiv [\hat{\alpha},\hat{\beta},\hat{\gamma},\hat{\delta}]'$ and the gradient vector $\nabla \equiv [\partial CV/\partial\alpha(\hat{\alpha}),...,\partial CV/\partial\delta(\hat{\delta})]'$, a consistent approximation to the variance of $CV(\hat{\phi})$, $V[CV(\hat{\phi})]$, is

$$V[CV(\hat{\phi})] \approx \nabla'\Omega_\phi\nabla, \tag{7}$$

provided $\hat{\phi}$ is a consistent estimate of the true underlying parameters ϕ (e.g., Greene).

The approximation in (7) is based on a first-order Taylor's series approximation to the variance of a nonlinear function of random variables. We use the Taylor's series approximation approach because of our interest in comparing results when underlying quasi-preferences are known and an analytic solution can be used, to the case where underlying quasi-preferences are unknown. In this latter case, the numerical approximation methods used to assess standard errors employ a Taylor-approximation methodology (e.g., Vartia; Porter-Hudak and Hayes; Breslaw and Smith). Thus for consistency of comparison, we use the Taylor's approximation in (7) for the analytic case as well.

Approximating the Standard Error of CV when Quasi-Preferences are Unknown

In many cases, the quasi-expenditure function underlying the estimated demand function may not be known analytically, particularly as it changes with quality. The *numerical* approach is used for cases like this. Algorithms presented by Vartia, Porter-Hudak and Hayes, and

Breslaw and Smith for evaluating price changes can be adapted to the case of quality changes with weakly complementary preferences.

The principle behind these algorithms is straightforward, as they simulate the standard error of compensating variation for a series of small changes $\triangle z^i$ for i=1,...,n steps covering the interval from z_0 to z_1, following the approach of Vartia. Given the variance-covariance matrix $\Omega_\phi(\hat{\phi})$ for the estimated parameter vector $\hat{\phi} \equiv [\hat{\alpha},\hat{\beta},\hat{\gamma},\hat{\delta}]'$, for each step i, the updated estimate of the expenditure function at each step i is

$$e^i(\hat{\phi}) = e^{i-1}(\hat{\phi}) + \tfrac{1}{2}[(\partial e(\hat{\phi})/\partial z)_0 + (\partial e(\hat{\phi})/\partial z)_1] \triangle z^i \tag{8}$$

and the variance of the new estimate of the expenditure function at the ith step, $e^i(\hat{\phi})$, is

$$\mathrm{var}[e^i(\hat{\phi})] = \begin{bmatrix} 1 & \frac{\Delta z}{2} & \frac{\Delta z}{2} \end{bmatrix} \begin{bmatrix} \sigma_{ee} & \sigma_{e0} & \sigma_{e1} \\ \sigma_{e0} & \sigma_{00} & \sigma_{01} \\ \sigma_{e1} & \sigma_{01} & \sigma_{11} \end{bmatrix} \begin{bmatrix} 1 \\ \frac{\Delta z}{2} \\ \frac{\Delta z}{2} \end{bmatrix} \tag{9}$$

where the σ_{ij}, for i, j = e, 0, 1, are the variances and covariances of, and between, the previous estimate of the expenditure level, $e^{i-1}(\hat{\phi})$, and the estimates of the expenditure slope at the previous $[(\partial e(\hat{\phi})/\partial z)_0]$ and current $[(\partial e(\hat{\phi})/\partial z)_1]$ levels of quality z. Each of these is a nonlinear transformation of the regression parameters $\hat{\phi}$, with gradients denoted $\nabla_i(\hat{\phi})$, respectively, for $i = y, 0, 1$. Estimates of the asymptotic variances and covariances are then obtained from

$$\sigma_{ij} = \nabla_i(\hat{\phi})\, \Omega_\phi(\hat{\phi})\, \nabla_j(\hat{\phi})$$

following Rao and Porter-Hudak and Hayes. The compensating variation is $\mathrm{CV}(\hat{\phi}) = \mathrm{M} - e^n(\hat{o})$, by analogy to (6), so with initial income taken to be fixed, the variance of the compensating variation measure is

$$V[CV(\hat{\phi})] = \mathrm{var}[e^n(\hat{\phi})],$$

from (9).

The difficulty with applying this numerical approximation technique directly when the underlying weakly complementary quasi-preferences are not known (i.e., when a different demand function is used) is that the constant of integration in (4) is not identified analytically, so the quality slopes $(\partial e/\partial z)_0$ and $(\partial e/\partial z)_1$ are unknown. The weak complementarity condition can still be imposed as part of the numerical approximation of compensating variation and its standard error, but the process is more involved than the direct simulation of a quality change in (8). The reason is that the weak complementarity condition is a statement about what happens to the expenditure function when z changes and x is *not* being consumed. That is, when the price of x is at the choke level p' and consumption of x is zero, weak complementarity of x with z means that there is no change in value and the expenditure function is stationary. Thus, weak complementarity is a condition that holds at a different set of prices (p') than those which hold at the reference point ($p < p'$, where there is positive consumption of x), so to impose it as part of the numerical approximation of CV and its standard error, prices must be changed from p to p'.

This is the three-part strategy suggested by Mäler for measuring welfare for general weakly complementary preferences (p. 173-176). To measure the welfare change associated with a change in z from z^0 to z^1, given prices p^0, one must

(a) change p from p^0 to p', given z^0; (10a)

(b) change p' to maintain consumption of x at zero as z changes from z^0 to z^1; (10b)

(c) change p from p' back to p^0 given z^1. (10c)

The compensating variation of the quality change can be measured this way because steps (a) and (c) are the area under the Hicksian demand for x given z^0 and z^1 respectively, each of which can be measured by a numerical approximation algorithm such as that given in (7), though for changes in price rather than z. The weakly complementary welfare measure is the difference between the two, since by assumption the welfare change in part (b) is zero.[4]

When underlying quasi-preferences are unknown, the welfare measures and price changes in each of these steps (a)-(c) can be measured numerically. But the result is that three numerical approximations are required, rather than just one as suggested by (7). The expectation is that having to sequentially raise and lower price in addition to changing quality to measure the welfare effects of the quality change will inflate the standard error of the resulting CV measure, resulting in more frequent occurrences of statistical insignificance for given quality changes.

Parameterizing the Simulation Model

Our starting point for developing a simulation model to compare the analytic and numerical approaches is the information which might be observed in the field about the demand for recreation by a "typical" individual, that is, the trips taken, price paid, income, and quality expected. Both the form of the demand model and its statistical significance (i.e., the variance-covariance matrix for a given parameter vector) are likely to be important to the significance of the compensating variation of a quality change, which is a nonlinear transformation of demand model parameters.

Demand Parameters

Demand model parameters are chosen to be representative of those found in recreation demand studies. Such a model might be motivated in terms of recreational fishing, where quality (e.g., expected fishing sucess) can play a prominent role. For example, consider an individual with income of $60,000 who takes 4 trips per year to a single fishing site, at a price of $55 per trip, with expected fishing success of 2.8 fish per trip. Assumptions about the demand elasticities with respect to each of these variables will yield coefficients $\phi=[\alpha,\beta,\gamma,\delta]$ for the model in equation (3).

The baseline model is one with unitary elasticities, with price elasticity of -1 and quality and income elasticities equal to +1. This implies the demand specification

$$x = -.07273p + 1.429z + .00006667m, \tag{11}$$

with $\alpha = 0$ by coincidence. For contrast, it is natural to consider coefficient values corresponding to both elastic and inelastic versions of the model. The inelastic version of the model has price elasticity of -.80, and income and quality elasticities of .80, yielding a demand model parameterized as

$$x = .80 - .05818p + 1.143z + .00005333m. \tag{12}$$

The elastic version of the model was also constructed for the same levels of trips, price, income, and quality, but for price elasticity of -1.2 and income and quality elasticities of 1.2. This results in a demand model of the form

$$x = -.80 - .08727p + 1.714z + .00008m. \tag{13}$$

Variance-Covariance Matrix

A key issue is how given levels of significance in the demand model translate to significance of the CV for a given quality change. We explore this by considering a variety of statistically-significant demand models, as measured by the Student's-t statistics on individual coefficients. To see how the assumptions we make about coefficient significance can be built into the model, note that the asymptotic standard error of coefficient $\widehat{\phi}_i$ can be written as

$\hat{\sigma}_{\hat{\phi}_i} = \hat{\phi}_i / t_{\hat{\phi}_i}$, where $t_{\hat{\phi}_i}$ is the Student's-t statistic for coefficient $\hat{\phi}_i$ under the common hypothesis of no association for a model with given degrees of freedom and significance level.

Using this definition of the standard error, the variance-covariance matrix for the parameter vector is

$$\Omega_o = \begin{bmatrix} \sigma^2_{\hat{\alpha}} & \cdots & \sigma_{\hat{\alpha}\hat{\delta}} \\ \vdots & \ddots & \vdots \\ \sigma_{\hat{\delta}\hat{\alpha}} & \cdots & \sigma^2_{\hat{\delta}} \end{bmatrix} = \begin{bmatrix} (\hat{\alpha}/t_{\hat{\alpha}})^2 & \cdots & \rho_{0m}(\hat{\alpha}/t_{\hat{\alpha}})(\hat{\delta}/t_{\hat{\delta}}) \\ \vdots & \ddots & \vdots \\ \rho_{0m}(\hat{\alpha}/t_{\hat{\alpha}})(\hat{\delta}/t_{\hat{\delta}}) & \cdots & (\hat{\delta}/t_{\hat{\delta}})^2 \end{bmatrix} \tag{14}$$

so the effect of varying significance level of coefficients, for a given parameter vector, on the variance-covariance matrix can be seen. The coefficients $\hat{\alpha},\ldots,\hat{\delta}$ for the simulations are identified in equations (11)-(13). As is well known, the magnitudes of the elements of the variance-covariance matrix will vary inversely with the precision of measuring the coefficient (i.e., its Student's-t statistic) and directly with the magnitude of the absolute value of the correlation coefficients.

Equation (14) points out the potential importance of the correlation parameters ρ_{ij}, beyond the issue of how precisely they are estimated. To isolate the effects of each separately, we present results on the standard error of the quality change welfare measure for different levels of precision of estimating coefficients (given by $t_{\hat{\phi}_i} = t_{\hat{\phi}_j} = 2, 3, 5$ all i,j) and for partial correlation coefficients between price, quality, and income coefficients ranging from -1 to 1.

The first threshold of Student's-t, t=2, corresponds roughly to the asymptotic t for the 95% confidence level, as a guide to a common rule of thumb used in practice. One could, obviously, use many different thresholds for determining "significance" of the quality change welfare measure, corresponding to specific sample sizes (which imply higher threshold Student's-t statistics) and alternative significance levels. It is worth noting that our choice of a test statistic based on asymptotic distributions is conservative, in the sense that we will find more "significant" welfare measures, and hence fewer problems, than would be found in small samples.

For increments of 0.25 for the partial correlation between price and quality coefficient (ρ_{pz}) in the interval [-1,1], we consider increments of 0.1 in the partial correlations between income and quality (ρ_{mz}) and between price and income (ρ_{pm}). In addition, the constant term is uncorrelated to price, income, and quality ($\rho_{0m} = \rho_{0p} = \rho_{0z} = 0$). It is important to note that not all combinations of ρ_{pm}, ρ_{mz}, and ρ_{pz} in the unit sphere are valid representations of a partial correlation matrix, because such a matrix must be positive semidefinite; that is, it must satisfy the equation

$$\sum_i \sum_j x_i x_j \rho_{ij} \geq 0$$

for all non-zero vectors $x = [x_1,...,x_n]$.

Rousseeuw and Molenberghs analyze this feasible set and their Figure 1 (reproduced here as Figure 1) illustrates the set visually. For a three-dimensional partial correlation matrix, they show that any horizontal cross section, obtained by fixing one partial correlation, is an ellipse. If, for instance, one partial correlation, say ρ_{pm}, is held at a value within the (0,1) interval, we obtain an ellipse with a major axis in the direction of the line $\rho_{pz} = \rho_{mz}$ and a minor axis in the direction of $\rho_{pz} = -\rho_{mz}$. Conversely, if ρ_{pm} is fixed at a value between -1 and 0, the major and minor axes of the resulting ellipse are reversed. Of particular note are the extremum values, where a value of $\rho_{ij} = \pm 1$ for one partial correlation reduces the set of feasible combinations of the remaining two partial correlations, to a line of equal slope (when $\rho_{ij} = +1$) or opposite slope (when $\rho_{ij} = -1$).

When the feasible set for valid representations of a correlation matrix is imposed on the unit sphere, there are 1,893 combinations of the three pairwise correlations ρ_{pm}, ρ_{pz}, and ρ_{mz} for each parameterization of the demand model.

Comparative Statics of the Analytic Approach

When quasi-preferences are known, the asymptotic variance of the quality change welfare measure is a function of the parameters of the problem, as (7) indicates. In particular, one can examine the effects of changes in the partial correlation coefficients ρ_{ij} or the level of precision t_i with which coefficients are measured. In our simulations, the precision of each coefficient is equal, though the signs may differ, so we can simplify by writing $|t_{\phi_i}| = |t_{\phi_j}| = t$. To preserve consistency with the use of the ratios ϕ_i/t_{ϕ_i} to represent standard errors (which are non-negative) in (13), it is necessary to redefine the coefficients also in terms of absolute magnitudes, not signs; i.e., we have $|\phi_i|/|t_{\phi_i}| = \sigma_i$. Letting ∇_i denote the ith element of ∇, $\partial CV/\partial\phi_i$ (where i, j = α, β, γ, δ), and ∇_{ij} denote the second partial derivative $\partial^2 CV/\partial\phi_i\partial\phi_j$, then (7) can be written out more fully (dropping the "hats" for simplicity) as

$$V[CV(\phi)] = t^{-2}[\nabla^2_{\alpha\alpha}\alpha^2 + \nabla^2_{\beta\beta}\beta^2 + \nabla^2_{\gamma\gamma}\gamma^2 + \nabla^2_{\delta\delta}\delta^2 + 2\nabla_\beta\nabla_\gamma\rho_{pz}|\beta\gamma|$$

$$+ 2\nabla_\beta\nabla_\delta\rho_{pm}|\beta\delta| + 2\nabla_\delta\nabla_\gamma\rho_{mz}|\delta\gamma|] \qquad (15)$$

Differentiating (15) with respect to t, the Student's t-statistic, the change in variance of the welfare measure as t increases is

$$\partial V[CV(\phi)]/\partial t = -(2/t) \cdot V[CV(\phi)] \quad \leq 0$$

since $V[CV(\phi)] \geq 0$ and $t > 0$ by construction. Not surprisingly, increasing the statistical significance of all demand parameters (by increasing the magnitude of the t-statistic) unambiguously decreases the variance of the welfare measure. Equivalently, the coefficient of variation of the welfare measure decreases as the significance of the estimated demand parameters increases.

Similarly, one can analyze how changes in the degree of correlation between variables affects the variance of the compensating variation welfare measure by differentiating (13) with respect to ρ_{pz}, ρ_{pm}, and ρ_{mz}.[5] This yields (16), (17), and (18), respectively.

$$\partial V[CV(\phi)]/\partial \rho_{pz} = 2|\beta\gamma|t^{-2}\nabla_{\beta}\nabla_{\gamma} \tag{16}$$

$$\partial V[CV(\phi)]/\partial \rho_{pm} = 2|\beta\delta|t^{-2}\nabla_{\beta}\nabla_{\delta} \tag{17}$$

$$\partial V[CV(\phi)]/\partial \rho_{mz} = 2|\delta\gamma|t^{-2}\nabla_{\delta}\nabla_{\gamma} \tag{18}$$

To determine these signs, one must know the signs of ∇_{β}, ∇_{γ}, and ∇_{δ}. For the linear demand model examined in this paper, one can differentiate the CV(ϕ) expression in equation (6) to determine ∇_{β}, ∇_{γ}, and ∇_{δ}. For this model, $\nabla_{\gamma} > 0$ always, and both ∇_{β} and $\nabla_{\delta} > 0$ when the combined intercept and income term is non-negative; i.e., when $\alpha + \delta m > 0$.

It is easily verified that all three versions of the demand model in (11)-(13) meet this condition. Therefore, since ∇_{β}, ∇_{γ}, and ∇_{δ} are all positive, V[CV(ϕ)] increases with each of the partial correlation coefficients.

Simulation Results

Analytic Approach

Table 1 presents results for the case of known quasi-preferences, where the welfare evaluation strategy in equations (6) and (7) is used. This is a tabulation of the results of simulations of the compensating variation for a 50 percent increase in the quality variable, from $z_0 = 2.8$ to $z_1 = 4.2$. These simulations cover 1,893 combinations of feasible partial correlation coefficients between the price, income, and quality coefficients, for each of the three types of demand functions (inelastic, unitary elastic, and elastic) and each of the three levels of precision in measuring coefficients (t = 2, 3, and 5). For each combination, the coefficient of variation of CV is calculated (as the standard error divided by estimated CV). When the coefficient of variation is greater than 0.5, the value of zero is within the 95% confidence bounds on CV. A conventional classical hypothesis test of difference of the CV estimate from zero would fail to reject that hypothesis.

The results are quite striking. For a model with elastic demand parameters, all statistically significant at the 95% (2-tailed) level (i.e., with t-values of 2), less than 2% of the CVs were had coefficients of variation less than 0.5. Results were comparable for the unit-elastic and inelastic cases, where slightly more than 2%, and less than 3%, of the CVs had coefficients of variation less than 0.5, respectively. When the significance of all demand coefficients increases to the .9987 level (Student's-t = 3), only 6% of the elastic demand, 7% of the unit elastic demand, and 10% of the inelastic demand simulations had coefficients of variation less than 0.5. When the t-value on all demand coefficients was increased to 5 (corresponding to a p-value of .9999997), 31% and 39% of the elastic and unit elastic demand model simulations satisfied this condition, respectively, and almost half of the inelastic demand model simulations did. The observed increase in precision of CV in all models with increases in the t-statistic is consistent with the comparative static results derived above.

Figure 2 illustrates the results from the unit-elastic case visually, with plots of the coefficient of variation of CV (COV) against the price-income [ρ(p,m)] and quality-income [ρ(q,m)] correlations, for given price-quality correlations [ρ(p,q)]. As one moves from left to right and top to bottom in each figure, the price-quality correlation relationship goes from perfect inverse correlation to perfect direct correlation. The combinations with coefficient of variation on CV less than 0.5 are illustrated with boxes, and those with higher COV are marked with pyramids. The coefficient of variation surfaces are smooth with different tilts depending on the values of the conditioning correlation, between price and quality.

These graphs emphasize the relationships found in the comparative statics results on the effects of partial correlation relationships on the precision of CV. As Figure 2 shows, the more positively correlated price and quality are (ρ_{pz} approaches 1), the less likely is there a significant coefficient of variation (i.e., COV < 0.5). Likewise, COV is higher as the correlation between price and income gets more positive (ρ_{pm} approaches 1), while COV falls as the correlation between income and quality gets more negative (ρ_{mz} approaches -1).

Other regularities are also apparent in the results. The elasticity of the demand parameters appears to play an important role in the precision of welfare measures. Table 2 highlights this effect by focusing on changes in the magnitude of all elasticities simultaneously, varying them from 0.2 to 2.0 in magnitude for a given (50%) quality change and significance level (t=2). Not surprisingly, the CV for a 50% quality change varies, from \$115 to \$165. As Table 2 indicates, the elasticity of the parameters clearly affects the significance of compensating variation, but it does not appear to be monotonic with the own-price elasticity. This can be seen by noting that for the linear model, changing the own-price elasticity of a demand function running through a fixed price-quantity point is equivalent to changing β; and similarly, changes in quality and income elasticities and equivalent to changes in γ and δ. But in (15) it is clear that the effect of changing the magnitude of β, γ, and δ will depend in non-trivial ways each of their magnitudes and on the pairwise correlations ρ_{pz}, ρ_{mz}, and ρ_{pm}. The non-monotonic relationship between magnitudes of elasticities and precision of the welfare measure appears to be due to the fact that all are changing simultaneously and they all affect the welfare measure precision in different ways.

Table 3 follows up with a consideration of the effects of changing individual elasticities, *ceteris paribus*. The first four columns of data in Table 3 contains the results from changing the own-price coefficient, $\hat{\beta}$, in the unit-elasticity model to reflect price elasticities of $-.2$, $-.8$, -1, and -2, while keeping all other coefficient values the same. These results suggest that the more own-price elastic the demand model, the lower the precision in measuring the welfare change for a 50% increase in quality. In contrast, the last four columns in Table 2 illustrate the effect of increasing quality elasticity, *ceteris paribus*, which shows a non-monotonic relationship with the precision of the welfare measure, with the most frequent occurrences of significant results at the intermediate elasticity levels.

These results highlight the important role that parameter correlations and elasticities play in determining the magnitude of empirical standard errors of quality change measures. But the overriding point is the generally low frequency with which willingness to pay models with all significant coefficients lead to "precise" welfare estimates for quality changes (in the sense that zero is not included in the 95% confidence bounds). For a large fraction of the feasible pairwise correlations, statistically significant (at the $\alpha = .05$ level of significance) parameters of the willingness to pay function translate to imprecise quality change welfare measures.

Numeric Approach

The second issue of interest is the degree to which the standard error of compensating variation is inflated because the analytic quasi-expenditure function is unknown and the 3-step Mäler/Vartia algorithm must be used instead. Table 4 provides some perspective on this question for all the cases where the coefficient of variation was less than 0.5 in the analytic solution results, for the case where all demand parameters have t-values of 2. Of these 40 cases (resulting from 1,893 different feasible correlation combinations), only 1 (highlighted in bold) had coefficient of variation less than 0.5 under numerical approximation of the standard error. The ratio of numerically-approximated standard error to analytic standard error showed considerable variation, depending on the particular correlations between explanatory variables. The ratio of standard errors ranged from 1.42 to 6.52, with an arithmetic mean of 3.3. This suggests that, on average, the numerical approximation routinely inflates standard errors by roughly a factor of 3.

Priors on the Correlation Combinations

One might suspect that not all correlation combinations are equally likely. In fact, one might have as priors that price and income are positively correlated (because, for example, higher income people might travel to a recreation destination in motor homes with lower gas mileage and slower travel speeds); that price and quality are positively correlated, if it is possible to price differentially for quality; and that quality and income are positively correlated, if quality is a normal good. From Table 4, one can see that none of the 40 correlation combinations that had coefficients of variation less than 0.5 were from this orthant of the correlation spheroid. Thus the problem we identify may be worse for the cases likely to occur more frequently in practice than our overall results suggest. It illustrates the decreased likelihood of "precise" welfare measures when the underlying quasi-expenditure function is unknown and numerical approximation methods must be used.

The problems we find are not due solely to the presumption of some correlation structure betweeen parameter estimates. Focusing attention on the subset of simulations where 1, 2, or 3 of the pairwise correlations are zero, none of these indicate a coefficient of variation for the welfare measure of less than 0.5 even though each demand parameter is statistically significant (with Student's-t of 2).

Conclusions

In the recreation demand literature, it is common to find benefit estimates associated with a variety of different quality characteristic changes, but much less common to find standard errors attached to these calculations. Our results suggest that were researchers to do this more frequently, they would be surprised by how routinely they are calculating statistically-insignificant welfare measures for a wide range of null hypotheses. We highlight two of the reasons why this occurs: the role of pairwise correlations between the model parameters, which is an issue with both revealed and stated preference methods of assessing willingness to pay for quality changes; and the fact that one sometimes has to numerically approximate through simultaneous price and quality changes in revealed preference studies, when the underlying quasi-expenditure function behind estimated demand is unknown.

The results suggest that strong caution is appropriate in interpreting welfare estimates of quality change for which no standard errors are provided. As a corollary, good empirical practice has to include an assessment of the precision of the welfare measure for such estimates to be taken seriously.

The results here clearly are only suggestive and not definitive. Further work is needed to assess how robust the findings from this simple, but commonly used, function really are. It is possible that the parameter values themselves, or the initial levels of consumer income, price, and quality make a difference to the magnitudes of the effects described here, though we doubt they would be reversed.

It is important to note that these results come from *correct* practice in measuring welfare change when a quality characteristic changes. This is more involved than calculating a change in consumer's surplus area from Marshallian demands, as it involves evaluating a quasi-expenditure function selected based on some prior restriction on preferences (e.g., weak complementarity). This evaluation is done either numerically or analytically depending on whether the quasi-expenditure function itself is known, or only the preference restriction used to identify the welfare measure is known. Others have noted the difficulties inherent in using consumer's surplus instead of the theoretically-correct compensating variation (or surplus) measure for quality changes(e.g., Kling 1988a,b; Bockstael and McConnell, 1993), or in incorrectly assuming weak complementarity (Bockstael and Kling; LaFrance). One would expect that the problems we identify would compound, not alleviate, these other difficulties with correct measurement of welfare changes with quality characteristics.

Footnotes

1. Furthermore, it corresponds to a story about how consumers value environmental quality that is plausible in some (though not all) contexts: if the consumer is not consuming any of the weak complements, he or she is indifferent to the quality change. This is probably reasonable for many localized resources with plentiful substitutes, that are unlikely to generate nonuse value.

2. Stated preference estimation, by contrast, typically directly estimates a compensating variation function so, in principle, the quasi-preferences are known.

3. This implicitly defines a choke price function which varies with quality and all parameters of the problem to keep quantity consumed identically at zero.

4. Graphically, the three-step procedure traces out the change in area under the Hicksian demand for the related market good. The weak complementarity assumption assures that this is the total value of the quality change. Extension to multiple goods related to quality is straightforward (Bockstael and Kling).

5. The magnitude of the variance of the welfare measure will depend on the standard errors of, and correlations among, the demand parameters, as well as the gradients of compensating variation with respect to the parameters.

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Table 1. Simulation Results for Analytic CV Estimates for 50% Increase in Quality

	Demand Elasticity								
	Inelastic[a]			Unit Elastic[b]			Elastic[c]		
Student's-t	2.00	3.00	5.00	2.00	3.00	5.00	2.00	3.00	5.00
No. "significant"	48	185	918	40	138	731	27	107	582
% "significant"	0.03	0.10	0.48	0.02	0.07	0.39	0.01	0.06	0.31
CV ($)	132	132	132	138	138	138	142	142	142

[a]Absolute value of all elasticities is 0.8. The implied demand is x = .8 - .05818p + 1.143z + .00005333m.
[b]Absolute value of all elasticities is 1.0. The implied demand is x = -.07273p + 1.429z + .00006667m.
[c]Absolute value of all elasticities is 1.2. The implied demand is x = -.8 -.087p + 1.714z + .00008m.

Table 2. Effect of Demand Elasticities on Precision of CV, for Student's-t = 2[a]

	Elasticity				
	0.2	0.8	1	1.2	2
No. "significant"	21	48	40	27	2
% "significant"	1.11	2.54	2.11	1.43	0.11
CV ($)	115	132	138	143	165

[a] All coefficients have the same elasticity.

