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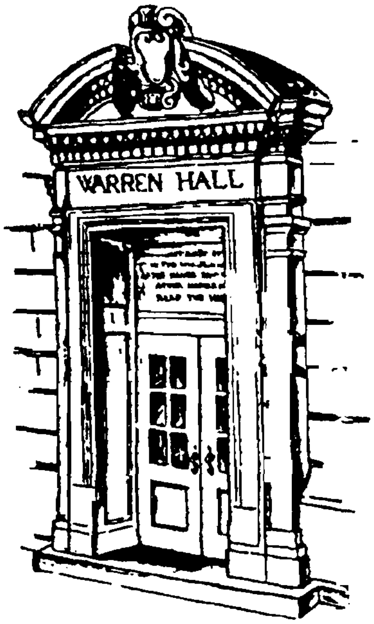

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Valuation of Groundwater Quality: Contingent Values, Public Policy Needs, and Damage Functions

by

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Abstract: In a departure from past contingent valuation research of groundwater quality, this paper estimates a damage function for nitrate exposures based on actual water test results of individual wells. From the perspective of reliability, it is argued that such a full information approach more closely represents the goal of valuation research in this area — to estimate the economic values that people would place on improving water quality if they were actually experiencing contaminated water. The adoption of a damage function approach linking willingness to pay to actual exposures is also more useful to policy makers at the study site because it potentially provides benefit information to a broad range of policy options. Finally, because the damage function is based on objective data that could be obtained from other sources such as local well test programs, such an approach may be desirable from a benefits transfer perspective. Damages, as measured by willingness to pay for protecting individual well supplies within a 10 mg/L NO_3-N health standards are estimated to be a concave function of nitrate exposure levels.

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Valuation of Groundwater Quality:
Contingent Values, Public Policy Needs, and Damage Functions

1. Introduction:

In recent years the need for valuation of groundwater resources has been identified as a critical national research and policy issue [USEPA 1990; National Research Council, 1997]. Corresponding to this need, there has been intensive research effort in the last decade to estimate contingent values for groundwater quality [Edwards, 1988; Schultz and Lindsay, 1990; Powell, 1991; Caudill and Hoehn, 1992; McClelland *et al.*, 1992; Sun *et al.* 1992; Jordan and Elnagheeb, 1993; Poe, 1993; Sparco, 1995; Barrett *et al.* 1996; Delavan, 1996; Randall and deZoysa, 1996; Crutchfield *et al.*, 1997]. Over much of the same period, a renewed interest in assessing the accuracy of benefits transfers has emerged [USEPA, 1993; Loomis, 1992; Downing and Ozuna, 1996; Kirchhoff *et al.*, 1997], with some attention paid specifically to water quality [Vandenberg *et al.,* 1995; Crutchfield, 1995; Crutchfield *et al.*, 1997].

This paper argues that there is an inherent incompatibility between groundwater contingent valuation research as it has developed in the last decade, and groundwater management policy and benefits transfer needs. Past contingent valuation groundwater research has provided important, policy relevant information to decision makers. Yet the objective hypothetical exposure ("Suppose your home tap water is contaminated by nitrates to a level that exceeds the EPA's minimum standard by 50%", Crutchfield *et al.*) and the subjective risk ("How safe do you feel about your household drinking water supply?", Powell) approaches utilized in past research are not directly amenable to the variety of policy outcomes needed to be considered by water managers in studied sites.

Moreover, even though values across groundwater studies and sites have been shown to vary in a systematic manner [Boyle *et al.,* 1994], the value information provided by the original studies precludes transfers to unstudied sites unless fairly restrictive assumptions about identical nature of preferences, perceptions, and exposure levels are made. An alternative to meeting these policy needs would be to reorient groundwater contingent valuation research so that the focus is on actual, objectively obtainable, exposure levels experienced at a study site. Towards this goal, this paper provides the results from a groundwater contingent valuation study that tested individual wells for nitrates, and then solicited WTP values for a groundwater protection program.

The organization of the paper is as follows. Section 2 expands on the arguments introduced in the previous paragraph. The third and fourth sections provide a summary of a contingent valuation study of willingness to pay for a rural well water program that maintains nitrate levels within government standards of 10 mg/1 NO_3-N. The critical difference between this and previous CV research is that the values are directly linked to actual exposures as measured by nitrate test results in the studied wells, allowing the estimation of a damage function consistent with theoretical, management and policy needs. The final section discusses the implications of this research.

