United States
Department of Agriculture

Economic
Research
Service

Agricultural
Economic
Report
Number 701

The Benefits of Protecting Rural Water Quality

An Empirical Analysis

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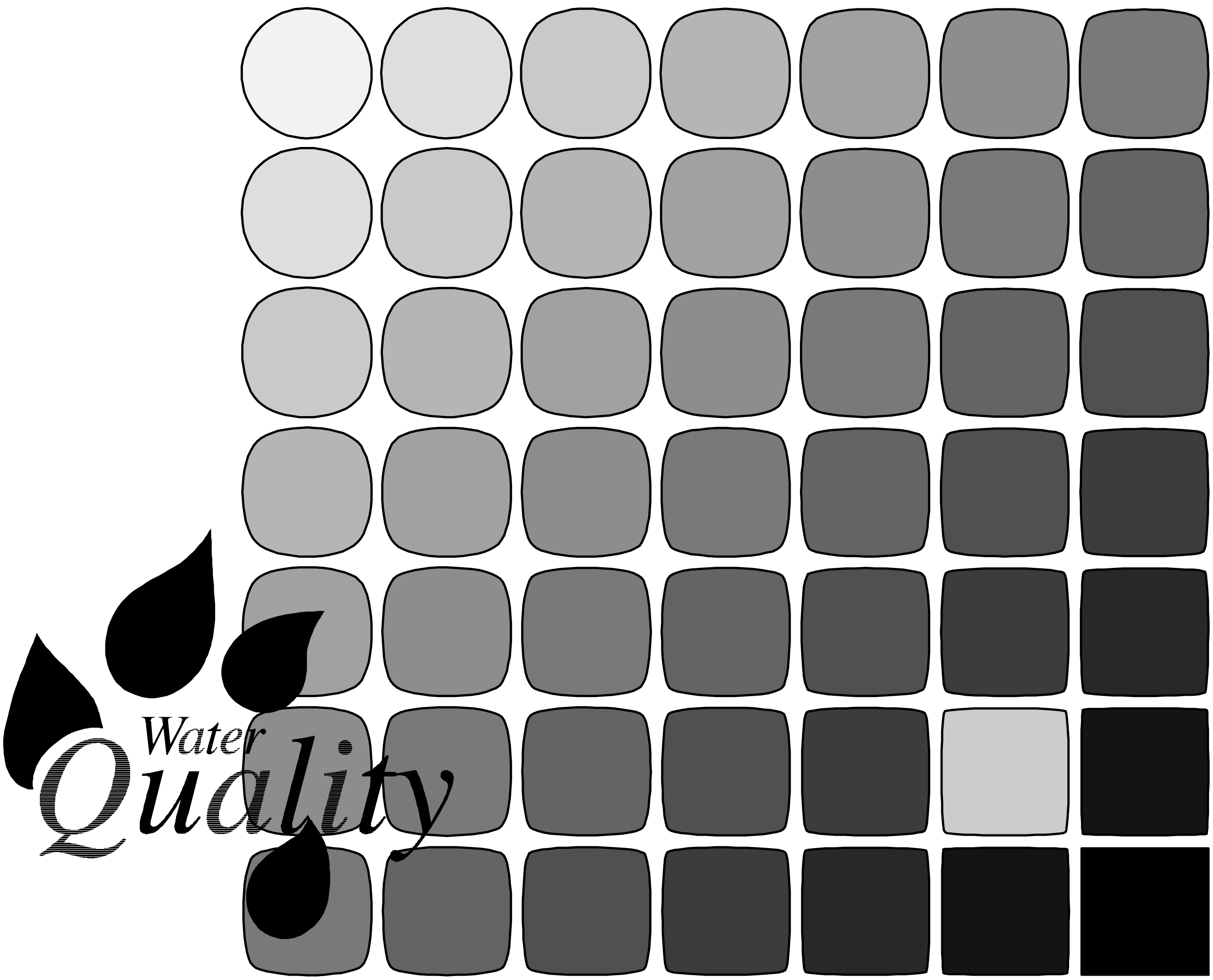

The Benefits of Protecting Rural Water Quality: An Empirical Analysis. By Stephen R. Crutchfield, Peter M. Feather, and Daniel R. Hellerstein. Natural Resources and Environment Division, Economic Research Service, U.S. Department of Agriculture. Agricultural Economic Report No. 701

Abstract

Concerns about the impact of farm production on the quality of the Nation's drinking and recreational water resources have risen over the past 10 years. Because point sources of pollution were controlled first, agricultural nonpoint sources have become the Nation's largest remaining single water-quality problem. Both public and private costs of policies that address the conflict between agricultural production and water quality are relevant, but measuring the off-farm benefits and costs of changing water quality is difficult. Many of the values placed on these resources are not measured in traditional ways through market prices. This report explores the use of nonmarket valuation methods to estimate the benefits of protecting or improving rural water quality from agricultural sources of pollution. Two case studies show how these valuation methods can be used to include water-quality benefits estimates in economic analyses of specific policies to prevent or reduce water pollution.

Keywords: Water quality, nonpoint source pollution, environmental quality, agricultural production, costs, benefits

1301 New York Avenue, NW.
Washington, DC 20005-4788

January 1995

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Summary

Concerns about the potential impact of farm production on the quality of the Nation's drinking and recreational water resources have risen over the past 10 years. Agricultural sources are now the largest single contributor to the Nation's surface water quality problem, and there is evidence that some ground water supplies may be vulnerable to leaching chemicals in agricultural areas. This report explores the use of nonmarket valuation methods, such as travel cost to a recreational lake, to estimate the benefits of improving or protecting rural water quality from agricultural sources of pollution.

Food and fiber production can impair surface and ground water resources. Fertilizers and pesticides used to grow crops may leach through soils and contaminate ground water supplies. Dissolved chemicals in drinking water may then pose a human health risk. Runoff of chemicals from sediment and crop-land, as well as soil erosion, may impair the quality of streams, lakes, rivers, and wetlands. Most early efforts to protect water quality were directed at municipal and industrial sources of pollution, where a single pollutant source could be identified (point-source pollution). The cumulative effect of more than 20 years of investment in such point-source pollution control is that nonpoint-source pollution, particularly from agricultural sources, has become the largest single remaining water-quality problem in the Nation.

Both public and private costs are relevant in resolving conflicts between agriculture and water quality. When making production decisions, farmers balance their expected production costs with expected returns from crops produced. However, farmers' decisions may have unintended long-range effects.

Economic losses from impaired water quality reflect, in part, how important the resources are to society. One case study is used to illustrate the relationship between agricultural production and the costs of impaired surface water quality. Changes in farm production practices may lead to changes in the quality of nearby lakes, affecting recreation activities. A case study of lakes in Minnesota shows the economic benefits of reducing soil erosion and improving lake clarity. Another case study shows the regional benefits of protecting ground water from agricultural chemicals. Using survey data from USDA's Area Studies Program, estimates of willingness to pay for ground water protection are developed for four specific regions.

Some of the approaches that can be applied to valuing water resources are discussed, and a historical review of previous studies shows how the procedures and methods for valuing water-quality benefits have evolved in recent years. Estimated water-quality benefits associated with policies and programs that prevent pollution can be used to more comprehensively assess the overall benefits and costs of farm policies.

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Introduction

Over the past 10 years, concern has risen about farm production adversely affecting the Nation's water quality. Considerable Federal and State resources have been committed to reducing agricultural sources of water pollution, such as sediment and nutrients in runoff and leaching chemical residuals. Agricultural sources now form the largest single contributor to the Nation's surface water quality problem, and there is evidence that some ground water supplies may be vulnerable to leaching chemicals in agricultural areas (Crutchfield, Hansen, and Ribaudo, 1993).

It is important that policies to improve water quality be designed to account for all costs and benefits of such policies in order to make the most effective use of scarce resources. The costs of agricultural policies that are intended to reduce or prevent degradation of surface water or ground water supplies may be readily estimated using conventional micro- and macroeconomic models of farm production. The benefits of improved water quality, however, are more difficult to assess. Since much of the benefit from improved water quality are environmental services not sold in conventional markets, valuation techniques that do not rely on market prices must be used to estimate these benefits.

This report explores the use of nonmarket valuation methods to estimate the benefits of improving or protecting water quality from agricultural sources of pollution. Some of the valuation methodologies that can be applied to water resources are discussed, emphasizing their practical application to the issues of valuing recreational uses of surface water bodies and the value of preventing ground water contamination. Our objective is to highlight some of the practical considerations that influence the choices analysts must make when applying theoretical models of resource valuation to real-world situations. We review some of the key studies on the costs of water pollution to illustrate how these valuation techniques have been applied in the past. Two case studies are used to show how some of the available valuation methods can be applied and how some of the tradeoffs and compromises that may be necessary to adapt these tools to available data. We close with a brief discussion of the implications of our findings and of future research and data needs.

Policy Setting: Agricultural and Water-Quality Conflicts

Food and fiber production can impair surface and ground water resources. Fertilizers and pesticides used to grow crops may leach through soils and contaminate ground water supplies. Dissolved chemicals in drinking water may then pose a human health risk. Runoff of chemicals from sediment and cropland, as well as soil erosion, may impair the quality of streams, lakes, rivers, and wetlands. Most early efforts to protect water quality were directed at municipal and industrial sources of pollution, where a single pollutant source could be identified. The cumulative effect of more than 20 years of investment in point-source pollution control is that nonpoint-source pollution, particularly from agricultural sources, is the largest single remaining water-quality problem in the Nation (U.S. EPA, 1992).

Both private and public costs must be considered before agricultural and water-quality conflicts can be resolved. When making production decisions, farmers balance their expected private costs of production options, including tillage practices and chemical use, with returns from crops produced. However, farmers' decisions may have unintended long-range effects. Consumers of water resources or other environmental services, such as recreation, may bear the costs when agricultural runoff, sediment, or farm chemicals degrade the quality of these resources. Though the public may place a value on these lost services, this value is not fully reflected in private costs farmers pay for farm inputs or

Table 1--Types of benefits from improving rural water quality

Benefit class	Benefit category	Examples
Use value	Instream services	Recreational uses, such as swimming, boating, and fishing. Commercial/municipal uses, such as fishing, navigation, and water storage facilities.
	Consumptive services	Drinking water from municipal water systems and private wells. Irrigation and other agricultural uses.
	Aesthetic value	Near-water recreation, such as picnicking and sightseeing. Property value enhancement.
	Ecosystem value	Preservation of wildlife habitat and promotion of ecosystem diversity.
Nonuse value	Vicarious consumption	Value placed on enhanced use of clean water by others.
	Option value	Desire to preserve opportunity to enjoy clean water at some future time.
	Stewardship value	Protection of environmental quality and desire to improve water quality for future generations.

in farmers' cost/benefit calculations in deciding how to produce a crop.