Table 3. The Effect of Different Price and Quality Elasticities on Precision of CV, for t = 2

	Elasticity Combinations							
Price	-0.2	-0.8	-1	-1.2	-1.0	-1.0	-1.0	-1.0
Quality	1.0	1.0	1.0	1.0	0.2	0.8	1.0	1.2
Number "significant"	79	50	40	1	3	47	40	27
Percent "significant"	4.17	2.64	2.11	0.05	0.16	2.48	2.11	1.43

Table 4. A Comparison of Standard Errors for Analytic Versus Numerical Methods Calculation of the Quality Change Welfare Measure[a]

Partial Correlations			Analytic Results		Numeric Results		Ratio of
ρ(p,q)	ρ(p,m)	ρ(q,m)	SEA(CV)[b]	COV	SEN(CV)[c]	COV	Std. Errors
-0.75	0.4	-0.9	59.66	0.43	219.40	1.58	3.68
-0.75	0.3	-0.8	60.92	0.44	213.15	1.53	3.50
-0.75	0.2	-0.7	62.15	0.45	206.70	1.49	3.33
-0.75	0.1	-0.7	49.96	0.36	170.68	1.23	3.42
-0.75	0.1	-0.6	63.36	0.46	200.05	1.44	3.16
-0.75	0	-0.6	51.45	0.37	162.56	1.17	3.16
-0.75	0	-0.5	64.55	0.47	193.17	1.39	2.99
-0.75	-0.1	-0.5	52.91	0.38	154.01	1.11	2.91
-0.75	-0.1	-0.4	65.72	0.48	186.03	1.34	2.83
-0.75	-0.2	-0.4	54.33	0.39	144.96	1.04	2.67
-0.75	-0.2	-0.3	66.86	0.48	178.61	1.28	2.67
-0.75	-0.3	-0.4	39.8	0.29	86.12	0.62	2.16
-0.75	-0.3	-0.3	55.71	0.40	135.30	0.97	2.43
-0.75	-0.3	-0.2	67.99	0.49	170.87	1.23	2.51
-0.75	-0.4	-0.3	41.66	0.30	68.64	**0.49**	1.65
-0.75	-0.4	-0.2	57.05	0.41	124.90	0.90	2.19
-0.75	-0.5	-0.1	58.37	0.42	113.55	0.82	1.95
-0.75	-0.6	0.1	59.66	0.43	100.94	0.73	1.69
-0.75	-0.7	0.1	60.92	0.44	86.50	0.62	1.42
-1	1.0	-1.0	41.42	0.30	270.00	1.94	6.52
-1	0.9	-0.9	43.21	0.31	264.94	1.91	6.13
-1	0.8	-0.8	44.93	0.33	259.79	1.87	5.78
-1	0.7	-0.7	46.59	0.34	254.52	1.83	5.46
-1	0.6	-0.6	48.2	0.35	249.15	1.79	5.17
-1	0.5	-0.5	49.75	0.36	243.66	1.75	4.90
-1	0.4	-0.4	51.25	0.37	238.04	1.71	4.64
-1	0.3	-0.3	52.71	0.38	232.29	1.67	4.41
-1	0.2	-0.2	54.13	0.39	226.39	1.63	4.18
-1	0.1	-0.1	55.52	0.40	220.33	1.59	3.97
-1	0.0	0.0	56.87	0.41	214.10	1.54	3.76
-1	-0.1	0.1	58.19	0.42	207.69	1.49	3.57
-1	-0.2	0.2	59.48	0.43	201.07	1.45	3.38
-1	-0.3	0.3	60.74	0.44	194.22	1.40	3.20
-1	-0.4	0.4	61.98	0.45	187.13	1.35	3.02
-1	-0.5	0.5	63.2	0.46	179.75	1.29	2.84
-1	-0.6	0.6	64.39	0.47	172.06	1.24	2.67
-1	-0.7	0.7	65.55	0.48	164.01	1.18	2.50
-1	-0.8	0.8	66.7	0.48	155.54	1.12	2.33
-1	-0.9	0.9	67.83	0.49	146.58	1.05	2.16
-1	-1.0	1.0	68.94	0.50	137.04	0.99	1.99

[a]This comparison is made for correlation combinations for which the coefficient of variation for the welfare measure was less than 0.5.

[b] SEA(CV) = Analytic standard error of compensating variation.

[c] SEN(CV) = Numerical approximation of standard error of compensating variation.

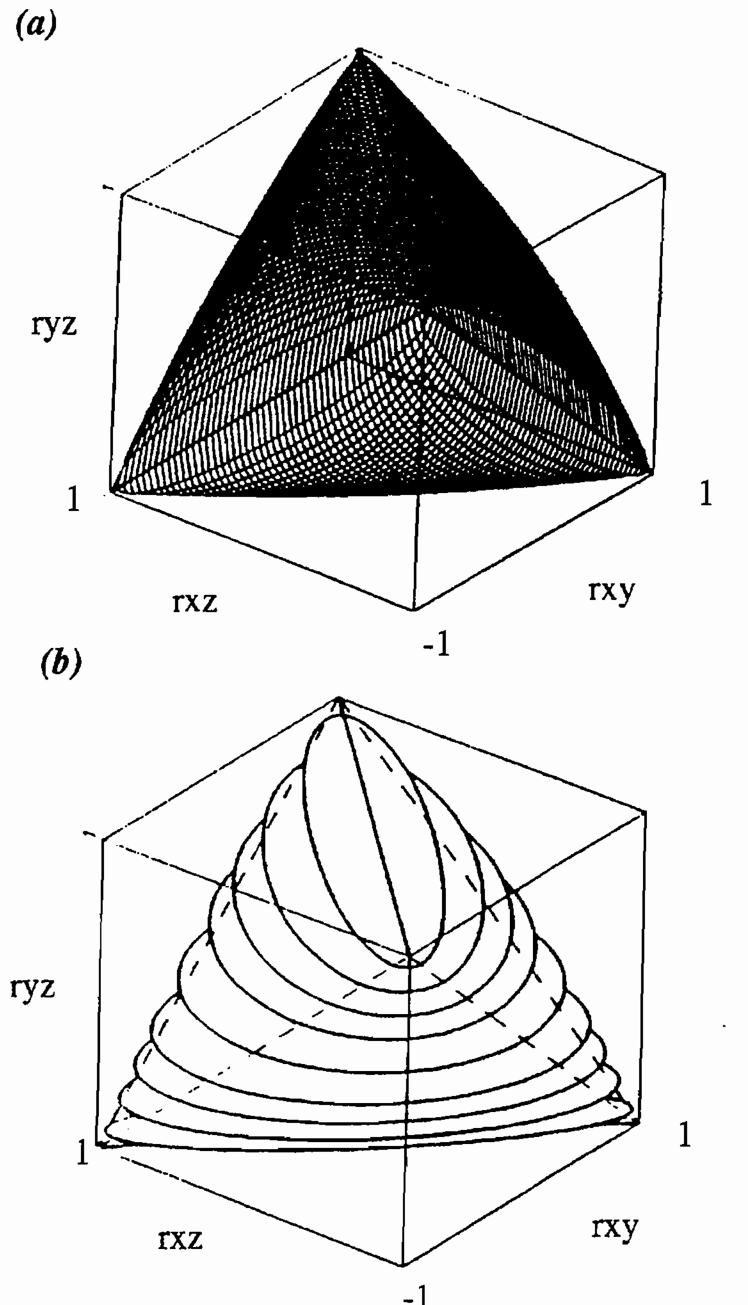
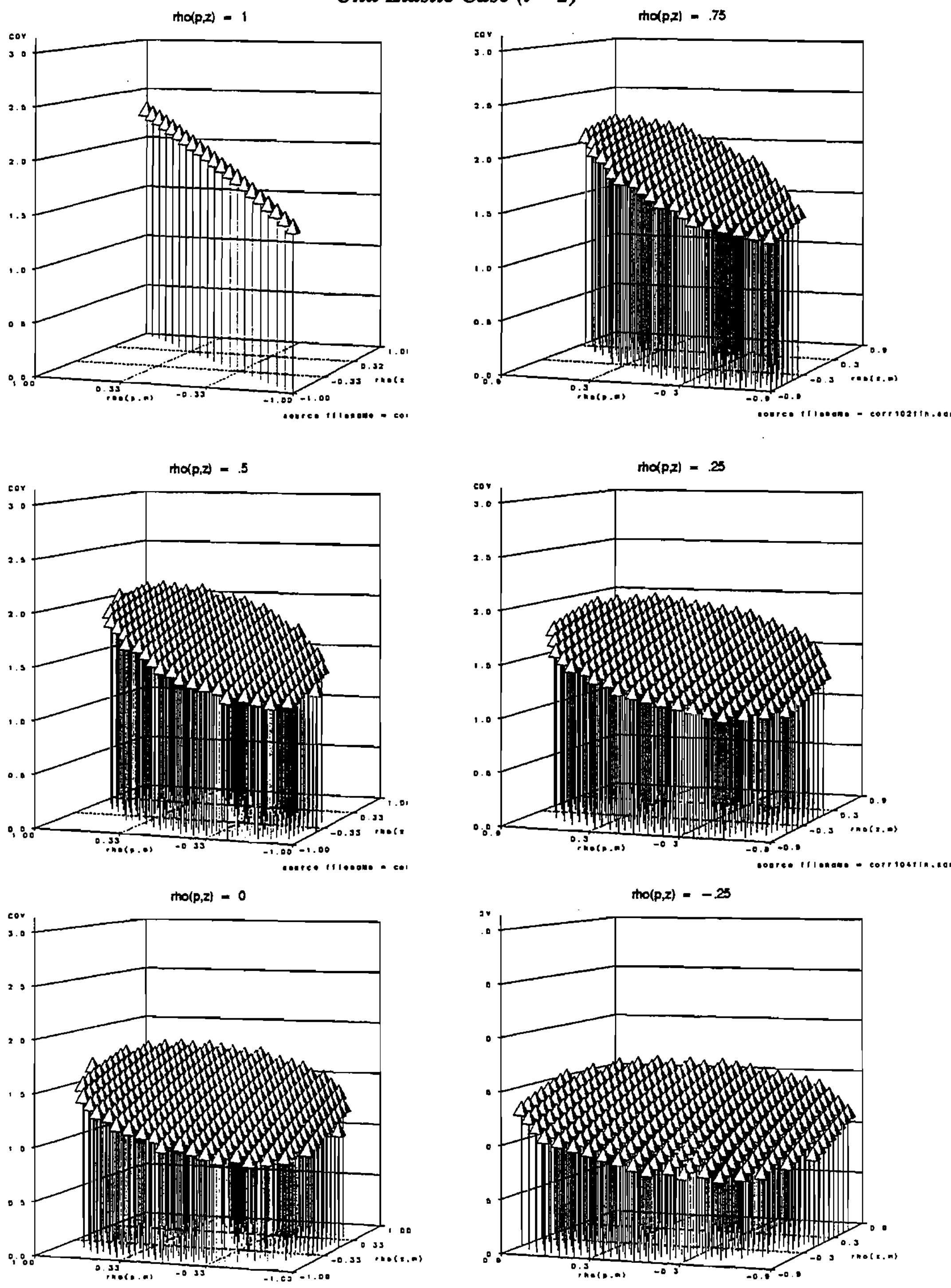


Figure 1:
(a) Set of all possible correlations between X, Y, and Z.
(b) Slicing this set at ryz yields ellipses.

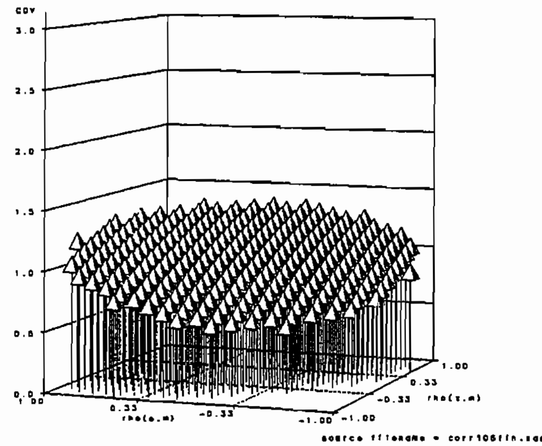
Figure 2

Scatter Plots of Coefficient of Variation for ρ_{pz}

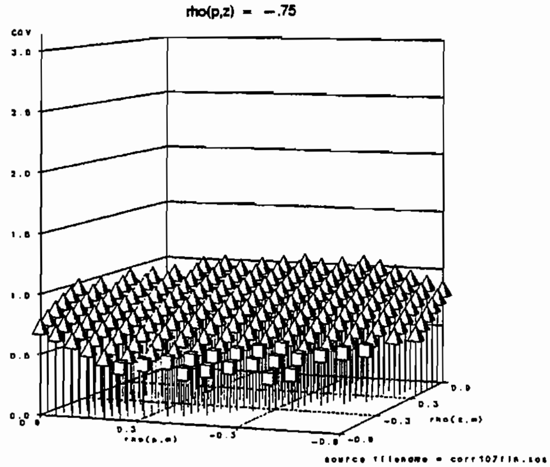
Unit Elastic Case ($t = 2$)



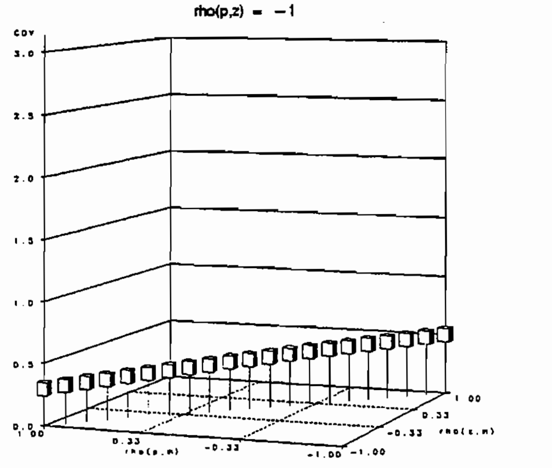
rho(p,z) = −.5



rho(p,z) = −.75



rho(p,z) = −1



Empirical Specification Requirements for Two-Constraint Models of Recreation Demand

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Empirical Specification Requirements for Two-Constraint Models of Recreation Demand

The literature on recreation demand is gradually becoming more sophisticated as researchers respond to the myriad conceptual and empirical challenges that are associated with this particular area of demand analysis. One of the most challenging and important areas of research is how to consistently integrate the role of time into recreation choices. The importance of modeling time in recreation demand has been known by applied researchers since early in the development of the literature (e.g., Clawson; Knetsch). The empirical literature has followed a distinct progression, from ignoring time entirely, to assuming time has a value which is a researcher-chosen fraction of the wage rate (e.g., following suggestions by Cesario), to allowing the data to determine the fraction (McConnell and Strand), to recognizing the differences between the values of time for salaried and hourly workers (Bockstael *et al.*).

Interestingly, there has been relatively little formal guidance about how to specify recreation demand models where time is an important constraint, beyond the basic case originally analyzed by Becker where time can be converted to money according to an exogenous labor supply function. The intuition behind the Becker analyis is that all demands should be functions of "full prices" and "full budgets," where time valued at the wage rate is included in the price and budget terms. One of the contributions of the Bockstael *et al.* paper was to point out that not all recreationists have the opportunity to "reveal" their marginal wage rate through participation in a discretionary labor activity, and that for these individuals the relevant value of time is endogenous. However, their paper does not provide any guidance on how to specify the value of time in such "corner solution" cases where the individual offers zero discretionary labor supply.

Perhaps partly as a result of the paucity of theoretical guidance, the literature has focused almost exclusively on the role of time "prices" (travel costs, typically, in the recreation demand model), while the role of the time budget in demand has been largely ignored. No doubt this is because researchers are well aware (thanks to the work of Knetsch and Cesario, among others) that consumer's surplus estimates of the net economic benefits of recreational activities are heavily influenced by the own-price coefficient, which will be biased if a systematic part of the cost of a recreational activity (the opportunity cost of time spent) is ignored. However, the common practice of forming a full "price" of recreation, and including this variable in demand with money income alone (i.e., omitting the time budget) cannot be a correct procedure as it violates the requirements of theory.

This paper develops the implications of the two-constraint recreation demand model that give rise to this and other insights for empirical practice. We develop the theoretical restrictions implied by the two versions of Roy's Identity when any consumption choice is made subject to two binding constraints. These restrictions are analogous to the Slutsky-Hicks equations of standard (single-constraint) consumer choice problems, though derived from a different conceptual basis in the choice problem.

In the context of choice subject to money and time constraints, three sets of necessary conditions provide additional symmetry structure for estimation and testing of recreation demand models. One relates cross-equation money price and money budget terms alone, one relates cross-equation time price and time budget terms alone, and one set of restrictions relates time and money price and time and money budget coefficients and the marginal value of leisure time. The first two sets of restrictions are fully observable, which means they can be imposed or tested for in estimation. The third set can be used to "reveal" the marginal value of time from properly-specified empirical recreation demand models.

These results provide the structure necessary to correctly specify two-constraint recreation demand models. Two points about their applicability are worth noting. First, they hold for all recreationists, whether or not they are making marginal labor supply choices along with recreation choice. They are of particular use in identifying the demand structure for recreationists with endogenous marginal values of leisure time, where the literature does not generally advance any particular requirements for specification. They also suggest ways that one can specify a marginal value of leisure time function as part of the structure of the demand model, and estimate its parameters as part of the model. Workers making marginal labor leisure choices in response to exogenous marginal values of leisure time (the "interior solution" case of Bockstael *et al.*) can be seen to represent a special case of the general two-constraint choice theory.

Second, it is important to emphasize that this paper is about the relationships between the covariates in the systematic part of recreation demand models. Because of this, they are applicable to all empirical recreation demands where time plays a role, whether single-equation or multiple-equation, whether continuous or discrete.

The basic results are developed in the context of a demand systems approach to recreation, because it is within this framework that much of the literature of how to treat recreation time has been developed. However, because many recent analyses have used count data or random utility formulations of the recreation choice model, we also show how the theoretical two-constraint requirements apply to these models.

Two-Constraint Recreation Choice Models

The standard consumer choice problem with two binding constraints provides the appropriate theoretical foundation for developing the specification requirements for recreation demand models when time has an opportunity cost.[1] Let $\mathbf{x} \equiv (x_1,...,x_n)$ be consumption goods with corresponding non-negative money prices $\mathbf{p} \equiv (p_1,...,p_n)$ and time prices $\mathbf{t} \equiv (t_1,...,t_n)$, and choices are made subject to a money budget constraint $M = \mathbf{px}$ and a time constraint $T = \mathbf{tx}$, both of which are strictly binding. The money and time budgets M and T can be thought of as resulting from a labor supply decision by the individual, which results in discretionary income and time to be allocated to leisure time activities and goods consumption.

Note that binding time and money constraints must characterize the model used whenever researchers argue that time spent in recreation has a "value" or opportunity cost. If the time constraint is non-binding, the marginal value of time is zero, the standard consumer choice problem results, and there is no bias to recreation benefit estimates from ignoring time. Intuitively, though, time must always be "spent" in some activity, so binding time constraints are highly plausible. Nonsatiation and the presence of numeraire activities with only one price (i.e, a positive money price and zero time price, or vice versa)[2] are sufficient for both constraints to bind.

Consider a consumer with utility function u($\mathbf{x}$,$\mathbf{s}$), with s a vector of shift parameters. The primal version of the choice problem is solved by the Marshallian demands $x_i = x_i(\mathbf{p},\mathbf{t},\mathrm{s},M,T)$ which are functions of both time and money prices and time and money budgets. The indirect utility function V($\mathbf{p}$,$\mathbf{t}$,s,M,T) for this problem is

$$V(\mathbf{p},\mathbf{t},s,M,T) \equiv \max_{\mathbf{x}} u(\mathbf{x}) + \lambda\{M - \mathbf{px}\} + \mu\{T - \mathbf{tx}\} \tag{1}$$

where, with both constraints binding, the ratio of the Lagrange multipliers on the time and money constraints, $\mu/\lambda = V_T(\cdot)/V_M(\cdot)$[3], is the money value of time.

Much of the recreation demand literature based on utility-theoretic foundations for the value of time notes that individuals observed at "interior" solutions with respect to labor supply effectively reveal their marginal value of time through their observed trades of time for money at a marginal or discretionary wage rate. This exogenous parameter can be used to collapse the two-constraint choice problem into a single-constraint problem of maximizing utility subject to full prices and full budgets, with the wage acting as the terms of trade between time and money (e.g., Becker). On the other hand, individuals at "corner solutions" work fixed hours and do not (or are not observed to) trade time for money marginally. Their marginal value of leisure time is endogenous, not observable as an exogenous parameter.

Empirical Implications of the Two Roy's Identities

The presence of an additional binding (time) constraint implies additional structure on the consumer choice problem. This structure can be developed by noting that with two constraints on choice, there are two versions of Roy's Identity, relating the price and budget slopes within each constraint.

Empirical recreation demand analysis is often based on incomplete demand systems estimated on a subset of consumption goods. In the two-constraint case, following Bockstael *et al.* we assume that the incomplete demand system estimated by the researcher is augmented by a time numeraire good which has a positive time price and a zero money price, and a money numeraire good with zero time price and a positive money price. As LaFrance and Hanemann have shown, welfare analysis can be conducted on incomplete demand systems conditional on the prices of goods excluded from the estimated system remaining unchanged. This is generally not true of partial demand systems, where separability of preferences leads to demand systems based on group budget, unless one also explains the allocation of overall income to group budgets (Hanemann and Morey).

Let goods 1,...,n (where $n \geq 1$) be the goods in the estimated incomplete demand system, with all having strictly positive time and money prices, and let good n+1 be the money numeraire and n+2 be the time numeraire good. The symmetry conditions for price and budget coefficients which follow apply to the n goods in the estimated demand system.[4]

From the envelope theorem applied to (1), we can see that $V_{p_j} = -\lambda x_j$, $V_{t_j} = -\mu x_j$, $V_M = \lambda$, and $V_T = \mu$, so that for all goods in the estimated incomplete demand system one can write

$$x_j(\mathbf{p,t,s},M,T) \equiv -V_{p_j}/V_M \equiv -V_{t_j}/V_T, \qquad \text{for } j=1,\ldots,n. \tag{2}$$

The two Roy's Identities in equation (2) are a source of parameter restrictions in the empirical demand system and prove useful for specification and identification of the marginal value of leisure time from demand system coefficients.[5]

Cross-Price Restrictions

Differentiating (2) with respect to p_i, one obtains two expressions for the Marshallian cross-money price slope $\partial x_j/\partial p_i$,

$$\partial x_j/\partial p_i = -[V_T \cdot V_{t_jp_i} - V_{t_j} \cdot V_{Tp_i}]/V_T^2 = -[V_M \cdot V_{p_jp_i} - V_{p_j} \cdot V_{Mp_i}]/V_M^2.$$

Noting that $V_{Tp_i} \equiv \mu_{p_i}$ and $V_{Mp_i} \equiv \lambda_{p_i}$, replacing the partial derivatives V_M and V_T with their respective shadow values λ and μ from (1), and using (2), this can be simplified to

$$\partial x_j/\partial p_i = (V_{t_jp_i} - x_j \cdot \mu_{p_i})/\mu = (V_{p_jp_i} - x_j \cdot \lambda_{p_i})/\lambda. \tag{3}$$

Similarly, the two expressions for the cross-time price derivative $\partial x_i/\partial t_j$ that follow from (2) are

$$\partial x_i/\partial t_j = (V_{t_it_j} - x_i \cdot \mu_{t_j})/\mu = (V_{p_it_j} - x_i \cdot \lambda_{t_j})/\lambda. \tag{4}$$

Since the middle term of (3) and the right term of (4) have the common term $V_{p_i t_j}$ ($\equiv V_{t_j p_i}$ by Young's Theorem), each can be solved for this term and equated, yielding a restriction on the cross-time and cross-money prices,

$$\partial x_i/\partial t_j = (\mu/\lambda) \cdot \partial x_j/\partial p_i + (x_j \cdot \mu_{p_i} - x_i \cdot \lambda_{t_j})/\lambda. \tag{5}$$

As a special case of (5), when i = j, the own-time and money price slopes are related by

$$\partial x_i/\partial t_i = (\mu/\lambda) \cdot \partial x_i/\partial p_i + x_i \cdot (\mu_{p_i} - \lambda_{t_i})/\lambda. \tag{6}$$

Equations (5) and (6) show how the marginal value of leisure time relates the time price slopes $\partial x_i/\partial t_j$ and the money price slopes $\partial x_j/\partial p_i$. This is not, in general, a simple relationship, as it is affected by the difference in quantity-weighted effects of each price change on the shadow value of the other constraint.

Because of the unobservables, (5) and (6) are not directly useful as sources of empirical restrictions on two-constraint demand models. However, by comparing with cross-budget effects, it becomes possible to derive such restrictions.

Cross-budget Restrictions

The Marshallian cross-budget effects are also derived by differentiating both versions of Roy's Identity in (2) with respect to M and T, yielding

$$\partial x_j/\partial M = -(\lambda_{t_j} + x_j \cdot \mu_M)/\mu = -(\lambda_{p_j} + x_j \cdot \lambda_M)/\lambda \tag{7}$$

and

$$\partial x_i/\partial T = -(\mu_{t_i} + x_i \cdot \mu_T)/\mu = -(\mu_{p_i} + x_i \cdot \lambda_T)/\lambda. \tag{8}$$

Because the cross-derivatives $\mu_M \equiv \lambda_T \equiv V_{MT}$, when (7) is solved for μ_M and (8) for λ_T, the two expressions can be equated. When this equality is simplified, the result can be written as

$$\partial x_i/\partial T = (\mu/\lambda) \cdot (x_i/x_j) \cdot \partial x_j/\partial M - (1/x_j) \cdot (x_j \cdot \mu_{p_i} - x_i \cdot \lambda_{t_j})/\lambda. \qquad (9)$$

Parameter Restrictions On Two-Constraint Demands

When (9) and (5) are compared, the general form of the Marshallian cross-equation restrictions in the two-constraint problem emerges as

$$\partial x_i/\partial t_j + x_j \cdot \partial x_i/\partial T = (\mu/\lambda) \cdot [\partial x_j/\partial p_i + x_i \cdot \partial x_j/\partial M], \qquad (10)$$

and again as a special case where $i = j$, the own-price and own-budget slopes must be related by

$$\partial x_i/\partial t_i + x_i \cdot \partial x_i/\partial T = (\mu/\lambda) \cdot [\partial x_i/\partial p_i + x_i \cdot \partial x_i/\partial M]. \qquad (11)$$

Equations (10) and (11) take a form comparable to the Slutsky-Hicks equations from standard consumer theory, and express necessary conditions which follow from utility maximization subject to two binding constraints. They are conceptually distinct from, though closely related to, the two sets of Slutsky-Hicks equations that result from the two expenditure minimization problems dual to the two-constraint utility maximization problem. The advantage of casting the requirements of theory in a form such as (10), though, is that all quantities $x_i(\mathbf{p},\mathbf{t},\mathbf{s},M,T)$ and $x_j(\mathbf{p},\mathbf{t},\mathbf{s},M,T)$ in (10) and (11) are Marshallian, not Hicksian, so they represent directly observable levels and slopes of ordinary demand.

To complete the comparative statics, when cross-money price slopes are compared to cross-money budget slopes, and cross-time price slopes are compared with cross-time budget slopes, the cross-equation restrictions are

$$\partial x_i/\partial p_j + x_j \cdot \partial x_i/\partial M = \partial x_j/\partial p_i + x_i \cdot \partial x_j/\partial M \tag{12}$$

and

$$\partial x_i/\partial t_j + x_j \cdot \partial x_i/\partial T = \partial x_j/\partial t_i + x_i \cdot \partial x_j/\partial T. \tag{13}$$

The necessary conditions represented in (12) and (13) further illustrate the empirical advantages of developing the symmetry requirements of two-constraint choice theory from Roy's Identities. All terms are observable, so these conditions can be directly tested for or imposed in estimating empirical recreation demand models.

Equations (10), (12), and (13) provide the general symmetry structure which empirical two-constraint consumer models must follow.[6] This has several implications for how leisure time enters the specification of recreation demand models. The analysis of these implications begins with the simplest case, most familiar in the literature, of exogenous values of time revealed through auxiliary choices recreationists make regarding labor supply. One finding is that the linear-in-parameters demand equation used by Bockstael *et al.* does not generalize readily to multiple equation systems, though the two-constraint theory helps identify alternative empirical functional forms, involving symmetric full prices and multiplicative full budgets, that do work.