2. Limitations of Past Groundwater Quality Contingent Valuation (CV) Research:

Since the publication of the Edwards study, a body of CV research has emerged for valuing improvements in groundwater quality. These studies can be categorized by how the valuation "scenarios" are structured[1]. One group follows Edwards' lead by specifying an "objective

[1] The McClelland *et al.* study deviates from other research by focusing on quantity shortfalls associated with shutting down contaminated sources, and is thus not included in this categorization.

hypothetical" initial exposure condition and an alternative hypothetical improvement [Jordan and Elnagheeb; Sparco; Crutchfield *et al.*; Delavan]. Other studies have allowed respondents to specify their own "subjective" probabilities of exceeding health standards in a specific time frame [Sun *et al.*; Poe] or perception of current safety levels [Powell], with the target being the reduction of the probability of exceeding standards to zero or the improvement of water quality to safe levels. Still other research has respondents value broadly defined groundwater protection programs and policies [Schultz and Lindsay; Caudill and Hoehn; Randall and deZoysa; Barrett *et al.*]. While providing useful information about willingness to pay for hypothetical programs, each of these approaches has limitations from a valuation, management, or policy perspective. These limitations are discussed here.

The first issue is how well this entire body of groundwater valuation literature represents willingness to pay if the households' water were indeed contaminated and the respondents were actually faced with decisions about public intervention and averting opportunities. To make such decisions, individuals need an "adequate" amount of information [Arrow *et al.*, 1993; Fischoff and Furby, 1988]. Information gathering has opportunity costs, and individuals may ration scarce information gathering resources by choosing to ignore information that is not relevant to current choices [Bishop and Welsh, 1992]. Such rationing appears to be the norm for specific environmental risks. For example, in a baseline study of radon, about 25 percent of respondents were unable to answer whether their current household exposure was serious or not serious or some level in between [Smith *et al.*, 1990]. With respect to groundwater quality, two water testing studies indicate that most households are unsure about their nitrate exposure levels relative to health standards, and that about 40 percent of rural residents who rely on their own wells are unable to attach a safety level to their water supplies [Poe *et al.*, 1996]. This evidence strongly implies that reliance on subjective perceptions of exposure and health risks may not provide a reliable reference point for valuing a protection policy. People simply do not have well-formed reference conditions, and thus it is unlikely that values collected under these conditions would reliably predict WTP values for a population actually experiencing contamination. The alternative approach of providing participants with an "objective hypothetical" exposure levels also has limitations. Both the radon and groundwater literature indicates that individuals do update their risk perceptions, and consequently their WTP for protection, with new information. Importantly, they also place weight on their prior perceptions in assessing new information about risks -- even when these priors are erroneous. Given this evidence of updating, it is not known how a household that believes their water to be safe reacts to being asked to assume that their water violates government health standards (or vice versa). At issue is whether adding a hypothetical reference exposure level is meaningful: Do households actually experiencing contaminated water at a given level react similarly to households that are asked to assume that they are experiencing contamination at the same level? At this point in time this question remains unanswered by the CV literature, representing a plausible but yet unquantified bias.

Beyond the reliability of individual values, there is a need to design research so as to provide critical information to groundwater managers and policy makers. A recent National Research Council panel notes that what is most relevant for decision making regarding groundwater pollution policies or management is knowledge of the how economic values will be affected by a decision affecting levels of contamination. This policy perspective reflects, in part, the theoretical requisites for identifying optimal groundwater pollution policies for groundwater, which rest on the notion of damage functions across nitrate exposure levels [e.g. Conrad and Olson, 1992]. Conceptually, it also reflects the necessary information for evaluating the welfare effects of alternative land use practices on the distributions of pollutants [Boisvert, Schmidt, and Regmi, 1997; Wu and Babcock, 1995; Lichtenberg and Zilberman, 1988]. What mangers need in order to meet these policy and managerial issues is information that would allow them to compare the benefits and costs associated with a range of alternative shifts in exposure distributions. To a large extent, past research has been fairly limited with respect to providing such a range of information. Research into specific policies or specific changes in exposures provide little information beyond those specific changes, and thus, has limited relevance to managers interested in exploring a range of alternative programs. The coarse percentages (0, 25, 50, 75, and 100) utilized in much of the "objective hypothetical" and "subjective" groundwater research also do not facilitate such comparisons: for example, given a health standard of 10 mg/l NO_3-N, how is a move from 9 to 7 mg/l or from 15 to 10 mg/l NO_3-N to be evaluated? More generally, how are shifts in entire distributions to be assessed? Clearly, for management purposes a damage function approach linking actual exposures to values would be useful for linking social benefits to the control of pollutants.