Economic losses from impaired environmental quality reflect, in part, the value of the services the resources provide. Table 1 shows the different kinds of benefits society derives from improving water quality. Typically, economists characterize the values placed by society on environmental services as *use value*, where a natural resource is directly consumed or used by individuals, and *nonuse value*, where individuals may place a value on the current or potential existence of an environmental service, even though they may not directly use or consume it. Use values for clean water include recreational and commercial uses of lakes, rivers, and streams; consumptive services such as drinking water and irrigation; and aesthetic and ecosystem values, where water resources indirectly contribute to the overall well-being of society. Nonuse values are less tangible; they reflect more subjective preferences individuals may hold about water resources, such as a desire to leave clean water to future generations or a stewardship ethic which places a value on the existence of clean water resources, apart from any actual use of these resources.

Ideally, economic analysis of agricultural production should include the economic value of off-farm water-quality effects. This value provides a more complete picture of the contribution of the agricultural sector to the Nation's economy by accounting for these off-farm social costs. It also allows for a more complete policy analysis by accounting for more of the costs and benefits that may accrue to specific policy choices, including agricultural and environmental policy legislation. However, values placed on environmental quality changes cannot be measured by market prices. Instead, we must use nonmarket techniques for valuing changes in water quality. The next section briefly discusses some of these methods.

Valuation of the Benefits of Improving Water Quality

Benefits of improved quality are defined by the difference between a person's well-being, or utility, before environmental quality changes and that person's level of utility after the change in environmental quality. That is, the value of a change in water quality is the amount of income a person would be willing to pay that would leave them indifferent between the original situation (dirty water) and the new state (clean water).[1]

[1] Formally, the benefits of changes in water quality are given by the compensating surplus and are measured by the area under the compensated demand curve for environmental quality. In cases where an environmental change is foregone (such as preventing water pollution), the appropriate measure is the equivalent surplus. This is the amount of income required to move an individual to the level of welfare that would have been achieved had environmental quality changed. (For a more detailed exposition and explanation of technical terms, see Ribaudo and Hellerstein, 1992.)

We cannot directly obtain demand functions for goods that are not traded in markets. Estimating the benefits or costs of changes in water quality involves using analytic techniques that are intended to elicit the values people place on clean water. Many different estimation procedures have been used to develop valuation functions for changes in environmental quality. These procedures are comprised of two different approaches. The first approach relies on indirect methods, where choices individuals make when using or consuming water resources are examined to obtain a measure of how these services are valued. The second approach uses "structured conversations" to directly elicit the values the respondent places on these services (Smith, 1993).

Indirect approaches are based on the premise that the values people place on goods and services are revealed by the choices they make in purchasing or consuming them. Under certain assumptions, these values can be retrieved using information on consumer choices about marketed goods and services that are complementary to the nonmarketed good or resource service in question.[2] The most commonly used indirect approach to valuing changes in water quality is the travel cost model and its many variants, where people's expenditure to enjoy recreational uses of water (as influenced by changes in water quality) is used to value those uses. Other revealed-preference approaches have been used to value water-quality changes, including averting expenditures models (where the value of clean drinking water is measured by expenditures on substitutes such as bottled water) and hedonic property analysis (where the variation in property values across sites with differing water quality provides a measure of the value of clean water).

Direct approaches to valuing water-quality benefits identify values people place on water quality from survey responses. The most widely used technique is the contingent valuation method (CVM), where respondents are presented with information about water quality and relationships between water quality and usability of the resource. They are then asked to tell the researcher how much a given change in water quality would be worth to them. Numerous examples of using CVM to value water quality changes are available, although they primarily deal with valuing recreational use of surface water resources. The methodology has been subject to considerable controversy. Some analysts argue that asking people hypothetical questions only gives you hypothetical answers that cannot be meaningfully used to value environmental quality changes. Generally, though, the profession has given the technique qualified acceptance (see Smith, 1993; Ribaudo and Hellerstein, 1992; Cooper, 1994; and Arrow and others, 1993).

Another approach to valuing water-quality changes does not involve estimation of benefits but instead uses benefits estimates derived in one location to value water-quality changes in another. This procedure, termed benefits transfer, makes determinations about economic value or tradeoffs in one context using information (price elasticities, demand parameters, and so on) obtained in another. For example, if an analyst were asked to assess the benefits of preventing ground water pollution in a particular setting, the analyst could commission a new study (averting expenditures or CVM). However, doing so would take time and money. As an alternative, if a valuation study had been done in another area with similar characteristics (demographics, scale, and extent of environmental problems) the analyst could transfer the benefits estimates from the original site to obtain benefits measures at the new site.

The process of benefits transfer introduces another layer of uncertainty and imprecision because measurement error implicit in the original case study may be compounded when applying benefits measures (per household willingness to pay) or valuation functions (travel cost or CVM equations) in the new situation. The appeal of this process, though, is that it allows the analyst to obtain some insight into the magnitude of environmental benefits and costs without the time and expense required for a new, original study.

Table 2 summarizes available valuation methodologies. While a complete review of the existing literature of water pollution benefits is beyond the scope of this report, it is instructive to look at a few of the key studies of water-quality benefits, especially as they relate to agricultural sources. Doing so serves two purposes: first, it provides a context within which we can judge the significance of agricultural sources of water pollution. Second, we can see how benefit estimation procedures have been refined over the years, thus giving us guidance as to how we can continue to improve our techniques for quantifying and valuing the benefits of protecting rural water resources from agriculture-related impairments.

The Value of Clean Water: A Historical Appraisal

Economists have conducted numerous studies of the value of water quality over the years. For example,

[2]These restrictions have to do with weak complementarity of the marketed goods with environmental quality. (See Ribaudo and Hellerstein, 1992.)

Table 2--Alternative approaches to water-quality benefits estimation

Valuation approach	Valuation procedure	Examples
Indirect, revealed preference	Travel cost	Ribuado and Piper, 1991; Smith and Desvousges, 1985; Caulkins, Bishop, and Bouwes, 1986.
	Hedonic travel cost	Mendelsohn, 1987.
	Zonal travel cost	Brown and Navas, 1973.
	Discrete choice	Parsons and Kealy, 1992; Bockstael, Hanneman, and Strand, 1984.
	Averting expenditures	Nielsen and Lee, 1987; Spofford, Krupnick, and Wood, 1989; and Abdalla, Roach, and Epp, 1992.
	Hedonic property values	Young and Teti, 1984, Mendelsohn and others, 1982; Michaels, 1993; McConnell, 1990.
Direct estimation	Open-ended contingent valuation	d'Arge, 1985; Mitchell and Carson, 1984; Sutherland and Walsh, 1985.
	Dichotomous choice CVM	Gramlich, 1977; Loomis, 1987.
Benefits transfer	Transfer per unit benefits estimates to new site	Ribaudo, 1986, and 1989.
	Transfer valuation equations	Loomis, 1992.

one recent publication identified 287 separate studies on the value of recreation, over half of which dealt with some form of water-based recreation (Walsh, Johnson, and McKean, 1992). A database of environmental benefits studies developed by the U.S. Environmental Protection Agency (EPA) identifies several hundred studies of water-quality benefits. Most of these studies, however, were for specific sites or “local” water-quality issues (river basins or lakes), and are of limited use in evaluating the national benefits of changes in water-quality policies. Relatively few studies have presented a comprehensive look at the costs of water pollution and the benefits of pollution reduction on a nationwide scale.

One of the first comprehensive assessments of the benefits of pollution control was published by Freeman (1982) (table 3). Drawing on a number of secondary studies of the costs of water pollution, he estimated four types of benefits associated with removal of water pollutants: recreational benefits, nonuser benefits, commercial fishing, and consumptive uses. He estimated the total benefits to be between $3.8 and $18.4 billion (1978 dollars), with a ‘most likely point estimate’ of $9.4 billion. The largest single category of benefits was recreation with a point estimate of $4.6 billion. The recreation benefits were largely drawn from travel cost studies. Other benefits were derived from a synthesis of various nonmarket benefit studies, including averting expenditures and surveys of willingness to pay.

Russell and Vaughan (1982), citing the importance of recreation benefits in the totality of water-quality benefits, used a travel cost model to estimate the national economic benefits to recreational fisheries of controlling water pollution. A key feature of this study was a participation model, which allowed for increased rates of participation as water quality improved. That is, improving the quality of lakes and streams could encourage existing fishermen to fish more days per year as well as attracting new participants. This participation model was linked to a travel cost model for measuring per fisherman benefits of improved water quality. The estimated total benefits of cleaner lakes and streams was between $300 and $966 million (1982 dollars), depending upon the amount of pollution abatement.

Neither the Freeman nor the Russell and Vaughan studies addressed water-quality problems related to agricultural sources of pollution. Clark, Haverkamp, and Chapman (1985) identified and quantified damages associated with soil erosion, particularly from cropland. They took existing studies, particularly the Vaughan and Russell and Freeman estimates, and prorated them to account for the amount of pollution thought to be related to soil erosion from all sources and from cropland. They

Table 3--Evolution of water quality benefits estimation

Study/year	Scope	Approach	Findings
Freeman (1982)	National benefits of water pollution control	Synthesis of existing studies, predominately travel cost and recreation participation models.	Total damages to recreational water uses from all forms of pollution: $1.8-$8.7 billion, "best guess" of $4.6 billion (1978 dollars per year).
Russell and Vaughan (1982)	National recreational fishing benefits from controlling water pollution	Linked travel cost valuation and recreation participation models with predictions about changes in water quality due to pollution control.	Total benefits of $300-$966 million, depending on level of pollution control instituted.
Clark (1985)	National damages from soil erosion from cropland	Prorated existing damage measures (particularly Vaughan and Russell) to reflect cropland's share of total erosion.	Damages to all uses: $3.2-$13 billion, 'best guess" of $6.1 billion (1980 dollars). Cropland's share of erosion-related damages: $2.2 billion.
Ribuado (1986)	Regional and national estimates of the benefits of reducing soil erosion	Disaggregated Clark's estimates among farm production regions. Linked benefits estimates with regional changes in water quality by reducing soil erosion.	1983 soil conservation programs, which reduced soil erosion, implied $340 million in offsite benefits. Benefits per ton of erosion reduced were from $0.28 to $1.50.
Nielsen and Lee (1987)	National estimates of the costs of ground water contamination	Averting expenditures estimates of the costs for monitoring for nitrates and pesticides in drinking water supplies, and the costs of providing alternative clean drinking water supplies.	Monitoring costs for presence of agricultural chemicals put at $890 million-$2.2 billion for private wells, and $14 million for public wells.
Ribaudo (1989)	Regional and national estimates of the water quality benefits from the Conservation Reserve Program	Added a fishing participation model to more directly tie off-farm benefits to erosion-related changes in water quality, which allowed for improved water quality to affect both scale and intensity of recreation.	Reducing erosion via retirement of 40-45 million acres of highly erodible cropland would generate from $3.5-$4 billion in water quality benefits.
Carson and Mitchell (1993)	National benefits of freshwater pollution control	Based on a nationwide contingent valuation survey. Respondents asked to value incremental changes in water quality (such as improvement from 'boatable' to 'swimmable').	Annual household willingness to pay for maximum water quality improvement of $205-$279 per household per year.

reported the total economic cost from impairments of surface waters related to soil erosion to be around $6.1 billion (1980 dollars), with cropland's share of erosion-related damages amounting to $2.2 billion.