Next the general "corner solution" case is considered, where there are no auxiliary labor supply decisions that reveal an exogenous value of time for the individual. We show that specifications involving symmetric full prices and multiplicative full budgets also satisfy the two-constraint restrictions, even though the marginal value of time in this case is endogenous and is, itself, a function of all parameters of the problem. The power of this result is that it shows how researchers can estimate value of time functions jointly with the recreation demand equations in models that satisfy the requirements of utility theory.

Implications for Models with Exogenous Marginal Values of Leisure Time

First it is shown how the general two-constraint restrictions in (10)-(13) encompass as a special case the most common formulation of time in the literature, where individuals are at interior solutions in the labor market, optimizing with regard to an exogenous discretionary wage w^D and offering a positive hours supply. If one of the goods in (1) is taken to have money price $-w^D$, time price 1, and zero marginal utility, that good corresponds to the hours supplied variable in the Bockstael *et al.* "interior solution case." As they show, its first order condition relates the two constraint shadow values as

$$\mu(\mathbf{p} + w^D \cdot \mathbf{t}, \mathbf{s}, M + w^D \cdot T) \equiv w^D \cdot \lambda(\mathbf{p} + w^D \cdot \mathbf{t}, \mathbf{s}, M + w^D \cdot T). \tag{14}$$

where all optimized choice variables are functions of full prices and full budget. From (14), it is clear that

$$\mu_{p_i} = w^D \cdot \lambda_{p_i} = \lambda_{t_i}, \tag{15}$$

and in light of (15), the term $(\mu_{p_i} - \lambda_{t_i})/\lambda = 0$ in (6) and $\partial x_i/\partial t_i = w^D \cdot \partial x_i/\partial p_i$; that is, as Bockstael *et al.* point out, all the Marshallian demands $h_i^I(\mathbf{p}+w^D\mathbf{t},\mathbf{s},M+w^DT)$, i=1,...,n, are functions of full prices and full budget. For this special case, (14) implies that (10) collapses to either (12) or (13), which are equivalent statements, depending on whether one wishes to characterize the demand restrictions in money terms or time terms.

This empirical model provides a useful illustration of the principles. For individuals at interior solutions, they estimated a single-equation model of the form

$$x_1 = \alpha + \gamma_1 \cdot (M + w^D \cdot T) + \beta_1 \cdot (p_1 + w^D \cdot t_1) + \gamma_2 \cdot q + \epsilon \tag{16}$$

where γ_1 and β_1 are the full budget and full price coefficients, respectively, and q is a quality argument. Clearly their model satisfies the own-price version of the two-constraint restrictions, given in equation (11), because $\partial x_1/\partial t_1 = w^D \cdot \beta_1$, $\partial x_1/\partial T = w^D \cdot \gamma_1$, $\partial x_1/\partial p_1 = \beta_1$, $\partial x_i/\partial M = \gamma_1$, and $\mu/\lambda = w^D$. For this model, (11) is then

$$w^D \cdot \beta_1 + x_1 \cdot w^D \cdot \gamma_1 = (w^D) \cdot [\beta_1 + x_1 \cdot \gamma_1],$$

which always satisfies the two-constraint requirement.

Multiple-Equation Interior Solution Models

Equation (10) goes beyond the single-equation incomplete demand case empirically estimated by Bockstael *et al.* to indicate the cross-equation restrictions on Marshallian demand coefficients required in multiple-equation systems of recreation demands where time is a constraint on choice. The linear-in-parameters specification does not work in the multiple-equation context because the cross-equation restrictions in (10) are violated, unless consumption quantities are constrained or there are no income effects. To see this, define a two-good incomplete demand system as

$$x_1 = \alpha_1 + \gamma_1 \cdot (M{+}w^D T) + \beta_{11} \cdot (p_1{+}w^D t_1) + \beta_{12} \cdot (p_2{+}w^D t_2) + \gamma_{12} \cdot q + \epsilon$$

$$x_2 = \alpha_2 + \gamma_2 \cdot (M{+}w^D T) + \beta_{21} \cdot (p_1{+}w^D t_1) + \beta_{22} \cdot (p_2{+}w^D t_2) + \gamma_{22} \cdot q + \epsilon$$

and for this system, equation (10) is

$$\beta_{12} \cdot w^D + x_2 \cdot (\gamma_1 \cdot w^D) = w^D \cdot [\beta_{21} + x_1 \cdot \gamma_2]$$

which defines a linear dependence between x_1 and x_2. If budget terms are zero ($\gamma_1 = \gamma_2 = 0$) and Marshallian cross-price effects are symmetric ($\beta_{12} = \beta_{21}$), equation (10) can hold without a linear dependence of consumption quantities.

It is no surprise that a linear Marshallian demand system in general fails to satisfy the two-constraint requirements, especially in light of LaFrance's work on integrability of single-constraint linear demand systems, which found that cross-price and income coefficients must be highly linearly dependent for integrability to be satisfied. The interesting thing about the result here is that the failure comes from a different facet of the integrability problem, namely satisfying the maintained hypothesis of two binding constraints on choice.

Satisfying the Two-Constraint Requirements in Multiple Equation Demand Systems

One can devise empirical interior-solution demand systems that satisfy the two-constraint requirements of (10)-(13), as for example with the system

$$x_i = h_i(\mathbf{p} + w^D \cdot \mathbf{t},\mathbf{s}) \cdot g(M + w^D \cdot T,s), \qquad \text{for } i = 1,\dots,n \qquad (17)$$

where the cross-partial price slopes are symmetric (i.e., $\partial h_j/\partial p_i = \partial h_i/\partial p_j$). The demand functions in this system have individual full-price effects $[h_i(\mathbf{p} + w^D \cdot \mathbf{t},\mathbf{s})]$ and a common full budget effect $[g(M + w^D \cdot T,s)]$. The price and budget slopes are

$$\partial x_j/\partial p_i = \frac{\partial h_j}{\partial p_i} \cdot g \qquad (18)$$

$$\partial x_i/\partial t_j = w^D \cdot \frac{\partial h_i}{\partial p_j} \cdot g \tag{19}$$

$$\partial x_j/\partial M = h_j \cdot g_M \tag{20}$$

$$\partial x_i/\partial T = w^D \cdot h_i \cdot g_M, \tag{21}$$

where $g_M \equiv \partial g(\cdot)/\partial M$. Using (17) to substitute out the $h_i(\cdot)$ and $h_j(\cdot)$ terms, (20) and (21) can be written as $\partial x_j/\partial M = x_j \cdot g_M/g$ and $\partial x_i/\partial T = x_i \cdot w^D \cdot g_M/g$, respectively. Using these with the price slopes in (18) and (19) and the fact that $\mu/\lambda = w^D$, equation (10) for this model is

$$w^D \cdot \partial h_i/\partial p_j \cdot g + x_j \cdot (x_i \cdot w^D \cdot g_M/g) = w^D \cdot [\partial h_j/\partial p_i \cdot g + x_i \cdot (x_j \cdot g_M/g)]$$

which holds given the symmetric Marshallian cross-price effects $\partial h_i/\partial p_j \equiv \partial h_j/\partial p_i$.

Clearly it is possible to design multiple-equation empirical demand systems to satisfy the two-constraint hypothesis implicit in models of recreation demand where the value of time plays an important role. An important question for further work is which forms of $h(\cdot)$ and $g(\cdot)$ are consistent with the other aspects of integrability (i.e., the negative definiteness and rank conditions identified by Partovi and Caputo).

Implications for Models with Endogenous Marginal Values of Leisure Time

The previous sections discussed the implications of the two-constraint choice structure for the special "interior solution" case where individuals reveal their marginal value of leisure time by making a discretionary labor supply choice. Because equations (10)-(13) hold for general marginal value of leisure time functions μ/λ, they describe the structure that must also apply to the system of demands $x_i = h_i^C(\mathbf{p},\mathbf{t},\mathbf{s},M,T)$ for those at corner solutions rather than interior solutions in the labor market. In this case, the marginal value of time (μ/λ) is an endogenous variable, which in general is a function of all parameters of the problem. What problems does the endogeneity of the marginal value of leisure time cause for specification of two-constraint demand systems?

Denoting this marginal value of leisure time function as $\mu/\lambda \equiv \rho(\mathbf{p},\mathbf{t},\mathbf{s},\mathrm{M},\mathrm{T})$, a set of sufficient conditions for (10)-(13) to hold is for the price and budget slopes to be related as

$$\partial x_i/\partial t_j = \rho(\mathbf{p},\mathbf{t},\mathbf{s},\mathrm{M},\mathrm{T}) \cdot \partial x_j/\partial p_i \qquad \text{for all i, j} \tag{22}$$

and

$$\partial \log(x_i)/\partial \mathrm{T} = \rho(\mathbf{p},\mathbf{t},\mathbf{s},\mathrm{M},\mathrm{T}) \cdot \partial \log(x_j)/\partial \mathrm{M} \qquad \text{for all i, j.} \tag{23}$$

One might anticipate problems with models using full prices $[p_i + \rho(\mathbf{p},\mathbf{t},\mathbf{s},\mathrm{M},\mathrm{T}) \cdot t_i]$ and full budget $[\mathrm{M} + \rho(\mathbf{p},\mathbf{t},\mathbf{s},\mathrm{M},\mathrm{T}) \cdot \mathrm{T}]$, because of the dependence of $\rho(\,\cdot\,)$ on prices and budgets. In deriving the price and budget slopes in (22) and (23), terms involving changes in $\rho(\,\cdot\,)$ with those prices and budgets must be accounted for.

For the case of endogenous marginal value of leisure time, equation (17) is

$$x_i = h_i(p_1 + \rho(\,\cdot\,) \cdot t_1,\ldots,p_n + \rho(\,\cdot\,) \cdot t_n) \cdot g(\mathrm{M} + \rho(\,\cdot\,) \cdot \mathrm{T},\mathbf{s}), \quad \text{for } i = 1,\ldots,n. \tag{24}$$

Demand equations of this form satisfy (22) and (23), which are sufficient conditions for (10)-(13) to hold, despite the dependence of $\rho(\mathbf{p},\mathbf{t},\mathbf{s},\mathrm{M},\mathrm{T})$ on the full set of prices and budgets. For this demand system, again assuming symmetric cross-partial price derivatives ($\partial h_j/\partial p_i = \partial h_i/\partial p_j$), the price slopes are

$$\partial x_j/\partial p_i = \tfrac{\partial h_j}{\partial p_i} \cdot g + \tfrac{\partial \rho}{\partial p_i} \cdot (\sum_k t_k \cdot \tfrac{\partial h_j}{\partial p_k} \cdot g + h_j \cdot g_M \cdot \mathrm{T}) \tag{25}$$

and

$$\partial x_i/\partial t_j = \rho \cdot \tfrac{\partial h_i}{\partial p_j} \cdot g + \tfrac{\partial \rho}{\partial t_j} \cdot (\sum_k t_k \cdot \tfrac{\partial h_j}{\partial p_k} \cdot g + h_j \cdot g_M \cdot \mathrm{T}) \tag{26}$$

while the budget slopes are

$$\partial x_j/\partial M = h_j \cdot g_M + \tfrac{\partial \rho}{\partial M} \cdot (\sum_k t_k \cdot \tfrac{\partial h_j}{\partial p_k} \cdot g + h_j \cdot g_M \cdot T) \tag{27}$$

and

$$\partial x_i/\partial T = \rho \cdot h_i \cdot g_M + \tfrac{\partial \rho}{\partial M} \cdot (\sum_k t_k \cdot \tfrac{\partial h_j}{\partial p_k} \cdot g + h_j \cdot g_M \cdot T). \tag{28}$$

Homogeneity of degree zero of Marshallian demands in the price and budget arguments of each constraint imply that the term in parentheses in each of (25)-(28) is identically zero. The terms $h_i \cdot g_M$ are the specific form of the income budget slope $\partial x_i/\partial M$ (for i=1,...,n) for the multiplicative demand given in (24), while the terms $(\partial h_i/\partial p_k) \cdot g$ are the money price slopes $\partial x_i/\partial p_k$ for all i,k=1,...,n. The term in parentheses is then

$$(\sum_k t_k \cdot \partial x_i/\partial p_k + \partial x_i/\partial M \cdot T) \equiv 0$$

by homogeneity.[7] Thus, for general value of time functions, (25)-(28) simplify to

$$\partial x_j/\partial p_i = \tfrac{\partial h_i}{\partial p_i} \cdot g \tag{29}$$

$$\partial x_i/\partial t_j = \rho \cdot \tfrac{\partial h_i}{\partial p_j} \cdot g \tag{30}$$

$$\partial x_j/\partial M = h_j \cdot g_M \tag{31}$$

$$\partial x_i/\partial T = \rho \cdot h_i \cdot g_M, \tag{32}$$

and as with (18)-(21), these slopes satisfy (22) and (23) and, hence, the two-constraint choice restriction in equation (10).

Thus the endogeneity of the marginal value of leisure time in the general corner solution case causes no additional problems beyond those raised in the interior solution case. The two-constraint restrictions must hold, and equations (17) and (24) are examples of how these restrictions can be satisfied with Marshallian recreation demand functions. Equation (24) further suggests how researchers can incorporate hypotheses about the structure of the marginal value of leisure time, as it may depend on prices, budgets, and other shifters **s**, directly into the demand model and estimate the marginal value of leisure time directly as part of the model.

This can be useful in the "interior solution" case as a validity check on the maintained hypothesis of the marginal value of leisure time (which is assumed to be w^D). It can often be difficult to measure the discretionary wage accurately even when people indicate they are trading time for money at the margin. For interior solution models, the researcher can specify w^D as one of the elements of **s** [*viz.*, $\rho(\mathbf{p},\mathbf{t},w^D,\mathbf{s},M,T)$] and test whether or not the empirically-measured discretionary wage is the best explainer of the marginal value of time and recreation demanded.

Implications for Current Practice

The two-constraint requirements have significant implications for current practice. One concerns the acceptability of formulating recreation demand models with full prices of travel and money income alone, which is common in the literature, both in conceptual and empirical models. The practice occurs in a wide variety of models, from standard recreation demand models (e.g., McConnell and Strand; Smith *et al.*) to count data models (e.g., Creel and Loomis, 1990; Englin and Shonkwiler; Hellerstein) to random utility models (e.g., Adamowicz *et al.*; Creel and Loomis 1992; Feather *et al.;* Morey *et al.*). Such formulations are inconsistent with the two-constraint requirements.

A second implication is that the value of time is "revealed" from coefficient estimates of correctly-specified models. This point is illustrated using empirical estimates from the Bockstael *et al.* model.

A Problem with Common Practice in Modeling Time

It is common in the literature to find recreation demand models that include a time price of recreation but no corresponding time budget variable. That is, full price (money cost plus time cost) and money income are included in the specification. This may be a concession to the difficulty of determining what the relevant time budget is for a recreation choice occasion,[8] or to data limitations. And it may be based on an assumption that the major specification issue is to avoid bias in the full price coefficient, on which welfare calculations are based. However, the point which may not be fully appreciated is that omission of the time budget variable invalidates the use of full prices in the model. This can be seen by considering each of the major types of models (continuous demand models, count data models, and random utility models) in light of the two-constraint requirements in (10)-(13).

Continuous Demand Models

The inconsistency of using full prices and money budget alone can be seen by recalling equation (11) for the single-equation demand model with exogenous marginal value of leisure time. This equation must hold in the empirical model if the researcher includes a time price (thereby invoking the maintained hypothesis of two constraints on choice). The rationale for omitting time budget must be an assumption that $\partial x_i/\partial T=0$, and when this is imposed on (11) the two-constraint restriction for the interior solution case is

$$\partial x_i/\partial t_i = w^D \cdot [\partial x_i/\partial p_i + x_i \cdot \partial x_i/\partial M]. \tag{33}$$

If the money income effect on demand is nonzero, then a demand model based on full prices and budgets, such as (16) or (17), would not satisfy (33). An obvious problem is the dependence on a consumption quantity (x_i), but any term beyond $\partial x_i/\partial p_i$ on the right side invalidates the use of full prices.

Time budgets play an integral role in the two-constraint recreation demand model, in maintaining the theoretical justification for the use of full prices. To avoid estimating incorrect models based on full prices and full budgets, they must be included in the empirical specification.

Count Data Models

Count data models are often used for single-equation demand models, to more realistically depict the distribution of the dependent variable, which is non-negative integer-valued. The principal difference from standard demand models is in the choice of the stochastic term of the model, which is usually assumed to be either Poisson or negative binomial (e.g., Greene). For example, the Poisson count model of recreation trips assumes that for individual i, the random trips variable X_i takes on the value x_i with $\text{Prob}(X_i = x_i) = e^{-\lambda_i}\lambda_i^{x_i}/x_i!$, with λ_i most commonly specified as $\lambda_i = e^{\mathbf{z}_i\boldsymbol{\alpha}}$, where $\mathbf{z} \equiv$ $[\mathbf{p},\mathbf{t},\mathbf{s},\mathrm{M},\mathrm{T}]$ is the vector of all demand covariates (time and money prices, time and money budgets, and shifters s) and α is the corresponding parameter vector.[9] The systematic part of this demand model is

$$E[X_i|\mathbf{z}_i] = \lambda_i = e^{\mathbf{z}_i\alpha}.$$

The analysis that develops equations (2)-(13) for this model is the same as for the demand systems case; equation (11) for the single count model (with general value of time function $\rho(\mathbf{z})$) is

$$\partial \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]/\partial \mathrm{t}_i + \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i] \cdot \partial \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]/\partial \mathrm{M} = \rho(\mathbf{z}) \cdot \big\{\partial \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]/\partial \mathrm{p}_i + \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i] \cdot \partial \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]/\partial \mathrm{M}\big\}.$$

This will always be satisfied if the systematic part of the demand model is a function of full prices ($\mathrm{p}_j + \rho(\mathbf{z}) \cdot \mathrm{t}_j$) and full budgets ($\mathrm{M} + \rho(\mathbf{z}) \cdot \mathrm{T}$). This can be seen from the fact that (denoting the full price i coefficient α_{p_i}) the relevant derivatives are $\partial \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]/\partial \mathrm{t}_i = \alpha_{p_i} \cdot \rho(\mathbf{z}) \cdot \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]$ and $\partial \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]/\partial \mathrm{p}_i = \alpha_{p_i} \cdot \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]$. Similarly, denoting the full budget coefficient as α_b, the money and time budget slopes are $\partial \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]/\partial \mathrm{T} = \alpha_b \cdot \rho(\mathbf{z}) \cdot \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]$ and $\partial \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]/\partial \mathrm{M} = \alpha_b \cdot \mathrm{E}[\mathrm{X}_i|\mathbf{z}_i]$.

Note that, as with the demand systems model above, specifications with full price and money budget alone are not consistent with these requirements. For multiple-equation count models with time constraints, the specification for the λ_j for each good j can be formulated along the lines of equation (24).

Random Utility Models

Random utility models are becoming very common in the literature, to explain the choice of which site, or recreation alternative, is chosen on a given choice occasion. The model is usually motivated based on a comparison of (indirect) utilities of the different alternatives, with the highest-valued alternative being chosen.

To see how the results on including time variables extend to this class of models, we can re-motivate equation (1) to describe the optimization of a continuous choice x_j

associated with discrete alternative j, for j=0,...,J.[10] The $J+1$ indirect utilities $V_j \equiv V(p_j,t_j,s_j,M,T)$ in (1) then describe the optimal utility derivable from each alternative, based on its own prices (p_j and t_j) and characteristics s_j, and on the consumer's money and time budgets. Incomplete observation by the researcher leads to an error ϵ_j for each alternative, and the optimal choice i is such that

$$V_i + \epsilon_i > \max_{j \neq i} V_j + \epsilon_j.$$

Given functional forms for the V_j and a distributional assumption for the ϵ_j (commonly, as iid extreme value or Generalized extreme value variates), the model can be estimated.

For the random utility model to validly represent economic behavior, it must be consistent with the requirements of theory, including the two-constraint requirements when time and money variables both enter the specification. For this model, the requirements can be seen most clearly from the two Roy's Identities in equation (2), since the indirect utility functions V_j are specified directly to motivate estimation of this model. Rearranging (2) slightly, for each alternative j, it must be true that

$$V_{t_j}/V_{p_j} \equiv V_T/V_M \tag{34}$$

and, therefore, the indirect utility functions of the different alternatives are linked as well; it must be true that

$$V_{t_i}/V_{p_i} = V_{t_j}/V_{p_j} \tag{35}$$

for all $i \neq j$.

A specification using full prices and full budgets again is sufficient to satisfy the two-constraint requirements in (34) and (35), regardless of whether the value of time is endogenous or exogenous. For example, writing

$$\mathrm{V}_j = \mathrm{V}_j[(\mathrm{p}_j + \rho \mathrm{t}_j),\mathbf{s}_j,(\mathrm{M} + \rho \mathrm{T})] \qquad \text{for all j,} \qquad (36)$$

it is easy to see that[11]

$$\mathrm{V}_{t_i}/\mathrm{V}_{p_i} = \mathrm{V}_{t_j}/\mathrm{V}_{p_j} = \mathrm{V}_T/\mathrm{V}_M = \rho.$$

Two points should be noted. First, since the most common specifications of the V_j are linear in parameters, e.g.,

$$\mathrm{V}_j = \alpha_{jp} \cdot (\mathrm{p}_j + \rho(\mathbf{z}) \cdot \mathrm{t}_j) + \alpha_{jM} \cdot (\mathrm{p}_j + \rho(\mathbf{z}) \cdot \mathrm{t}_j) + \boldsymbol{\alpha}_{jp} \cdot \mathbf{s}_j,$$

it is well-known that the budget terms drop out of the choice probability for alternative j, since they do not vary across alternatives. Thus a model with full prices of the alternatives, but no budget terms (e.g., Jakus *et al.*, Parsons and Hauber), is consistent with the two-constraint requirements.

A number of recent random utility formulations have postulated that the indirect utility of an alternative is the difference between the available budget and the cost of the alternative itself; e.g., using full prices and budgets,

$$\mathrm{V}_j = \mathrm{V}_j[(\mathrm{M} + \rho \mathrm{T}) - (\mathrm{p}_j + \rho \mathrm{t}_j),\mathbf{s}_j] \qquad \text{for all j.} \qquad (37)$$

It is easy to see that (37) satisfies the two-constraint requirements, because it is a special case of (36), which satisfies them.

However, in a number of cases in the literature, the income remaining after cost is calculated not like (37), but instead as a difference between money income and full cost, so that $V_j = V_j[(M - (p_j + \rho t_j), s_j]$ (e.g., Feather *et al.*; Parsons and Kealy; Kaoru *et al.;* Herriges, Kling, and Phaneuf; Montgomery and Needelman). Here, $\partial V_j/\partial M = \partial V_j/\partial p_j$ and $\partial V_j/\partial t_j = \rho \cdot \partial V_j/\partial p_j$, but $\partial V_j/\partial T = 0$, so the two-constraint requirements (34) and (35) cannot be satisfied in this conceptual formulation. Whether this raises a problem empirically depends on how the budget enters indirect utility: in the linear-in-parameters specification, the individual's budget (whether full or just money budget) cancels from the choice probabilities. With increasingly-sophisticated specifications being made possible by advances in computation speed and estimation techniques, including the development of linked participation-site choices, the specification of the individual's budget becomes more important to achieve consistency with utility theory.

Inferring the Marginal Value of Leisure Time from Utility-Theoretic Demands

A second empirical point is that the marginal value of leisure time can be measured from the demand coefficients of a properly-specified system. Perhaps the easiest way to make this point is to return to the empirical model of Bockstael *et al.*, this time using instead their corner solutions model, which was

$$x_1 = \alpha + \gamma_1 \cdot M + \gamma_2 \cdot T + \beta'\gamma_1 \cdot p_1 + \beta'\gamma_2 \cdot t_1 + \gamma_3 \cdot q + \epsilon$$

where q is an exogenous quality variable and $\beta' \equiv \beta/(\gamma_1+\gamma_2)$. Because this system is utility-theoretic, it satisfies (29-32) and, therefore, the two-constraint choice restriction in (11). From (22) and (23), it can be seen that the marginal value of time can be measured directly from the demand coefficients, as

$$\rho = (\partial x_1/\partial t_1)/(\partial x_1/\partial p_1) = (\partial \log(x_1)/\partial T)/(\partial \log(x_1)/\partial M).$$

For this model, $\partial x_1/\partial p_1 = \beta'\gamma_1$, $\partial x_1/\partial t_1 = \beta'\gamma_2$, $\partial \log(x_1)/\partial M = \gamma_1/x_1$, and $\partial \log(x_1)/\partial T = \gamma_2/x_1$, so (34) becomes

$$\rho = \beta'\gamma_2/\beta'\gamma_1 = \gamma_2/\gamma_1.$$

Bockstael *et al.* estimated the money price slope to be $\hat{\gamma}_1 = .024$, with a time price slope of $\hat{\gamma}_2 = 2.982$. Thus the marginal value of time in this model is a constant, $\rho \approx$ (2.982 units x/hour)/(.024 units x/\$) $\approx$ \$124/hour.[12] This contrasts with the estimate of the authors, who infer an estimate of \$60/hour for the marginal value of leisure time by comparing compensating variation estimates of welfare loss from eliminating the resource, denominated in dollar and time units.[13]

Conclusions

This paper develops a number of the structural requirements for the specification of recreation demand models where time is thought to be an important choice constraint. Coefficient restrictions take a form similar to the Slutsky-Hicks equations from standard consumer theory of choice subject to a single constraint, but arise from a different facet of the consumer choice problem when multiple constraints bind. The Slutsky-Hicks equations arise from the identity of Hicksian and Marshallian demands when income or utility is chosen appropriately, where the two-constraint restrictions arise from the equivalence of the two Roy's Identities that govern the response of Marshallian demands to parameter changes. Thus the two constraint restrictions relate observable Marshallian demand slopes and the generally-unobservable marginal value of leisure time. The restrictions relating cross-money price and money budget effects are fully observable, as are the restrictions relating cross-time price and time budget effects, so they can be implemented and tested for easily in practice. They provide guidance in two important areas not addressed by the existing literature: specification of how time should enter systems of demand equations, and how to deal with endogenous marginal values of leisure time. The two-constraint requirements apply to all types of empirical demand models where time is a second constraint on choice, whether motivated as systems of continuous demands, count data models, or random utility models. We show how these requirements can be applied to the specification of each of these classes of models.

An important finding is that the basic intuition of the simple model where time is an exogenous function, and the resulting demand is a function of full prices and full budgets, carries through to models where the value of time is endogenous. This should enable researchers to estimate value of leisure time functions auxiliary to the recreation demand model of interest. Individuals with exogenous values of time (those at "interior solutions" in the labor market) represent a special case where the marginal value of time is a constant or a known exogenous function.

Use of the structure required by the hypothesis of choice subject to two binding constraints is also helpful in empirical practice. We show that the approach used by much of the current literature on valuing time, to include full price of the activity but only money income, cannot be consistent with the requirements of consumer theory. We also show how the theory can also be used to infer the marginal value of time from properly specified two-constraint models. Thus the empirical two-constraint restrictions should be of considerable use in specifying theoretically-consistent demand systems and in inferring marginal values of leisure time from their empirical implementation.

Footnotes

1. This formulation is common in the recreation demand literature with utility-theoretic formulations for the value of time, such as Bockstael *et al.* Smith, in particular, examines some of the primal and dual properties of the two-constraint problem.

2. Examples of such goods include taking walks on the beach (positive time price but no money price) and making charitable contributions (positive money price but no–or nearly no–time price).

3. Parameters appearing as subscripts refer to partial derivatives; e.g., $V_{Tp_i} \equiv \partial^2 V(\mathbf{p},\mathbf{t},z,M,T)/\partial T \partial p_i$. The subscripts i and j index the consumption goods and their corresponding prices.