A second area of policy need is benefits transfers. Following Boyle and Bergstrom [1992] and Desvousges *et al.* [1992], benefit transfers can be defined in the groundwater context as the transfer of existing benefit estimates from an original study site to a change in exposure at an unstudied policy site. The need for such transfers is motivated by relatively high cost and time considerations of conducting original research at the policy site. One way to minimize costs of transfers would be to limit the covariates used in statistical analyses of willingness to pay functions to those that might be readily obtainable from prior research at the policy site: demographic and socio-economic variables (e.g., age, household composition, and income) used in estimating WTP functions could be limited to those corresponding to census records; distributions of groundwater contaminants might be available from hydrologic research in the area [e.g. Portage County Groundwater Plan, 1987; Baker, 1990]. Obviously, studies in which the original research focuses on a localized site-specific issue or policy option will not be likely candidates for benefits transfers. The objective hypothetical or subjectively defined probability also has limited value from a benefits transfer perspective. Given that past research has not linked these values to actual exposure levels, transferring these values to an unstudied site poses a difficulty without conducting a second survey at the study site to determine the range of distributions of probabilities exceeding standards.

In all, from the perspective of obtaining informed values that reflect the best interests of individual decision makers actually experiencing contamination, the need to provide policy makers with valuation data to explore a range of management decisions and the need to conduct benefits transfers, it is argued here that groundwater valuation studies should be based on actual exposures levels and informed respondents. The remainder of this paper describes the first groundwater CV research to be based on actual exposure and to provide a fully informed damage function amenable to local management decisions and benefit transfers.

3. Conceptual Framework

Groundwater valuation of quality changes can be depicted in a standard option price framework [Boyle *et al.*] in which uncertainty is expressed over health states. With respect to nitrates (N) found in well water, the consumer's choice problem can be characterized by the minimization of the planned expenditure function [Smith, 1986]:

$$\dot{e}(g(h;N),p,\overline{EU}) = \min_{X} p'X \quad \text{subject to } EU = \overline{EU} \tag{1}$$

where: $\dot{e}(\cdot)$ is the planned expenditure function; g(h;N)) is the subjective distribution of health outcomes (h) for a given nitrate exposure levels N; p is the corresponding state-independent vector of prices for all goods (X) including the explicit or implicit prices for substitute water sources, and $\overline{EU}$ is the reference level of expected utility. *Ex ante* willingness to pay (i.e., before the health risk is resolved) for a groundwater protection program that shifts the exposure distribution from N to N' is given by the difference in the planned expenditure function with the project and the planned expenditure function without the project:

$$WTP_{N'|N} = \dot{e}(g(h;N),p,\overline{EU}) - \dot{e}(g(h;N'),p,\overline{EU}) \tag{2}$$

More typically however, groundwater protection projects are defined, as in this research, in terms of truncating the nitrate distribution f(N′) at some health standard or threshold (T). For example, most nitrate studies to date [e.g., Sun *et al.;* Crutchfield *et al.*] have formulated the target nitrate level in terms of a zero probability of exceeding standards. In this case, the willingness to pay is given by:

$$WTP_{T|N} = \dot{e}(g(h;N),p,\overline{EU}) - \dot{e}(g(h;\int_0^T f(N')dN),p,\overline{EU}) \tag{3}$$

where f(N′) depicts a distribution of exposures given the project. Using this expenditure difference, a damage function relative to the threshold level could be obtained from cross-sectional data with varying initial exposure levels. To isolate effects of moving along a damage function, Equation (2) could be approximated by the differencing of Equation (3) across initial nitrate levels under the assumption that the truncated distribution f(N′) is independent of the initial level of exposure and that health risk perceptions across nitrate levels are independent of reference nitrate levels[23].

4. Survey Implementation:

The groundwater survey was conducted for private wells in Portage County, Wisconsin, an area known to have a wide range of nitrate distributions based on previous hydrologic research and water testing programs. Prior water testing indicated that approximately 18 percent of the private wells exceed the government health standards of 10 mg/l NO_3-N designed to protect infants from *methemoglobinemia*.

In order to test individual wells and obtain values based on well test results, a two-stage survey design was created. In the first stage (Stage 1), individual households received the following survey package: a cover letter; a Wisconsin State Laboratory of Hygiene mailable nitrate test kit; instructions for collecting a water sample for nitrates; a question and answer sheet providing further information about the study; a business reply return envelope; and an initial survey about respondent socio-demographic characteristics, prior knowledge of groundwater, and safety perceptions. This

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[2] Both these assumptions may be questionable. For example, it is likely that an individual whose nitrate well test is 2 mg/l will likely have a different perception of f(N') truncated at T = 10 mg/l NO3-N than an individual whose reference nitrate level is 20 mg/l. Similarly, prospect reference theory [Viscusi, 1989] suggests that individuals will formulate perceptions of health risk based on their exposure level. Nevertheless, given that the magnitude of possible biases is not known, it is argued that the willingness to pay values for a shift in distributions as suggested in the text could be used as a rough approximation to evaluate incremental shifts in reference nitrate levels.