The Clark, Haverkamp, and Chapman (referred to as the Clark study) estimates only identified the total damages from soil erosion. While revealing, they did not address the related issue of the marginal benefits of *reducing* these damages by reducing erosion. Ribaudo (1986) used the Clark study estimates as a starting point in a study of the benefits of reducing soil erosion. He disaggregated the total damage estimates by farm production region and created estimates of water-quality benefits by linking these damage estimates with regional water-quality changes induced by reducing soil erosion. The estimated off-farm benefits of soil conservation programs in place in 1983 were put at $340 million. A subsequent study by Ribaudo in 1989 added on a travel cost and recreation participation model to improve the recreational fishing components of his water-quality benefits estimates. A key finding of Ribaudo's work was that the off-farm damages of soil erosion varied regionally, from $0.57 per ton of erosion in the Northern Plains to over $7 per ton in the Northeast. This finding implies that the economic efficiency of conservation programs could be improved by directing erosion control programs to regions where the off-farm water-quality benefits were greatest.

A shortcoming of all of the studies mentioned so far was that, for the most part, they relied on indirect measures of water-quality benefits, such as travel cost-based recreational values or averting expenditures or costs of remediation. As such, they did not reflect the value placed by individuals on clean water (or incremental improvements in water quality) when those individuals do not directly use the water resources. In 1993, Carson and Mitchell published what is currently the only comprehensive, nationwide estimate of the benefits of freshwater pollution control which is based on direct estimation of water-quality benefits. In a contingent valuation survey, over 800 respondents were asked to indicate their willingness to pay for various levels of water-quality improvements. They conclude that the national benefits of improving surface water quality from a baseline of 'nonboatable' to 'swimmable' quality to be about $29 billion per year (1990 dollars), or about $240 per household. ('Boatable' and 'swimmable' are standards used by the EPA and the States to measure water quality, and represent the recommended safe uses of rivers, lakes, and streams.) A direct comparison of this finding with early estimates would be questionable, since they apply different methodologies to measure different types of benefits at different points in time.

All of these studies were directed at the economic dimensions of surface water pollution. Few studies have measured the costs of ground water pollution from agricultural sources. The only existing study that takes a nationwide perspective was done by Nielsen and Lee in 1987. Since valuation of the costs of exposure to potentially toxic substances represents such a difficult challenge (as will be discussed below), Nielsen and Lee took a more modest approach, choosing to measure the avoidance and averting expenditures costs of ground water contamination. Using a simple screening approach to identify the potential extent of ground water vulnerability to leaching farm chemicals, they then defined a partial measure of the costs of potential contamination as the amount of money necessary to test for the presence of nitrates and chemicals in those vulnerable areas. They placed these one-time monitoring costs at between $890 million and $2.2 billion. Although it was the first comprehensive attempt at measuring the costs of ground water contamination, it was not based on any direct or indirect assessment of individual preferences; as Abdalla, Roach, and Epp (1992) point out, averting expenditures measures typically understate the true willingness to pay to prevent exposure to environmental pollutants. On the other hand, later information developed by the EPA indicates that the extent of actual contamination of drinking water supplies by agricultural chemicals may be significantly less than the extent of vulnerability (U.S. EPA, 1990).[3]

Practical Considerations for Estimating Rural Water-Quality Benefits

Three general conclusions may be drawn from this brief look at the water-quality benefits literature. First, economists are using increasingly sophisticated analytic techniques to estimate the economic consequences of changes in environmental quality. The various forms of the travel cost and contingent valuation models bring a richness and rigor to the analysis that were not available 10 or 15 years ago. Second, a broader class of water-quality benefits can now be estimated; use of contingent valuation and other "structured conversation" techniques may enable us to at least qualitatively assess nonuse values placed on improved or protected water quality. Third, there has been increased attention in recent years to the need to make the linkages between policy changes, environmental outcomes, and the economic benefits and costs of those policies.

What implication does this hold for our task of valuing changes in rural water quality? The greatest economic benefit to be found from improved water quality is instream uses, primarily recreation. In principle, any one of the nonmarket valuation techniques described in table 2 could be used to value changes in rural surface water quality associated with changing agricultural practices. While hedonic methods and contingent valuation hold promise, travel cost methods are most frequently used in recreation valuation. Cropper and Oates (1992) provide a good overview of the uses of travel cost models. The technical development of these methods can be found in Ribaudo and Hellerstein (1992), Mendelsohn (1987), and Cooper (1994).

We also need to account for the relationships between farm production choices and environmental outcomes. Despite the significance of Clark's earlier work, we need to move beyond simple assumptions that agricultural contributions to water degradation can be defined by a percentage share of cropland erosion to total erosion. Integration of models that make the connection between changes in farm practices and changes in water quality with our economic valuation models is necessary if

[3]Nielsen and Lee applied their monitoring cost estimates to the number of private and public wells in potentially contaminated areas, measuring roughly one-third of the counties in the United States. However, the EPA study of actual contamination found that while nearly half of the wells surveyed had detectable levels of nitrates, only about 1.2 percent of public wells and 2.4 percent of private wells had nitrates in excess of recommended levels. Only 10 percent of public wells and 4 percent of private wells had detectable levels of pesticides. Thus, the Nielsen and Lee study probably overestimates the costs of monitoring, at least for pesticides.

these valuation models are to successfully evaluate the costs and benefits of agricultural and environmental policies.

However, the task of connecting changes in agricultural practices to changes in water quality is made difficult by the likelihood that data to estimate complex physical and economic valuation models may be sparse. Information on the impact of agricultural activities on rural water quality may be limited. In addition, fundamental information about the quality of water bodies visited by recreationists and on the quality of potential substitutes may also be in short supply. Short of a massive data collection effort, two methods can be used to deal with these problems. The first involves aggregation, and the second involves two-stage models.

Aggregation involves combining information on individuals or on sites into zonal aggregates. Aggregating possible recreation sites will decrease the number of choices. This allows the analyst to use larger-scale indicators of land use and environmental quality instead of site-specific information. Similarly, aggregating individuals will increase the number of visits to each site (or zone), allowing the use of larger-scale census measures as explanatory variables.

Two-stage models incorporate the use of detailed data on a selected sample of sites in conjunction with more general data on land use and environmental quality to predict quality at unmeasured sites. In fact, a reduced-form model, which directly incorporates land use data, can sometimes be used to predict site visitation. With predicted site quality and information on site visitation (either individual data or aggregates of individuals), a quality-incorporating model can be estimated. (In a later section of this report, we illustrate how such a two-stage model can be applied using limited data.)

With respect to ground water, the available literature upon which to draw conclusions is much thinner than it is for surface water benefits. Despite the concern about potential ground water contamination from agricultural chemicals, the benefits of preventing ground water contamination are not well known.[4] In part, this is because many of the valuation studies that have been conducted have examined people's willingness to pay to avoid exposure to highly toxic substances present in high concentrations. For example, the benefits of ground water protection have been estimated in the context of landfills, toxic waste dumps, and leaking underground storage tanks. In those cases, the threat to human health is fairly evident and can be easily understood by the respondent in a contingent valuation setting. However, in the case of agricultural chemicals in ground water, the risks of exposure are less well known because in many instances the level of contamination is below the level thought to pose an immediate health risk to humans. The Nielsen and Lee study, while providing useful insight on the scope and extent of possible costs, did not directly measure the participants' willingness to pay to prevent ground water contamination. More work needs to be done to develop comprehensive estimates of the value of protecting ground water resources.

In summary, both the travel cost and contingent valuation approaches promise to refine and extend our knowledge of the benefits of protecting rural water quality from agriculture-related impairments. Benefits transfer offers the promise of developing better aggregate measures of water-quality benefit without the expense and time requirements of nationwide studies. To show how these approaches might work, we illustrate the use of two valuation methodologies to estimate water-quality benefits in the remaining sections. First, we apply a variation of the travel cost model to estimate the benefits of improving surface water quality by modifying agricultural nonpoint source pollution in rural areas. Next, we use CVM estimates of the benefits of protecting ground water from chemical contamination in a benefits transfer approach to obtain estimates of the value of protecting ground water from agricultural chemicals. Our objective is twofold: first, to illustrate some of the practical problems involved in estimating the benefits of reducing agriculture-related water-quality impairments; second, to show how water-quality benefits estimates can be used in policy analysis to evaluate the tradeoffs between agricultural production and environmental quality.

Case Study: Rural Water-quality Benefits in Minnesota

The value of rural water-based recreation flows from a number of different sources, including the use of such freshwater resources as lakes, streams, and wetlands for hunting and fishing, swimming, and nature viewing. Intangible aspects of a healthy (or degraded) ecosystem may be important as well. To some extent, the enjoyment of these activities is predicated on the cleanliness of the water. Thus, to estimate the value of clean water, it is also necessary to examine how the recreational use of rural water bodies varies under different levels of water quality. The quality of rural water, in turn, is largely dependent on the agricultural practices adopted on surrounding lands. As different

[4]Ground water is difficult or costly to clean up once contaminated by chemicals. Most economic valuation studies have therefore focused on the value of preventing contamination from occurring or preventing exposure to potentially hazardous substances rather than on the value of improving the quality of the resource itself.

agricultural practices induce different degrees of water quality, the value to society of these water bodies will vary.

This section provides an illustrative example of benefits estimation for changes in rural quality. A demand model for water-based rural recreation (angling) is developed, estimated, and used to assess the economic benefits associated with that recreational activity. A relationship between agriculture and water quality is estimated, which is used to explain changes in water quality associated with changes in erosion on cropland. Together, the recreation demand model and the erosion-water-quality linkages allows us to value the off-site benefits from changes in water quality associated with reduction in cropland erosion.