4. Thus, for example, a single-equation empirical demand function has n=1 and implies a three good world, with only the own-price and own-budget restrictions holding.

5. To minimize notational clutter, it is noted here that all restrictions developed below hold for goods i, j = 1,...,n; that is, they are restrictions which must be accounted for in the estimated incomplete demand system.

6. The results we develop here have also been derived by Partovi and Caputo, who examine the implications of the general K-constraint consumer choice problem. They also prove the negative semidefiniteness and rank conditions for the matrix of cross-equation restrictions for the general K-constraint problem.

7. It is well-known that the two-constraint Marshallian demand functions are homogeneous of degree zero in the parameters of each constraint (Partovi and Caputo; Smith). For general two-constraint demands, zero-degree homogeneity implies $\mathbf{x}(\theta\mathbf{p},\mathbf{t},\mathbf{s},\theta M,T) = \mathbf{x}(\mathbf{p},\mathbf{t},\mathbf{s},M,T)$, and differentiation with respect to θ yields $(\sum_k p_k \cdot \partial x_i/\partial p_k + \partial x_i/\partial M \cdot M)=0$. For the two-constraint model with full prices and full budgets [which has, as a special case, equation (24)], scale both money and time prices and budgets by θ (which leaves the ratio of Lagrange

multipliers, ρ, unchanged). Then homogeneity of degree zero implies $\mathbf{x}(\theta\mathbf{p}+\rho\cdot\theta\mathbf{t}$, s,$\theta$M+$\rho\cdot\theta$T) = $\mathbf{x}(\mathbf{p}+\rho\cdot\mathbf{t}$,s,M+$\rho\cdot$T), which upon differentiation with respect to θ yields $(\sum_k p_k\cdot\partial x_i/\partial p_k+\partial x_i/\partial M\cdot M)+\rho\cdot(\sum_k p_k\cdot\partial x_i/\partial t_k+\partial x_i/\partial M\cdot T)=0$. Since the first term in parentheses must be zero by homogeneity in the money budget alone, the second term in parentheses must be zero also.

8. In reality, it may not be too difficult to assess the time budget with at least as much accuracy as the relevant money budget variable, which is complicated by tax, credit, and household size differences.

9. Applications to recreation demand include Hellerstein, Creel and Loomis, and Englin and Shonkwiler.

10. The following arguments generalize readily to a set of continuous choices x_{ij}, i=1,...,I_j made to optimize the utility derived from the j^{th} alternative.

11. As with the demand systems case, homogeneity of degree zero of each alternative's indirect utility in the parameters of each constraint leads the terms involving ρ in the partial derivatives to cancel. There are fewer cross-equation restrictions in the random utility setting because typically researchers specify the utility of an alternative as a function of own price but no other prices.

12. Bockstael *et al.* note (p.298) that one of the undesirable features of the utility function they use for their illustration is that it implies a constant money-time tradeoff for the corner solution case.

13. The empirical magnitude of the difference is a secondary issue, as the denominator (money budget coefficient) is statistically insignificant anyway; the empirical estimate would also be affected if, for instance, separate parameters were estimated for people at corner solutions versus those at interior solutions. The main point is how knowing the structure of two-constraint models makes the value of time immediately available from demand coefficients for this model.

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Lessons Learned: A Systematic Look at Validity Issues in Conjoint Analysis

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Abstract

Contingent Valuation (CV) and Conjoint Analysis (CA) are both stated preference valuation techniques, and have a lot in common. To date, CV has been more widely used in the context of valuing natural resource amenities. Considerable attention has been focused on the validity of CV surveys, and over time we have learned much about how to assess and improve the validity of CV surveys. Many of the lessons learned from the debate over the validity of CV can be applied to the emerging use of CA to value natural resource amenities. This paper uses the content, construct and criterion validity framework to explore a variety of issues related to CA techniques. Among the issues receiving particular attention are: the types of information needed and the problem of information burden; the choice of valuation format (e.g., choice experiments or ranking or rating exercises); the appropriateness of CA for estimating nonuse values; issues associated with the multiple sources of uncertainty that may be present, including whether survey participants view the various alternatives within a CA exercise as equally plausible; and the ability of CA to estimate willingness to accept compensation. The goal of the paper is to use this framework to illuminate techniques to assess validity and to help CA survey designers improve the validity of their surveys. To this end, a variety of validity tests are discussed. The paper concludes with suggestions for future research on CA in the natural resource valuation setting.

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Introduction

Contingent Valuation (CV) and Conjoint Analysis (CA) are two similar methodologies that can be used to estimate economic values for natural resource amenities. While terminology has yet to be standardized (some authors use the terms stated preference or stated choice), by CA I am referring generally to the use of rating, ranking, or choice experiments that share the following similarities: survey participants are asked about their preferences for alternative bundles of natural resource amenities that are described by a set of attributes; and the levels of the attributes are varied across the alternatives. This allows us to tease out estimates of marginal values for each attribute, which is perhaps the major advantage of CA over CV. The purpose of this paper is not to compare the strengths and weaknesses of the two methods, but rather to adapt what we have learned about validity in CV studies to future research on CA.

The heated debate over the validity of welfare estimates obtained from CV studies has taught us a fair amount about assessing and improving validity, and many of these lessons have analogs for CA. While CA is relatively new to natural resource valuation, it has been widely used in marketing research. Validity research in the marketing literature has focused on predictive validity and test-retest reliability. Natural resource applications present different challenges, however. We are dealing with harder to define goods, which are not bought and sold, for which we often will not have observable behavior to compare predictions to, and for which high-quality surveys will be more expensive. A more useful validity framework for our purposes is the content-construct-criterion validity framework that has been discussed in the CV literature (Mitchell and Carson, 1989; Bishop, Champ and Mullarkey, 1995). This paper uses the 3-C's validity framework to discuss some validity issues that I think require attention as we invest more time and money in using CA for natural resource valuation.

Content Validity

Content validity basically deals with whether the structure of the choice problem and the information provided in the survey are conducive to measuring the economic object of interest. Content validity can be thought of as a necessary but not sufficient condition for the validity of a CA survey. If the survey fails to achieve content validity, there is little point in assessing construct or criterion validity since the survey is not measuring the object it is intended to. This section discusses several issues related to the content validity of CA studies that I would like to draw attention to.

Information

The most important requirement of any CV or CA survey is to provide participants with the proper information. For either type of survey, the two basic types of information needed are information about the amenity itself and information about the choice framework in which participants will operate. For CA, required information about the amenity includes specifying the status quo level of each attribute (each alternative will provide a target level for each attribute), the source of the change to the amenity, the geographic and temporal extent (i.e., the timing) of the change, and the certainty of the change. Information about baseline levels and any changes to substitutes and complements of the amenity should also be specified where possible.

The key to CA is getting the attributes right. Omission of key attributes will obviously bias the welfare estimates. Focus groups and other preliminary design techniques should be used to develop a survey that includes all the relevant attributes in a clear and useful manner. Experts can and should be used to determine levels of attributes in many applications, but to a large degree the set of attributes should be determined by listening to what lay people say is important to them. It should be noted that the number of attributes included in a survey may need to be restricted in some cases due to cognitive burden (which will be discussed further below). More research is needed into how many attributes participants can effectively deal with in the environmental context.

In addition to selecting the proper attributes, the survey designer will need to select the range of levels and the number of levels to be used across alternatives for each attribute. This is similar to the issue of the number and range of bids to use in a CV survey. In the CA context, Louviere and Timmermans (1990) suggest that in choice experiments, where the individual is asked to choose the preferred alternative from a set of two or more, attributes with a large range of levels may receive more attention that attributes with small ranges. An interesting experiment would be to offer two or more subsamples different ranges for the same attributes. If this results in different marginal values for the same attribute, CA estimates will be open to criticism as being arbitrary.

It is also important to get the economic setting specified accurately. Features of payment mechanism that have been identified as important in the CV literature include the payment vehicle (e.g., income taxes, user fees, etc.), the decision-making unit, the timing of the payment(s), and the prices of substitutes and complements. Three aspects of the context of valuation need to be addressed. The survey should specify all parties that will pay for the change as well as who is expected to be affected by the change. Second, the survey designer needs to decide whether to measure willingness to pay (WTP) or willingness to accept (WTA), or both. Third, the value elicitation device needs to be selected. In CV, this refers to the choice between open-ended, payment card, or referendum questions. In CA, the researcher needs to choose between ranking, ratings, pairwise or three-way choice experiments, or some combination of these formats.

Ranking experiments ask participants to rank a set of alternatives according to their preferences. Ratings experiments typically ask for a numerical rating, often using a 10-point scale, for each alternative. Choice experiments ask the participant to select a preferred alternative from a set of two (pairwise) or more (three-way, four-way, etc.) alternatives. Each format has its strengths and weaknesses. I do not go into much detail here (see Louviere and Timmermans (1990), McKenzie (1993), and Roe, Boyle and Teisl (1996)), but I would like to argue that we should explore the differences carefully. Choice experiments utilize the random utility model that we are used to working with, and therefore offer some advantages over ordinal ratings or rankings, which the researcher must then translate into welfare estimates. However, choice experiments may suffer from biases related to the task of making comparisons, such as greater attention being paid to attributes with high variances. Research on decision heuristics for each form of CA would be very useful in helping us understand what survey participants are actually doing.

Nonuse Values

I want to touch briefly on how nonuse values are handled in CA studies. When participants are evaluating any particular alternative, their nonuse value for the amenity in question will likely influence their response, so CA analysis does incorporate nonuse values. We should avoid the temptation to try to measure nonuse values directly, however, since it is not very tenable to tie nonuse values to any particular attribute. One issue I do see for CA researchers to deal with is how nonuse values affect the choice of functional form specified for the indirect utility function. The recent paper by Rollins and Lyke (1998) formalized the idea that nonuse values are likely to exhibit diminishing returns to scale. This is inconsistent with the common CA practice of specifying an indirect utility function that is linear and additive in the attributes. With nonuse values present, we would not expect a doubling of the change in attributes to result in a doubling of WTP. Thus other functional forms should be considered when nonuse values are expected to be nontrivial. To the degree that this alters the design of the choice sets, the functional form issue needs to be addressed prior to survey design rather than after the data are collected.

Uniform Plausibility

The different structures of the choice tasks in CV and CA studies raises an additional issue for content validity. There is an implicit assumption in CA that participants find each alternative equally plausible. This assumption makes sense for marketing applications, where there is no uncertainty regarding the well-defined attributes of the product. However, anecdotal evidence from a number of CV surveys supports the notion that people are skeptical that humans can successfully implement large scale improvements in environmental amenities. Specifically, some people are likely to find a small-scale improvement much more likely to be achieved than a large-scale change. People do not believe humankind can perfectly replicate natural processes, thus they will not believe, for example, that manmade wetlands will work as well as natural wetlands. In CV, there is no such issue since while people may be uncertain about the change being offered, there is only one possible change, not many, each with different subjective probability of success. In CA, attribute differences are assumed to be the only differences between alternatives. Hence, the presence of plausibility differences between alternatives will bias estimates of marginal attribute values. In applications with relatively little uncertainty, this may not be much of an issue, but many environmental applications do involve considerable uncertainty. Thus it seems incumbent upon the CA researcher to invest effort in the design phase aimed at determining whether people perceive plausibility differences across scenarios, and to minimize them to the extent possible.

Information Burden

In terms of accuracy of welfare estimates, I think that CA sacrifices precision in favor of flexibility when compared to CV. If we look at it from the perspective of the information burden placed on the survey participant, CV generally asks people to consider two situations – a reference or baseline situation and a target situation. CA, on the other hand, often asks people to evaluate between six and sixteen or more alternatives. If you accept that participants have a fixed burden capacity for either CV or CA, after which point more information is either ignored or the participant starts forgetting or confusing information, then CA must either provide less information per alternative or risk lowering the quality of responses due to information overload. Clearly some information will be constant across conjoint alternatives, so there are some economies of scale. However, some information will not be constant across alternatives. For example, the conjoint surveys I have seen have not specified differences in the time required to implement various alternatives, but it is quite conceivable that larger scale changes will take longer to implement. This type of information may be omitted in an effort to limit the number of attributes included in the survey, and may not matter to some participants, but it is information that can be more easily included in CV, and increases the precision of the definition of the economic construct being considered. My intent here is not to condemn CA since some of this information may not be very important to people, but I do wish to suggest that due to information burden, CA seems to offer less precision than CV, all else equal.

Assessing Content Validity

Assessing content validity is largely a subjective endeavor. As with CV, it will be very important for CA researchers to heavily invest in the design phase. Input and feedback from lay people will be critical elements in developing a successful CA survey. That being said, there are two techniques that I and others have found helpful for developing some quantitative information on content validity.

First, True-False quizzes preceding the actual valuation or choice questions can be used to achieve several goals. This type of exercise causes many participants to review the information provided. This translates into better understanding and absorption of information, and therefore better informed responses. It also provides the researcher with evidence of how effectively the survey communicated important information. CA appears to offer less opportunity to fully employ True-False questions than CV since key attribute information varies by alternative. However, information that could be included in True-False questions is the status quo level of each attribute, and facts that are assumed constant across alternatives (perhaps the timing of the change and the source of the change, and questions about the payment vehicle).

The second type of quantitative information that can be obtained on content validity comes from follow-up questions that explore whether participants accepted key elements of the scenario. If large numbers of participants reject key elements of the scenario, the survey clearly suffers from a lack of content validity. Potentially important issues include whether the participant believed the payment vehicle, such as whether they really believed the tax would be a one-time only payment; whether they believed the alternatives would actually cost the amount specified; whether certain changes that were presented as certain to occur were perceived as certain or uncertain; and whether the scenarios were equally plausible. Questions about plausibility would need to be carefully worded. Simply asking if they found all scenarios equally plausible would probably yield lots of "yes" responses in an effort to please the researcher. A better way to ask the question might be, "Were there one or more alternatives that seemed less likely to occur than the others?" This could be followed up questions asking for identification of these alternatives, and asking how much it affected their responses.

Construct Validity

Construct validity centers on the relationship between the measure of interest and other economic variables (referred to as theoretical construct validity) or other measures of the value of the amenity (convergent construct validity). Much of the CV validity literature has focused on testing construct validity. With a little forethought, CA surveys can be structured to gather the types of information that will allow for a variety of tests of construct validity.

<u>Theoretical Construct Validity</u>

Theoretical construct validity is concerned with how well the relationship between the measure and other economic variables conforms to economic theory. To assess theoretical construct validity, the survey needs to be designed to collect the appropriate information. This can involve both gathering socio-demographic information and selecting a set of questions that allows for testing of various hypotheses. Below I discuss how some of the tests that have been proposed in the CV literature can be applied to CA studies, as well as some additional tests that are possible due to the multiple-question format of CA.

Scope Tests

Theory predicts that up until the point of satiation, larger quantities of a normal good should engender larger WTP estimates. Critics of CV contend that CV estimates are not sensitive to the scope of the amenity (Kahneman 1986, Diamond et al. 1993, Desvousges et al. 1993, Schkade and Payne, 1994). A number of CV studies have included scope tests by having one subsample value a larger change in one or more attributes than a second subsample. The vast majority of these studies show that CV estimates can in fact be sensitive to the scope of the construct being measured (e.g., Boyle, Welsh and Bishop 1993, Carson, Wilks and Imber 1994, Mullarkey 1997). With conjoint studies, scope tests are essentially built into the empirical analysis. If the coefficients of the attributes are significantly different from zero and of the proper sign, then the welfare measure will be affected by changes in attributes as predicted by theory. If the coefficient on an attribute is not statistically significant, but the attribute was identified by focus groups as being important, something is amiss.

Demographic Variables

Relative to CV, it is harder for CA studies to demonstrate that welfare estimates are related to demographic variables in the expected manner. With CV, demographic variables can be included as regressors in bid equations to help explain variation in WTP. Unfortunately, this will not work for CA since these variables are constant across an individual's choices. However, the data can be split into demographic groups (kids vs. no kids, urban vs. rural resident, income above vs. below some amount, etc.), and likelihood ratio tests can be used to check for differences in coefficients.[1]

[1] Likelihood ratio tests determine whether the estimated coefficients are equal across samples by comparing value of the log likelihood function (LL) for each the two samples to the value of the log likelihood function for a pooled sample. The likelihood ratio test statistic (LR) is calculated as LR = -2 * [LL(group A) + LL(group B) - LL(pooled)], and compared to the chi-square distribution with degrees of freedom equal to the number of estimated parameters (Judge et al.. 1988).

Adding-up Tests

Economic theory implies that WTP for good A, plus WTP for good B conditional on already having purchased good A, should equal WTP for A and B together. A test of this hypothesis is called an adding-up test. With CV, adding-up tests can be done using split samples. The major difficulty lies in accounting for income and substitution effects. It will be more difficult to conduct adding-up tests with CA. The prospect of asking an individual to evaluate two separate amenities, and then to evaluate them together as part of a CA study, is not promising. Aside from the usual within-sample issue of independence, this task would double the information burden placed on the individual and could be somewhat confusing.[2] Therefore, three subsamples would be needed. Group 1 could be asked about good A, group 2 about good B, and group 3 about A and B combined. Group 3 would obviously face a larger information burden, but it is hard to see how this could be avoided. Given the complexity of the task, amenities that can be described by relatively few attributes would be preferred candidates for this type of test. As with CV, income and substitution effects will need to be accounted for. An adding-up test of this sort would considerably increase the expense of a CA study (and some funding sources may not be interested in paying for this type of methodological experiment), but could be very informative.

Transitivity

With choice experiments, one can test the transitivity assumption. The survey can be designed to ask an individual to choose between alternatives A and B, then between B and C, and finally between A and C. For those that preferred A to B and B to C, transitivity requires them to prefer A to C. The converse is also true; however, if A and C are both preferred to B, then theory offers no prediction on the relationship between A and C. This type of test would be fairly easy to include in a CA study. Inevitably some participants will violate the transitivity assumption (they should be removed from the sample). The larger the percentage of participants that fail this test, the more questionable would be the theoretical construct validity of the survey.

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2 Within product adding-up tests are not appropriate since functional form will dictate the results. For example, WTP to improve the first three attributes plus WTP to improve the second three attributes will always equal WTP to improve all six attributes if the common linear additive utility function is specified. Similarly, other specifications will drive the welfare estimates.

Monotonicity or Dominance

A similar test for violations of the monotonicity hypothesis can be incorporated into the design of the alternatives. If one alternative offers equal or greater levels of each (good) attribute at the same cost as a second alternative, the first alternative is said to dominate the second, and should always be preferred by the participant. With ranking or rating exercises, testing this is straightforward. With choice experiments, there are several options available. The simplest way would be to ask the participant to choose between a dominating alternative and the alternative that it dominates. Since this might render the choice too obvious, a less direct test can be devised. Consider three alternatives, where F strictly dominates D, and E is any alternative that neither dominates nor is dominated by D. Ask participants to choose between D and E, and between E and F (the order of questions should not matter). Monotonicity requires those participants who prefer D to E to also prefer F to E (the test is indeterminant for those that prefer E to D). One possible explanation for violations of the monotonicity assumption is that the participant does not place equal probabilities of success on each alternative. If the participant does not believe that the dominant alternative is politically or physically feasible at the cost specified, or feels that it is less likely to be realized than the dominated strategy, she may feel less inclined to choose the dominant strategy.

Ranking Attributes

One simple piece of evidence that can be collected in every CA survey is a non-monetary ranking of attributes. Participants could be asked before the central questions to simply rank the importance of the various attributes. This ranking should match or be very close to the ranking of attributes revealed by the marginal attribute values. A lack of consistency between the two rankings could result from a number of factors, ranging from a lack of content validity to inappropriate econometric analysis. Unfortunately, the small sample sizes found in focus groups may not provide an early warning of this problem, but perhaps the larger sample sizes of pilot studies would allow identification of a problem before the final survey instrument is fielded. Subsequent focus groups could then be conducted to uncover the root of the problem.

Convergent Construct Validity

Convergent construct validity is the degree to which the estimate in question resembles other measures of the same construct. CA estimates can be compared to either CV estimates or revealed preference (RP) estimates. Comparisons of CA to CV include Hanley et al (1998), Boxall et al (1996), Adamowicz et al (1998). A few studies comparing dichotomous-choice CV to CE show that the CV estimates tend to be larger than the CA estimates. Comparisons to RP estimates may be possible when the construct has little or no nonuse value. In fact, TCM and CE share the RUM framework, and can be combined into one data set, as Adamowicz, Louviere and Williams (1994) did. These types of comparisons are helpful and should be encouraged where applicable.

Another test of convergent construct validity would be to use multiple conjoint-type formats to estimate values for the same amenities. Thus a study that compared welfare estimates of an amenity derived from various forms of ratings, rankings, and choice experiments would be one type of convergent validity test. There are a few studies that do this to some degree, including McKenzie (1993) and Roe, Boyle and Teisl (1996). Coupled with research on the types of decision heuristics participants use in each format, this would be a most informative study.

Criterion Validity

Criterion validity considers the relationship between the measure and an alternative measure that is closer to the underlying construct. In order to assess the criterion validity of a CA study, one would therefore need to have an external welfare measure that is unequivocally closer to the "true" total value than the CA estimate.[3] As with CV, it will be quite difficult to assess the criterion validity of CA studies. For the applications we are concerned with, there will seldom be an observable criterion that offers a measure that is unequivocally closer to true total value. This is particularly true for applications dealing with nonuse values.

Some authors may argue that CA studies of market goods can be compared to actual market behavior or to simulated markets as a test of criterion validity. While this is appropriate

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[3] The existence of a measure that is considered closer to total value does not preclude the use of CA in cases where CA studies would be less expensive. If estimates from a less expensive technique consistently approach the criterion measure of a more expensive technique, it may prove cost effective to adopt the less expensive measure.

for market goods, it is not clear that these comparisons shed any light on the criterion validity of CA applications that deal with nonmarket and nonuse amenities. The contexts are quite different. There is likely to be supply-side uncertainty with environmental amenities, and the alternatives being considered may involve irreversible impacts on the amenities. Respondents may have less experience with the amenity or similar amenities than in market-goods contexts, and they may have different incentives to carefully search their preferences. The penalty, in terms of welfare loss, of making a poor decision for an inexpensive private good is likely to be small, potentially reversible, and the loss is borne solely by the respondent or the respondent's household. With public goods, on the other hand, not only might a poor decision be irreversible, but it potentially affects millions of other lives, human and nonhuman. This is an added responsibility that some civic-minded people take very seriously. Thus while it can be argued that if CA does not work well for private goods it would be unlikely to work well for public goods, it can also be argued that respondents have greater incentives to carefully search their preferences in studies of public goods. Therefore I am not too comfortable with using private goods to learn about how people value public goods, and even less so when the public goods have nonuse value components.

A second avenue for criterion validity assessments is the use of simulated market experiments that deal with nonmarket amenities. Simulated markets do produce observable behavior, which may be considered to lead to better welfare estimates as long as the simulated market is appropriately structured (simulated markets need to meet the same content validity standards as CV or CA studies).[4] However, it is exceedingly difficult for most researchers to have sufficient control over natural resources and payment collection mechanisms to actually construct and utilize an appropriately structured simulated market for a nonmarket amenity (particularly for those that evoke nonuse values). It is also necessary to have a large enough sample to fund the project in order for participants to find the survey realistic.

Simulated markets for CA studies will be slightly different than those used in conjunction with CV surveys. In each case, split samples are used, with the simulated market survey instrument differing from the hypothetical CV or CA instrument only in that participants are told

[4] It should be noted that unless the simulated market can be shown to have completely optimal incentive structure (e.g., no incentive to free-ride), it should be viewed as a test of convergent construct validity rather than criterion validity.

that they will actually have to pay for the project being evaluated. Given the structure of CA studies, the data will have to be analyzed before payments are made in order to identify the alternative that provides the largest welfare gain (it does not make much sense to first collect for each individual's preferred alternative and then make refunds or require additional payments from those participants that did not initially select the best alternative). Thus the simulated market participants are told that they will be required to pay for whichever alternative is chosen as the best by the group. Assuming the enforcement mechanism is effective[5], several questions can be asked of the data. Do the CA and simulated market instruments identify the same alternative as the best? Do they provide identical rankings of the marginal attribute values? And are these marginal values statistically different between treatments?

Laboratory experiments, typically using college students, are another form of simulated market. These experiments are useful as a relatively inexpensive method for exploring survey design issues, such as the range of levels or the number of levels to use per attribute. However, they have several features that make them suspect for measuring welfare changes. First, students are not a representative sample of the target population for most environmental projects. Second, a fairly large sample would typically be needed to actually pay for an environmental project of any magnitude. A sample size that is too small (or too large) for the project weakens the plausibility and validity of a survey. Third, it is unlikely that managers of natural resources will allow a handful of students to determine the availability of the type of public amenity that we are interested in.

Conclusions and Future Research

Conjoint Analysis has the potential to be quite useful for economic analyses of natural resource and environmental amenities. The ability to measure marginal values for individual attributes is useful both for determining the design of public projects and for use in benefits transfer studies. CA studies face many of the same challenges as CV studies, as well as some that specific to the structure of CA. To help realize its potential, CA could benefit from the same focus on validity that CV has.

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5 If payments are not collected from each individual, the validity of the simulated market results is questionable.

There is much to be explored as we apply CA to the natural resources – environmental context. Uncertainty, irreversibility, nonuse values and the public-goods nature of natural amenities are issues not generally dealt with in the marketing literature. The essence of CA is the tradeoffs between attributes. We need to know more about the decision heuristics participants use to make these tradeoffs. Decision heuristics may well vary between different forms of CA. For example, differences in the variance of levels between the attributes may cause some attributes to receive too much or too little attention in choice experiments, but this may not be a problem for ranking exercises. We also need to know whether participants consider each alternative as equally plausible, and if not, how that affects their decision-making process.

In general, issues related to information burden need to be explored. Given that the information burden can be large, we should explore such design considerations as how many attributes are participants willing to consider, and how many levels for each attribute should be used. It may be that the number of attributes presents a greater cognitive burden than the number of levels per attribute. Increasing the number of levels may therefore increase statistical power without substantially increasing burden, and may also help avoid the potential variance bias discussed above. These types of considerations are critical to developing CA surveys with strong content validity. Content validity needs to receive serious attention, perhaps even more so than with CV since it has been argued that due to greater information burden, CA offers improved flexibility at the potential expense of precision.

One area that is very intriguing is the possibility that CA may produce reasonable estimates of WTA compensation for environmental decrements. Surveys in which some alternatives offer improvements and some offer deteriorations, thereby allowing the participants to evaluate both "goods" and "bads", may reduce the reluctance people have shown in CV studies to explicitly trade the environment for money. Studies that combine WTP and WTA questions should test for status quo effects, as in the Adamowicz et al. (1998) study. As for analysis of the data, be it WTP or WTA or both, sensitivity analysis of the choice of functional form for the indirect utility function would be useful. Most studies to date have relied on linear functions, but the presence of nonuse values makes it unlikely that marginal attribute values are constant.

CA studies can and should be designed to allow for multiple validity tests. The basis of CA is to observe tradeoffs, and thus including both a transitivity test and a monotonicity test may be overkill. But half the sample could receive a transitivity test and the other half a monotonicity test without compromising statistical power. One test that can easily be included in any study is to ask participants to rank the attributes in order of importance. The aggregate ranking and the ranking based on marginal attribute values should be consistent. Where appropriate, comparisons to either CV or RP studies can be useful for establishing convergent construct validity. However, as with CV studies that include nonuse values, there is no good way to test CA studies for criterion validity when nonuse values are present. Hopefully, as more attention is focused on assessing and improving validity, additional validity tests will be identified that are appropriate to the structure of CA.

