From: elimoore@sonic.net
Sent: Monday, September 27, 2010 11:25 PM
To: ILRP Comments
Cc: emoore@pacinst.org
Subject: ILRP comments
Attachments: PI comments ILRP DEIR.docx; benefit of reducing nitrates in private wells.pdf; USDA_ 1995.PDF

Hello Ms Smith,

Please accept the attached comments on the DEIR for the Irrigated Land Program. Also attached are two key documents cited in the comments and suggested to staff for their review.

Thank you,

Eli Moore
Senior Research Associate
Pacific Institute

Via Electronic Mail

ILRP Comments
Ms. Megan Smith
630 K Street, Suite 400
Sacramento, CA 95814

Re: Comments on the Central Valley Regional Water Quality Control Board's Staff-Recommended Long-Term Irrigated Lands Regulatory Program and Draft Technical Memorandum Concerning the Economic Analysis of the Irrigated Lands Regulatory Program

September 27, 2010

Dear Ms. Smith:

We commend the State Water Board and staff for developing alternatives to the Irrigated Land Regulatory Program and are excited to see potential solutions to address a key source of the significant environmental health problem of nitrate contamination of ground water.

We are pleased to submit the enclosed comments pertaining to the Technical Memorandum Concerning the Economic Analysis of the Irrigated Lands Regulatory Program. Pacific Institute is currently conducting primary and secondary research on the health, social and economic effects of nitrate contamination of drinking water in the San Joaquin Valley. Our comments draw from this research and the nearly 25 years of previous work on water issues in California.

By modifying the Irrigated Lands Regulatory Program, the Board can take a major step forward toward addressing a persistent and detrimental problem affecting our state: nitrate contamination of drinking water. Nitrate contamination of ground water drawn by private and public water systems currently adds significant expenses to already strapped water boards, local and state agencies, private well owners, and consumers. According to an analysis of public water system monitoring data by Balazs (2009), there were 93 systems in the San Joaquin Valley serving 1.3 million consumers that had nitrate levels above the MCL during at least one quarter between 2005 and 2008 (see Appendix 1). The Groundwater Ambient Monitoring and Assessment Program estimates that 10% of the 600,000 private domestic wells in the state are also above the legal limit for nitrates, affecting another 169,000 residents.[1]

Studies have found exposure to nitrates to result in serious illness and death, including significant increased risk of: neural tube defects, premature birth, intrauterine growth restriction, anencephaly, increased methemoglobin levels causing pregnancy complications, central nervous system birth defects, and congenital malformations.[2] Nitrate exposure at excess levels can cause methemoglobinemia, also known as “blue baby syndrome”, a cause of illness and death in infants. Additional known or suspected health effects include: respiratory tract infections in children, thyroid disruption, pancreatitis, sudden infant death syndrome (SIDS), and cancers of the digestive system.[3]

Because California has no systematic monitoring of run-off and ground water quality, it is difficult to estimate the extent of nitrate contamination attributable to agricultural activities. However, several studies point to a widespread and severe problem with nitrate contamination from agricultural sources. UC Davis researcher Thomas Harter analyzed the use of fertilizers on California farms in 2007 and estimated that on average more than 80 lbs N/acre/year may leach into the groundwater beneath irrigated lands, usually as nitrate.[4] Harter concludes that “without attenuation, 80 lbs N/acre/year would lead to groundwater NO3-N concentrations at the water table that are two to four times higher than the MCL.” Even though subsurface attenuation does occur in some areas, this is a remarkably high amount of unabsorbed nitrate released on irrigated lands.

General Comments on the Economic Analysis of the ILRP

The current draft Economic Analysis ignores several categories of costs and underestimates others, producing an artificially low finding of overall economic impact. Revisions to the analytical approach and the use of additional data sources can remedy this. Methods for the revised approach can be adapted from previous studies by the U.S. EPA, the USDA, and leading scientists.

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[1] State Water Resources Board, Groundwater Ambient Monitoring & Assessment Program (2010). *Summary of Detections Above a Drinking Water Standard, GAMA Domestic Well Project.* Accessed on September 20, 2010 from http://www.swrcb.ca.gov/gama/domestic_well.shtml.