Our objective here is not just to estimate water-quality benefits, but also to illustrate the application of, and problems with, benefits estimation procedures in an agricultural context. The model best serves as an example to build on rather than a definitive expression of the state of the art in benefits estimation. In particular, the benefits estimates we obtain are for illustrative purposes; any generalization of the quantitative results to different region or broader scale should be done with caution. Since we link changes in resource use to changes in water quality and use values which are particular to one region, extrapolation of our results to other regions without accounting for different resource conditions is not recommended. This case study illustrates the type of analysis that can be done, how models may need to be adapted to fit available data, and points to additional data needs and other approaches that may work better.

This model is dictated by the nature of the available data. This is often a problem resource economists face. Recreational data are collected in ways that are not designed for the purpose of valuing the impacts on rural recreation benefits. In addition, we are often faced with the difficult task of making explicit the linkages between onfarm practices and off-farm water-quality impacts. Such fate-and-transport relationships can be estimated at a site-specific level using models, but aggregate relationships between agricultural practices and water quality must often be based on limited resource data. Since the limitations of the data in our case study are illustrative, we describe them first before we discuss implementation of the models.

Data and Sources

Ideally, the information needed to estimate a water-based recreation demand model includes:

1. The agricultural conditions affecting water quality;
2. Survey data describing the location and intensity of recreational activities; and
3. Water-quality data.

Specifically, detailed information is needed to describe the water quality and the physical conditions affecting water quality near the destinations that are visited.

The recreational activity considered in this section is lake-based angling trips in the rural, agriculturally intensive sections of southern and western Minnesota (areas with 50 percent or more land in agricultural use). Agricultural data are available on a county basis, while water-quality and recreational information are available for individual lake basins. This is a weakness in the data — one source is micro while the other is aggregated. Nonetheless, the aggregated agricultural data and the micro lake quality data must be used to provide a link between agricultural practices and water quality.

A survey of lake-based angling activity in Minnesota conducted by the Minnesota Center for Survey Research was used to measure the demand for fishing locations. Two measures of participation were considered in the survey: locations of “long” trips requiring over 30 minutes of travel time and the number of “short” trips taken to a “favorite” location that is within 30 minutes travel time from the respondent’s residence. Because the location of the majority of the long trips are to counties with little agriculture, the short trip information is used to estimate the model. Excluding counties where less than 50 percent of the land use is agriculture and discarding individuals who do not participate left 178 observations. The average intensity of participation was 12 trips throughout the year with a range of 1-99 trips.

A lake water-quality data set was constructed using information from the Minnesota Department of Natural Resources (MDNR) and the Minnesota Pollution Control Agency (MPCA). The combined MPCA and MDNR data sets contain information describing approximately 3,500 individual lakes. Both data sets contain numerous missing values and a large variance in observations per lake. Acreage is known for almost all lakes, but other quality measures are known for only a small percentage of the lakes in the State. In light of this, only two additional physical measures are used to describe the quality of the fishing locations: lake depth and secci disk depth measurements (SDM) — a commonly used measure of water clarity. SDM observations collected during the “open water season” (June 24-September 11) over 1985-89 were averaged for each lake to describe the expected lake water clarity. The “open water season” has been used by the MPCA in previous water-quality analysis because "summer data are preferred for assessment purposes as they generally correspond to the maximum productivity of the lake, yield the best agreement between trophic variables, and reflect the period of maximum use of the resource" (Heiskary, Wilson, and Larsen, 1987, pp. 5).

In what follows, we take a two-step approach to capturing the linkage between agricultural production choices, water quality, and the benefits of water-based recreation. First, we specify a simple relationship between agricultural practices around Minnesota lakes and the quality of those lakes. We do this because it is assumed that the recreationist's decision to visit any particular recreation site depends, in part, on the quality of the water at that site. Next, we specify a simple recreational demand model to explain choices made by visitors to Minnesota lakes, with quality included as one explanatory variable in this demand model. This, then, enables us to determine how changing agricultural practices will indirectly affect water quality, recreation choices, and economic benefits from improved water quality.

Water Quality-Agricultural Characteristics Links

In order to capture this relationship, we constructed a simple model relating agricultural activities to lake clarity. The link between water quality and agricultural characteristics is based on the assumption that the SDM (water clarity) of a lake is affected by the extent and type of agricultural activity surrounding it. Changing agricultural practices will change water clarity, which is assumed to be valued by anglers using these lakes.

Ideally, we would like to model this relationship using information about the characteristics of land in close proximity to each lake. This would allow us to accurately specify a relationship between agriculture and water quality. Unfortunately, the data are limited to county-level agricultural measures. The only reasonable way to proceed given these less than ideal data is to regress individual lake SDM observations on county agricultural observations.

Our estimated secci disk relationship is:[5]

$$SDM = f(ER, \%AG, \%FERT, CORN\ DUM, SOY\ DUM) \quad (1)$$

where

SDM = Secci disk measurement,
f(·) = a tobit functional form,
ER = cropland erosion: average tons/acre/year per county,
%AG = percentage of total county area used for agriculture,
%FERT = percentage of total county cropland receiving fertilizer, and
CORN DUM, SOY DUM = dummy variables,each equal to 1 if corn or soybeans is the predominant crop in the county.

Estimation results appear in table 4. For purposes of comparison, the model is estimated using all data and using data from counties where 50 percent or more of the land use is agricultural. In general, the variables have the expected sign and are significantly different from zero. Two types of agricultural descriptors are included in the equation. The first type describes physical features, such as the percentage of county land used in agriculture or the predominant crop grown in the county. The second type represents agricultural conditions that may be influenced by policies that, for example, change erosion rates or areas fertilized. These variables can be used to estimate nonmarket costs or benefits resulting from changes in agricultural practices affecting water quality.

Table 4--Water clarity estimation[1]

	All land[3]		50 percent agricultural land[4]	
Variable[2]	Parameter estimate	T-value	Parameter estimate	T-value
Constant	3.1409	59.90	4.2047	13.40
Erosion	-0.0815	-2.00	-0.1329	-13.19
%AG	0.2921	1.22	-1.5960	-3.33
%FERT	-1.5773	-3.81	-0.7947	-1.69
CORN DUM	-0.8074	-6.80	-0.6718	-5.40
SOY DUM	-1.3775	-5.91	-1.0097	-4.72
SIGMA	1.3589	57.70	1.1579	35.30

[1]Estimation results of secci disk (water clarity) regressed against agricultural variables. Secci disk is measured in meters and is specific to an individual lake. The agricultural variables are observed on a county basis.
[2]Constant is the constant term. Erosion is average tons per acre of farmland erosion per county. %AG is the percentage of total county area used for agriculture. %FERT is the percentage of total county farmland that receives fertilizer applications. CORN DUM equals one if corn is the predominant crop grown in the county; zero otherwise. SOY DUM equals one if soybeans are the predominant crop grown in the county; zero otherwise. SIGMA is the variance parameter associated with the tobit model.
[3]Results using all (87) counties in Minnesota; number of observations is 1,667.
[4]Results using counties (32) in Minnesota where 50 percent or more of the land area is for agricultural production; number of observations is 624.

[5]Because SDM takes on strictly positive values, a truncated tobit model is used to estimate the water clarity/agricultural practices model (Maddala, 1983). This estimation method results in positive predicted values of SDM which will be used to estimate the recreational demand model.

Recreational Demand Model

Having established a link between land use and water quality, we now need to establish the link between water quality and recreational choices. As previously discussed, the data available to estimate the model are limited, allowing only a very rudimentary travel cost model to be estimated. The model is specified as:

$$T_i = f(P_i, Y, Q_i, E_i, S), \qquad (2)$$

where

- T_i = The number of trips the i-th person took to their "favorite" fishing location
- $f(\cdot)$ = a linear function,
- P_i = the travel cost to the location, computed as the round-trip distance from the center of the individual's zip code zone to the lake times $0.305 (the American Automobile Association's estimate of the average mileage cost of driving a midsize automobile),
- Y = income,
- Q_i = a vector of quality variables describing the location: lake area in acres, lake depth in meters, and predicted water clarity as indicted by the secci disk depth measurements (SDM),
- E_i = a vector of socioeconomic variables: the respondent's age and sex, and
- S = a measure of substitutes: the number of lakes within 20 miles of the individual's residence excluding the lake visited.

The model follows conventional specification for the travel cost method found under indirect revealed-preference methods in table 1. It is hypothesized that lake clarity is one determinant of overall recreation demand. We make the linkage between agricultural activity, water quality, and recreational demand by treating lake clarity as a separate variable, dependent upon agricultural practices.

Estimation of Recreation Demand Function

Having estimated the water clarity model, we then use it in the recreational demand equation. Water clarity is predicted at the observed quality level and then used as an independent variable in a travel cost equation. To value the effects of a change in agricultural practices, levels of water clarity resulting from the change are predicted and substituted into the travel cost equation. The resulting welfare measure is the difference in consumer's surplus before and after the change.

Results of the estimation appear in table 5. The equation suggests that younger, predominantly male individuals tend to be participants. Lakes that are close to the respondents' residence, larger, and shallower tend to be visited more often. Linear and quadratic predicted SDM terms are used to capture the nonlinear relationship between water clarity and trophic status (Heiskary, Wilson, and Larsen, 1987). The parameter associated with NUM20, the number of lakes within 20 miles of the respondents residence, is unexpectedly positive. Although it was intended to be a measure of substitutes, large values of NUM20 may indicate that the "favorite" lake is of high quality. If an individual has several alternatives in close proximity to his residence, the lake frequently visited may be the best in this set of alternatives. As the number of alternatives increases, the probability of the existence of an outstanding lake may also increase.

Policy Analysis: The Benefits of Reducing Soil Erosion

One policy option frequently suggested to reduce agricultural nonpoint source pollution is reducing cropland erosion. Doing so aids water quality in two ways: first, reduction of erosion reduces the amount of nitrogen, phosphorus, and chemical pesticides that may reach surface water bodies via eroding soil particles. Second, reducing erosion also reduces sediment delivery to lakes and streams, thereby enhancing water clarity. We focus on the second aspect.

Table 5--Recreational demand estimation[1]

Variable[2]	Parameter estimate	T-value
Constant	0.9654	2.46
Age	-0.0108	-6.25
Sex[2]	0.7750	10.80
Income	-0.0000064	-4.94
Cost	-0.0114	-3.88
Area	0.0073	7.05
Depth	-0.0034	-4.04
PSDM	1.5122	3.64
$PSDM^2$	-0.3895	-3.61
NUM20	0.0042	5.10

[1]Dependent variable is the number of trips taken to the respondent's favorite fishing lake. Number of observations is 178. A truncated Poisson model is used to estimate the equation. T-statistics appear in parenthesis below the parameter estimates.