[3] In specifying Equation (3), it is of course recognized that, even with the nitrate test results, the reference conditions may also be characterized by a distribution of exposures, say F(N). Previous research, suggests that nitrate levels in individual wells may fluctuate over time [e.g. Baker]. Adding such a redefinition would not change the essence of the analysis -- it merely suggests that the single test approach adopted in this research lies somewhere along the continuum of uninformed to fully informed. Similarly, it is possible to regard f(N') as a normalized distribution wherein the observed CDF F(N′) is adjusted to reflect the mass at the truncation point.

survey also contained a subjective/uninformed CV question about a 10 mg/l groundwater protection program. Water samples from the Stage 1 respondents were tested for NO_3-N at the Wisconsin State Laboratory of Hygiene. These results were returned to the Stage 1 respondents in a second survey package, which also contained a nitrate information sheet, a Stage 2 CV questionnaire, and a stamped first class return envelope. The information sheets were based on information readily available at the local extension offices and other State and County agencies, and included background information on sources of nitrates, health effects of nitrates, and a listing of possible averting opportunities available to individuals.

The contents of the survey received design input from other CV practitioners and were evaluated in three individual in-person debriefing sessions. The two stage survey design was pre-tested on 20 Portage County households. Based on these pre-tests and other inputs, only minor wording changes were made in the final questionnaire.

Implementation of the survey followed Dillman's total design method [Dillman, 1978], employing an initial survey package, a thank you/reminder post card to all respondents, and a follow-up survey package to those who had failed to reply to the initial survey package. [4] No financial incentives were provided, but participants were informed that the free nitrate test had a $9.00 value.

A zip-code based sample list was obtained from Americalist, and cross checked with local plat books to isolate residences not connected to public water supplies. The survey was initially sent to 480 addresses in rural areas of Portage County that did not have public water supplies. After

[4] For Stage 1, in place of Dillman's suggested registered mail third follow-up, telephone contacts were made with survey recipients whose telephone numbers could be identified. A third mailing was sent to those contacted who indicated on the telephone that they would consider completing a questionnaire.

accounting for bad addresses and addresses outside of the desired area (n = 47), the adjusted Stage 1 response rate was approximately 77% (n = 332). The conditional response rate for the Stage 2 survey was about 83% (n = 275). Each of these individual response rates exceeded the present CV standard of 70%, and the combined response rate across the two stages was about 64%. Even though the 64% response rate reflects non-participation across both survey stages, this ratio still lies at the upper end of the range of single stage groundwater valuation studies [Jordan and Elnagheeb (35%); Barrett *et al.* (45%); Powell (50%); Randall and deZoysa (51%); Sun *et al.* (51%); Schultz and Lindsay (58%); McClelland *et al.* (60%); and Hoehn (66%); Edwards (78.5%)].[5]

Nitrate test results reflected prior water testing results for Portage County. In this study about 16 percent of the wells exceeded government standards of 10 mg/l, with the highest values being 43 mg/l. This corresponds closely with the 18% figure obtained from previous sampling in the area. About 28 percent fell below the highest natural levels of 2 mg/l. The majority of respondents, about 56 percent, had some evidence of human impact on nitrate levels but did not exceed government standards. Thus, a wide range of exposure levels was available to serve as input for a damage function[6].

The two stage questionnaire complicates discussion of the flow of the survey. The first stage was constructed as a standard stand-alone CV questionnaire, obtaining information about personal

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[5] Heckman type selection tests were conducted across stages. Nitrate test levels, demographic and socio-economic variables were included in a probit analysis across stages. Only the age of the respondent (+) and bottled water users (-) were significant factors in explaining whether a Stage 1 respondent completed a Stage 2 questionnaire. However, inverse Mills ratios derived from this analysis were not a significant explanatory variable in estimating Stage 2 willingness to pay response functions, and are, thus, not included in the econometric analysis below.