[2] Manassaram, Deana M., Lorraine C. Backer, and Deborah M. Moll (2006) A Review of Nitrates in Drinking Water: Maternal Exposure and Adverse Reproductive and Developmental Outcomes in *Environ Health Perspectives*, 114:320–327. doi:10.1289/ehp.8407 available via http://dx.doi.org/ [Online 3 November 2005]

[3] Various, see for example Ward, Mary H., Theo M. deKok, Patrick Levallois, Jean Brender, Gabriel Gulis, Bernard T. Nolan, James VanDerslice (2005) Workgroup Report: Drinking-Water Nitrate and Health-Recent Findings and Research Needs. *Environmental Health Perspectives*, Vol. 113, No. 11 (Nov., 2005), pp. 1607-1614.

[4] Harter, Thomas (2009) Agricultural Impacts on Groundwater Nitrate. *Southwest Hydrology*, volume 8, number 4.

In this letter we provide a preliminary analysis of the costs to domestic well owners, public water systems, and water consumers using available data. With this analysis, we find that, currently, the total estimated costs for these three impacted stakeholders are between $40,169,276 and $89,600,723 (See Table 1). We also analyze trends in nitrate levels in monitored wells in Kern County, and find that levels are increasing in a third of locations, and the number of wells where nitrate levels exceed federal health standards is likely to double in the next ten years.

Table 4. Additional Cost Estimates for ILRP

	Cost estimate		**Number of systems/projects**			**Cost per project**		
	Low range	**High range**	**Low**	**High**	**Notes**	**Low**	**High**	**Notes**
Public drinking water systems	$24,000,000	$60,000,000	60		Source: 2007 Compliance Reports (considered low because they are known to under-report.	$400,000	$1,000,000	Source: Paul Boyer, Self-help Enterprises
Domestic well owners	$5,615,734	$12,011,486	16,713	16,713	Assuming 10% of wells are above MCL (GAMA), and 60% of those have agriculture as a source of contamination.	$336	$719	Low range source: Culligan (2010); High range source: EPA (2002)
Users of public water systems	$10,553,542	$17,589,237	161,074	268,456		$65.52		Source: Pacific Institute Nitrate Survey (2010)
TOTAL	**$40,169,276**	**$89,600,723**						

It should be noted that, even with our proposed revisions to the economic analysis, a lack of data will continue to severely limit the economic impact assessment. As the Technical Memo makes clear, much of the data necessary for understanding the economic impact of the current program and proposed alternatives is not available. Monitoring of ground water quality in California is neither systematic nor comprehensive, making the extent of contamination and the identification of sources extremely difficult. Ironically, this is in part due to the current regulations under the ILRP. In a strange twist, we observe that only if an ILRP alternative were implemented would there be some of the data needed to analyze the full costs of the current program.

Chapter 1, Analytical Objectives and Approach

To fully assess the costs of the current ILRP and proposed alternatives, the analytical approach of the Economic Analysis must include several key costs, including the costs to all affected public drinking water systems, drinking water consumers, and private well owners. A rich literature has documented the range of potential costs. The USDA report, *The Benefits of Protecting Rural Water Quality, An Empirical Analysis*, provides a succinct summary of the types of benefits from improving rural water quality (see Table 2).

Table 2. Types of benefits from improving rural water quality[5]

Use Value	In-stream 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 place 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.

The analytical approach in the ILRP Technical Memo focuses solely on the costs to Growers and Land Owners, and Administrative Costs of the Program. While an assessment of each of the above types of benefits as they relate to the ILRP may not be necessary, we urge staff to at least integrate the costs related to *consumptive services, given the profound implications for public health and quality of life of millions of California residents impacted by this program*. To do so, the analytical approach should be revised to include the following question: *What are the costs to water system operators, well owners and drinking water consumers due to agricultural activities potentially*