[2]Constant is the constant term. Age is the respondent's age in years. Sex equals one if the respondent is male; zero if female. Income is the respondent's income. Cost is the round trip travel cost to the lake in dollars. Area is the area of the lake in acres. Depth is the maximum depth of the lake in meters. PSDM is the predicted secci disk measure from the tobit equation. $PSDM^2$ is the predicted secci disk squared measure from the tobit equation. NUM20 is the number of lakes 20 miles or less from the respondent's home excluding the lake visited.

Table 6 - Changes in consumer's surplus (CS) from decreases in erosion on agricultural land[1]

Measure	10-percent reduction		25-percent reduction		50-percent reduction	
	No choke	Choke	No choke	Choke	No choke	Choke
			Dollars			
Per person change:[2]						
Minimum	-9.37	-4.43	-23.52	-11.17	-47.75	-22.68
Maximum	23.03	10.51	56.35	25.71	107.23	48.93
Average	1.52	0.67	3.20	1.40	4.24	1.86
Standard deviation	5.54	2.42	13.51	5.90	25.72	11.24
Per trip change:[3]						
Minimum	-4.25	-2.02	-11.42	-5.42	-25.55	-12.13
Maximum	23.03	10.51	56.35	25.71	107.23	48.93
Average	0.47	0.21	1.02	0.44	1.48	0.63
Standard deviation	2.58	1.16	6.31	2.85	12.07	5.46
Total change in CS[4]	531,950	217,919	1,206,005	529,820	1,843,478	879,699

[1]Erosion on agricultural lands reduced by three levels (10 percent, 25 percent, and 50 percent) in each county. Changes in CS is the difference (in dollars) between CS at the reduced erosion rate and CS at the observed erosion rate. Welfare measures are calculated with and without a choke price (labeled "Choke" and "No choke" respectively)

[2]Difference in CS per individual using Minnesota lakes.

[3]Difference in CS per trip.

[4]Difference in aggregate CS. Aggregate measures are defined as the sum of the average per person measures in each county multiplied by county population, adjusted by an aggregation factor.

For illustrative purposes, we examine the expected effects of reducing the rate of farmland erosion in three stages: reductions of 10 percent, 25 percent, and 50 percent in Minnesota farmland. We use the estimated equation 1 to predict the changes in water clarity from reducing erosion and insert this new value into estimated equation 2. The changes in consumers surplus (found by integrating our estimated equation 2) give us a measure of the benefit associated with recreation under the changed conditions.

If fishing trips slowly approach zero as price increases, then large amounts of consumer's surplus may be attributed to unrealistically high prices. To avoid this, it may be more realistic to use a choke price (such as the largest observed trip cost). In the policy analysis that follows, we do both. The largest observed travel cost, which is used as a choke price, is $57.95.[6]

Summary statistics of these changes are displayed in table 6. The change per person is an annual measure of consumer's surplus change; the per trip change (change per person divided by number of trips taken) measures the welfare impact on a per occasion basis. Table 6 also presents estimates of the total change in consumer's surplus associated with reductions in soil erosion; these are found by aggregating the individual changes across affected counties.

The most surprising numbers on this table are the negative minimum changes, which suggest that increases in water clarity leave some individuals worse off. The model is developed to allow for the fishing quality (in terms of desirable species habitat) at a lake to diminish when a lake becomes too clear. Although the model captures the nonlinear relationship between trophic status and water clarity, this result is unanticipated, especially when small changes in erosion are considered.[7]

To put the changes in consumer's surplus in perspective, total consumer's surplus prior to any hypothetical changes in erosion appears in table 7. Comparing the changes with the total reveals that even a 50-percent reduction in erosion only translates into an approximately 1-percent change in total consumer's surplus. The welfare estimates appearing in tables 6 and 7 also indicate a sensitivity to a non-infinite choke price. Over half of the consumer's surplus in the "No Choke"

[6]Although the "short" trips are supposed to be limited to 30 minutes travel time, the straight line distance exceeded 30 miles in a few cases.

[7]It should be noted that this measure neglects potential increases in participants and trips due to improvements in water quality. Predicting changes in participants is yet another shortcoming of our model and data, which cannot accommodate nonparticipants.

Table 7--Total consumer's surplus measures[1]

	Per trip[2]		Per person[3]	
	No choke	Choke	No choke	Choke
	Dollars			
Minimum	13.50	6.01	315.01	43.25
Maximum	1,522.10	722.76	2,708.06	1225.56
Average	285.25	125.23	1,060.32	463.17
Standard deviation	304.84	138.24	424.21	201.42

	Consumer's surplus[4]	
	No choke	Choke
	Dollars	
Total	185,700,000	80,390,000

[1]CS measures prior to changes in erosion rates. Welfare measures are calculated with and without a choke price (labeled "Choke" and "No choke" respectively)
[2]CS per trip.
[3]CS per individual.
[4]Total CS at initial erosion rates.

columns is attributable to trip costs exceeding the choke price of $57.95.

The results of this exercise show that reducing soil erosion on agricultural cropland may be expected to yield economic benefits by improving water quality in southern Minnesota lakes. In principle, we could use the benefit measures in a cost-benefit analysis to evaluate the tradeoffs that accompany these reductions in erosion, such as increased production costs, lower yields, and the like. However, our case study also shows the limitations placed on the analysis by our data sources. Better georeferenced physical data, which would more directly tie water quality to resource conditions and agricultural practices, could allow a better estimation of the agriculture-water quality relationships. Also, a more comprehensive survey of respondents that includes more information about choices available to rural recreationists might enable us to construct a more sophisticated resource valuation model.

Case Study: The Benefits of Protecting Ground Water in Four Geographic Regions

Over the past 10 years, a considerable amount of public interest has arisen about the quality of the Nation's ground water resources. This is especially true for agricultural chemical residuals, which may potentially degrade ground water quality. Discovery of nitrates and pesticides in ground water during the late 1970's and early 1980's dispelled a commonly held view that ground water was protected from these chemicals by layers of rock, soil, and clay.

Ground water is an important source of drinking water, especially in rural areas. Concern about agricultural sources of ground water contamination is driven by fears that exposure to agricultural chemicals in drinking water may pose a risk to human health. In this case, the travel cost approach isn't applicable, and typically either averting expenditures or CVM formulations are used. (For a more detailed discussion of how CVM can be used to value ground water protection benefits, see Cooper, 1994.)

Existing Studies of Ground Water Protection Benefits

Table 8 summarizes the available CVM studies on the value of protecting ground water from chemical contamination. The estimated benefits of ground water protection vary widely, as might be expected given the variety of procedures used and differences in the way the studies were conducted. The CVM estimate of ground water protection benefits range from about $40 per household per year (Caudill and Hoehn, 1992) to over $1,000 per household per year (Edwards, 1988; Sun, Bergstrom, and Dorfman, 1992).

Given these results, can the estimated values be used for policy analysis? Any attempt to draw more general conclusions about the benefits of preventing agricultural contamination of drinking water based on these few studies must be done carefully. Fortunately, the emerging literature on benefits transfer procedures offers a way to make use of these benefits measures for policy analysis while maintaining the viability of the analysis. Before applying the existing benefits measures to a case study, we discuss in the next section the concepts and procedures of benefits transfer as they relate to our analysis.

Table 8--Estimates of ground water protection benefits

Study	"Good" being valued	Estimated willingness to pay (WTP)	Description of valuation procedure
Caudill, 1992, and Caudill and Hoehn, 1992	Protection of ground water subject to pesticides and nitrates.	Rural: \$43-\$46/household (hh)/year. Urban: \$34-\$69/hh/year.	Open-ended means.
Powell, 1991	Ground water subject to contamination by toxic chemicals and diesel fuel.	All data: \$61.55/hh/year. Respondents with a history of contamination: \$81.66/hh/year. Respondents with no contamination: \$55.79/hh/year.	Method of computation not specified. WTP for private well users exceeds WTP for public water supply users by \$14.04.
McClelland and others, 1992	Ground water, type of contaminant not specified.	Complete sample: \$84/hh/year.	Predictions from Box-Cox model.
Shultz, 1989, and Shultz and Lindsay, 1990	Ground water, type of contaminant not specified.	Mean WTP: \$129/hh/year.	Computed from logit model.
Jordan and Elnagheeb, 1992	Drinking water subject to contamination by nitrates.	Public water systems: \$146/hh/year. Private wells: \$169/hh/year.	Averages computed at midpoints from CVM payment card.
Poe, 1993, and Poe and Bishop, 1992	Drinking water subject to contamination by nitrates.	\$168-\$708/hh/year.	Computed from logit models. WTP estimates vary depending on water quality information given respondent.
Edwards, 1988	Ground water subject to contamination by nitrates and pesticides.	\$286-\$1,130/hh/year.	Derived from fig. 2 in journal article.
Sun, 1990, and Sun, Bergstrom, and Dorfman. 1992	Ground water subject to contamination by agricultural fertilizers, nitrates and pesticides.	Mean WTP: \$641/hh/year, ranges from \$165-\$1,452/hh/year.	Computed from logit model.

Transferring Benefits Estimates from One Site to Another

As mentioned earlier, benefits transfer is a procedure whereby measures of resource value developed in one context or geographic location are used to assess the benefits or costs of environmental policies in another location or for another resource issue. Benefits transfer is not necessarily a valuation methodology (like travel cost or contingent valuation methodologies), it is instead a way of carefully defining analytic procedures to ensure that benefits estimates in the new situation (the 'policy site') satisfy statistical and theoretical guidelines for defensibility and validity.

Suppose some resource exists for which a valuation study has been conducted, such as a travel cost model of recreational fishing in a river basin or a CVM model of the value of protecting drinking water in an area subject to ground water contamination. Typically, these valuation studies yield estimates of unit values (willingness to pay per person or per household for improved or protected environmental quality) that are functions of a set of exogenous variables.

How can these values be transferred to a new site? Suppose we knew that the individual willingness to pay for ground water protection were given by the following:

$$WTP = f(p, x, q, z) \qquad (3)$$

where

WTP = Willingness to pay for access to or consumption of the resource, on a per-household basis,
$f(\cdot)$ = some valuation function (for example,an estimated CVM equation),
p = price of access to the resource (for example, cost of drilling a well),

x = quantity of resource consumed,
q = a measure of quality of the resource, and
z = other demand determinants (income, demographic characteristics, etc).

Now, suppose we had full information about all the individuals in the 'policy site.' Then, if we can assume that individuals in the study site *and* the policy site have the same basic preferences (that is, that the same valuation function applies to both populations), we could obtain a measure of willingness to pay in the policy site by substituting socioeconomic and water-quality data from the policy site into the valuation function we obtained from the study site. A measure of total benefits, then, could be obtained by aggregating this valuation function over all individuals in the policy site.

Typically, however, we do not have information for each individual in the policy area. More often we have data at some aggregate grouping, such as average household income broken down by race at the county level. Another approach would be to substitute group means for p, x, q, and z in equation (3), and then sum over the number of groups in the policy site.