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WELFARE IMPLICATIONS OF SITE AGGREGATION: A COMPARISON OF CONDITIONAL LOGIT AND RANDOM PARAMETERS LOGIT ESTIMATES

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A COMPARISON OF CONDITIONAL LOGIT AND RANDOM PARAMETERS LOGIT ESTIMATES

ABSTRACT

This paper investigates the relationship between site aggregation and calculated welfare effects, comparing a random parameters logit (RPL) specification with a conditional logit (CL) specification. An empirical application to a Montana angling data set where the site definition varies from less aggregate river sites to more aggregate river sites is presented. In this application, the RPL models produce substantially different welfare estimates across the two alternative site definitions, while the results from the CL model are similar. These results indicate important links among IIA violations, site definition, and model specification, where less aggregate sites may cause larger deviation from the IIA property and hence necessitate a more flexible model specification such as the RPL.

1. INTRODUCTION

The effects of model specification on the estimated value of sites or site attributes has received considerable attention in the recreation-demand literature. This interest is in part because of legislation that holds polluters liable for environmental damages making the magnitude of the estimated damage subject to intense peer and court scrutiny.[1] Defining the alternatives that comprise an individual's choice set is fundamental to estimating any random utility model (RUM).[2] This paper focuses on defining alternatives and analyzes the welfare impacts of site aggregation, by comparing two alternative definitions of river sites.[3]

The link between site definition and the Independence of Irrelevant Alternatives Property (IIA) is considered by estimating both a CL and RPL model, which handle error correlation differently. The RPL model does not require the IIA property, which the simple CL does.[4] The sensitivity of the RPL model to alternative site-definition strategies is of further interest given the considerable computer resources required to estimate the more flexible RPL.[5] A smaller choice set also can substantially reduce model estimation resources, but unlike the simple CL model, an RPL model cannot be consistently estimated using known random-draw techniques.[6]

Ben-Akiva and Lerman (1985) label alternatives as either "aggregate" or "elemental". They define elemental sites to be a set of mutually exclusive and collectively exhaustive sites considered by individuals. In other words, a spatial area

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[1] Under Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) and the Oil Pollution Act of 1990 (OPA) trustees may seek damages for the cost of restoring, replacing, or obtaining the equivalent of an injured natural resource.

[2] See Feather (1994) and Parsons and Hauber (1998) for discussion of the larger question of deciding which alternatives belong in an individual's choice set.

[3] The term "sites" and "alternatives" are used interchangeably in this paper.

[4] A "simple" CL in this paper refers to a one-level conditional logit model that includes no interaction terms to allow for heterogeneous preferences.

[5] The RPL model reported later, with the finer definition of sites and hence larger choice set, took five days to converge on a 450 megahertz PC with 518 meg of available memory.

[6] McFadden (1978) exploits the IIA property of CL to prove that estimating a RUM using random draws of the alternatives provides consistent estimates. Since the RPL does not exhibit the IIA property, the current random-draw techniques are not readily transferable to the RPL.

can be partitioned into elemental sites. Aggregate sites are formed by grouping together elemental sites. Parsons and Needelman (1992) and Feather (1994) define lakes to be elemental sites in their recreational fishing models. This "elemental" site-definition approach is intuitively appealing but not readily transferable to situations where large water bodies or rivers need to be modeled. To model river, Great Lake, bay, or ocean fishing requires defining alternative sites, which is less straightforward than defining an inland lake as a site.[7]

Studies showing the effect of aggregation bias (Parsons and Needelman, 1992 and Feather, 1994), can potentially lead to the conclusion the "smaller is better". While this conclusion is clearly justified in more extreme cases of aggregation, where aggregation includes every lake within a county or a large district, it is less clear what impact lesser levels of aggregation will have. It also is harder to determine whether a site is an aggregate or elemental site when moving towards smaller site definitions. A river, for example, could potentially be divided into extremely small segments. Every 50 miles of a river could be labeled a site, every 10 miles, every mile, every quarter mile, or at the extreme every segment wide enough for an angler to stand. The same argument could be made for the Great Lakes or other large continuous water bodies. Sites that are "too small" will no longer be elemental sites, according to Ben-Akiva's and Lerman's definition, which requires that each trip be associated with only one elemental site. This will not hold if anglers can fish easily at several sites defined for a continuous water body either by walking or boating.

As pointed out by Ben-Akiva and Lerman (1985) the problem of site definition is not unique to recreation demand analysis.

> In other applications, such as the choice of car type, the alternatives are usually grouped by major characteristics of make, model, and vintage, and no distinction is made, for example, among cars of the same make, model, and vintage with different engines.

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[7] Inland lakes also may not be elemental sites. For example, the popular Lake Winnebago in Wisconsin is an inland lake of over 137,000 acres that spans three counties. Parsons and Needelman (1992) divide this lake into four sites. Chains of lakes, where lakes are inter-connected and in close proximity, also challenge this simple definition of elemental sites.

Site definitions affect the extent to which the model violates the IIA property. The IIA property for a simple conditional logit model holds that the relative choice probabilities of any two alternatives are unaffected by the addition or subtraction of another alternative.[8] This property implies that the error term associated with the utility of each alternative must be independent of the error associated with any other alternative. Manski (1973) identifies four sources of error in the portion of utility unobserved by the researcher: unobserved attributes, unobserved taste variations, measurement errors, and instrumental variables. Whenever these unobserved disturbances *systematically* affect alternatives and induce correlation among their error terms, the IIA property will not hold and the model estimates will be affected. Two common ways to reduce violations of IIA are to include interaction terms that control for heterogeneous preferences or to estimate a nested logit model.

The impact of IIA on welfare estimates is an empirical question because IIA violations can cause welfare estimates to be biased upwards, downwards, or not very biased at all. Researchers have shown that parameter and welfare estimates are sensitive to researcher decisions about alternative model specifications that relax the IIA property. Kling and Thomson (1996) find that specifying a nested model produces results significantly different from a conditional logit model. However, Train (1998) finds that the welfare estimates from a random parameters logit model are not significantly different from that of a conditional logit model. While these results are specific to the data used, the conclusion is that the handling of IIA can influence welfare measures.

The link between the IIA property and site definition arises from the well-know "red bus/blue bus" paradox. This paradox illustrates the inability of the simple logit model to handle a choice problem containing alternatives that are identical or nearly identical (Ben-Akiva and Lerman (1985)). Similar alternatives are likely to have similar unobserved attributes, unobserved taste variations, measurement errors in the attributes, and involve similar instrumental variables. This can produce strong correlation among the error terms associated with these alternatives and result in a strong violation of the IIA property.

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[8] See Ben-Akiva and Lerman (1985) for a complete discussion.

In this application, the effects of dividing a river segment into multiple sites are explored. In this case, one would imagine that the smaller segments are likely to have error terms correlated with each other because they share the same unobserved characteristics and/or are grouped together by individuals, which links them if these preferences are not observed. The working hypothesis is that smaller sites will result in more serious violations of the IIA property.

Section 2 considers RPL and CL model specifications, Section 3 discusses the data, Section 4 the model results, Section 5 the welfare implications, and Section 6 the conclusions.

2. MODEL SPECIFICATION

The RPL and the CL models differ in their treatment of site-attribute parameters. The simple CL model assumes that individuals choose the site that provides maximum utility and assumes a linear-in-parameters utility function

$$U_{nst} = \beta' X_{ns} + \varepsilon_{ns} \qquad \textbf{(2.1)}$$

where ε_{ns} has an i.i.d. extreme-value distribution and sites are indexed $\{1,\ldots,s,\ldots,S\}$, individuals $\{1,\ldots,n,\ldots,N\}$, and trips $\{1,\ldots,t,\ldots,T\}$.[9] The utility function underlying the RPL model looks quite similar to equation 2.1, except that the RPL specification allows β to vary by individual. Hence, the utility function underlying the RPL specification can be written

$$U_{nst} = \beta_n' X_{ns} + \varepsilon_{ns} \qquad \textbf{(2.2)}$$

Further, the RPL specifies a distribution for β_n, which in general notation can be written $f(\beta|\theta^*)$. The parameters θ^* characterize the nature of the distribution.

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[9] For ease of exposition it is assumed that the site characteristics (X) do not vary by trip, only by individual and site. In the application presented, trip cost is the only site characteristic that varies by individual, the remainder vary by site only.

One could rewrite (2.2) to separate the random component of the parameter distribution

$$U_{nst} = b' X_{ns} + \eta_n' X_{ns} + \varepsilon_{ns} \tag{2.3}$$

where b represents the non-stochastic mean of the distribution and η_n the random deviation from that mean. In equation (2.3), the error term, which corresponds to the two rightmost terms in the equation, contains X_{ns}. The interaction of the site attributes (X_{ns}) with the random error η_n allows correlation in the error terms among alternatives. This in turn ensures that the model does not require the IIA property associated with the simple CL. Further, because β is indexed by n and not by t, it induces correlation across trips taken by the same individual. Thus, the RPL model treats the data as a panel data set. The assumptions of homogeneous preferences and independent trip decisions by the same individual inherent in a simple logit model can be relaxed with an RPL model.[10]

The RPL model estimated in this paper assumes that all of the parameters are random and have independent normal distributions.[11] In other words, the RPL allows for heterogeneous preferences over each of the site attributes. This provides a flexible substitution pattern across sites and does not impose the IIA property.[12]

The simple CL and the RPL represent two extreme approaches to handling the IIA property. The simple CL model exhibits the most restrictive IIA property whereas

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[10] For more discussion of the underpinnings of the RPL model in a recreation demand context, including simulation techniques and specification issues, see Train (1998).

[11] Other distributions with either bounded or unbounded support could have been selected. Further, the RPL model does not require that all parameters share the same distribution. The normal distribution was selected for all parameters for simplicity, because it allows for both positive and negative reactions to attributes, and because it is a well-known and commonly used distribution to explain economic phenomena. Revelt and Train (1997) discuss in more detail some guidelines for model specification and distribution selection, illustrate the use of both the normal and log-normal distribution, and discuss the possibility of using a distribution with bounded support. Further, Train at the 1999 W-133 meetings suggested that perhaps a bounded support is a better choice because it gives the researcher the ability to prevent individuals from having counter-intuitive preferences.

[12] McFadden and Train (1998) show that under mild regularity conditions, any discrete-choice random utility model with any pattern of substitution and correlation among the error terms can be reproduced by a RPL model with an arbitrarily close degree of accuracy. This implies that the RPL can be made to mimic a nested logit or any other specification designed to handle IIA.

the RPL model allows for a very flexible pattern of substitution. This allows for a test of the sensitivity of models to site definition and its subsequent IIA implications. If a difference were found, future work could examine how more "intermediate" models, such as a nested model, perform in the same experiment.

3. DATA

The data include information on Montana fishing trips taken by Montana anglers during the period from July 1992 through August 1993. Respondents were selected through a random-digit-dial telephone solicitation and asked to return bi-monthly diaries detailing all of their fishing trips.[13] This analysis employs a subset of the data that includes only trips to rivers and single-day trips.

River sites are selected for three reasons. First, substantially better data on site attributes is available for the river sites compared to lake sites. Second, by selecting river sites we have isolated the problem of defining sites over continuous water bodies. Third, by limiting the model to a particular type of site some of the more obvious IIA issues are circumvented. Nested models have been estimated where water body type, fishing mode, or geographic area defines the nest.[14] By selecting only river sites, which attract shore anglers almost exclusively, some sources of potential IIA violations are avoided. The remaining likely cause of IIA violations is spatial. In other words, sites within a certain geographic area are likely to have correlated error terms. This allows a sharper focus on the research question of interest, which is the effect of the size of the defined site, an inherently spatial issue. Single-day trips are included to avoid complicating issues associated with multiple-day / multiple-purpose trips.

Under the less aggregate site specification, 182 unique fishing sites are identified. These sites are defined as the smallest stream segments identified in the Montana Rivers Information System (MRIS), which provides the important site-attribute data. The model contains 750 river trips taken by 199 anglers. With the more

[13] For more detailed information on the data please see Desvousges and Waters (1995).

[14] See Kling and Thomson (1996), Desvousges and Water (1995), Morey et al (1993), Morey et al (1991), Bockstael et al (1989) for examples of alternative nesting structures.

aggregate site specification, 53 unique sites are identified. These aggregate sites are defined by combining stream segments based on natural geography and the natural clustering of trips observed in the angler survey. The model contains 810 river trips taken by 210 anglers.[15] The average length of a less aggregate site is approximately 17 miles, whereas the average length of an aggregate site is approximately 57 miles.

Table 1 shows the variables included in the aggregate and less aggregate models and their source.

Table 1.
Variable Definitions

Variable	Less Aggregate Model	Aggregate Model
BIOMASM	Biomass measured as 100 pounds per 1,000 feet of river	Average biomass for river segments within aggregate site
AESMDUM1	Dummy variable for river segments given the highest aesthetics rating	Dummy variable for any river segments given the highest aesthetics rating within the aggregate site
LOGLNGTH	Natural log of the length of the river within the site	Log of the size of the site measured in USGS blocks (LOGSIZE) *
SRAMILE	The number of state recreation areas per mile of river	Number of State Recreation Areas per USGS block within the site (SRABLK)
MAJOR	Dummy variable for sites identified as major in the Angler's Guide to Montana	Dummy Variable for any segment within aggregate site identified as major in the Angler's Guide to Montana
RES_SPEC	Number of restricted species	Number of restricted species
CGMAPBLK	N/A	Number of campgrounds per USBS block in the site
TRIPCOST	Gasoline costs, maintenance costs, plus the opportunity cost of time (1/3 wage rate) to the town nearest the center of the site	Gasoline costs, maintenance costs, plus the opportunity cost of time (1/3 wage rate) to the town nearest the center of the site

* When different from the less aggregate model, the variable name in the aggregate data set is given in parentheses.

[15] The difference in the number of trips is believed to result from the added difficulty of assigning individual trips to smaller sites. Some respondents may not have provided enough information to be assigned to the less aggregate sites, but did provide enough information to be assigned to the aggregate sites.

4. MODEL RESULTS

Both RPL and CL models are estimated for the less aggregate and more aggregate sites. The RPL model assumes that all of the parameters are random and have independent normal distributions. The simulated log-likelihood is estimated using 500 draws from the parameter distribution associated with each variable.[16] This number of draws seems sufficient to ensure negligible estimation bias from too few draws.

Table 2 shows the estimation results from these alternative specifications. The log-liklihoods clearly support the RPL specification over the CL specification under both site-definition strategies. This suggests that there is important heterogeneity among preferences captured in the RPL model. With both site definitions, the CL model failed the Small and Hsiao (1982) test for IIA with greater than 99 percent confidence.[17]

The estimated signs of coefficients are stable across the RPL and CL models. Further, the biomass and aesthetic variables have the expected positive sign and the restricted species and trip cost variable have the expected negative signs in all models.[18] Comparing the mean RPL parameters with the CL parameters does not reveal any systematic differences in the estimates, however the significance of the parameter estimates are uniformly higher in the CL specification compared to the corresponding mean estimates in the RPL specification except in two cases.[19] Thus while these models differ substantially in their treatment of IIA the effects on the parameter estimates appear modest.

Turning now to the RPL models only, the standard deviation parameters are nearly all significant, the exception being the standard deviation associated with

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[16] Revelt and Train (1997) also use 500 draws and Train (1998) uses 1,000 draws.

[17] For a discussion of the test see Ben-Akiva and Lerman (1985). The test was applied using a subset of the sites that are considered major by the Angler's Guide to Montana. For the aggregate site model the test statistic was 75.6 and for the less aggregate model, 60.0.

[18] Given potential differences in tastes for seclusion and size of alternative, there was no expected sign for the major fishing site and log of size variables.

[19] The logsize variable in the less aggregate model is more significant in the RPL specification and the major variable in the aggregate model is more significant in the RPL specification.

biomass in both RPL models and the standard deviation of the log of size term in the aggregate model. The strong significance of the standard deviation terms in general, supports the hypothesis that preferences are in fact heterogeneous among anglers. Another interpretation is that these site attributes are measured with error. The RPL model cannot distinguish between heterogeneous preferences and measurement error, which are observationally equivalent. The hypothesis of measurement error may be supported by the very large standard deviation associated with the aesthetics variable and the insignificance of its mean estimate in both the aggregate and less aggregate models. Assuming a mean of zero, which cannot be statistically rejected, these parameter estimates imply that 50 percent of individuals find aesthetics to be an undesirable attribute. This seems contrary to intuition and suggests possible measurement problems with the aesthetics variable.[20]

The insignificance of the standard deviation of biomass implies that there is not much heterogeneity of preferences among anglers for fish catch. This is a surprising result given that biomass is intended to serve as a rough proxy for expected catch because true expected catch is unobservable. The insignificant sign on the estimated standard deviation implies that biomass may serve as a good proxy for expected catch. In contrast, the standard deviation of trip cost is significant in both the aggregate and less aggregate models. In this case, it is difficult to determine whether this result is driven by measurement error or differences in individual's tastes for travel. Given the simple assumptions built into the travel cost variable about the opportunity cost of time, speed traveled, and standard cost per mile one would expect considerable measurement error. However, the fact that more than 90 percent of individuals consider travel cost to be a negative attribute is encouraging.

[20] Of course the counter-argument that "beauty is in the eye of the beholder" cannot be rejected. However, these parameter estimates imply so much disagreement among individuals as to make this counter-argument unlikely to reflect the entire story.

Table 2.
Estimation Results for CL and RPL with Alternative Site Definitions
(Asymptotic t-ratios in parentheses)

	Less Aggregate model (182 sites)		Aggregate model (53 sites)	
	RPL	**CL**	**RPL**	**CL**
BIOMASM	0.014 (1.9)	0.192 (8.0)	0.304 (2.8)	0.266 (4.0)
SD(BIOMASM)	0.158 (1.5)	-	0.256 (1.6)	-
AESMDUM1	0.448 (1.6)	0.656 (6.4)	0.320 (1.9)	0.661 (6.0)
SD(AESMDUM1)	2.336 (5.4)	-	0.845 (3.7)	-
LOGSIZE	0.439 (3.4)	0.160 (2.3)	0.759 (4.3)	0.528 (5.1)
SD(LOGSIZE)	1.037 (8.4)	-	0.197 (0.4)	-
MAJOR	0.564 (2.2)	0.575 (5.3)	1.045 (3.6)	0.329 (1.9)
SD(MAJOR)	1.906 (5.9)	-	0.920 (2.2)	-
RES_SPEC	-0.285 (-1.7)	-0.346 (-5.1)	-0.831 (-4.3)	-0.484 (-6.9)
SD(RES_SPEC)	0.649 (2.9)	-	0.606 (3.7)	-
TRIPCOST	-0.133 (-21.7)	-0.090 (-32.2)	-0.121 (-15.4)	-0.097 (-30.8)
SD(TRIPCOST)	0.058 (11.6)	-	0.042 (5.6)	-
LOG-LIKELIHOOD	-1845	-2272	-1240	-1533
LIKELIHOOD RATIO INDEX	0.527	0.418	0.614	0.523

5. WELFARE RESULTS

This section compares the welfare impact of potential improvements across four alternative models. Two potential improvement programs are considered. The first increases the biomass by 100 pounds per mile at all sites. The average biomass at the aggregate sites is approximately 103 and at the less aggregate sites, approximately 83 pounds. The second program doubles biomass at all sites. The standard deviation of biomass is 110 at the aggregate sites and 155 at the less aggregate sites. These two programs do not differ much for the "average site", but may have very different affects overall because of the wide dispersion of the biomass among sites. In other words, a site with below average biomass will benefit more from the first program whereas a site with above average biomass will benefit more from the second program. The average value of an aggregate site is also compared across models. Valuing an aggregate site allows direct comparisons between the results from models estimated with the aggregate and less aggregate sites. For the less aggregate models, the group of sites contained within the aggregate site is valued. This estimate allows a more general comparison among the models that is not as dependent on the estimated biomass parameter as the two improvement programs.

The calculation of the welfare change for each individual, in terms of the compensating or equivalent variation, follows equation 5.1. In equation 5.1, X1 represents the individual's attribute matrix in the improved state, X0 represents the individual's attribute matrix in the original state, and the summations are over the alternatives in an individual's choice set.

$$CV_n = EV_n = -1/\beta_{TC} * \{ \ln(\sum e^{X1\beta}) - \ln(\sum e^{X0\beta}) \} \quad (5.1)$$

The calculation of welfare within the RPL model requires simulation over β to estimate equation (5.1) since the RPL model does not estimate β but rather a distribution of β. Estimates are obtained by randomly drawing 10,000 parameters from the normal distribution associated with each of the explanatory variables. Because the parameter on travel cost appears in the denominator of the welfare expression, care must be taken to avoid parameter draws near zero. Given the distribution of the travel cost parameter in both RPL models, truncation at plus and minus one standard deviation is chosen.[21] Welfare estimates computed using the mean of the travel cost distribution rather than the truncated distribution did not differ substantially from those reported.[22]

Table 3 shows the estimated welfare implications of the two improvement programs. One way to interpret these results is to compare the two less aggregate site models and then the two aggregate site models. With the less aggregate site definition, the estimated value of the two improvement programs differs substantially between the CL and RPL. The RPL estimates a gain of $0.90 per trip for program 1 whereas the CL estimates a gain of $2.13 per trip. The estimated gains for program 2 also differ by more than a factor of two between the RPL and CL specification, $2.23 and $4.57 per trip respectively. These results indicate that with the less aggregate site definition there are substantial differences between the RPL and CL welfare estimates. The models estimated with the aggregate sites tell a different story. Here, the RPL model predicts a slightly lower benefit from program 1, $2.60 versus $2.75 for the CL, but the RPL model predicts a slightly higher benefit from program 2, $4.47 versus $4.07 for the CL. The results with the aggregate site definition indicate no systematic difference between the RPL and CL models in terms of their welfare predictions.

The result that RPL and CL models perform similarly, at least in terms of predicted welfare changes, with aggregate sites but quite differently with less aggregate sites supports the original hypothesis that IIA violations may be more serious when sites are defined as smaller areas. Models that differ in their treatment of the IIA issues seem to differ in their reaction to site definition.

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[21] This is achieved by drawing 20,000 β_n 's, removing those β_n 's with travel cost parameters outside of one standard deviation, and then randomly keeping 10,000 of those remaining.

[22] For the less aggregate site RPL model the following welfare estimates are obtained using the mean of travel cost: $0.89 for program 1, $2.12 for program 2, and 0.28 is the average value of a site. For the aggregate site RPL model the following welfare estimates are obtained using the mean of travel cost: $2.60 for program 1, $4.43 for program 2, and 0.24 is the average value of an aggregate site. These results are very similar to those reported in Table 3.

Table 3.
Estimated Welfare Changes per Trip

	PROGRAM #1: Increase Biomass by 100 lbs. per mile at all sites	**PROGRAM #2:** Double Biomass at all sites	**AVERAGE TOTAL USE-VALUE OF AN AGGREGATE SITE**
LESS AGGREGATE MODELS			
RPL	0.90	2.23	0.27
CL	2.13	4.57	0.24
AGGREGATE MODELS			
RPL	2.60	4.47	0.24
CL	2.75	4.07	0.25

Alternatively, one could read the table by comparing the two CL models with each other and the two RPL models with each other. The RPL model appears to be sensitive to the two alternative definitions of site. Under program 1, the RPL model predicts a smaller welfare gain of $0.90 with the less aggregate site definition compared to $2.60 with the aggregate definition. Similarly, under program 2, the RPL model predicts a smaller welfare gain of $2.23 with the less aggregate site definition compared to $4.47 with the aggregate definition. The RPL model detects a difference of more than a factor of two, in terms predicted welfare, between the alternative site definitions. In contrast to RPL, the CL model does not appear to be sensitive to the two alternative definitions of site. Under program 1, the CL model predicts a smaller welfare gain of $2.13 with the less aggregate site definition compared to $2.75 with the aggregate definition. However, under program 2, the CL model predicts a larger welfare gain of $4.57 with the less aggregate site definition compared to $4.07 with the aggregate definition. For the CL, the welfare estimates across site specifications are quite close for both programs and do not appear to differ in a systematic manner. The CL model does not appear to detect a difference, in terms predicted welfare, between the alternative site definitions. This result indicates that violations of IIA may mask welfare differences resulting from alternative site definitions. This could lead to the faulty conclusion that the size of the defined sites is not important to the calculation of welfare gains.

The third column of Table 3 shows the estimated average value of an aggregate site. This value does not seem to vary substantially across models. However, as noted for the policy improvement scenarios, the difference between the RPL and CL with the less aggregate sites is greater than the difference between these two models with the more aggregate sites. Also consistent with the above findings is that there is a greater difference between the RPL models across site definitions than the CL models.

Some caution when interpreting the RPL model results is warranted. The desirable properties of the RPL are accompanied by the challenge of deciding which parameters should be allowed to vary and which distributions should be used. The effects of model specification issues, especially the choice of parameter distributions, has not been fully discussed in the literature. Additional research on specification decisions and their impact on parameter and welfare estimates is needed before strong conclusions can be made.

6. CONCLUSIONS

This paper investigated the relationship between site definition and the calculated welfare effects, comparing a Random Parameters Logit (RPL) specification with the Conditional Logit (CL) specification, with an empirical application to a Montana angling data set. The more aggregate sites correspond to approximately 57 miles of river and the less aggregate sites to approximately 17 miles of river.

In this application, the RPL models produce substantially different welfare estimates when aggregate sites are used compared to the less aggregate sites, whereas the results from the CL model are nearly identical for the two alternative site definitions. Thus, while the RPL model detects a difference between the aggregate and less aggregate site definitions, the CL does not seem to detect a difference. The failure of the CL model to detect a difference between the aggregate and less aggregate site definitions indicates that violations of IIA may mask welfare differences resulting from alternative site definitions. This could lead to the faulty conclusion that the size of the defined sites is not important to the calculation of welfare gains.

The result that RPL and CL models perform similarly, at least in terms of predicted welfare changes, with aggregate sites defined but quite differently with less aggregate sites defined supports the original hypothesis that IIA violations may be more serious when sites are defined as smaller areas. Dividing a larger site into several smaller sites may worsen IIA violations because the error components of these smaller sites are likely to be correlated. Random errors related to heterogeneous preferences, unobserved variables, and proxy variables are likely to effect the smaller sites in similar ways. Thus defining smaller sites may increase the severity of IIA violations. This can explain why a substantial difference between RPL and CL is only noted in the less aggregate site specification where IIA is more likely to be a serious concern and the added flexibility of the RPL more necessary. These results suggest that when smaller sites are identified more attention to IIA is warranted and a more flexible model specification such as the RPL may be necessary.

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WHAT DO WE KNOW ABOUT DON'T KNOWS?
ABSTENTION, AMBIVALENCE, DECEPTION

BY

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What do We Know about Don't Know Responses:
Abstention, Ambivalence and Deception[1]

Often, respondents in contingent valuation surveys do not provide definitive answers to all questions. Such *missing* or *don't know* responses present a theoretic and practical challenge. Don't know responses to the valuation (bid) questions create a particularly thorny conundrum when an estimate of total value is required. In disciplinary analysis, one can focus on characteristics of the definitive answers (yes or no for dichotomous choice format; zero or a bid value in open ended bid formats). Certainly, economists are supplied with ample puzzles even where responses are definite. However, in a policy situation, the applied analyst does not have the option of ignoring the don't know response. Some value must be imputed willy-nilly.