[6] About 10 percent of the respondents had levels less than measurable (i.e. < 0.15 mg/l) by the techniques used by the Wisconsin State Laboratory of Hygiene. These were excluded from the econometric analyses because they had a special sticker manually placed in their surveys indicating that it was not possible to improve their water quality.

perceptions of groundwater exposure and health risks, eliciting other background information on respondents' environmental concerns, eliciting a yes/no response to a dichotomous choice CV question for a 10 mg/l standard based on pre-existing, "subjective/ uninformed" values, and then obtaining socio-economic descriptors. The second stage questionnaire focused instead on personal impressions of their individual water test results and, given that information, the relative safety of their water. Individual averting options were discussed and a community-wide program was presented as an alternative to individual protection. Following a reminder that taxpayers, individuals, and farmers already pay for groundwater protection through government programs, higher prices, and lower profits, the following program was proposed:

- *With the groundwater protection program, nitrate levels in all Portage County wells will definitely be kept below the government health standard of 10 mg/l. In some areas this may be difficult, but suppose that it would be possible.*
- *Without such a groundwater protection program, present trends in nitrate levels in Portage County will continue and the number of wells with nitrate levels higher than the government standard will increase in Portage County in the next five years.*

Respondents were subsequently asked to vote in a "subjective/informed" or "fully informed" manner on the program with the following dichotomous choice contingent valuation question:

> *Would you vote for the groundwater protection program described above if the total annual cost to your household (in increased taxes, lower profits, higher costs, and higher prices) were $________ each year beginning now and for as long as you live in Portage County (CIRCLE ONE NUMBER)*
>
> *1. No*
> *2. Yes*

Dollar values were individually inscribed and ranged from $1 to $999. The range and distribution of these bid values were based on information obtained from the Stage 1 survey responses.

5. Econometric Methods

Corresponding to the expenditure approach described in Equations (1) to (3), estimation of the WTP function follows the expenditure difference random utility model initially described by Cameron [1988, 1991; McConnell, 1990]. In this framework, the possibility of a 'yes' response to the dichotomous choice bid value 'A' is given as:

$$\pi(yes) = \pi(\mathrm{WTP}_{\mathrm{TIN}} + \varepsilon \geq \mathrm{A}) \tag{4}$$

where the error term is assumed to have a zero mean. $\mathrm{WTP}_{\mathrm{TIN}}$ is unobserved but indicated by the 1/0, yes/no response to the dichotomous choice question. Assuming a logistic distribution for ε the following relation provides a first step in recovering an estimated $\mathrm{WTP}_{\mathrm{TIN}}$ function:

$$\pi(yes) = (1 + \exp(-(\alpha + \beta \mathrm{A} + \underline{\gamma}\underline{Z})))^{-1} \tag{5}$$

where $\underline{Z}$ is a vector including a function of nitrate levels and demographic characteristics of the respondent, and α,β, and $\underline{\gamma}$ are coefficients to be estimated. Estimated $\mathrm{WTP}_{\mathrm{TIN}}$ for an individual can be recovered by the following transformation:

$$\mathrm{WTP}_{\mathrm{TIN}} = \frac{\alpha}{\beta} + \frac{\underline{\gamma}}{\beta}\underline{Z} \tag{6}$$

Derivation of standard errors for the ratios of coefficients follows the standard logistic estimation procedures detailed in Cameron's 1992 article.

In the statistical analyses that follow, $\underline{Z}$ will be defined to consist of two components. The first component contains of socio-demographic variables of the type that could be linked to census type data for benefits transfers. These covariates, and their expected correlation with WTP, are: the age (-) and gender (?) of the respondent; presence of children less than 4 years of age in the household (+); involvement in farming (-); education level (+); and household income (+). These variables are further defined in Table 1. Expectations of the sign of the estimated coefficients were taken from other CV research on valuing risks.[7]

Importantly, from the perspective of this paper, $\underline{Z}$ also includes a nitrate exposure variable, for which the derivation of the conditional WTP is the objective of this research. Two approaches to characterizing exposure levels are evaluated. The first corresponds with the "subjective/informed" probability of exceeding standards approach. Immediately preceding the valuation question, the following question about exposures was posed:

> *Without such a groundwater protection program, do you expect that your own well will have more nitrates than the government standard of 10 mg/l during the next five years? If you are not sure, please give us your best guess. (CIRCLE ONE NUMBER)*
>
> 1. ***Yes, my well already has more nitrates than the 10 mg/l standard and I expect it to remain above the standard.***
> 2. ***Yes, definitely (100 percent chance)***
> 3. ***Probably (75 percent chance)***
> 4. ***Maybe (50 percent chance)***
> 5. ***Probably not (25 percent chance)***
> 6. ***No, definitely not (0 percent chance)***

Responses to this question were recoded according to their probability of exceeding standards to form the covariate Pr(NO3-N > 10 mg/l) with a range between 0 and 1 in 0.25 increments. The expectation is that the coefficient on this variable would be positive, reflecting the well established result that people with a higher perceived likelihood of exposure will have a greater WTP for

[7] See Poe and Bishop (1997) for a more detailed discussion of these variables.

protection. Given this formulation, there is no direct link to exposure levels. Although, as discussed below, such a relationship might be obtained by linking expectations to exposures in a secondary analysis.