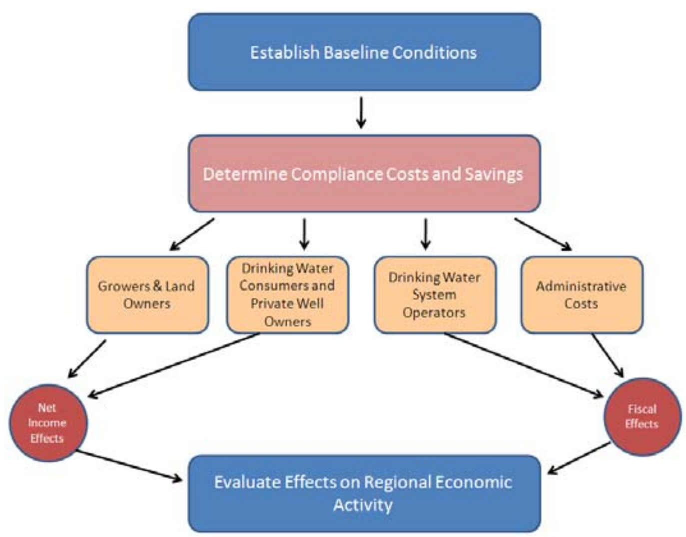

Figure 1. Suggested Revision to Economic Analysis Approach to the Irrigated Lands Regulatory Program

[5] United States Department of Agriculture Economic Research Service (1995). *The Benefits of Protecting Rural Water Quality, An Empirical Analysis*. Accessed September 20, 2010 from http://www.ers.usda.gov/Publications/AER701/.

regulated under long-term ILRP alternatives?

Addressing this question would allow staff to estimate savings associated with ILRP alternatives, such as the potential savings to drinking water systems no longer having to invest in nitrate mitigation. These savings are both fiscal, such as in the case of grants made by the California Department of Public Health to mitigate nitrate contamination of ground water, and they are economic, such as the case of consumers with unsafe drinking who may no longer have to purchase bottled water in addition to paying flat fees for tap water. Figure 1 here represents a revised version of the diagram of the analytical approach to the economic analysis on page 1-1.

This approach to assessing costs and benefits was undertaken in the 2002 U.S. EPA analysis, *The Benefits of Reducing Nitrate Contamination in Private Domestic Wells Under CAFO Regulatory Options.* For each regulatory option being considered, the EPA reported the Expected Reductions in Number of Households with Well Nitrate Concentrations above 10 mg/L. In this case, staff used existing research on Willingness to Pay for such drinking water quality improvements to estimate the economic benefit to these households.[6] In the following section we adapt this methodology to estimate the costs of the ILRP to drinking water consumers.

ILRP Costs to Domestic Well Owners

According to the Groundwater Ambient Monitoring & Assessment Program, there are an estimated 600,000 private domestic wells in California and 10 percent of those tested have nitrate levels above the legal limit.[7] According to the USGS, there is a population of 813,390 in Central Valley counties who rely on domestic wells (See Table 3).[8] The percentage of wells contaminated per county in the Central Valley ranged widely, from less than 1% in Tehama to 40% of those tested in Tulare County. The extent to which contamination originates from agricultural run-off is not known, in part due to a lack of systematic monitoring of run-off and ground water quality. Most researchers agree that agriculture is the leading source of nitrate contamination of ground water in the Central Valley.[9]

Table 3. Population Served by Domestic Wells in Central Valley Counties

County	Total Population	Population served by domestic wells	As percentage of total population
Butte	203,170	38,400	19%

[6] U.S. EPA (2002) *The Benefits of Reducing Nitrate Contamination in Private Domestic Wells Under CAFO Regulatory Options.* Accessed online September 20, 2010 from http://www.epa.gov/npdes/pubs/cafo_benefit_nitrate.pdf.

[7] State Water Resources Board, Groundwater Ambient Monitoring & Assessment Program (2010). *Summary of Detections Above a Drinking Water Standard, GAMA Domestic Well Project.* Accessed on September 20, 2010 from http://www.swrcb.ca.gov/gama/domestic_well.shtml.

[8] USGS (2000) Estimated Use of Water in the United States County-Level Data for 2000. Online at http://water.usgs.gov/watuse/data/2000/index.html

[9] United States Geological Survey (1995) *Water Quality in the San Joaquin-Tulare Basins, California, 1992-95.* Accessed on September 20, 2010 from http://pubs.usgs.gov/circ/circ1159/sec6.html.