In some cases we may not even have information on the independent variables at a group level. The analyst is sometimes faced with the task of transferring benefits measures at some very broad aggregate level, where the only information available is the number of affected households. In that case, a third measure of willingness to pay in the policy site can be obtained by multiplying the average willingness to pay in the study site by the number of affected households in the policy site.

Many existing water-quality benefits studies have taken this last approach. For instance, Ribaudo estimated per unit value measures for access to recreational fishing benefits and multiplied this value across households to measure the damages associated with erosion-related impairment of freshwater fishing resources (Ribaudo, 1989). (This approach does not, of course, address diminishing marginal returns.)

How good are these types of approximations? There are two general sources of error in our estimates: errors in estimating benefits in the original study site and errors associated with transferring these estimates to the policy site. McConnell (1992) lists five main sources of error in estimating benefits in the study site: choosing the wrong functional form for the benefits function, omitting important variables in the benefits function, measuring the arguments incorrectly (income, for example), measuring the dependent variable incorrectly, and misspecifying the random process that generates the data (for example, truncating trips in a travel cost model). He also identifies additional sources of error in the calculation of benefits at the policy site: incorrect handling of the random components of the valuation function, errors of aggregation in calculating the group means for the independent variables, and errors in calculating the number of affected households and the extent of the market for the environmental service being considered. Given those multiple sources of error, the transferred benefits in the policy site must be used very carefully in evaluating the tradeoffs among different environmental policies. He concludes:

"There is no simple, acceptable way mechanically to transfer a model. Just as the chief ingredient in model construction is judgment, it is also the most important ingredient in transferring benefits. Consequently, the transfer of benefits makes considerable sense. But the nature of the value of nonmarket benefits, and what we know about that value, preclude simple cataloging values to be drawn out as the next natural resource valuation problem arises" (p. 700).

In summary, benefits transfer can be viewed as one additional tool for providing the policymaker with additional information about the benefits and costs of water-quality policies. If the studies that form the basis of the transfer are carefully conducted and sufficient detailed information is obtained about the policy site that supports the theoretical bases of the original studies, then transfer of environmental benefits measures from one site to another can provide useful information to policymakers at a smaller cost.

Description of the Policy Sites

We chose as our case study the issue of protecting rural drinking water from possible contamination by agricultural chemical residuals. We used data from the USDA Area Studies Program as the basis for our exercise in ground water benefits transfer, with the objective of showing how estimates of water-quality benefits can be used to evaluate policies in areas beyond the original study sites.

At issue is the possibility that leachable chemical or nitrogen fertilizer use on cropland could reduce ground water quality and possibly pose a risk to rural families, who may be exposed to elevated levels of chemical residues in drinking water. In this case study, we pose the hypothetical question: what is the extent of the possible willingness to pay to prevent ground water contamination from farm chemicals in these regions? Since we do not, at present, have primary survey data to compute the valuation of ground water protection benefits in these areas, we transfer existing benefits measures to the policy sites.

The four policy sites selected for this exercise are: Central Nebraska, the White River Basin in Indiana, the Mid-Columbia Basin in the Pacific Northwest, and the Lower Susquehanna Basin in Pennsylvania and Maryland. We chose these study sites because they are also part of the U.S. Geological Survey's National Water Quality Assessment (NAWQA) program. NAWQA is a 5-year program created to describe status and trends of the Nation's water resources that specifically addresses the issue of ground water impairments from agricultural chemicals.

The four sites encompass 126 counties, with an estimated rural population of about 1.1 million households (based on the 1990 census). All four sites rely on ground water for public water supplies, particularly for self-supplied sources of private drinking water (U.S. Geological Survey, 1990). Agricultural chemicals in ground water supplies have been identified by the Geological Survey as major water-quality issues in these regions (U.S. Geological Survey, 1991).

Data Sources

The USDA Area Studies Program is designed to develop farm-level data that link production activities with environmental characteristics in selected regions of the country. The objective is to support the assessment of environmental policies affecting agriculture. As part of the program, surveys were conducted to identify production technologies, chemical use, and cropping strategies in the four regions described above. The survey sample points correspond with the Soil Conservation Service's Natural Resources Inventory, providing a linkage to soil, water, and other natural resources data.

Following procedures suggested by McConnell (1992) and Boyle and Bergstrom (1992), we examined the available ground water valuation literature to identify benefits estimates for possible use in this benefits transfer exercise. To ensure comparability of results, we limited our effort to contingent valuation studies of the benefits of preventing agricultural contamination of drinking water supplies. Other criteria for selection included considerations of sample size, theoretical appropriateness of the benefits measure, correct specification of the valuation equation, validation through peer-reviewed publication, and ease of transfer of the valuation equation, (such as right-hand side variables for which equivalent data could be obtained for our policy sites).

Eight studies met our initial selection criteria, and these are listed in table 5. Of the eight studies, seven addressed agricultural chemical contamination issues, and one (Powell) dealt with trichloroethylene and diesel fuel. Five studies (Caudill and Hoehn; Edwards; Jordan and Elnagheeb; Poe; and Sun, Bergstrom, and Dorfman) examined willingness to pay to prevent contamination from nitrates. Two studies (Caudill and Hoehn; and Sun, Bergstrom, and Dorfman) also included agricultural pesticides in the CVM questionnaire. The McClelland and Shultz studies did not specify the type of contaminant.

The consensus of the benefits literature seems to be that the valuation equations should be transferred from study areas to site areas wherever possible (Loomis, 1992, McConnell, 1992, and Smith, 1993). The wide disparity in mean willingness to pay per household (from $45 to over $1,000) would imply an equally wide disparity in estimated regional benefits when applied to our for geographic regions. Any attempt to transfer the actual willingness to pay equations, however, is complicated by the wide disparity between studies in the choices of functional form and independent variables. For example, Poe reported 3 to 5 significant variables and 12 to 13 insignificant variables, depending on the specification of the model, while Edwards and Shultz and Lindsay used only 3 exogenous variables in their willingness to pay equations. Most of the studies included the socioeconomic variables income, age, and education levels as determinants of willingness to pay. Most also included attitudinal variables designed to capture the respondent's level of information and preferences towards water quality (awareness of contamination and its causes, risk perception, altruistic or bequest motives, and so forth). In addition, information in the policy sites that corresponds to the right-hand side variables in the valuation equations must be obtained. This may eliminate some studies for benefits transfer if we have no information in the policy sites about these attitudinal questions.

Accordingly, in the following analysis we selected three studies for transfer of the actual valuation equation: Shultz and Lindsay; Jordan and Elnagheeb; and Sun, Bergstrom, and Dorfman. We did so, in part, because of the relatively small amount of information needed to compute the willingness to pay estimate for these three studies and because these three studies have been published in peer-reviewed journals. In this section, we perform a direct benefits transfer using data collected in the policy areas to substitute for right-hand side variables of the three selected studies.

The estimated equations for the three chosen studies, including coefficient estimates and variable descriptions, are found in the appendix. All models include per household income and age as explanatory variables. Jordan and Elnagheeb include race, gender, education, urban vs. rural, and two dummy variables for water-quality perceptions and risk as explanatory variables.

Shultz and Lindsay include the land value of the respondent's home, but no additional variables in their 'reduced form' equation. Sun, Bergstrom, and Dorfman include dummy variables for concern about the health effects of pollution, subjective probability of ground water contamination, and subjective probability of demand for clean water as explanatory variables, in addition to income and age.

USDA surveyed farmers in the four Area Studies regions in 1991. Farm operators in the four areas were asked detailed questions about their farm operations, including questions designed to yield socioeconomic information for further analysis. The total sample size was 3,428. The sample was designed to represent the agricultural sector based on land use, so it is not directly representative of the farm population (that is, households on large farms may be over-represented in a sample based on acreage rather than one based on farmers). Nevertheless, the surveys do provide more detailed information about current farm conditions than do county-level aggregates and permit linkage of benefits measures to resource conditions.

Two variables present in the demand equations for the three studies may be found directly on the survey: age of respondent and education level of the respondent. Obviously, the value for the dummy variable "FARM" in the Jordan study was one. It was not possible to directly estimate income for each respondent, since the survey did not ask for farm household income. The only information available from the survey was the total value of farm sales (in discrete levels from $5,000 to over $1,000,000). To create a proxy variable for income, we calculated the average farm household income for farm operators in each area for each sales level using data from the USDA's Farm Costs and Returns Survey using procedures developed by Ahearn, Perry, and El-Osta (1993). Income estimates derived in this manner, by farm sales class, were used as proxies for the household income of the individual survey respondents.

Finally, we needed to develop estimates of the remaining right-hand side variables. Jordan and Elnagheeb include dummy variables for sex and race. We used the county-level percentages of farm households headed by Blacks and the county-level percentages of women in the population as reported by the latest Census of Agriculture. All studies included explanatory variables relating to the subjective probability of pollution and health concerns. Since we did not have any information about farmer's attitudes about pollution probabilities in our policy sites, we followed recommended practices and transferred the mean values from the study sites to the policy sites. Average values of the right-hand side variables for the Jordan and Elnagheeb, Shultz and Lindsay, and Sun, Bergstrom, and Dorfman study sites, along with the average values for each of the four Area Studies Program sites, are found in the appendix.

To calculate the total willingness to pay for ground water quality in the four policy sites, we computed mean values of the independent variables on a county-by-county basis. These county averages were then used in the respective valuation functions to obtain mean willingness to pay on a per household basis. Finally, these per household values were multiplied by the number of rural households in each county to obtain an estimate of aggregate willingness to pay for ground water quality at the county level. Table 9 presents results for the four Area Studies regions as a whole. For two of the three studies considered, the value estimated based on transfer of the valuation function was about the same as the value based on transfer of the mean willingness to pay. Aggregate willingness to pay for ground water protection was estimated at $197 - $730

Table 9--Benefits of ground water protection, based on transfer of valuation functions

	Willingness to pay	Area Studies region: Central Nebraska	Lower Susquehanna	Mid-Columbia Basin	White River	Total
				Number		
Rural households		135,746	627,125	57,436	296,889	1,117,195
	$/household/year			*Million dollars*		
Study:						
Shultz and Lindsay	128	15	132	9	41	197
Jordan and Elnagheeb	233	32	118	18	73	241
Sun, Bergstrom, Dorfman	639	81	402	52	195	730

million per year, with the Shultz and Lindsay equation giving the lowest value, and the Sun, Bergstrom, and Dorfman equation giving the highest.