Consider the two most obvious treatments of the missing values: omitting the observation with the missing value or assigning the response a value of zero. Omitting an observation with a missing value is a standard approach. The consequence of omitting the observation is that the respondent is implicitly assigned the value of the average response when the sample results are generalized back up to the population. A simple example using an open-ended valuation question will illustrate. Suppose the average response to the willingness-to-pay (bid) question in a survey is $50, counting zero and non-zero responses, but with don't know responses omitted. Suppose the relevant population contains one million people. Using the average response gives a total value of $50 million for the valuation object. But using the $50 average value in the total value estimate implicitly assumes that the valuation of the don't know group is the same as that of the "typical" or average respondent in the survey.

Another straightforward approach to treating the missing value is to argue that, since the value of the don't know respondent is unknown, no positive value can be inferred; and therefore a default value of zero should be assigned. Now suppose that 20% of respondents answer don't know to the bid question. If these respondents are assigned a zero value, then the weighted sample average value becomes $40 (.8*50 + .2*0). When the revised value is generalized to

[1]Without attaching blame for any errors, the authors wish to acknowledge stimulating and insightful discussions with Ron Mittelhammer and many of the W-133 family in addition to those formally acknowledged by inclusion in the cited literature.

the population the total value becomes $40 million. Treatment of don't know responses has created a swing of $10 million in the estimate of total value. In the case of a dichotomous choice or closed ended bid question the mechanics are a little different, but the basic pattern holds[2].

Therefore, while much disciplinary research in contingent valuation is appropriately focused on methods for eliciting a valid bid value, the approach used to treat the don't know responses to the bid question can be at least as important for practical and policy research. The major purpose of this paper is to explore the possible meanings of the don't know response and suggest a systematic approach to modeling the responses which emphasizes the heterogeneity of the don't know responses. We will also briefly review some of the literature on treating missing/don't know responses to the bid question, analyze some empirical evidence regarding the nature of don't know responses in a case study of the contingent valuation of the off-site benefits of reduced agricultural dust emissions, and, finally, offer some suggestions for future contingent valuation studies.

Meaning of the don't know response

Respondents in a contingent valuation survey might fail to give definitive responses to the bid question for at least four reasons: insufficient consideration/information, rejection of response categories, uncertain (ambivalent) preferences, and deception. (See, e.g., Arrow, et al, 1993; Alberini and Champ, 1998; and Wang, 1997, for other discussions.) In the first case the respondent might have insufficient information about the valuation topic or be unwilling or unable to invest sufficient time in the decision process to determine a definitive value. The respondent's preference set is incomplete.

In the second case the respondent rejects the response space provided by the survey instrument. For instance, the respondent may have negative values for the valuation topic and

[2]Recently the willingness-to-pay elicitation method in which the respondent is offered a bid amount and then asked whether or not he will pay that amount in a voting/referendum framework has become known as the dichotomous choice method. For reasons that will become obvious the term closed ended will be used in this paper.

so cannot legitimately respond with an amount he or she is willing to pay. While some respondents in this situation may choose a zero value in an open-ended survey or a NO in a dichotomous choice referendum, others may not find these options satisfactory. Another reason for rejecting the response options might be that the respondent objects to monetarization of his or her values. Essentially the don't know response may be another form of "protest" response. (Zero and NO responses are sometimes examined for "protest" responses which are then omitted from the data.)

In the third case the respondent is truly uncertain or ambivalent about his or her true valuation of the valuation topic. Indifference surfaces are thick or fuzzy. This is an area of current development in instrument design and analysis.

A final case of indefinite or missing response might occur when the respondent feels that his or her true response would not be socially acceptable or would not be strategic. For instance, a respondent might feel that a "don't know" response is more appropriate than zero for a valuation object that has positive social value. Thus, a respondent might feel embarrassed to respond with a low or zero value in a survey concerning a policy that affects public health although the respondent truly has little personal value for the proposed policy.

Do motives matter?

In standard economic demand analysis, economists do not worry about the motives behind consumer decisions; their analysis is based on behavior. All the analyst needs to be concerned about is whether the consumers' underlying preferences satisfy basic rationality requirements. However, in non-market valuation the analyst has the task of reconstructing the respondents' values from the stated information. Generally, the values inferred from the survey information will differ depending on how preferences are modeled. In this case inquiring into motivation may assist in constructing valid survey instruments, in modeling preferences, and in inferring appropriate values.

For instance, if don't know responses are primarily a result of insufficient information or insufficient decision effort, then increased emphasis should be placed on the construction of the survey instrument to provide information and context to assist, but not bias, the decision process. In modeling and analyzing the data, the analyst would look for information biases and any selection bias that may arise if the values of those whose preferences are not clearly formed tend to be lower than those who do respond. In contrast, if don't know responses arise primarily because of ambivalence of preferences, then the survey instrument should either try to minimize the source of this ambivalence or explicitly provide for indifference/ambivalence responses.

In summary, more information about the nature of the don't know responses would facilitate the construction of valid contingent valuation survey instruments and the development of valid methods to infer values.

Modeling the don't know response process

For purposes of modeling, the four cases cited above can be associated with three response patterns: abstention, ambivalence, and deception. Each of these three response patterns requires a different modeling approach, and perhaps a different instrument design.

Abstention might occur in either of two cases: 1)when respondents have insufficient information to answer or are unable or unwilling to formulate a definitive response; and 2)when respondents find the question inappropriately formulated or the response categories don't fit their preferences. Abstention implies a two stage or nested decision process. The respondent decides whether or not he or she will formulate a value and then calculates the value. Therefore, any stated value is conditional on the decision to respond. This means that abstention has two potential effects: a potential bias in the observed responses and a missing value problem. One must determine if one has the correct response for those who do provide a definite value, and one must assign a value to those who do not provide a definite value.

Abstention is essentially a problem of non-response, a problem well-known in the survey literature. Specifically, these abstention-don't know responses are an instance of item non-response (see, e.g., Mitchell and Carson, 1989)[3]. As indicated in the introduction, non-response is often treated by simply omitting the observation and calculating the value for the population based on the remaining sample. However, omitting these observations creates the risk of bias: either (item) non-response bias or selection bias (e.g., Mitchell and Carson, 1989). A non-response bias occurs if some identifiable groups are more likely to respond to the question than other groups. The non-response circumstance can become a sample selection problem if those who do not respond differ systematically from those in comparable groups who do respond. A sample selection bias presents a more serious problem than the non-response bias. One does not generally know how the answers of non-respondents would differ from those who actually did respond.

Simply omitting the missing values does not capture the nested nature of the decision process. A better approach is to re-weight the values of the sample respondents according to their incidence in the population. However, there may be unobserved characteristics that distinguish the respondents from the non-respondents. An explicit two stage estimation process in which the decision to respond and the value response conditional on the response would therefore be better still. Cameron, Shaw, and Ragland (1999) have recently applied this approach for the case of unit non-response (failure to return the whole questionnaire) for a mail travel cost valuation study.

After one corrects the value of the respondents with definite values, one must still deal with the values of those who respond don't know in a contingent valuation study. In the Cameron, Shaw and Ragland study, if respondents do not take trips, they do not have behaviorally revealed economic value. In contrast, in a contingent valuation study, respondents who abstain may have economic value that they are not revealing for one reason or another. The situation is somewhat analogous to missing income data; most respondents who refuse to provide income data have an income.

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[3] The analyst faces a similar problem of potential bias when the survey is not answered at all, or when the whole observation must be dropped because irregularities in responses. See, for instance, Mitchell and Carson, or Cameron et al, for discussion of "unit non-response."

A standard, "missing data," remedy for don't know responses is to impute some value to the don't know (DK) respondent based on information about the respondent. Thus, suppose respondents with lower incomes are more likely to answer "don't know" (or some other responses coded as "missing") than are other groups. Rather then omitting the observation and implicitly assuming the simple average value, one can estimate the mean responses by income -- and other socio-demographic characteristics -- for those who do respond and use these estimates to reweight. Equivalently, one can estimate a model of bid values statistically, and use the model to predict the values for the missing (don't know) data, using the revised sample totals to estimate total value.

The abstention model of don't know responses suggests that we examine motives for abstaining in order to predict the don't know values. In the two stage decision process we are predicting the don't know response as much as we are predicting the definite answer response. The value of the don't know respondents is conditional on their don't know response as much as the value of the definite value respondents is conditional on their decision not to abstain. For instance, the non-response literature suggests that respondents for whom the survey topic has low saliency will be less likely to respond to the survey. An obvious analogy is that don't know responses to value questions are likely to come from those with lower interest in the survey topic--and therefore with lower value for the value object. In this case simply omitting the don't know respondents' observations would lead to an overestimation of the program's value. It may be that less informed respondents are also likely to have lower education and income. In this case one can adjust the results provided that one has the socio-economic data. Of course the difficulty here is that low saliency may or may not be correlated with observable characteristics.

In summary then, the abstention model suggests that one first model the decision of whether to abstain (respond don't know or missing) or give a definite value. One then estimates a conditional value for those who give a definitive value and a conditional value of those who abstain. The problem is that one has little or no information with which to estimate the conditional value of the non-respondents. In essence, the conditional estimate of the willingness to pay for the abstention/don't know non-respondents is conditional on the assumption made by the analyst - whether that is an assigned zero value or some estimated value.

Supposing that the respondent is willing and able to respond, he or she still may be ***ambivalent*** about his or her value. Under standard micro theory assumptions, individuals know their own preference and should be able to make a definitive offer. Recently however, value uncertainty has been investigated in some contingent valuation studies. Value uncertainty may be a result uncertainty in the preference structure itself or it may enter somewhere else in the response decision process. For instance, it may be that the respondent is not really certain of the consequences of the program. Often the two cases will be indistinguishable for purposes of modeling[4].

The idea that respondents may have uncertain responses due to ambivalent preferences has been developed by a number of authors including Opulach and Segerson (1989), Ready et al (1995), Wang (1997), and Li and Mattson (1995). Loomis and Ekstrand (1998) have a short survey of the recent empirical literature. Much of this literature has been focused on the uncertainty of stated responses. In one line of research, analysts attempt to use post-bid question expressions of the degree of confidence in the bid to refine the estimates of value (e.g., Li and Mattson, 1995; Loomis and Ekstrand, 1998). Another line of research explicitly incorporates uncertainty in the response categories of the survey instrument. Providing an explicit option for an uncertain/unsure response was one of the NOAA panel recommendations (Arrow, et al). For instance, Ready et al, provide six options ranging from definitely yes to definitely no. Recently a new valuation method has been developed which explicitly incorporates uncertainty in the response set (e.g., Welsh and Poe, Cameron et al). The multiple bound method comprises a matrix of bid levels by degree of certainty (for instance, values of $10, 20, etc and certainty levels: definitely yes, probably yes, not sure, probably no, definitely no).

A third line of research concerns the idea that the item non-response or don't know response to the bid valuation question is itself an expression of ambivalence (Wang, 1997; Alberini and Champ, 1998). Dichotomous choice becomes tri-chotomous choice with YES, DON'T KNOW/NOT SURE, and NO, being the possible response categories - even where the don't

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[4]However, in a theory of adaptive utility, preference uncertainty might be reduced with increased experience (Cyert and DeGroot, 1987).

know response is not explicitly provided[5]. Wang uses ordered probit and related methods to estimate a willingness-to-pay values based on this model.

In summary, the ambivalent/don't know response is generated by a different kind of decision process than the abstention/don't know process. For the ambivalent/don't know preference, the don't know response represents one of an ordered set of possible responses. The respondent is choosing one from among several alternative responses rather than abstaining from answering. In the case of the closed ended elicitation format, the choices are yes, no, or don't know. In the case of the open ended bid format, the choices are zero, a positive value, or don't know. In the closed-end bid question the respondent answers don't know if his or her value is close to the proffered amount. In the open-ended case the respondent answers don't know if the variance is so great it cannot be resolved into a discrete amount (including zero). One implication is that the don't know/ambivalence respondent is different for the open-ended and the closed-ended surveys.

The third type of don't know response is motivated by ***deception*** or strategy. For instance, it may be that the respondents true value is at odds with what is perceived to be the "proper value." Or perhaps, as Opaluch and Segerson (1989) suggest, the respondents personal values are at odds with his or her moral values; the respondent is not lying, but the response does not truly reveal personal willingness to pay. For these cases, the true value may be zero for the open ended question and NO for the dichotomous choice format, but the socially accepted or morally correct value is non-zero or YES respectively. If such deception can be verified, then assigning the zero or NO response is the correct remedy. One obvious procedure is to assess whether or not the don't knows resemble the zero or NO respondents. For instance, Alberini and Champ (1998) found that don't know respondents in a closed-ended contingent valuation study resembled the no respondents. Essentially, Alberini and Champ tested the ambivalence hypothesis of Wang against the deception hypothesis for explaining don't know responses.

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[5]Ready et al label their multiple response categories with explicit degree of confidence, the "politomous choice" (PC) format.

One problem is that no test to distinguish the deceptive from the non-deceptive can be definitive. It may be that one finds a difference between the don't knows and the actual zero or no respondents but the difference is due to the fact that the prevaricators are, in fact different - but they are indeed prevaricators. It may be that one finds no difference between the don't knows and the actual zeros or NO respondents, but that the similarity on the particular measures used is coincidental and would be discovered if other, currently unobserved characteristics were measured.

More recently Blamey, Bennett and Morrison (1999) have suggested that the "yea-saying" tendency many analysts believe are characteristic of closed-ended responses can be countered by offering an appropriate set of response categories. Yea-saying is a kind of deception in which respondents to closed-ended bid questions are assumed to say yes when offered bids higher than their true willingness to pay. BBM offer respondents the opportunity to say they support a socially valued policy with or without financially supporting it. BBMs results are particularly interesting because they compare three models: one which explicitly accounts for uncertainty/ambivalence (Ready et al, 1995), the standard dichotomous choice (closed ended), and their proposed multiple choice bid elicitation format. BBM find that accounting for ambivalence does not change value estimates in the same way that accounting for deceptive responses (yea-saying) does. It appears that the BBM approach of incorporating multiple response categories may address several three don't know response types: inappropriate response categories, ambivalence, and deception/strategy. On the other hand, it strays from the a primary raison d'etre of the dichotomus choice format: an attempt to replicate a voter referendum.

The above discussion suggests that the reasons that respondents fail to answer the willingness to pay question, or give a don't know response are heterogeneous. The implication is that one is really dealing with more than one phenomenon and that modeling approaches based on addressing just one of the possible generators of don't know values may be incomplete. A single strategy for dealing with don't knows may still leave some aspects of the problem mis-specified.

In the remainder of this paper we will present some preliminary analysis of don't know heterogeneity based on an empirical contingent valuation study. The topic of this study was the off-site benefits of agricultural dust reduction.

The agricultural dust study

The Columbia Plateau covering much of Eastern Washington and parts of neighboring states is characterized by high levels of windblown agricultural dust. This is both a soil conservation problem and an air quality problem. The Northwest Columbia Plateau Wind Erosion PM-10 Project is a comprehensive, multi-disciplinary research and education program whose purpose is to study wind erosion from area cropland, analyze its impacts on air quality, and discover and implement strategies for reducing the wind erosion. A contingent valuation study was initiated to measure the off-site benefits of possible improvements in air quality from reductions in wind erosion. (For study details see Scott and Wandschneider, 1997.) It is important to note that local particulate air quality is a public good with direct impact on the area population. Issues of existence/passive use should play a minor role in responses. However, there is opportunity for the influence of altruism and of social approval on the bid revelation process.

The contingent valuation questionnaire was developed in a multistage process: interviews with experts, formal focus groups, medium sized panel response groups[6] (25 participants responding electronically to 90 minutes of questions) and pre-testing of the survey instrument. The survey instrument was administered by a professional survey organization (Social Survey Research Unit at the University of Idaho). The survey used a random telephone survey frame. An advance letter was sent and a lottery based reward was offered to respondents to help increase participation. A total of 1802 interviews were completed with 868 from Spokane and 934 from the tri-cities area of Washington (Richland-Kennewick-Paco). The overall cooperation rate (ratio of completes to completes + partials + refusals) was 74%. The overall cooperation rate (the ratio of completes to all eligible sample numbers) was 59%.

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[6]The panels were conducted by Tell-Back, Inc, of Spokane Washington.

Some general attitudinal and environmental perceptions questions revealed that the survey population was fairly well informed. For instance Spokane residents reasonably viewed motor vehicles as the biggest contributor to air pollution whereas respondent from the tri-cities viewed agricultural dust as the largest contributor. About 9% of the respondents reported that at least one member of their household had asthma and about 20% reported at least one member with some chronic respiratory or heart condition that would place them at risk for health effects from poor air quality.

A two stage split sample valuation format was used. In the first stage all respondents were asked if they would support a program designed to improve air quality by reducing the incidence of agricultural dust during strong wind events. In the second, split sample stage, about one-third of the sample were asked an open-ended format bid question and the remainder were asked a closed ended (dichotomous choice) question. Bids for the dichotomous choice question were based on the distribution of early responses to the open-ended questions. Follow-up questions including follow-up bid questions were asked based on initial responses. For instance, those who could not provide a definite WTP response (zero or a value) were branched into the closed-ended question sequences.

Results, Abstention

The literature suggests that the open ended bid format should generate more decision-process related abstentions than the closed ended format (see, e.g., Mitchell and Carson, 1989). An examination of the don't know respondents to the open ended question in the dust study supports this general finding. Table 1 shows that 60 of the don't know/unsure respondents had responded to the first stage (no valuation) question with an indication of support for the dust reduction program. (Those who answered NO in the first stage question were assumed to possess zero value and were not asked to express a willingness to pay value - except for a small number based on responses to follow up questions .) The table also shows that, of the 108 respondents who initially reported a don't know/unsure response, 54 (exactly half) reported a value when asked a follow-up dichotomous choice question. Therefore, at least some of the don't know respondents have a positive willingness to pay value -- although they are reluctant to express it in the initial open-ended situation. For those who eventually express a value, it is not clear whether their bid value is lower than the initial open-ended respondents as the low saliency model would predict. For comparison, the initial open-ended respondents had an overall average of $54 dollars but an average of $85 excluding the zero values.

Table 1: Open-ended don't know responses v program support			
		Follow-up Valuation	
Program Support	**N**	**N**	**Mean Bid**
For program	60	40	$66
Depends on cost	25	7	$56
Not Sure/no opinion	20	7	$35
No answer	3		
Total	108	54	

The table is also interesting for the negative finding that some people remain in the don't know/missing value category after the follow-up dichotomous choice bid question and other follow-up questions designed, for instance, to identify "true zeros" from don't knows. The residual don't know respondents may remain in the don't know for any of the reasons discussed above although the ambivalence rationale seems least likely. Respondents have had at least two chances to reveal some level of willingness to pay.[7]

We asked follow-up questions which identified, and reclassified some respondents who said they weren't sure but, for instance, gave inability to pay as the reason for not responding. Thus, at least some of the low saliency, don't know responses are revealed by properly phrased follow-up questions to be true zero responses.

Our survey provides some evidence regarding the "answer categories inappropriate" rationale for the abstention-don't know. The first stage of the dust questionnaire was explicitly designed

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[7]Since the open-ended don't know respondents were branched into the dichotomous choice (DC) routine, they also were asked the open-ended follow-up questions asked of the DC branch.

to allow people who opposed the program to express their opposition in a voting framework. We also included a branch allowing a negative valuation for those who opposed a dust reduction program. We received a small number of these negative bids -- too small to reliable use for estimating the costs of the proposed program[8]. In another recent study, Blamey, et al (1998), allowed respondents to choose among degrees of support for a project, with and without financial commitment. The use of multiple response categories with qualitatively different dimensions appears to have potential for responding to some of the abstention-don't know response processes.

Ambivalence and deception

The closed-ended bid elicitation format, including dichotomus choice and a growing number of variations, has been the major venue for investigating ambivalence and deception as a cause of don't know/unsure responses. Alberini and Champ (1998) suggest analyzing data from dichotomus choice studies in two stages. In the first stage the characteristics of the respondents who gave don't know responses are compared to the characteristics of those who gave no responses in order to test whether the don't know is really a separate response. In the second stage the actual willingness to pay estimation is made based on the appropriate treatment of the don't know group. Alberini and Champ use a multinomial logistic model for the first stage test. If the first stage results indicate that don't know responses are not distinct from NO responses, then the don't know responses are reclassified as NO responses. If the don't know and NO response groups are different, then Wang's procedures for trichotomous choice should be used. Alberini and Champ found that, for their sample group, the don't knows were very similar to the NO group, but they suggested that these might not be general results; each circumstance may need to be evaluated separately. Alberini and Champ estimated willingness to pay for their sample under both the don't knows reclassified as NOs and the trichomous Wang method.

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[8] Costs of the program could be directly estimated using forgone agricultural net revenues. Values for opposition to the program by non-farmers might be used to calculate any off farm costs of the dust reduction program. Such values would be primarily existence and altruistic values introducing more problematic valuation questions.

We have not completed our analysis following the Alberini and Champ method, but a "quick and dirty" stage one test using discriminant analysis suggested difficulty in classifying the don't knows as a separate group. Discriminant analysis treats the response categories as simply three groups to be classified by their characteristics. It does not explicitly model the choice process. In future research we will determine if these rough results hold up using a multinominal logistic model and more formal testing. Still the classification approach may be a useful quick screening method to investigate the different response groups.

The results from the dust study show that the discriminant functions for the don't know and the NO response groups were similar in a three way classification, though not for all classification variables. (See table A1.) Interestingly, in a test to see how well the classification scheme worked on the same data that generated it, the discriminant functions assigned no respondents to the don't know group when using the observed prior proportions. A discriminant analysis classifying only don't know and NO responses assigned all responses to the No group when using observed prior proportions. In sum, the evidence from the discriminant analysis is mixed, but provides some support for classifying at least some of the don't knows as NOs.

We have also made some preliminary estimates using a standard 2-way logistic model and the trichotomous ordered probit model following Wang (1995). Appendix table A2 compares logistic estimates of dichotomous choice models with the don't know respondents reclassified to NO responses and the ordered probit trichotomous of YES-DK-NO responses in two specifications: one with only the bid explanatory variable, one with other co-variates - all of which were significant, though statistics are not reported in the table (available from the authors). While tests to formally compare the models have not yet been done, visually, the coefficients are little different between the reclassification-dichotomous models (1 and 3) and the trichotomous, ordered probit models (2 and 4).

A direct examination of the individual responses has something to say also. A notable feature of the don't know responses is that the don't know responses in the dichotomous choice sub-sample are remarkably stable compared to the don't knows in the open-ended sub-sample. Only one of 232 don't know respondent switched to a definite willingness to pay value under an open-ended follow-up payment question - and that for just $2! If the don't knows reflect ambivalence, the ambivalence is very stable.

This combined group of don't knows also includes 53 don't know respondents who branched in from the open-ended sub-sample. (The don't knows from the open-ended sub-sample were branched into the dichotomous choice questions as follow-up questions, and about half responded with a definite response while half remained don't know.) These "triple don't knows" still included 20 who had answered the program screening support question positively and 18 who said it depended on cost. More generally, of the 232 don't know responses to the dichotomous choice question, 112 had supported the dust reduction program in the screening vote and 47 reported it depends on cost. These program supporters are might be expressing either indifference or deception over the payment choice. Still 70 of the total group of 232 and 15 of the 53 "triple don't knows" had given no answer or an unsure response to the program support question. These truly uncertain respondents seem to be unlikely candidates for deception or payment indifference. They seem more likely to be abstentions.

Conclusions

Our analysis of the dust survey data is preliminary, but this preliminary evidence indicates heterogeneity in the don't know/unsure respondents. This analysis supports the generally accepted idea that the open-ended bid format is more likely to induce don't know responses due to incomplete decision processes than the dichotomous choice format. A telling point here is that some of the open-ended format sub-sample don't knows gave bid values in follow-up questioning where virtually none of the dichotomous choice format respondents did. Those who don't respond seem to include people with low saliency and lower than average values.

The analysis also supports the point that don't know responses to the dichotomous choice question should not simply be omitted. While some of these respondents may be truly ambivalent about the support level, some appear to be better described as NO respondents. If these results hold up, they suggest that the Alberini and Champ (1998) results for a small convenience sample are supported, at least in part, in one larger, random sample survey.

These results also render some support for the Opaluch and Segerson (1989) framework but raise a question about how to interpret some don't knows. Suppose a respondent has an internal conflict between his or her moral values and personal values and cannot decide whether or not to pay. Does this imply a real NO value that the individual is reluctant to reveal, or does it imply a real ambivalence which is best interpreted as indifference as Ready et al (1995) and Wang (1995) do. The difference between these interpretations may very easily be related to an unobserved variable so that tests such as those described by Alberini and Champ will not be able to distinguish the two cases.

Overall this analysis suggests that there is work to be done in determining what to do with don't knows. The analysis suggests that some rough idea of the magnitudes of the don't know respondents values can be gained by follow-up questions and additional analysis. The results also suggest that simple solutions won't work: omitting the don't know observations is likely to produce an overestimate of total value and assigning a zero value will produce an underestimate.

What is to be done? Certainly the tests suggested by Alberini and Champ need to be conducted. But these tests alone will not suffice. Further development of bid formats and follow-up questions are likely to be required for additional progress to be made in dealing with the don't know respondents. It is unlikely we will ever totally eliminate the don't know/uncertain respondent group. For those remain don't knows, the best practical solution in policy applications, after making the best possible estimate, is to provide a range of values from assigning a zero value at the lower bound to assigning the best weighted average value at the upper end.

Appendix

Table A1: Linear discriminant functions for three classes of responses to CVM bid question

Variable	YES bid group	Don't know/unsure	NO bid group
Bid	0.01973	0.02293	0.033660
ReClean	1.36915	1.72316	1.68815
Income	1.81787	1.67792	1.63707
ReDeath	1.44393	1.40759	1.69174
Other issues	2.07324	2.05177	1.85987
Male	0.92913	0.35510	0.50782
AgDust	2.16043	2.25880	2.37374

Table A2: Logit models of bid responses: dichotomous & trichotomous ordered probit (Y=0 in all models)

Variable	Model 1	Model 2	Model 3	Model 4
Intercept	0.7253	0.7454	0.4456	0.3652
Intercept 2		1.0486		0.7019
Bid	-0.0115	-0.0118	-0.0127	-0.1300
Jobscale			0.0814	0.0743
AgDust			-0.1790	-0.1792
ReClean			-0.3237	-0.2967
ReCosts			-0.2006	-0.1788
OtherIssues			0.1432	0.1815
Male			0.4618	0.3937
Income			0.1735	0.1706

Variables:
Bid: bid amount offered
Jobscale: jobs v environment attitudinal scale
AgDust: Agricultural dust is major air pollution contributor
ReClean: Cleaner air is important for reducing cleaning costs
ReCosts: Cleaner air is important for reducing medical costs
Male: Respondents' sex
Income: Household income
ReDeath: Reduce risk of death
Other Issues: Other issues more important

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CONTROLLING FOR CORRELATION ACROSS CHOICE OCCASIONS AND SITES IN A REPEATED MIXED LOGIT MODEL OF RECREATION DEMAND*

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Proposed Running Head: Correlation in Repeated Mixed Logit

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Controlling for Correlation Across Choice Occasions and Sites in a Repeated Mixed Logit Model of Recreation Demand

Abstract

The repeated nested logit model (RNL) introduced by Morey, Rowe and Watson [19] provides a utility consistent approach to controlling for the count nature of recreation demand data. However, it requires strong assumptions on cross-site and cross choice occasion correlation patterns. We examine the use of the mixed logit framework (e.g., [17]) in its place to generalize the available correlation patterns. A Monte Carlo experiment is used to illustrate the ability of the repeated mixed logit model (RXL) to capture quite general correlation patterns, and demonstrate its importance with an application to sport fishing in the Wisconsin Great Lakes.