Colusa	18,800	7,060	38%
Fresno	799,410	41,730	5%
Glenn	26,450	12,260	46%
Kern	661,650	76,050	11%
Kings	129,460	20,990	16%
Madera	123,110	49,070	40%
Merced	210,550	53,140	25%
Placer	248,400	25,920	10%
Sacramento	1,223,500	64,030	5%
San Joaquin	563,600	102,340	18%
Shasta	163,260	25,560	16%
Stanislaus	447,000	85,170	19%
Sutter	78,930	21,310	27%
Tehama	56,040	32,590	58%
Tulare	368,020	103,420	28%
Yolo	168,660	33,460	20%
Yuba	60,220	20,890	35%
TOTAL	**5,550,230**	**813,390**	**15%**

The cost of ensuring safe drinking water to the users of these wells must cover strategies for reducing nitrate levels or accessing an alternative water source. This may include installing treatment technology or a filter, drilling a new well, or buying bottled or vended water. According to Culligan, one of the leading purveyors of filter systems in the Valley, a typical nitrate filter costs $336 per fixture per year including maintenance.[10] Our cost estimate assumes that only 10 percent of the Central Valley population relying on domestic wells have high nitrates. Assuming only 60% of the contamination affecting these 16,713 households have agricultural run-off as a contaminating activity, the costs for each of them to install a Culligan filter total at $5,615,734. In the above-mentioned EPA report on CAFOs, a domestic well owner's Willingness to Pay for nitrate levels being brought down to the MCL is valued at $718.67 per year (inflation adjusted from $583 in 2001 dollars). Using this as the annual cost per household, the annual costs to domestic well owners amount to $12,011,486.

ILRP Costs to Drinking Water Consumers

It has been well documented that households impacted by groundwater contamination incur significant costs to avoid contaminated tap water. A series of studies using the "avoidance cost" method—that is, "assessing the costs of actions taken to avoid or reduce damages from exposure to groundwater contaminants"—have demonstrated that household responses to contamination of domestic water supplies is far from inexpensive and that these expenditures must be taken into consideration in valuing the costs and benefits of groundwater protection.[11,12,13] To avoid nitrate-

[10] Culligan (2010) Personal Communication 9/17/10
[11] Abdalla, Charles W. *Measuring Economic Losses from Ground Water Contamination: An Investigation of Household Avoidance Costs.* Water Resources Bulletin Vol. 26 No. 3, 451-463.
[12] Collins, Alan R. and Scott Steinback (1993). *Rural Household Response to Water Contamination in West Virgina.* Water Resources Bulletin Vol. 29 No. 2, 199-209.
[13] Laughland, Andrew S., Musser, Lynn M., Musser, Wesley N., and James S. Shortle (1993). *The Opportunity Cost of Time and Averting Expenditures for Safe Drinking Water.* Water Resources Bulletin Vol. 29 No. 2, 291-299.

contaminated tap water, households must install costly reverse osmosis filters, order domestic water service to their home, or buy gallons of vended and bottled water for consumptive household uses such as cooking and drinking.

In the summer of 2010, Pacific Institute conducted a survey of 21 out of the 28 households connected to the community water system, Beverly Grand Mutual Water Company, which was in violation of the 45 mg/L MCL for nitrate concentration. Respondents were asked a series of questions about household socioeconomic and demographic information, perception of contamination, household water use, and expenditures on tap water, filters, and alternative sources of water (such as vended and bottled water).

Preliminary analysis of the survey shows that households that are aware of contamination in their water and that drink and cook with exclusively non-tap sources of water pay on average 77% more than they would have had they solely used tap water for these consumptive household uses. On average, non-tap water expenditures for these households constituted 2% of household income, although some households spent up to 4.2% of their income on bottled and vended water for use in the home. On average, households that exclusively use non-tap sources of water for cooking and drinking spend $5.46 per person per month on vended and bottled water for use in the home (although some households spent up to $14.08 per person per month). This suggests that, collectively, the 1.3 million people connected to water systems with nitrate-contaminated groundwater supplies between 2005-08 spent approximately $7.1 million per month, or $85.2 million per year to avoid nitrate-contaminated water. How much of these costs of nitrate contamination can be attributed to agriculture is impossible to know without effective ground water monitoring, so we are left with an upper figure on the costs associate with the ILRP. A GIS analysis of land use surrounding the systems in violation would allow staff to identify systems in close proximity to agricultural land uses, a methodology regularly employed by researchers.

Costs to Public Water Systems

The costs of nitrate contamination of ground water extend to all public water systems with high nitrates and agriculture as a contaminating activity. The Economic Analysis in its current form only looks at the impact on community water systems, one subset of public systems. This analysis should be expanded to include other types of public water systems.