Policy Analysis

Aggregate measures of willingness to pay for ground water quality may be somewhat misleading since they do not take into account the distribution of actual environmental risk across households. Asking a person how much they would be willing to pay for clean ground water in a hypothetical context may give a meaningless result if that person's water supplies are neither contaminated at present nor at risk of future contamination.[8] A more relevant measure from a policy standpoint would be to determine the willingness to pay to protect ground water supplies that are currently or potentially at risk, rather than applying per household values derived from surveys to all resource users.

One attractive feature of the Area Studies Survey is that information is collected about resource conditions that can help us define measures of environmental risk. In a separate study, Crutchfield, Keim, and Vandeman (1994) have used an environmental risk assessment procedure developed by Weber and Warren (1992) to create a qualitative index of the likelihood of pesticide leaching in the Area Studies regions. This procedure, which incorporates information about soil quality, agricultural pesticide chemical qualities, and chemical application methods, derives an index of chemical leaching potential, ranging from 'Safe' to 'Hazardous.' A risk potential is assigned to each sample point, which enables us to link willingness to pay for ground water quality to qualitative measures of environmental risk.

Suppose each individual knew whether or not local ground water were at risk of contamination. The results of the available studies indicate that providing information about actual ground water quality changes the valuation individuals place on the resource, but in opposite directions. If individuals exhibit risk aversion, decreasing the uncertainty about their water quality will, other things being equal, decrease willingness to pay. Jordan and Elnagheeb found that the coefficient on their UNCERTAINTY variable carried a positive sign, which is consistent with risk aversion. However, informing an individual that local ground water supplies are considered 'Hazardous' to pesticide leaching would have a positive effect on willingness to pay in the Sun, Bergstrom, and Dorfman study, since providing this information would (presumably) increase the individual's subjective contamination probability, and possibly increase concern about possible health effects. Jordan and Elnagheeb found weak, though not statistically significant, evidence that the willingness to pay for those who rated water quality as poor was slightly higher than for those who rated water quality as good.

To examine this issue, we recomputed the willingness to pay for ground water quality in the Area Studies region, accounting for the different environmental risk measures across sample points. The cropland in each region was stratified according to whether it was considered hazardous, risky, slightly risky, or safe from potential pesticide leaching. We assumed that those respondents whose land was ranked either hazardous or risky would have an increased health concern and a higher estimate of the probability of contamination if they were informed of their water-quality risk, compared to those whose land was ranked as slightly risky or safe. We also assumed that for cropland identified as having no chemicals applied the survey provided no *additional* information about the probability of contamination. In those instances, the average values of subjective probability of contamination from the original studies were retained.

Table 10 shows the assumptions made in imputing new values for subjective probability. New average values for these variables were calculated for each county, and plugged into the valuation models. (Because the Shultz and Lindsay model does not include subjective risk or health assessment probabilities in their reduced-form model, we do not use that valuation equation in this policy analysis.) Table 11 shows the distribution of ground water leaching potential in the four study areas.

Table 12 presents two sets of results. The first part of the table shows aggregate willingness to pay for ground water protection, based on adjustment of perceived risk measures based on cropland vulnerability described in table 10. Adjusting the perceived risk measures to account for cropland vulnerability increases the aggregate willingness to pay for ground water protection 9 percent (using the Sun, Bergstrom, and Dorfman function) and 36 percent (Jordan and Elnagheeb).

However, it may be overstating the matter to assume that people whose water supplies are not at risk will be as willing to pay as these estimates might indicate. Such individuals may have some existence or option value for protection of ground water even though their water supplies may not currently be at risk. Even setting the perceived risk measures to zero in our studies yields estimates of willingness to pay of $202 per household per year (Jordan and Elnagheeb) and $89 per household per year (Sun, Shultz, and Dorfman), which may be too high and not an accurate representation of the true value. An alternative assumption is if

[8] There may, of course, be elements of bequest or nonuse value in people's valuation of ground water quality. None of the studies currently available, with the exception of Jordan and Elnagheeb, distinguish between use value and total value (which is the sum of use, existence, and bequest values).

people know their water supplies are not at risk, then they have no willingness to pay for further ground water protection. If we make this assumption, then we might consider the true benefits of ground water protection as the willingness to pay for individuals living on or near cropland considered ‘at risk.’

The second part of table 12 gives those values, which range from $600 per household per year to $1,166 per household per year for the four regions taken together. Under this assumption, the aggregate benefits of protecting ground water supplies in areas thought to be at risk from leaching pesticides is between $76 million and $153 million per year, which is substantially lower than without this assumption.

Assuming people living on or near land deemed ‘safe’ derive no benefit from ground water protection programs denies the existence of motivations such as bequest or altruistic values in people’s preferences for environmental quality. On the other hand, assigning a full willingness to pay to all residents in an area overstates the value of ground water protection; the effect of giving respondents information that their water supplies are currently safe is to lower estimated willingness to pay substantially (Poe, 1993).

These results illustrate one of the shortcomings of the benefits transfer approach. In an original case study, where we would actually survey individuals in the study area, we could control for variation in resource conditions and cropland vulnerability directly. Here, we are forced to make some assumptions about the connection between resource vulnerability and willingness to pay in order to make the analysis fit the available data sources. One interesting extension of this research would be to survey individuals in the policy site about their preferences for ground water protection, and compare the results with estimates derived from benefits transfer.

In conclusion, then, we can see an indication that rural residents in the four Area Studies regions might be willing to pay for assurance that their ground water supplies were protected from agricultural contamination. The estimates of the total willingness to pay vary widely, but most likely lie between $73 and $780 million per

Table 10 -- Assumptions about changes in perceived risk

Variable	Average value in original study	New value: hazardous/risky	New value: slightly risky	New value: safe
Probability of future ground water contamination (Sun, Bergstrom, and Dorfman)	0.54	1.00	0.50	0
Subjective estimate of risk from pollution (1 = not concerned, 4= very concerned (Sun, Bergstrom, and Dorfman)	3.89	4	2	1
Risk of ground water contamination: 1 = poor water quality, 0=good water quality (Jordan and Elnagheeb)	0.13	1	0.25	0
Uncertainty about current water quality: 1 = uncertain, 0 = otherwise (Jordan and Elnagheeb)	0.14	1	0.25	0

Table 11 -- Distribution of cropland vulnerability

Vulnerability measure	Central Nebraska	Lower Susquehanna	Mid-Columbia Basin	White River	All regions
			Percent of all acres surveyed		
Hazardous	6	3	16	1	7
Risky	10	9	12	18	11
Slightly risky	11	24	7	41	14
Safe	9	14	8	16	10
No chemicals applied	63	49	56	24	56
No data or unknown risk	1	2	1	1	1

Table 12--Benefits of ground water protection, accounting for resource vulnerability

Study	Willingness to pay	Central Nebraska	Lower Susquehanna	Mid-Columbia Basin	White River	Total
	Benefits: All households					
	$/household/year	*Million dollars*				
Jordan and Elnagheeb	318	44	157	28	105	32
Sun, Bergstrom, and Dorfman	701	85	438	64	196	783
	Benefits: Vulnerable land only					
	Willingness to pay	Central Nebraska	Lower Susquehanna	Mid-Columbia Basin	White River	Total
	$/household/year	*Million dollars*				
Jordan and Elnagheeb	600	8	29	9	30	76
Sun, Bergstrom, and Dorfman	1,166	16	62	19	56	153

year. If we knew the costs of preventing ground water contamination, such as through limitations on pesticide use or farm practices, the results of our analysis indicate that the environmental benefits may be considerable, and should be weighed against the costs to producers and consumers of modifying farm practices.

Conclusions and Suggestions for Further Research

We have shown how nonmarket benefits estimation techniques can be used to value the benefits of improving or protecting water quality. We focused on two resource issues: the benefits of reducing agricultural nonpoint source pollution of surface water bodies, and the value to the public of preventing ground water contamination from agricultural chemical residuals. We showed, using a case study of surface water recreation in Minnesota, how a simple travel cost model can be linked to agricultural land use choices to evaluate the water-quality benefits associated with reducing soil erosion. We used the technique of benefits transfer coupled with existing CVM studies of household willingness to pay to prevent ground water pollution to illustrate how existing water-quality benefits measures can be applied to new study areas and to show how these new benefits measures can support cost-benefit analyses when primary data are unavailable.

Techniques for valuing environmental services, such as clean water, have been steadily refined and improved in recent years. There is a growing consensus that benefits measures obtained from travel cost, contingent valuation, and benefits transfer approaches can be used in benefits/cost calculations in support of policy analysis (Smith, 1993). Our valuation methodologies, at least at a site-specific level, are now based on economic models of consumer behavior, with benefits estimates obtained that are consistent with consumer theory. Even so, work remains to be done to increase our understanding of the benefits associated with improving water quality. Our case studies developed here, while hardly conclusive about the overall cost of agriculture-related water-quality impairments, highlight several important issues that need to be addressed in future efforts to measure water-quality benefits.

First, and foremost, comprehensive estimation of water-quality benefits requires more complete and comprehensive data than are currently available. Data are needed both on the uses of water resources (consumptive and recreation) and on the quality of the resources themselves. Particularly in the case of ground water, the lack of a comprehensive and uniform database on drinking water or aquifer quality makes a global assessment of the benefits of protecting ground water difficult. The available literature, thin as it is, supports the conclusion that an important consideration when asking people their willingness to pay for clean drinking water in a CVM context is the current quality of their water supplies and the health implications this holds for them.

Second, we must be able to make a close linkage between our measures of water-quality benefits and the actions taken on the farm to reduce pollutant loadings. We explored one approach to this issue in our case study of surface water benefits in Minnesota, where we estimated a functional relationship between lake clarity and cropland erosion. However, better information on resource conditions, such as soil quality, distance to water bodies, agricultural chemical use and production practices, and water use on cropland which are then linked spatially to recreational or consumptive use data would enable us to make use of pollutant fate and transport models and models of agricultural production to develop the spatial and temporal ties between changes in farm production, changes in water quality, and changes in benefits.

Finally, economists will never completely escape the need to extrapolate benefits estimates from one site to another and from one geographic scale to another. Our case study of benefits transfer in the context of ground water protection benefits shows how existing studies can be used in new settings. However, our new benefits measures will only be as good as the original studies themselves. Care should be taken by researchers to evaluate the quality of published research work before using empirical results in a benefits transfer exercise. Some possible factors in making such an evaluation might include a subjective assessment of the quality of the journal, examination of the statistical reliability of the results, acquisition of the data to replicate the results, and consultation with the original researchers to verify that the proposed use of their research results would be appropriate.

This is relevant when we consider extrapolating CVM studies of water-quality benefits. Considerable controversy remains on the value of CVM measures, particularly where the qualitative measures of resource quality are ill-defined or nonuse values may form a large proportion of the estimated total benefit. Although the results of benefits transfer studies can provide useful insight to guide policymakers when considering the tradeoffs of alternative environmental policies, they are best used in preliminary evaluations. When site-specific measures are required, or when legal issues of compensation and liability for damages arise, primary studies using new data may be required.