The second approach instead focuses on establishing a direct damage function relationship between WTP responses for the 10 mg/l protection program and nitrate levels. Little prior empirical evidence exists about the shape of this function. All else equal, we would expect that people with low reference exposures would have low WTP for a protection project, while households with high exposures would have a relatively high WTP for such a project. However, when linking WTP directly to exposures, concern must be given to the convexity of the damages between these two extremes. On one hand, the standard value of life literature would suggest a convex damage function [Jones-Lee, 1974]. However, when substitutes or defensive expenditures such as bottled water are included as decision options, the damage function may become non-convex [Burrows, 1995]. In all, convexity of the damage function is an empirical question [Shogren and Crocker, 1991; Quiggen 1992]. Ignoring for the moment all other elements of $\underline{Z}$, convexity is investigated by assuming the following reformulation of Equation 6:

$$WTP_{TIN} = \frac{\alpha}{\beta} + \frac{\gamma_N}{\beta} N^{\tau} \qquad (7)$$

In this specification, $\tau>1$ implies a convex damage function and $\tau<1$ corresponds to concavity. In the analyses that follow, an optimal τ is determined by a grid search with the objective of minimizing the likelihood function. Once determined, τ is fixed and the remaining coefficients are estimated using standard logistic maximum likelihood techniques.

6. Results

The results of the estimation process are summarized in Table 2. The first column of the table defines the coefficient or the variables to be estimated. The second column provides the mean values and standard deviations for relevant variables. The third through fifth column reports coefficients and estimated summary statistics for maximum likelihood estimates corresponding to Equations (4) through (6). Different columns in this set correspond to different specifications of $\underline{Z}$. In the first specification, the Pr(NO3-N >10 mg/l) is the only variable in $\underline{Z}$. The second specification expands the definition to include all the socio-economic variables except income. The third specification includes income as an element of $\underline{Z}$, at the cost of losing about 10 percent of the observations. The final three columns of Table 2 report the model demand function defined by Equation 7 for the same sequence of covariates.

Within each specification of the nitrate variable, the three formulations of $\underline{Z}$ exhibit similar trends. Coefficients on the nitrate variables are highly significant, with appropriate signs in all specifications. In the estimates excluding INCOME, the coefficients on OWNAGE and DCGRAD are negative and positive respectively, as expected. The other coefficients are not significant. When INCOME is included, all the coefficients for the remaining non-nitrate covariates become insignificant. This suggests that estimation of a WTP function will be dominated by income and the level of exposure. Should this result be supported by future research, benefits transfers might be accomplished by relatively simple models of income and exposure.

Although both specifications are significant at the 1% level, a comparison of the informed subjective probability models with the corresponding nitrate exposure model indicates that the former provides a better statistical estimate of WTP: the variance of the WTP estimate (given by κ)

is smaller, the χ^2 goodness of fit statistic is higher, and the percent of responses correctly predicted is higher. Thus, if the sole objective is goodness of fit, then the subjective/informed approach based on the likelihood of exceeding standards would dominate.

However, as discussed in Section 2, such an approach is limited by its indirect linkage to nitrate levels. From the perspective of local management policies and the potential for benefits transfers, it is policy useful to have WTP estimated as a function of nitrate levels. Such a direct estimate is provided in the last three columns of Table 2. In this analysis, $\frac{\gamma_N}{\beta}N^K$ indicates an increasing concave function of nitrates. That is, $WTP_{T|N}$ rises with N but in a decreasing manner. Given the grid search approach adopted here, direct statistical tests of concavity cannot be performed. However, support for this conclusion is found by bootstrapping the data set and identifying an optimal τ for each bootstrap sample. Using this approach, 87 of 100 bootstrap estimations provided τ values of less than 1.

Figure 1 provides a graphical depiction of this damage function based on the simple model $\underline{Z}$ model in Column 6 of Table 2. As depicted, the direct nitrate exposure model provides a concave function that levels off at higher reference exposure levels. Such a result is consistent with opportunities for substitutes (part of the information packet provided with the Stage 2 survey). Taking averages of expectations about the probability of exceeding standards across ranges of nitrate levels provides point estimates at various levels of a derived damage function. In contrast to the concave damage function, the estimated damages rise relatively slowly across low levels of NO_3-N contamination, jump sharply as reference exposures cross 10 mg/l, and then level off as the expectations of exceeding approach 100 percent. The resulting damage mapping suggests an 'S' shaped function of damages, wherein a convex function corresponding to standard value of life hypothesis occurs across lower values, but the WTP values are eventually truncated from above.