The assumption of the size of wells that small community water systems must replace is also flawed. Our understanding is that even small water systems must install wells that pull 2,000 gallons per minute due to fireflow requirements. This may explain why the cost estimate from Newkirk and Dewby is significantly lower than the costs of projects in applications for proposition 50 and 84 funding. We suggest you use the latter figures and abandon the Newkirk and Dewby figures for this reason.

Even with these corrections, the resulting cost estimates will represent a lower bound because of several categories of costs that are not quantifiable. These include the costs of treatment for health problems resulting from exposure to agriculture-related nitrates. They also include the future costs to water systems that may no longer have the option of simply digging a new well. Several systems have reported that they dug deeper wells to avoid nitrates only to then find ground water with high arsenic levels and, as a result, incurred the additional costs of treatment for arsenic.

Regional Economic Impacts

With the same rational that the economic analysis expects costs of the ILRP to agricultural businesses to have a ripple effect on the region, costs to well owners, public systems, and water consumers will have indirect economic effects. The current economic analysis excludes costs to community water systems from the analysis of regional economic impacts, removing them as a factor in the analysis of regional economic impacts. Although the limited data linking nitrate contaminated drinking water to agricultural activities constrains such a quantitative analysis, excluding them from the modeling relegates these significant effects to being inconsequential in the comparison of program alternatives.

As the technical memo states,

> Because businesses in a local economy are linked together through purchases and sales of goods and services produced in the region, an action that has a direct effect on one industry is likely to have an indirect effect on firms providing production inputs and support services, as the demand for their products also changes. As household income is affected by the changes in regional economic activity, additional induced effects are generated by increased household spending.

Similarly, changes to agricultural run-off brought about by the Program will have an economic ripple effect on drinking water consumers, domestic well owners, water system operators, and water system funding agencies, which in turn will have indirect effects on local economies.

In current form, the analysis of Regional Economic Impact focuses on the “value of agricultural production and spending to comply with program requirements and to implement management”, and measures impact with economic indicators for Total industry output, Personal income, and Employment. This implies that the only changes in economic conditions resulting from the IRLP will be limited to within the farm properties and related businesses. Significant economic gains could also result for local drinking water users, water agencies, and local governments, among others. These gains will have a multiplier effect as they free up revenue for increased spending in other areas. As household expenses on avoiding nitrate-contaminated water are reduced, disposable income increases and allows for a rise in consumer spending. Public revenue currently dedicated to drinking water improvements necessary because of nitrates could be invested in public services or infrastructure projects, both of which would contribute to employment and profits.

Recognizing the Trend of Increasing Ground Water Nitrate Levels

The current economic analysis and Draft EIR assume that future nitrate levels in ground water will mirror current levels, but data suggests otherwise. Our analysis of the data shows that nitrate levels are increasing in a third of locations, and the number of wells where nitrate levels exceed federal health standards is likely to double in the next ten years. Looking at wells monitored by GAMA in Kern County, we carried out a regression analysis to estimate the number of wells currently under the MCL that can be expected to rise above this threshold in the next ten years. Using a database including all nitrate measurements from 1980 to present in the GAMA database for Kern County, we selected wells that had ten or more samples recorded (678 wells), and fit a trend line of nitrate concentration versus time, using ordinary least squares regression. We used the uncertainty associated with this relationship to calculate the percent likelihood of exceeding the 45 mg/L threshold in 2010, 2015, and 2020.

Table 3. Trend analysis of nitrate levels in Kern County wells

Groundwater Basin	Total number of Wells	Number of wells with greater than 75% likelihood of exceeding MCL in 2010	Number of wells with greater than 75% likelihood of exceeding MCL in 2015	Number of wells with greater than 75% likelihood of exceeding MCL in 2020
Antelope Valley (6-44)	29	0	0	0
Brite Valley (5-80)	4	0	0	0
Castac Lake Valley (5-29)	6	0	0	0
Cuddy Canyon Valley (5-82)	5	0	0	0
Cuddy Ranch Area (5-83)	4	0	0	0
Cuddy Valley (5-84)	6	0	0	0
Cummings Valley (5-27)	14	2	2	3
Fremont Valley (6-46)	11	0	0	0
Indian Wells Valley (6-54)	36	0	0	0
Kern River Valley (5-25)	55	4	7	8
Mil Potrero Area (5-85)	2	0	0	0
No Basin Found	67	1	2	2
San Joaquin Valley - Kern County (5-22.14)	417	24	37	50
Tehachapi Valley East (6-45)	3	0	0	0
Tehachapi Valley West (5-28)	18	2	2	2
Walker Basin Creek Valley (5-26)	1	0	0	0
TOTAL	**678**	**33**	**50**	**65**