1. INTRODUCTION

The task of modeling recreation demand is complicated by both the count nature of the data and the prevalence of corner solutions (i.e., households typically choose to visit only a subset of the available sites and set the demand for the remaining sites to zero). Over the past few decades, a variety of frameworks have evolved attempting capture these features of demand. For example, the so-called linked model segments the consumer's decision into two components: (a) the discrete choice of site selection for a given trip and (b) the participation decision regarding the number of trips to be taken.[1] Yet, while the linked model is intuitively appealing and relatively easy to estimate, it cannot be derived from an underlying set of preferences (see, e.g. [13], [27]), making the resulting welfare calculations approximations at best. In contrast, the Kuhn-Tucker (KT) model, initially proposed by Hanemann [10] and Wales and Woodland [30], provides a unified utility theoretic framework within which to model both the site selection and participation decision, while controlling for corner solutions.[2] The KT model does not, however, explicitly control for the count nature of recreation demand data and, as yet, has only been estimated using relatively simple functional forms. Of the competing frameworks currently available in the literature, only the repeated nested logit (RNL) model (introduced by Morey, *et al.* [19]) integrates the site selection and participation decisions in a utility theoretic framework *and* controls for the count nature of recreation demand data. The RNL model is not, however, without its drawbacks. Two key assumptions are employed in its development. First, individuals are assumed to decide whether and where to recreate during a fixed number of choice occasions in a season. For example, it is common practice to assume that households face 52 weekly choice occasions during a year. Second, choice occasion decisions are assumed to be independent not only across individuals, but also across choice occasions for the same individual. This latter assumption precludes habit formation or learning on the part of the recreator and it is the relaxation of this assumption that is the focus of this paper.

Recently, Phaneuf. Kling, and Herriges [24] proposed introducing correlation across choice occasions into the RNL framework by adopting the mixed logit (or random parameters logit) specification developed in McFadden and Train [17].[3] The mixed logit model represents a generalization of multinomial logit, introducing additional error components into the preference structure underlying consumer choices. These error components can modify specific parameters of the individual's preference function (resulting in a "random parameters" specification) or they can be used to capture complex correlation patterns across alternatives and/or choice occasions. The purpose of this paper is to examine the potential for the resulting repeated mixed logit (or RXL) model to address criticisms of the RNL model.

The remainder of the paper is divided into five sections. In Section 2 and 3, we provide an overview of the both the standard RNL specification and the RXL generalization, respectively, illustrating how the latter can be used capture correlation across both choice occasions and alternatives. In the RNL model, correlation across alternatives is captured by identifying *a priori* a nesting structure for the available set of alternatives. While the mixed logit approach can be used to mimic a given nesting structure, we demonstrate how it can be used to allow for more general correlation patterns among alternatives and, indeed, test for specific nesting structures. Furthermore, individual specific error components can be used to introduce correlation across choice occasions. A Monte Carlo experiment is employed in Section 4 to illustrate the properties of the RXL model. Section 5 provides an application of the RNL and RXL models to the demand for recreational angling in the Wisconsin Great Lakes region. Finally, conclusions and recommendations for applying the RXL model are provided in Section 6, along with suggestions for future research.

2. REPEATED NESTED LOGIT

The RNL model begins with the assumption that individuals face a fixed number choice occasions (T) during the course of a season, deciding on each choice occasion whether to stay at home or visit one of the M available sites. The conditional indirect utility that individual i receives from visiting site j during choice occasion t is assumed to take the form

$$
\begin{aligned}
U_{ijt} &= V_{ijt} + \varepsilon_{ijt} \\
&= V(y_i - c_{ij}, \mathbf{q}_{jt}; \beta) + \varepsilon_{ijt}, \quad i = 1,\ldots,N; j = 1,\ldots,M; t = 1,\ldots,T,
\end{aligned} \tag{1}
$$

whereas the utility associated with staying at home is given by

$$(2)\qquad \begin{aligned} U_{i0t} &= V_{i0t} + \varepsilon_{i0t} \\ &= V_0(y_i;\beta) + \varepsilon_{i0t}, \qquad i = 1\ldots,N; t = 1,\ldots,T, \end{aligned}$$

where y_i denotes individual i's income per choice occasion, c_{ij} denotes the cost for individual i to visit site j, $\mathbf{q}_j$ is a vector of quality attributes for site j, and β is a vector of unknown parameters. The ε_{ijt}'s are random terms used to capture heterogeneity of preferences in the population, with the ε_{ijt}'s treated as known by the individual but unobserved by the analyst. Finally, the individual is assumed to select on each choice occasion that alternative providing the highest level of utility.

The above framework, in fact, applies to a variety of random utility models. The RNL model emerges by imposing further structure on the way in which preferences vary across individuals (and choice occasions). In particular, the vectors $\varepsilon_{i\cdot t} \equiv (\varepsilon_{i0t},\ldots,\varepsilon_{iJt})'$ are assumed to be independent and identically distributed across individuals *and* choice occasions and drawn from a Generalize Extreme Value (GEV) distribution. Thus, each choice occasion is treated as an independent event characterized by a nested logit model. Nested logit is used at the choice occasion level to allow for correlation (nesting) patterns among the alternatives. Alternatives within the same nest are more similar (i.e., better substitutes) than alternatives in different nests. It is typical, for example, to assume that the recreation sites are grouped into a nest separate from the stay at home alternative (i.e., j=0). Figure 1a illustrates the implied nesting structure. The corresponding choice probabilities are then given by:

$$(3)\qquad P_{ijt} = \begin{cases} 1 - Q_{it} & j = 0 \\ Q_{it} P_{ijt|trip} & j = 1,\ldots,M \end{cases}$$

where

$$(4) \qquad Q_{it} = \frac{\left[\sum_{k=1}^{M} \exp\left(V_{ikt}/\theta\right)\right]^{\theta}}{\left[\sum_{k=1}^{M} \exp\left(V_{ikt}/\theta\right)\right]^{\theta} + V_{i0t}}$$

denotes the probability that individual i chooses to take a trip on choice occasion t and

$$(5) \qquad P_{ijt|trip} = \frac{\exp\left(V_{ijt}/\theta\right)}{\sum_{k=1}^{M} \exp\left(V_{ikt}/\theta\right)} \qquad j = 1,\ldots,M$$

denotes the conditional probability that individual i chooses to visit site j on choice occasion t *given* that they have decided to take a trip. The parameter θ is known as the dissimilarity coefficient and is required to lie in the unit interval (see, eg., [18]). As θ declines towards zero, greater similarity and correlation exists among choice alternatives within the same nest, whereas as θ approaches one the model reduces to multinomial logit, with independence among the utilities associated with the various choice alternatives ($j = 0,\ldots,M$). This two-level nested logit model imposes the following block diagonal structure on the variance-covariance matrix of the underlying error components:

$$(6) \qquad \Sigma \equiv \begin{bmatrix} \sigma_{00} & \sigma_{01} & \cdots & \sigma_{0M} \\ \sigma_{10} & \sigma_{11} & \cdots & \sigma_{1M} \\ \vdots & \vdots & \ddots & \vdots \\ \sigma_{M0} & \sigma_{M1} & \cdots & \sigma_{MM} \end{bmatrix} = \begin{bmatrix} \sigma_{00} & 0 & \cdots & 0 \\ 0 & \sigma^2_{trip} & \cdots & \rho\sigma^2_{trip} \\ \vdots & \vdots & \ddots & \vdots \\ 0 & \rho\sigma^2_{trip} & \cdots & \sigma^2_{trip} \end{bmatrix} = \begin{bmatrix} \sigma_{00} & \mathbf{0}'_M \\ \mathbf{0}_M & \Sigma_{trip} \end{bmatrix},$$

where $\sigma_{ii'} \equiv Cov\left(\varepsilon_{ijt}, \varepsilon_{i'jt}\right) \ \forall j,t$, $\mathbf{0}_n$ is an $n \times 1$ vector of zeros,

$$(7) \qquad \Sigma_{trip} = \sigma^2_{trip} \begin{bmatrix} 1 & \rho & \cdots & \rho \\ \rho & 1 & \cdots & \rho \\ \vdots & \vdots & \ddots & \vdots \\ \rho & \rho & \cdots & 1 \end{bmatrix}$$

and $\rho \in [0,1)$ measures the correlation among the various trip alternatives. To understand the implications of this model, consider the simplest case in which $V_{ijt} = V \ \forall i,j,t$, so that only the stochastic terms matter. Intuitively, the structure in (6) argues that if an individual prefers to travel to site 1 rather than staying at home (i.e., because $\varepsilon_{i1t} > \varepsilon_{i0t}$) then they are also more likely to prefer site 2 to staying at home since, given the positive correlation between ε_{i1t} and ε_{i2t}, it is more likely that $\varepsilon_{i2t} > \varepsilon_{i0t}$.

More complex correlation patterns among the choice alternatives can be imposed in the RNL model by further dividing the trip nest (i.e., sites 1 through M) into sub-nests. For example, Figure 1b illustrates a three-level nested logit model that groups alternatives 1 through M_A into a sub-nest A of similar sites and alternatives M_A+1 through M into a sub-nest B of similar sites. The resulting choice probabilities take the form:

$$(8) \qquad P_{ijt} = \begin{cases} 1-Q'_{it} & j=0 \\ Q'_{it} P_{iAt|trip} P_{ijt|iAt} & j=1,\ldots,M_A \\ Q'_{it} P_{iBt|trip} P_{ijt|iBt} & j=M_A+1,\ldots,M \end{cases}$$

where Q'_{it} denotes the probability of taking a trip, $P_{iAt|trip}$ denotes the conditional probability of visiting one of the sites in sub-nest A *given* that a trip is take, and $P_{ijt|iAt}$ denotes the conditional probability of visiting site j in sub-nest A *given* that the individual chooses to visit sub-nest A.[4] As in the case of the two-level nested logit model, this implies a specific block structure to variance-covariance matrix of the underlying error components:

$$(9) \qquad \overline{\Sigma} \equiv \begin{bmatrix} \sigma_{00} & \mathbf{0}'_{M_A} & \mathbf{0}'_{M_B} \\ \mathbf{0}_{M_A} & \Sigma_A & \Sigma'_{AB} \\ \mathbf{0}_{M_B} & \Sigma_{AB} & \Sigma_B \end{bmatrix},$$

with $M_B \equiv M - M_A$,

$$(10) \qquad \Sigma_k \equiv \sigma_k^2 \begin{bmatrix} 1 & \rho_k & \cdots & \rho_k \\ \rho_k & 1 & \cdots & \rho_k \\ \vdots & \vdots & \ddots & \vdots \\ \rho_k & \rho_k & \cdots & 1 \end{bmatrix}, k = A, B.$$

$\Sigma_{AB} = \sigma_{AB} J$, where $\sigma_{AB} \equiv \rho_{AB}\sigma_A\sigma_B$, $\rho_{AB} = Corr(\varepsilon_{ijt}, \varepsilon_{ij't})\ \forall j \in A, j' \in B$, and J is an $M_B \times M_A$ matrix of ones). Two alternatives within the same sub-nest are typically assumed to be more correlated than alternatives from different nests (i.e., $\rho_A > \rho_{AB} \geq 0$ and $\rho_B > \rho_{AB} \geq 0$).

The intuition for the three-level nesting structure in (9) is more complicated but similar to that for the two-level nest. Once again, if an individual prefers site 1 to staying at home then they are also more likely to prefer one of the other sites to staying at home, since the "trip" ε_{ijt}'s are positively correlated. Moving down the nesting structure, if the individual prefers the first site in sub-nest A (e.g., j=1) to sites in sub-nest B (i.e. $\varepsilon_{i1t} > \varepsilon_{ij't}, j' \in B$), then it is likely that any site in sub-nest A would be preferred to any site in sub-nest B. This is because the values of the sub-nest A's ε_{ijt}'s are more highly correlated ($\rho_A > \rho_{AB}$), reflecting the belief of the analyst that sites in a sub-nest are close substitutes.

Regardless of the chosen nesting structure, the resulting log-likelihood function is then given by:

$$(11) \qquad LL_{RPL}(\beta) = \sum_{i=1}^{N}\sum_{j=1}^{M}\sum_{t=1}^{T} I_{ijt} \ln P_{ijt},$$

where $I_{ijt} = 1$ if individual i chose to visit site j on choice occasion t; = 0 otherwise.

3. REPEATED MIXED LOGIT

Similar to the RNL model, the basic repeated mixed logit (RXL) model begins with the specification of conditional indirect utility functions for the various alternatives. The utility received by individual i during choice occasion t from visiting site j is given by

$$(12) \qquad \begin{aligned} \widetilde{U}_{ijt} &= \widetilde{V}_{ijt}(\widetilde{\beta}_{it}) + \widetilde{\varepsilon}_{ijt} \\ &= V(y_i - c_{ij}, \mathbf{q}_{jt}; \widetilde{\beta}_{it}) + \widetilde{\varepsilon}_{ijt}, \quad i = 1,\ldots,N; j = 1,\ldots,M; t = 1,\ldots,T, \end{aligned}$$

whereas the utility associated with staying at home is given by

$$
(13)\qquad \begin{aligned}\widetilde{U}_{i0t} &= \widetilde{V}_{i0t}\left(\widetilde{\beta}_{it}\right) + \widetilde{\varepsilon}_{i0t} \\ &= \widetilde{V}_0(y_i;\widetilde{\beta}_{it}) + \widetilde{\varepsilon}_{i0t}, \qquad i = 1\ldots,N; t = 1,\ldots,T.\end{aligned}
$$

The model deviates from the RNL in two respects. First, the additive error terms (i.e., the $\widetilde{\varepsilon}_{ijt}$'s) are assumed to be i.i.d. extreme value variates. Second, the parameter vector $\widetilde{\beta}_{it}$ is now assumed to be random, rather than fixed, potentially varying both across individuals and choice occasions. As with the RNL model, all of the random components (i.e., the $\widetilde{\varepsilon}_{ijt}$'s and $\widetilde{\beta}_{it}$'s) are assumed to be known by the individual, but unobserved by the analyst.

Conditional on the parameter vector $\widetilde{\beta}_{it}$, the probability of observing that individual i chooses alternative j on choice occasion t follows the standard logit form:

$$
(14)\qquad Y_{ijt}(\widetilde{\beta}_{it}) = \frac{\exp\left[\widetilde{V}_{ijt}\left(\widetilde{\beta}_{it}\right)\right]}{\sum_{k=0}^{M}\exp\left[\widetilde{V}_{ikt}\left(\widetilde{\beta}_{it}\right)\right]}.
$$

The corresponding unconditional probability, $\widetilde{P}_{ijt}(\varphi)$, is obtained by integrating over an assumed probability density function for the $\widetilde{\beta}_{it}$'s. Typically, the $\widetilde{\beta}_{it}$'s are assumed to be i.i.d., so that

$$
(15)\qquad \widetilde{P}_{ijt}(\varphi) = \int Y_{ijt}(\beta) f(\beta|\varphi)d\beta,
$$

where $f(\beta|\varphi)$ is the pdf for β, parameterized by φ. The log-likelihood is then given by

$$
(16)\qquad LL_{RXL}(\varphi) = \sum_{i=1}^{N}\sum_{j=1}^{M}\sum_{t=1}^{T} I_{ijt}\ln \widetilde{P}_{ijt}(\varphi).
$$

While the conditional choice probabilities (i.e., the $Y_{ijt}\left(\widetilde{\beta}_{it}\right)$'s) are easy to compute, simulation methods are typically required to compute the unconditional probabilities (the $\widetilde{P}_{ijt}(\varphi)$'s) in the process of constructing the maximum likelihood estimates of φ.[5]

An important advantage of the RXL specification over its RNL counterpart is that it allows for greater heterogeneity in individual preferences. The RNL model implicitly allows for shifts in utility in terms of the intercepts (through the ε_{ijt}'s), but restricts marginal effects (such as the marginal utility of income) to be the same across individuals. The mixed logit framework relaxes this latter assumption by treating the $\widetilde{\beta}_{it}$'s as random.

There are two key disadvantages of the basic RXL model outlined above. First, unlike the RNL model, it assumes that, for a given individual and choice occasion, alternative specific utilities are uncorrelated (since the $\widetilde{\varepsilon}_{ijt}$'s are i.i.d.). The nesting structure employed in the RNL framework to group similar alternatives is missing.[6] Second, like the RNL model, there is no correlation in utilities across choice occasions for a given individual.[7] In the following two subsections, the RXL model is generalized so as to relax these two restrictions.

3.1. Cross-site correlation

The RNL model imposes a specific correlation (or substitution) pattern across sites on a given choice occasion by nesting similar alternatives, assuming that the error vector $\varepsilon_{i\cdot t}$ is drawn from the appropriate GEV distribution. As Train [29, p. 127] notes, the analogue to a nest emerges in the mixed logit framework when a random dummy variable is introduced to group certain alternatives. For example, the counterpart to the nested logit model in Figure 1a emerges if equation (12) is replaced by

$$
\begin{aligned}
\widetilde{U}_{ijt} &= \widetilde{V}_{ijt}(\widetilde{\beta}_{it}) + \delta_{it} + \widetilde{\varepsilon}_{ijt} \\
&= \widetilde{V}_{ijt}(\widetilde{\beta}_{it}) + \widetilde{\eta}_{ijt}, \quad i = 1,\ldots,N; j = 1,\ldots,M; t = 1,\ldots,T,
\end{aligned}
\tag{17}
$$

where $\widetilde{\eta}_{ijt} \equiv \delta_{it} + \widetilde{\varepsilon}_{ijt}$ and δ_{it} is an i.i.d. random variable that is also independent of $\widetilde{\varepsilon}_{ijt}$. The composite error term $\widetilde{\eta}_{ijt}$ has two components, one that is independent across alternatives ($\widetilde{\varepsilon}_{ijt}$) and one that is shared by all of the trip alternatives (δ_{it}).[8] It is the latter term that captures the "similarity" of the M trip alternatives. Individuals with a large positive realization of δ_{it} tend to prefer taking some sort of trip on choice occasion t, since the corresponding $\widetilde{U}_{ijt}$'s ($j = 1,\ldots, M$) will, *ceteris paribus*, be larger than $\widetilde{U}_{i0t}$. Similarly, when δ_{it} is negative, all of the trip alternatives will be less attractive. The variance-covariance matrix for $\widetilde{\eta}_{ijt}$ has the same structure as in equation (7).

More complex nesting structures can be created by incorporating additional random dummy variables into the model. For example, equation (17) can be replaced by:

$$(18)\qquad \begin{aligned}\widetilde{U}_{ijt} &= \widetilde{V}_{ijt}(\widetilde{\beta}_{it}) + \delta_{it} + \sum_{k=1}^{M} \tau_{it}^{jk} + \widetilde{\varepsilon}_{ijt} \\ &= \widetilde{V}_{ijt}(\widetilde{\beta}_{it}) + \widetilde{\eta}_{ijt}, \quad i = 1,\ldots,N; j = 1,\ldots,M; t = 1,\ldots,T,\end{aligned}$$

where now $\widetilde{\eta}_{ijt} \equiv \delta_{it} + \sum_{k=1}^{M} \tau_{it}^{jk} + \widetilde{\varepsilon}_{ijt}$ $(i = 1,\ldots,N; j = 1,\ldots,M; t = 1,\ldots,T)$ and $\tau_{it}^{kj} = \tau_{it}^{jk}\ \forall j,k$. The τ_{it}^{jk}'s capture pair-wise similarities of sites.

The structure in (18) can be used to mimic a wide variety of nesting structures. To illustrate this, consider the special case in which $M = 4$. Let $\tau_{it}^{jj} \sim N(\bar{\tau}^{jj}, \sigma^2_{\tau(j,j)})$, $\tau_{it}^{jk} \sim N(0, \sigma^2_{\tau(j,k)})$ $(j \neq k)$, and $\delta_{it} \sim N(0, \sigma_\delta^2)$. The implied variance-covariance matrix is then given by:

$$(19)\qquad \widetilde{\Sigma} \equiv \begin{bmatrix} \widetilde{\sigma}_{00} & \widetilde{\sigma}_{01} & \cdots & \widetilde{\sigma}_{04} \\ \widetilde{\sigma}_{10} & \widetilde{\sigma}_{11} & \cdots & \widetilde{\sigma}_{14} \\ \vdots & \vdots & \ddots & \vdots \\ \widetilde{\sigma}_{40} & \widetilde{\sigma}_{41} & \cdots & \widetilde{\sigma}_{44} \end{bmatrix} = \begin{bmatrix} \widetilde{\sigma}^2 & 0 & \cdots & 0 \\ 0 & \widetilde{\sigma}_{11} & \cdots & \widetilde{\sigma}_{14} \\ \vdots & \vdots & \ddots & \vdots \\ 0 & \widetilde{\sigma}_{41} & \cdots & \widetilde{\sigma}_{41} \end{bmatrix} = \begin{bmatrix} \widetilde{\sigma}^2 & \mathbf{0}_4' \\ \mathbf{0}_4 & \widetilde{\Sigma}_{trip} \end{bmatrix},$$

where $\widetilde{\sigma}^2 \equiv Var(\widetilde{\varepsilon}_{ijt})$, $\widetilde{\sigma}_{jj} \equiv Var(\widetilde{\eta}_{ijt}) = \widetilde{\sigma}^2 + \sigma_\delta^2 + \sum_{k=1}^{M} \sigma^2_{\tau(j,k)}$, and $\widetilde{\sigma}_{jk} \equiv Cov(\widetilde{\eta}_{ijt}, \widetilde{\eta}_{ikt}) = \sigma_\delta^2 + \sigma^2_{\tau(j,k)}$.

The nesting structures in Figure 2 are obtained using the following restrictions:

- Figure 2a: 2-level nest {0, (1,2,3,4)}. This nesting structure, analogous to Figure 1a, is obtained by imposing the restrictions:

(20) **R1**: $\sigma_{\tau(j,k)} = 0\ \forall j,k$,

so that

(21) $$\tilde{\Sigma}_{trip} = \begin{bmatrix} a & b & b & b \\ b & a & b & b \\ b & b & a & b \\ b & b & b & a \end{bmatrix},$$

where $a \equiv \tilde{\sigma}^2 + \sigma_\delta^2$ and $b \equiv \sigma_\delta^2$. This is the same pattern as in (7).

- Figure 2b: 3-level nest {0, [(1,2),(3,4)]}. This nesting structure, analogous to Figure 1b, is obtained by imposing the restrictions:

(22) **R2**: $$\sigma_{\tau\{j,k\}} = \begin{cases} \sigma_\tau & \{j,k\} \in \{\{1,2\},\{3,4\}\} \\ 0 & \text{otherwise} \end{cases},$$

so that

(23) $$\tilde{\Sigma}_{trip} = \begin{bmatrix} a & c & b & b \\ c & a & b & b \\ b & b & a & c \\ b & b & c & a \end{bmatrix},$$

where $a \equiv \tilde{\sigma}^2 + \sigma_\delta^2 + \sigma_\tau^2$, $b \equiv \sigma_\delta^2$, and $c = \sigma_\delta^2 + \sigma_\tau^2 > b$. This is the same pattern as in (9).

- Figure 2c: 3-level nest {0, [(1,2),3,4]}. This nesting structure is similar to the previous nest, except that alternatives 3 and 4 are not isolated into a separate sub-nest:

(24) **R3**: $\sigma_{\tau\{j,k\}} = 0 \ \forall\{j,k\} \neq \{1,2\}$,

so that

(25) $$\tilde{\Sigma}_{trip} = \begin{bmatrix} a & c & b & b \\ c & a & b & b \\ b & b & a & b \\ b & b & b & a \end{bmatrix}.$$

As the above examples illustrate, a wide variety of "nesting" patterns can be allowed for in the RXL framework. The pairwise nesting suggested above is just one alternative. Other possibilities would include grouping alternatives by type of recreation (e.g., shore versus boat fishing) or by geographical proximity. Furthermore, in contrast to nested logit, it is relatively straightforward to test competing nesting structures when using mixed logit. As Herriges and Kling [12] note, it is common practice in the literature to simply assume a nesting structure when employing nested logit models, without formally testing it against competing assumptions. At best, informal criteria are used, such as consistency with the utility maximization (e.g., [14]) or likelihood dominance (e.g., [12]). Yet, the assumed nesting structure can have a potentially significant impact on the resulting welfare measures ([12]). The mixed logit framework avoids this problem by allowing for *overlapping* nests. For example, after estimating the general specification in (18), the three-level nest {0, [(1,2), (3,4)]} can be explicitly tested for using the restrictions R2 in equation (22). The competing nesting structure {0, [(1,3), (2,4)]} can likewise be tested using the restrictions:

(26) **R4**: $$\sigma_{\tau(j,k)} = \begin{cases} \sigma_\tau & (j,k) \in \{(1,3),(2,4)\} \\ 0 & \text{otherwise.} \end{cases}$$

3.2. Cross Choice Occasion Correlation

Unlike the RNL, the RXL is capable of modeling cross choice occasion correlation by including individual specific error components that are constant over time. This is analogous to random effects models used in continuous panel data models. For example, equation (18) can be generalized as:

(27) $$\begin{aligned} \widetilde{U}_{ijt} &= \widetilde{V}_{ijt}(\widetilde{\beta}_{it}) + \gamma_{ij} + \delta_{it} + \sum_{k=1}^{M} \tau_{it}^{jk} + \widetilde{\varepsilon}_{ijt} \\ &= \widetilde{V}_{ijt}(\widetilde{\beta}_{it}) + \widetilde{\eta}_{ijt}, \quad i = 1,\ldots,N; j = 1,\ldots,M; t = 1,\ldots,T, \end{aligned}$$

where now $\widetilde{\eta}_{ijt} \equiv \gamma_{ij} + \delta_{it} + \sum_{k=1}^{M} \tau_{it}^{jk} + \widetilde{\varepsilon}_{ijt}$. The random component γ_{ij} can be viewed as the unobserved portion of an individual's site utility that does not vary over time.[9] One might, for example, assume that $\gamma_{ij} \sim N(0, \sigma^2_{\gamma(j)})$. With this specification, $Cov(\widetilde{U}_{ijt}, \widetilde{U}_{ijs}) = \sigma^2_{\gamma(j)}\ \forall t \neq s$.

4. MONTE CARLO EXPERIMENT

As the previous section suggests, the RXL model can be used to mimic the familiar nesting structures embodied in the RNL model of recreation demand. Moreover, unlike RNL, the mixed logit framework allows for explicit testing of competing nesting specifications. The purpose of this section is to illustrate these features of the RXL model through a simple Monte Carlo experiment. Data are generated using a RNL model with $M=4$ and the nesting structure in Figure 2b. We then examine how well the RXL model detects the underlying correlation pattern.

Specifically, we assume that 500 individuals (N=500) have four recreation sites to choose from, in addition to the option of not making a trip (M=4), during the course of 5 choice occasions (T=5). Preferences are generated by the simple conditional utility function:

$$U_{ijt} = -\beta c_{ijt} + \varepsilon_{ijt} \qquad i = 1,\ldots,N; j = 1,\ldots, M; t = 1,\ldots,T, \tag{28}$$

where $\beta = 0.003$ denotes the marginal utility of income. The cost of visiting a specific site are assumed to be fixed over time (i.e., $c_{ijt} = c_{ij}\ \forall t$), drawn from uniform distributions for each individual-choice occasion combination (i.e., $c_{ij} \sim i.i.d. U[50,90]$). The error vector $\varepsilon_{i\cdot t}$ is assumed to be i.i.d. over time and across individuals, drawn from a GEV value distribution with the three-level nesting structure depicted in Figure 2b. The dissimilarity coefficient for the upper level nest (i.e., trip versus non-trip) is set at $\theta = 0.5$, while the dissimilarity coefficient for the lower level nests (i.e., in choosing between sub-nest (1,2) versus (3,4)) is $\rho = 0.25$. This structure implies that there is greater similarity between alternatives in the same sub-nest (say 1 and 2) than between alternatives in different sub-nests (say 1 and 3). Likewise, there is greater similarity between any two trip options (say 1 and 4) than between a trip option and staying at home (e.g., 0 and 1). The choice probabilities have the form outlined in equation (8), with $M_A = M_B = 2$, $A = \{1,2\}$, and $B = \{3,4\}$.[10] These choice probabilities were then used to simulate choice outcomes for each of the 500 individuals over five choice occasions, yielding a data set with 2500 independent observations.[11] Two hundred such data sets were generated for this Monte Carlo experiment.