7. Discussion:

This paper suggests that CV research on groundwater quality and other environmental risks adopt a paradigm that WTP values should be based on actual exposure levels. Arguments underlying such a proposal center on the reliability of individual WTP responses as well as the need to provide land use and groundwater managers and policy makers with valuation data that can be linked to a range of decisions. Such an approach would also provide more flexible input for benefits transfers.

Towards this objective, this paper provides the results from the first CV survey of groundwater nitrate contamination to be based on actual exposure levels experienced by respondents. Willingness to pay for a program to protect groundwater at a 10 mg/l NO3-N standard was obtained from respondents who had been informed of their households' nitrate test results. Adopting an expenditure difference approach, a damage function was estimated linking WTP to actual exposure levels. In analyzing the dichotomous choice response, a relatively simple functional form for nitrates was estimated within a logistic framework, resulting in convex damages. An indirect approach, obtained by first estimating WTP as a function of subjective probabilities of exceeding standards and then linking these probabilities to exposure levels, suggests a damage function with convexities and concavities. Nevertheless, in contrast to standard presentations of damages, both approaches suggest that WTP eventually levels off. Such a result is consistent with opportunities for substitution. Examination of more sophisticated functional forms remains a critical area of future research.

In arguing that a fully informed approach should serve as the paradigm for future research, it is recognized that testing water quality may be expensive, perhaps prohibitively so in some situations. Nevertheless, it is incumbent upon researchers, policy makers, and funding agencies to recognize that values based on partial information will provide limited, and perhaps biased, information to decision makers. The benefits of obtaining values from a fully informed sample are likely to be more than marginal, and thus merit consideration in future policy relevant research.

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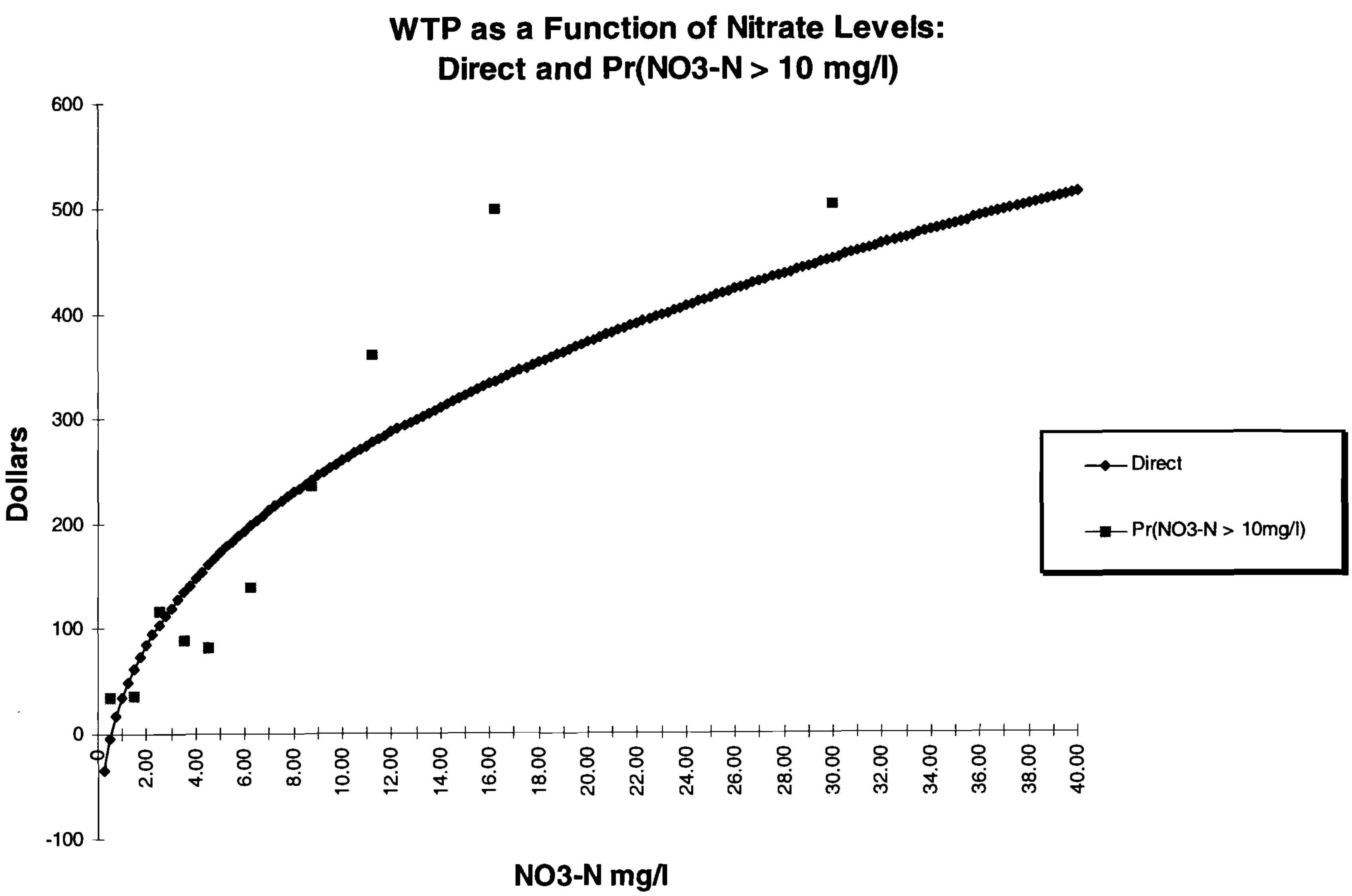
Table 1: Description of the Covariates for The Econometric Analysis