Based on our analysis, we found 33 wells where the likelihood of exceeding the MCL is 75%. In 2015, this increases to 50 and in 2020 rises to 65 (See Table 3). This is almost a doubling of the number of wells with nitrate levels above the MCL by 2020, an increase from 5% to 10% of monitored wells. Based on current trends, we estimate that the number of wells exceeding the MCL in Kern County will double in the next ten years.

This trend of increasing nitrates in one of the counties with the most intensive agriculture, combined with the significant numbers of water systems and users and private well owners encountering nitrate-contaminated ground water points to the need for a systematic approach to monitoring and mitigating this contamination at the source. Additional costs that we are not able to quantify include those related to health outcomes caused by exposure to nitrates – including health services and pain and suffering – as well as the costs to ecosystems. A recent study looked at the effect of nitrate levels on California ecosystems, and found that 35% of the state's conifer forests, chaparral and oak woodlands were "at risk of major vegetation type change" due to nitrates.[14] This will undoubtedly have an effect on local economies and quality of life.

Despite the limitations of our economic analysis due to limited data availability, there is no question as to the existence of significant costs resulting from regulated by the ILRP. Without an analysis of how the current program contributes to these costs and their indirect effects, and a comparison of the program alternatives' impact on these costs, the Board will not have the information it needs to make an intelligent and balanced decision on the program's future.

Thank you for considering these comments, please contact us with any questions or requests for additional information.

Sincerely,

Eli Moore, Eyal Matalon, and Matt Heberger

Pacific Institute

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[14] Fenn, M.E., et al. (2010), Nitrogen critical loads and management alternatives for N-impacted ecosystems in California, *Journal of Environmental Management*. Doi:10.1016/j.jenvman.2010.07.034

Appendix 1

Public Water Systems in the San Joaquin Valley with $\geq$ 1 Nitrate Violation, 2005-2008

System Name	Population Served	Number of quarters with nitrate violations
AKIN WATER CO.	50	1
Arvin Community Services Dist	11,847	1
ATWATER, CITY OF	28,100	1
BEAR VALLEY CSD	7,400	3
BEVERLY-GRAND MUTUAL WATER	108	10
BROCK MUTUAL WATER COMPANY	500	3
BUEHNER HOUSES	25	1
BUEHNER WATER SYSTEM - WEBER COMPLEX	100	1
California Water Service – Stockton	171,777	5
CANYON MEADOWS MUTUAL WATER	325	1
CENTRAL WATER CO.	170	1
CENTURY MOBILE HOME PARK	50	3
Ceres, City of	40,943	12
CHERRY LANE TRAILER PARK	100	5
City of Modesto, DE East Turlock	500	1
City of Modesto, DE Grayson	1,100	12
Corcoran, City of	25,528	3
COUNTRY WESTERN MOBILE HOME PARK	120	3
CWS – LAKELAND	789	16
CWS - North Garden	15,998	6
Del Oro River Island Serv Terr #1	975	12
Del Oro River Island Serv Terr #2	87	8
Denair Community Services District	3,225	2
Dinuba, City of	19,297	1
DUCOR CSD	850	1
EAST OROSI C.S.D.	106	5
EAST WILSON ROAD WATER COMPANY	35	6
EDMUNDSON ACRES WATER SYSTEM	550	3
EL MONTE VILLAGE M.H.P.	100	5
EL NIDO MOBILE HOME PARK	250	6
ENOS LANE PUBLIC UTILITY DISTRICT	250	2
Fairview Water Company, LLC	100	9
FAIRWAYS TRACT MUTUAL	250	4
FAWCETT FARMS	50	1
FCSA #32/Cantua Creek	230	3
FCWWD #42/Alluvial & Fancher	257	3
FRESNO, CITY OF	457,511	15
GOOSELAKE WATER COMPANY	102	2
GREEN RUN MOBILE ESTATES	100	7
HARVEST MOON MUTUAL WATER CO	180	1
HILLVIEW WATER CO-RAYMOND	243	12
HILMAR COUNTY WATER DISTRICT	5,000	2