In examining the performance of the RXL model, three specifications were estimated for each of the 200 data sets:[12]

- Unconstrained: This model contains dummy variables relating all possible pairs of sites (i.e. $\tau_{it}^{jk} \sim N\left(0,\sigma^2_{\tau(j,k)}\right)$, $\forall j \neq k; j,k = 1,\ldots,4$), in addition to the outer nest dummy variable $\delta_{it} \sim N\left(0,\sigma^2_{\delta}\right)$. That is,

$$(29) \qquad \widetilde{U}_{ijt} = -\beta c_{ij} + \delta_{it} + \sum_{\substack{k=1 \\ k\neq j}}^{4} \tau_{it}^{jk} + \widetilde{\varepsilon}_{ijt} \qquad i = 1,\ldots,N; j = 1,\ldots,M; t = 1,\ldots,T.$$

- "True": This model contains dummy variables relating sites (1,2), (3,4) and (1,2,3,4), imposing the restrictions that $\sigma_{\tau(j,k)} = 0 \;\forall (j,k) \notin \{(1,2),(3,4)\}$.

- "False": This model contains dummy variables relating site (1,3), (2,4) and (1,2,3,4), imposing the restrictions that $\sigma_{\tau(j,k)} = 0 \;\forall (j,k) \notin \{(1,3),(2,4)\}$.

A typical example of the resulting parameters is provided in Table 1. As expected, the unconstrained RXL model detects the correlation pattern implicit in the generated the data. The correlation among all trips (σ^2_{δ}) is statistically significantly at 1% level. Furthermore, the sub-nest correlations between sites 1 and 2 ($\sigma^2_{\tau(1,2)}$) and sites 3 and 4 ($\sigma^2_{\tau(3,4)}$) are also found to be significant using a 1% critical level. In contrast, the remaining pairwise correlations are generally insignificant, with only $\sigma^2_{\tau(2,4)}$ departing significantly from zero and then only at the 5% level. When the restrictions underlying the true nesting structure are imposed (i.e., $\sigma_{\tau(j,k)} = 0 \;\forall (j,k) \notin \{(1,2),(3,4)\}$), as in the third column of Table 1, the restrictions cannot be rejected at any reasonable confidence level.[13] The remaining parameters for the "True" model are all statistically significant. On the other hand, when we attempt to impose the wrong nesting structure, as in the last column of Table 1, the corresponding restrictions are soundly rejected. A likelihood ratio test statistic is χ^2_4=42.92 with a P-value of less than 0.001. Furthermore, the remaining $\sigma^2_{\tau(1,3)}$ and $\sigma^2_{\tau(2,4)}$ are insignificant.

This same pattern of results in Table 1 emerges in general for the 200 replications of this Monte Carlo experiment. Using a 5% critical level, the "False" model restrictions were rejected in 81% of the replications. whereas the true model restrictions were rejected in only 37% of the cases.[14]

5. APPLICATION

The RXL model is illustrated in this section with an application to recreational angling in the Wisconsin Great Lakes region. Results from a comparable RNL model are provided for comparison purposes.

5.1. Data

Data on angling behavior in the Wisconsin Great Lakes region during the 1989 season were gathered via mail surveys by Richard Bishop and Audrey Lyke at the University of Wisconsin – Madison.[15] The surveys provided detailed information about Wisconsin fishing license holders, including the number and destination of fishing trips to the Wisconsin great lakes region, the distances to each region, the type of angling preferred, and socio-demographic characteristics of the respondents. A total of 487 completed survey were available (i.e., $N = 487$), including responses from 240 individuals who visited at least one of the 22 Great Lakes destinations defined in the survey and 247 who fished only inland waters (non-users from the perspective of the Great Lakes region). We have aggregated the destinations of anglers into four sites:

1. South Lake Michigan
2. North Lake Michigan
3. Green Bay
4. Lake Superior

This aggregation divides the Wisconsin portion of the Great Lakes into geographical zones consistent with Wisconsin Department of Natural Resources' classification of the lake region.

The price of a trip to each of the four sites consists of both the direct cost of getting to the site and the opportunity cost of the travel time. Travel costs were computed based on the round trip cost of travel for the vehicle class, while the opportunity cost of time was computed using one-third of the wage rate. The price of a trip is the sum of these two components.

Three site and household characteristic variables are used in our application: fishing catch rates, toxin levels in fish, and an indicator variable for boat ownership. Catch rates were available for the relevant time period from creel surveys by the Wisconsin Department of Natural Resources. A catch rate index was formed as a weighted average of the catch rates for the four aggressively managed salmon species: lake trout, rainbow trout, Coho salmon, and Chinook salmon. In particular, we formed:

$$(30) \qquad R_j \equiv \sum_k w_k R_{kj} \,,$$

where w_k denotes the percentage of anglers indicating that they were fishing for the k^{th} fish species (k = lake trout, etc.) and R_{kj} denotes the catch rate for species k at site j.

Toxin levels in fish were obtained from De Vault *et al.* [8]. Toxins provide a good proxy for overall water quality, and are directly responsible for consumption advisories. Results from [8] were matched on the basis of proximity to the four sites defined above. However, toxin levels at the sites are likely to affect visitation rates only if the individual perceives them to create a safety issue. The Wisconsin angling survey provided information regarding this perception. We used this to form an "effective toxins" variable $E_{ij} = T_j D_i$, where T_j denotes the toxin level at site j and $D_i = 1$ if individual i was concerned about fish toxin levels and $D_i = 0$ otherwise.

5.2. Model Specification

Both the RNL and RXL models require specification of conditional indirect utility functions. For the RNL model we assume that conditional indirect utility that individual i receives from visiting site j during choice occasion t takes the form[16]

$$(31) \qquad U_{ijt} = -\beta_1 c_{ij} + \beta_2 R_{ij} + \beta_3 E_{ij} + \varepsilon_{ijt}, \quad i = 1,\ldots,N; j = 1,\ldots,M; t = 1,\ldots,T,$$

whereas the utility associated with staying at home is given by

$$(32) \qquad U_{i0t} = \beta_0 + \beta_4 B_i + \varepsilon_{i0t}, \quad i = 1\ldots,N; t = 1,\ldots,T,$$

where B_i=0 if individual i owns a boat; =0 otherwise. The error terms, ε_{ijt}'s, are assumed to be independent across individuals and choice occasions, with $\varepsilon_{i\cdot t}$ drawn from a GEV distribution with an assumed nesting structure of $\{0,[(2,3),(1,4)]\}$.[17]

A similar specification was used for the RXL model, with

$$(33) \quad \tilde{U}_{ijt} = -\beta_1 c_{ij} + \beta_2 R_{ij} + \beta_3 E_{ij} + \tilde{\eta}_{ijt} \qquad i = 1,\ldots,N; j = 1,\ldots,M; t = 1,\ldots,T,$$

for visits to a recreational site, whereas

$$(34) \quad \tilde{U}_{i0t} = \beta_0 + \beta_4 B_i + \tilde{\eta}_{i0t}, \quad i = 1\ldots,N; t = 1,\ldots,T,$$

for staying at home.[18] The error terms, $\tilde{\eta}_{ijt}$'s are assumed to take the form:

$$(35) \quad \tilde{\eta}_{ijt} = \begin{cases} \gamma_{ij} + \delta_i + \sum_{\substack{k=1 \\ k \neq j}}^{4} \tau_i^{jk} + \tilde{\varepsilon}_{ijt} & j = 1,\ldots,4; i = 1,\ldots,N; t = 1,\ldots,T \\ \tilde{\varepsilon}_{ijt} + \gamma_{ij} & j = 0; i = 1,\ldots,N; t = 1,\ldots,T, \end{cases}$$

where $\gamma_{ij} \sim iid\, N(0,\sigma_\gamma^2)$, $\delta_i \sim iid\, N(0,\sigma_\delta^2)$, $\tau_i^{jk} \sim iid\, N(0,\sigma_{\tau(j,k)}^2)$, and $\tilde{\varepsilon}_{ijt}$ drawn from an extreme value distribution. This specification assumes that the nesting structure, captured by the terms δ_i and τ_i^{jk}, remains constant across choice occasions. Thus, unlike the RNL model, there is correlation across choice occasions (i.e., when $t \neq t'$) for the same individual, since

$$(36) \quad Cov(\tilde{\eta}_{ijt}\tilde{\eta}_{ij't'}) = \begin{cases} \sigma_\gamma^2 + \sigma_\delta^2 + \sum_{\substack{k=1 \\ k \neq j}}^{4} \sigma_{\tau(j,k)}^2 & j = j' \\ \sigma_\delta^2 + \sigma_{\tau(j,j')}^2 & j \neq j' \end{cases}$$

In the application section below, we estimate both the unconstrained specification and three restricted versions of the model:

- <u>RXL-A: No nesting structure</u>. This model allows cross equation correlations (through γ_{ij}), but allows for no nesting structure (i.e., restricting $\sigma_\delta = 0$ and $\sigma_{\tau(j,k)} = 0\ \forall j,k$).
- <u>RXL-B: Limited cross-choice occasion correlation</u>: This model allows for a general nesting structure, but requires that there are no cross-choice occasion correlations beyond the nesting structure (i.e., $\sigma_\gamma = 0$).
- <u>RXL-C: Analogue to RNL model</u>: This model imposes an error structure analogous to the RNL's nesting structure of $\{0,[(2,3),(1,4)]\}$. In particular, it imposes the restrictions that there is no cross-choice occasion correlation (i.e., $\sigma_\gamma = 0$) beyond the nesting structure and that

$$(37) \qquad \sigma_{\tau\{j,k\}} = \begin{cases} \sigma_\tau & \{j,k\} \in \{\{2,3\},\{1,4\}\} \\ 0 & \text{otherwise.} \end{cases}$$

5.3. Results

The parameter estimates for both the RNL and RXL models are provided in Table 2. Beginning with the RNL results in column 2, we find that the parameter estimates have the expected signs and are uniformly significant at a one-percent critical level. The marginal utility of income (β_1) is positive, with a point estimate of 0.003. As expected, a high catch rate significantly increases the utility of a site, while higher effective toxin levels diminishes utility. Owning a boat reduces the probability of staying at home on a given choice occasion, with $\beta_4 = -1.55$. Finally, both the upper- and lower-level dissimilarity coefficients (θ and ρ, respectively) lie in the unit interval (with $0 < \rho < \theta < 1$) and are significantly different from one, indicating that a distinct correlation patterns exist among the alternative sites. In particular, visits to North Lake Michigan (2) and Green Bay (3) are more similar than visits to North Lake Michigan (2) and Lake Superior (4). Similarly, angling trips are more similar to each other than to the "stay at home" option.

Columns 3 through 6 of Table 2 provide the parameter estimates for the various RXL models. Beginning with the unconstrained specification, several results emerge. First, like in the RNL model, the parameters associated with nonstochastic portion of the RXL utility function (i.e., the β_k's) all have the expected signs and are statistically significant. Second, a complex nesting structure appears to exist among the various alternatives. Like RNL, the four site alternatives (j = 1, 2, 3, 4) are found to be correlated, although σ_δ is not significantly different from zero. Also like the RNL model, sites 2 and 3 (1 and 4) are even more correlated, with $\sigma_{\tau\{2,3\}}$ ($\sigma_{\tau\{1,4\}}$) significantly different from zero at a one percent critical level. This is analogous to the fact that ρ is significantly different from 1 in the RNL model. However, unlike the RNL model, these are not the only cross-site correlations that exist. Indeed, each pairwise correlation term ($\sigma_{\tau\{j,k\}}$) is statistically significant at a one percent critical level.

Third, additional correlation (i.e., beyond the nesting structure) exists across choice occasions, as indicated by the fact that the individual specific error component γ_{ij} is significant (with σ_γ significantly different from zero at a one-percent critical level). Fourth and finally, the unconstrained RXL specification yields a substantial reduction (31%) in the log-likelihood function over its RNL counterpart. Obviously, these two models are not nested, so that a likelihood ratio test does not apply. However, using the likelihood dominance criterion of Pollak and Wales [25], the RXL specification would clearly be preferred.

Columns 4, 5 and 6 of Table 2 represent natural restrictions on the more general RXL specifications. Column 4 considers elimination of the general nesting structure that establishes linkages (i.e., correlations) across alternatives, leaving only the correlation across choice occasions. This restriction is soundly rejected, with the corresponding likelihood ratio statistic of $\chi^2_{df=7} = 66$ and a P-value of less than 0.001. Similarly, when the cross-choice occasion correlation is limited by constraining $\sigma_\gamma = 0$, as in column 5, the restriction is rejected, with $\chi^2_{df=1} = 20$ and a P-value of less than 0.001. Finally, specifying the RXL model so that it mimics the correlation structure implicit in the RNL model (as in the final column of Table 2), the remaining parameters are all statistically significant, yet the restriction is rejected, with likelihood ratio statistic of $\chi^2_{df=5} = 570$ and a P-value of less than 0.001.

5.4. Welfare Analysis

The motivation for estimating models of recreation demand is typically to evaluate the welfare effects of changing site characteristics or availability. In this subsection, we consider two hypothetical changes to conditions in the Wisconsin Great Lakes region:

- A 20% reduction in toxins at each of the four sites,
- Loss of the South Lake Michigan site.

For each scenario, mean compensating variation is computed for the five models presented in Table 2, comparing and contrasting both across the RNL and RXL frameworks and within the RXL approach given different error specifications.

In standard RNL models, the task of computing the compensating variation (CV) associated with a change in site characteristics or in the mix of available sites is straightforward, as closed form equations exist. For the RNL model estimated above,

$$(38) \qquad CV = \frac{T}{\beta_1}\left[IV^1 - IV^0\right],$$

where IV^r is the mean inclusive value associated with conditions r ($r = 0$ for initial conditions; $= 1$ for final conditions), with

$$(39) \qquad IV_i^r = \frac{1}{N}\sum_{i=1}^{N} \ln\left\{\left[\left(\exp(V_{i2}^r/\rho) + \exp(V_{i3}^r/\rho)\right)^{\rho/\theta} + \left(\exp(V_{i1}^r/\rho) + \exp(V_{i4}^r/\rho)\right)^{\rho/\theta}\right]^{\theta} + \exp(V_{i0}^r)\right\}$$

and

$$(40) \qquad V_{ij}^r = \begin{cases} -\beta_1 c_{ij}^r + \beta_2 R_{ij}^r + \beta_3 E_{ij}^r, & r = 0,1; j = 1,2,3,4 \\ \beta_0 + \beta_4 B_i^r & r = 0,1; j = 0. \end{cases}$$

Computing compensating variation for the RXL model is not as clear-cut and depends upon the interpretation of the error components γ_{ij}, δ_i, and τ_i^{jk}. If, as is typically the case in the literature, these components are treated as representing variation in consumer preferences across individuals in the population, then that variation should be accounted for in calculating compensating variation. A randomly selected individual in the population will have an expected CV that depends upon γ_{ij}, δ_i, and τ_i^{jk}, given by:

$$(41) \qquad \widetilde{CV}(\gamma_i, \delta_i, \tau_i) = \frac{T}{\beta_1}\left[\widetilde{IV}^1(\gamma_i, \delta_i, \tau_i) - \widetilde{IV}^0(\gamma_i, \delta_i, \tau_i)\right],$$

where $\widetilde{IV}^r(\gamma, \delta, \tau)$ is the mean inclusive value associated with conditions r and error components $\gamma \equiv (\gamma_0, \gamma_1, \gamma_2, \gamma_3, \gamma_4)$, δ_i, and $\tau \equiv (\tau^{12}, \tau^{13}, \ldots, \tau^{34})$

$$(42)\qquad I\tilde{V}_i^r(\gamma,\delta,\tau) = \tfrac{1}{N}\sum_{i=1}^{N} \ln\left\{\sum_{j=0}^{M} \exp\left[\tilde{V}_{ij}^r(\gamma,\delta,\tau)\right]\right\}$$

and

$$(43)\qquad \tilde{V}_{ij}^r(\gamma,\delta,\tau) = \begin{cases} -\beta_1 c_{ij}^r + \beta_2 R_{ij}^r + \beta_3 E_{ij}^r + \gamma_j + \delta + \sum_{\substack{k=1 \\ k\neq j}}^{4} \tau^{jk}, & r = 0,1; j = 1,2,3,4 \\ \beta_0 + \beta_4 B_i^r + \gamma_j & r = 01,; j = 0. \end{cases}$$

The unconditional compensating variation is then constructed using numerical integration, with

$$(44)\qquad C\tilde{V} = \tfrac{1}{S}\sum_{i=1}^{n} C\tilde{V}(\gamma^s,\delta^s,\tau^s)$$

where the superscript s is used to denote the s^{th} draw ($s = 1,\ldots, S$) from the estimated distributions for γ, δ, and τ.

An alternative compensating variation results if the error components γ_{ij}, δ_i, and τ_i^{jk} are interpreted as capturing measurement error. In this case, our best estimate of the underlying preference structure for any one individual corresponds to setting the error components to zero. The appropriate welfare measure then becomes

$$(45)\qquad CV^* \equiv C\tilde{V}(0,0,0).$$

In general, it will not be the case that $C\tilde{V} = CV^*$.[19]

Given the parameter estimates in Table 2, compensating variation estimates are provided in Table 3 for the two scenarios. The columns labeled "Calculation A" use $C\tilde{V}$ for the RXL models, whereas those labeled "Calculation B" use CV^*. Several results emerge. First, compensating variation estimates vary between the RXL and RNL frameworks and, to a lesser extent, across the error specifications used for the RXL model. For example, using $C\tilde{V}$ for the RXL models, the compensating variation associated with a twenty- percent reduction in toxin levels differs by almost a factor of two between the RNL ($22) and unconstrained RXL ($41) specifications. The RXL estimates themselves range from $36 under the RXL-B model to $46 under the RXL-C specification.

Similar patterns emerge in the loss of South Lake Michigan scenario.

Second, the interpretation of the error components significantly alters the implied compensating variation. This is particularly true when the error components are used to capture cross-site correlations. For the unconstrained RXL model $C\widetilde{V}$ is over six times CV^{*}. Unfortunately, there is no observable basis for choosing between these two interpretations.

6. SUMMARY AND CONCLUSIONS

The mixed logit framework has recently garnered considerable attention in the literature, providing a mechanism for generalizing the variety and complexity of the error structures that can be practically built into discrete choice models. Our goal is writing this paper was to explore how this framework can be specifically used to address concerns with the repeated nested logit model of recreation demand. First, can it be used to both test for specific nesting structures and allow for a wider range of nests? Second, can it be used to relax the implicit assumption that individual choices are independent across choice occasions? The answer to both questions appears to be "yes". The Monte Carlo exercise indicates that the RXL model can identify the nesting structure implicit in an RNL model and can in general identify correlation patterns contained in site visitation data. The application indicates that more complex and more general correlation patterns exist in practice than is typically assumed in nested logit models. Furthermore, these correlation patterns matter in terms of the implied welfare effects from changes in site attributes. The correlation across choice occasion also appears to be a significant factor in recreation demand, both in terms of fitted choice probabilities and implied welfare effects.

This research also raises some issues regarding the application of the RXL framework. First, as one might expect, careful specification of the error components is important. Ignoring both correlations across sites and/or across choice occasions can significantly alter the estimated welfare measures. The error components employed here are by no means exhaustive. Additional research is needed into the specification process. Second, and perhaps of greater concern, is the fact that the welfare measures themselves depend upon our interpretation of their source. Traditionally, they have been treated as representing real variation in consumer preferences in the population.[20] However, if they in fact stem from measurement error, then the appropriate welfare calculations can be quite different. Unfortunately, neither the theory nor the data are likely to provide much guidance in choosing between these two interpretations. Instead, analysts should probably compute both measures, with the hope that they do not differ substantially in practice.

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Table 1: Example Results for Monte Carlo Experiment[a]

Parameter	Model: Unconstrained	"Truth"	"False"
β	0.011 (0.006)	0.012** (0.001)	0.008** (<0.001)
σ_{δ}	4.06** (0.97)	3.42** (0.74)	1.97** (0.49)
$\sigma_{\tau\{1,2\}}$	2.24** (0.48)	1.87** (0.32)	
$\sigma_{\tau\{1,3\}}$	0.57 (0.41)		0.40 (0.35)
$\sigma_{\tau\{1,4\}}$	0.35 (0.42)		
$\sigma_{\tau\{2,3\}}$	0.35 (0.40)		
$\sigma_{\tau\{2,4\}}$	1.04* (0.41)		0.08 (0.48)
$\sigma_{\tau\{3,4\}}$	2.99** (0.56)	2.52** (0.34)	
Likelihood	-3298.34	-3300.88	-3319.80

[a]Standard Deviations are given in parentheses.
*Significantly different from zero at a 5% level.
**Significantly different from zero at a 1% level.

Table 2: Application

Parameter	Model				
	RNL	Unconstrained	RXL-A No Nesting	RXL-B Limited Corr.	RXL-C Limited Corr. RNL Nesting
Intercept	2.94** (0.01)	8.31** (0.40)	8.66** (0.40)	8.30** (0.33)	6.96** (0.26)
Income	0.003** (<0.001)	0.008** (<0.001)	0.007** (<0.001)	0.007** (<0.001)	0.012** (<0.001)
Catch	1.90** (0.07)	17.68** (1.44)	19.40** (1.44)	17.90** (1.27)	13.38** (0.24)
Toxin	-0.04** (0.00)	-0.12* (0.06)	-0.13** (0.04)	-0.105* (0.04)	-0.19** (0.02)
Boat	-1.55** (0.01)	-2.73** (0.32)	-3.26** (0.27)	-2.99** (0.30)	-2.64** (0.40)
θ	0.24** (0.01)				
ρ	0.18** (0.01)				
σ_{δ}		0.22 (0.18)		0.49** (0.16)	1.18** (0.26)
$\sigma_{\tau\{1,2\}}$		1.22** (0.22)		1.37** (0.16)	
$\sigma_{\tau\{1,3\}}$		1.73** (0.24)		1.73** (0.17)	
$\sigma_{\tau\{1,4\}}$		2.24** (0.21)		2.39** (0.17)	3.22** (0.12)
$\sigma_{\tau\{2,3\}}$		2.64** (0.19)		2.48** (0.18)	3.22** (0.12)
$\sigma_{\tau\{2,4\}}$		1.10** (0.19)		1.45** (0.14)	
$\sigma_{\tau\{3,4\}}$		1.34** (0.18)		0.88** (0.14)	
σ_{γ}		0.73** (0.13)	2.44** (0.10)		
Log-likelihood	-8229	-5667	-5700	-5677	-5952

Table 3: Seasonal Welfare Gains

	20% Reduction in Toxins		Loss of South Lake Michigan	
	Calculation A[a]	Calculation B[b]	Calculation A	Calculation B
RNL	$22.30	$22.30	-$322.50	-$322.50
RXL Unconstrained	$40.72	$6.37	-$637.07	-$132.70
RXL – A: No Nesting	$45.67	$9.26	-$751.33	-$200.66
RXL – B: Limited Correlation	$35.99	$6.84	-$710.64	-$160.06
RXL – C: Limited Correlation & RNL Nesting	$41.10	$4.64	-$398.20	-$50.69

[a]Mean welfare estimates calculated using repeated draws from estimated parameter distributions.
[b]Mean welfare estimates calculated using means of estimated parameter distributions.

8. FOOTNOTES

[1] The linked model was originally developed by Bockstael, Hanemann, and Strand [4] and Bockstael, Hanemann and Kling [1], with subsequent modifications and applications by Hausman, Leonard, and McFadden [11], Feather, Hellerstein, and Tomasi [9], and Parsons and Kealy [21], among others. See Herriges, Kling and Phaneuf [13] for further discusion.

[2] See Herriges, Kling and Phaneuf [13] and Phaneuf, Kling and Herriges [23] for recent applications to the recreation demand literature.

[3] There have been numerous applications of mixed logit model appearing in the literature as of late, including Ben-Akiv, Bolduc and Bradley [1], Bhat [2], Brownstone and Train [7], Revelt and Train [26], and Train [28,29].

[4] For the sake of brevity, the exact forms of the choice probabilities in equation (7) are not reported here, but are available from the authors upon request.

[5] Descriptions of simulation methods for use with the mixed logit model can be found in [7], [17], and [29], among others. Gauss code incorporating these simulation procedures into a program to estimate mixed logit models (developed by Kenneth Train, David Revelt, and Paul Ruud at the University of California, Berkeley) can be found on Train's home page at http://elsa.berkeley.edu/~train. A modified version of Train's code used in the application section of this paper (for faster estimation in repeated choice situations when characteristics of individuals do not change over time) is available from the authors upon request.

[6] It is important to note, however, that despite the use of i.i.d. extreme value variates for the error term ε_{ijt} ,the basic RXL model does not suffer from the "independence of irrelevant alternatives assumption" that plagues standard logit models. See Train [28,29].

[7] It should be emphasized that throughout this paper when we speak of correlation it is from the perspective of the analyst. As noted above, for the consumer all of the error components are assumed to be known and hence $\widetilde{U}_{ijt}$ is nonstochastic.

[8] See Train [29, pp. 126-128] for additional discussion regarding the interpretation of mixed logit as an error-components model.

[9] Correlation across choice occasions can also be imposed by restricting $\tilde{\beta}_{it} = \tilde{\beta}_{i}\ \forall t$, as suggested, e.g., in [26], [28], and [29].

[10] The exact equations for the choice probabilities are left to an appendix, available from the authors upon request.

[11] Specifically, for a given individual and choice occasion, the choice probabilities $\{P_{i0t}, P_{i1t}, \ldots, P_{i4t}\}$ can be viewed as dividing the unit interval into 5 segments. A uniform random number generator was then used to select one of these segments and, hence, a specific choice alternative.

[12] The estimation itself was carried out using Gauss and the cross-sectional mixed logit code developed by Train, Revelt, and Ruud (See footnote 5). 100 replications were used in simulating the unconditional choice probabilities. The τ_{it}^{jk}'s and δ_{it} were treated as normally distributed in the mixed logit model, with their means constrained to be zero. The marginal utility of income coefficient β was treated as fixed.

[13] The likelihood ratio test statistic is χ_4^2=5.09 with a P-value of 0.27.

[14] While it is tempting to expect the former percent to be larger and the latter percentage to be smaller, it should be kept in mind that the RXL model provides only an *analogue* to the RNL, mimicking the desired correlation pattern. However, the shape of the underlying distributions are different.

[15] Details of the survey procedures and samples are provided in Lyke [15] and Phaneuf [22].

[16] β_1 denotes the marginal utility of income. Since the conditional utility functions are assumed to be linear in income, the household's base income level becomes irrelevant and is dropped for convenience.

[17] This nesting structure was chosen based upon prior studies using the same data, including [13] and [24].

[18] While the parameters β_k can be specified as random in the mixed logit framework, we have chosen to leave them as nonstochastic in this analysis so as to focus on the cross-site and cross-choice occasion correlations.

[19] This concern about the interpretation of the error term and its effect on welfare calculations is analogous to the concerns raised by Bockstael and Strand [5] in the continuous demand system setting. Previous applications of the mixed logit model in recreation demand (e.g. [28]) have in essence employed the measurement error interpretation in computing welfare effects.

[20] Furthermore, it is what implicitly underlies the CV calculations in equations (39-41) for RNL models.