ERS is continuing to conduct new research to further refine our understanding of water quality and other environmental benefits. The forthcoming National Survey on Recreation and the Environment (NSRE) will help support analysis of recreation benefits of reduced water pollution. Special study design, which samples specifically in the Area Studies regions, is intended to help us more completely model the agriculture-water quality relationship by linking recreational data with resources data. Finally, future research plans include comparison of ground water valuation measures derived from benefits transfer with original valuation studies using new data. This will enable us to compare how well benefits transfer performs compared to original CVM studies and help to develop new ways to make use of existing studies in new geographic settings.

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Appendix A: Calculation of Recreational Benefits in Minnesota Lakes

The welfare measure used to quantify recreation benefits is the area under the estimated recreational demand function (equation A2). Although the dependent variable used in the demand model described above takes on strictly positive (count) values, its expectation is continuous. This allows for expected consumer's surplus to be determined by simply integrating the expectation of T (number of trips) over the price (travel cost) (Mendelsohn, Hellerstein, and others, 1992)

$$E(CS) = \int_{P}^{P'} E(T|X)dP = \int_{P}^{P'} \exp(X\beta)dP = (1/\beta_p)[\exp(X\beta|P') - \exp(X\beta|p)], \tag{A1}$$

where E(CS) is the expected value of consumer's surplus (CS), X is a vector of independent variables (such as income, age, sex), P observed travel cost, P' is the price where no trips will be taken (the "choke" price) and where β_p is the travel cost parameter. If P' is set equal to infinity (assuming ß is negative), then consumer's surplus becomes:

$$E(CS) = \exp(X\beta)/-\beta_p. \tag{A2}$$

Since the survey sample is drawn from the population of registered anglers, it is representative of the population affected by water quality improvements and can be expanded to the angling population. These total changes in consumer's surplus per county and over the entire sample area appear in appendix table 3. Consumer's surplus in the i-th county (CS_i) is defined as:

$$CS_i = POP_i * E * CS_i \tag{A3}$$

where

POP_i = the population in the i-th county,
E = an expansion factor identifying the potential participants in each county[9],
CS_i = the average per person consumer's surplus in county i, and

$$CS_i = \sum_{k=1}^{N_I} CS_k/N_i, \text{ where}$$

N_i is the number of respondents from the survey who visited lakes in county i.

It should be noted that this measure neglects potential increases in participants and trips due to improvements in water quality. Predicting changes in participants is yet another shortcoming of our model and data, which cannot accommodate nonparticipants.

[9]The expansion factor is the probability that an individual is a registered angler (0.29441) times the probability that the individual would respond to the survey (0.61) times the probability that the individual participates in trips close to home (0.7378).

Appendix B: Summary of Ground Water Studies Used in Benefits Transfer Exercise

Jordan and Elnagheeb: Summary of valuation question

“The Environmental Protection Agency (EPA) has ranked the State of Georgia as second in the Nation for potential contamination of underground water. At the same time, underground water is a source of drinking water for almost 50% of the U.S. population. Results from EPA’s five-year study of wells in different States showed that over half of U.S. drinking water wells contain nitrates. Nitrates are chemical substances hazardous to human health if taken in large quantities. Most of the wells surveyed have nitrate levels below hazardous levels.

As farmers continue to apply more fertilizers to increase yields, the underground water may become contaminated with nitrates. Adoption of different agricultural practices can reduce the amount of nitrates in the ground water BUT may increase food prices. On the other hand, if agricultural practices did not change, the amount of nitrates in ground water would increase. So the costs of cleaning water from nitrates will go up. The local water companies have to clean pumped water to make it safe for drinking. Since the costs of cleaning water from nitrates will increase, the consumers will have to pay higher water bills.

Suppose you found that the amount of nitrates in your well water exceeds the safe level. Suppose also that a local water supplier offers to install AND maintain new equipment on your well. This equipment will clean you water from nitrates but the Water Supplier will charge for use of its equipment. If you do not want to pay to the water supplier, the equipment will NOT be installed and you will have to bear the risk of increasing nitrates in your drinking water.

To avoid the risk of increasing nitrate in my drinking water, the MOST I would permanently pay to the water supplier, ABOVE my current monthly water bill is: (Please circle ONE answer): $0.00, $1.00, $5.00, $10.00, $25.00, $50.00, or $100.00)."

Appendix table 1--Ground water study estimation results, Georgia

Variable	Coefficient	Mean	Standard error	T-value
Log of income	0.12571	1.80	N/R	1.526
Male (1 if male, 0 otherwise)	-0.82210	0.51	N/R	-2.145[1]
Black (1 if black, 0 otherwise)	1.26447	0.13	N/R	2.245[1]
Age (years)	-0.00877	53.00	N/R	-0.750
Education (1 if greater than high school, 0 otherwise)	1.00902	0.66	N/R	2.073[1]
Farm (1 if farmer/rancher, 0 otherwise)	1.23931	0.25	N/R	2.805[1]
Risk (1 if rated currrent water quality as poor, 0 otherwise)	0.00912	0.13	N/R	0.002
Sigma	1.06174	--	N/R	7.330[2]

Sample size = 40.
N/R = Not reported in journal article.
[1]Significant at the .05 level.
[2]Significant at the .01 level.

Shultz and Lindsay: **Summary of valuation question**

Dover's water supply comes from its ground water sources. Several other nearby N.H. towns have recently had their ground water supplies polluted. For example, 15 wells in Northwood were closed, and in Barrington 38 families have been forced to drink bottled water for the last five years due to ground water pollution. In almost all cases where ground water contamination has occurred, the costs of cleanup or finding an alternative supply of clean water have been very high.

On the other hand, many N.H. towns have never had any serious ground water pollution problems. Obviously, it is impossible to predict with complete certainty if and when ground water pollution will occur in any given N.H. town.

As you may already know, several towns in Strafford County along with the Office of State Planning, are now in the process of formulating specific ground water protection plans. Basically these plans are an attempt to protect community ground water supplies from future pollution by: purchasing land overlying sensitive ground water areas, formulating stricter zoning ordinances, hiring inspectors to enforce ground water pollution laws and standards, and a variety of other strategies. These protection plans cannot guarantee the prevention of ground water pollution, rather they are intended to reduce the risk of such a problem occurring.

Question: Would you be willing to pay $____ per year in extra property taxes for such a ground water protection plan in Dover? 1) yes, 2) no. (The range of dollar values for the bid variable was $1-$500, in $25 ranges)

Appendix table 2--Ground water study estimation results, Dover, New Hampshire

Variable	Coefficient	Mean	Standard error	T-value
Constant	0.13050	1.00	-0.6892	0.189
Land value ($1,000)	0.04070	10.42	0.0214	1.902[1]
Age (years)	-0.02780	52.02	0.0099	-2.808[2]
Household income ($)	0.00002	36,533.00	7.73e-06	2.567[2]
Bid value ($)	-0.00570	214.90	0.0011	-5.182[2]

Sample size = 346.
[1]Significant at the .10 level.
[2]Significant at the .01 level.

Sun, Bergstrom, and Dorfman: **Summary of valuation question**

Suppose with the [ground water protection] program, pollution by agricultural pesticides and fertilizers in Dougherty County will be definitely kept at safe levels for drinking and cooking (that is, below the EPA's health advisory levels). Given this assumption, please evaluate and give YOUR BEST ANSWERS to question (14) and (15).

(14) Would you vote to support the program for preventing ground water pollution from agricultural pesticides fertilizers, if the program reduces the amount of money you have to spend on other goods and services by $_____ per year? 1) yes, 2) no.

(15) What is the **highest** amount the program could reduce the amount of money you have to spend on other goods and services before you would vote against it? $_____ dollars per year.

Appendix table 3--Ground water study estimation results, Dougherty County, Georgia

Variable	Coefficient	Mean	Standard error	T-value
Constant	-1.0800	1.000	N/R	-0.510
Log of bid value	-0.8130	476.600	N/R	9.650[1]
Log of income	0.7370	42.517	N/R	5.510[1]
Log of subjective estimate of risk from pollution	1.4900	3.890	N/R	3.890[1]
Log of subjective estimate of probability of future ground water contamination	0.3630	0.541	N/R	3.180[1]
Log of future demand for clean water (1/0 dummy variable)	0.0732	0.675	N/R	0.817
Log of age	-0.7180	46.800	N/R	2.210[2]

Sample size = 591.
N/R = Not reported in journal article.
[1]Significant at the .01 level.
[2]Significant at the .05 level.

Appendix table 4--Definitions and weighted averages of variables used in benefits transfer exercise

Variable	Jordan	Shultz	Sun	Central Nebraska	Lower Susquehanna	Mid-Columbia Basin	White River	Total
Income[1]	22,008	36,533	42,517	34,419	35,540	55,127	37,627	36,924
Age[2]	53	52	47	50	47	49	49	49
Male[3]	0.51	N/A	N/A	0.49	0.49	0.51	0.49	0.49
Black[4]	0.13	N/A	N/A	0.004	0.027	0.012	0.015	0.015
Education[5]	0.66	N/A	N/A	0.4	0.16	0.67	0.4	0.36
Farm[6]	0.25	N/A	N/A	1	1	1	1	1
Risk[7]	0.13	N/A	N/A	0.13	0.13	0.13	0.13	0.13
Uncertain[8]	0.14	N/A	N/A	0.14	0.14	0.14	0.14	0.14
Land value[9]	N/A	10,420	N/A	8,760	11,798	13,968	12,415	11,987
Health[10]	N/A	N/A	3.43	3.43	3.43	3.43	3.43	3.43
Contamination probability[11]	N/A	N/A	0.54	0.54	0.54	0.54	0.54	0.54
Water demand[12]	N/A	N/A	0.675	0.675	0.675	0.675	0.675	0.675

N/A = Not applicable
[1]Thousand dollars
[2]Years.
[3] 1 if Male, 0 otherwise.
[4]1 if Black, 0 otherwise.
[5]1 if more than high school, 0 otherwise.
[6]1 if lives on a farm or ranch, 0 otherwise.
[7]1 if rated current water quality as poor, 0 otherwise.
[8]1 if uncertain about current water quality, 0 otherwise.
[9]Assessed land values of property, not including buildings.
[10]Index for concern over pollution effects on own heath: 1 = not concerned, 2 = somewhat concerned, 3 = concerned, and 4 = very concerned.
[11]Estimated subjective probability of ground water contamination within 5 years without a protection program.
[12]Estimated subjective probability of clean water demand within 5 years.