Variable	Description	Sign Expectation
OWNAGE	Categorical Variable for Years of Age: 1= less than 18; 2 = 18 to 44; 3=45 to 64; 4 = 65 or older.	-
DGENDER	Binary variable for gender of respondent: 0= male; 1= female.	?
DAGE<4	Binary variable for young children < 4 years of age in household: 0=no; 1=yes.	+
DFARM	Binary variable for involvement in farming:: 0=no; 1= yes.	-
DCGRAD	Binary variable for college grad: 0=no; 1=yes.	+
INCOME	Categorical variable for total household income before taxes: 1= < \$10,000; 2=\$10,000 to \$19,999; 3=\$20,000 to 29,999...10=\$90,000 to 100,000; 11= >\$100,000	+
P(NO3-N> 10 mg/l)	Probabilistic categorical variable: 0, 0.25, 0.50, 0.75, and 1.00 probability of exceeding standards.	+
N	Nitrate Level (NO3-N) in mg/l, continuous from 0.15 mg/l.	+

Table 2: Subjective Probability of Exceeding Standards and Nitrate Exposure Models[a]

Variable	Mean [b] (N= 185)	Subjective Probability Model[c]			Nitrate Exposure Model[c]		
1	2	3	4	5	6	7	8
Constant	1 [0.00]	-102.30 (91.02)	166.51 (203.26)	-284.29 (232.56)	-146.63 (168.99)	237.37 (246.55)	-206.94 (271.27)
OWNAGE	2.67 [2.76]		-108.81 (63.87)*	-21.67 (59.86)		-147.58 (68.52)*	-46.94 (61.16)
DGENDER	0.38 [0.48]		-45.77 (88.95)	46.46 (83.53)		-14.33 (95.28)	43.15 (87.45)
DAGE<4	0.18 [0.39]		-35.22 (114.46)	-8.93 (106.04)		-44.46 (126.58)	8.57 (112.25)
DFARM	0.20 [0.40]		103.67 (104.24)	117.22 (98.69)		61.31 (113.13)	117.14 (104.43)
DCGRAD	0.25 [0.43]		289.54 (106.60)***	99.08 (104.18)		293.70 (115.24)**	109.18 (107.02)
INCOME	4.07 [2.07]			58.49 (22.90)**			55.77 (24.20)**
Prob (>10)	0.45 [0.35]	618.07 (166.96)**	729.38 (195.75)***	528.51 (138.62)***			
τ					0.352	0.353	0.346
N^{τ}					180.75 (89.34)**	146.69 (85.18)*	131.11 (77.92)*
κ^{e}	265.71[d] [276.76]	282.79 (56.36)***	253.91 (48.17)***	213.85 (39.73)***	309.39 (65.03)***	287.24 (58.16)***	234.91 (45.64)***
Obs.	185	210	210	185	210	210	185
χ^2		53.08***	67.85***	72.96***	36.12***	49.42***	55.67
Perc. Pred.		77	73	78	69	73	72

a. ***, **, * denote 1, 5 and 10 percent significance levels, respectively.
b. Numbers in [ ] are standard deviations.
c. Numbers in ( ) are asymptotic standard errors.
d. Mean and standard deviation for the dichotomous choice bid value.
e. $\kappa = 1/\beta$ following Cameron.



WTP as a Function of Nitrate Levels:
Direct and Pr(NO3-N > 10 mg/l)
Dollars
600
500
400
300
200
100
0
-100
0
2.00
4.00
6.00
8.00
10.00
12.00
14.00
16.00
18.00
20.00
22.00
24.00
26.00
28.00
30.00
32.00
34.00
36.00
38.00
40.00
NO3-N mg/l
Direct
Pr(NO3-N > 10mg/l)

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