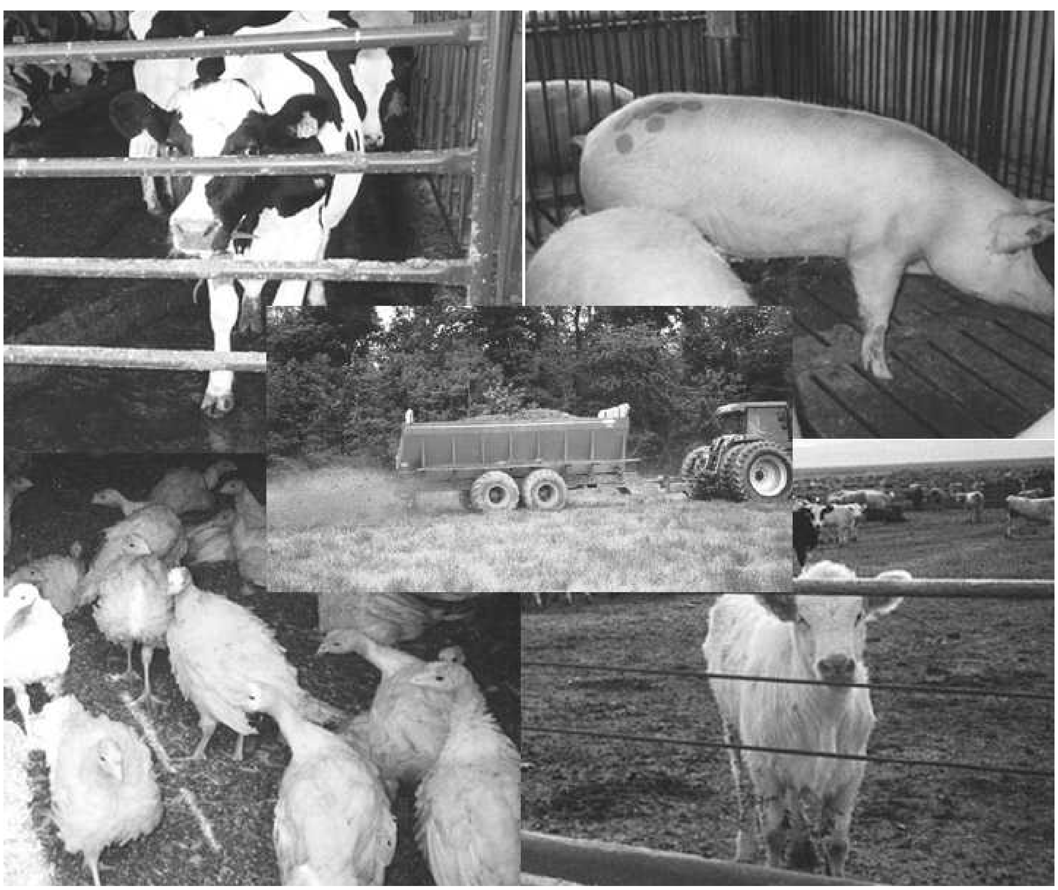
Ivanhoe Public Utility Dist	4,474	12
Josephina and Enrique Water System	32	2
KERN VALLEY MUTUAL WATER	100	1
LEMON COVE WATER CO	200	7
Lindsay, City of	11,185	1
LSID – Tonyville	400	14
Madera County M.D. #10A - Madera Ranchos	2,255	2
MALAGA COUNTY WATER DISTRICT	900	2
MANTECA, CITY OF	66,000	2
MD#43 MIAMI CREEK KNOLLS	100	3
MD#85 VALETA MUTUAL WATER COMPANY	45	3
MERCED, CITY OF	80,453	1
MODESTO MOBILE HOME PARK, LLC	200	6
Modesto, City of	212,000	8
MOJAVE PUD	3,900	1
MONTEREY PARK TRACT COMMUNITY SERVICE DI	186	3
MOUNTAIN MESA WC	1,035	3
NORSEMAN M.H.P.	70	1
OASIS PROPERTY OWNERS ASSOCIATION	80	1
Orosi Public Utility District	7,318	1
PATIO VILLAGE MOBILEHOME PARK	75	2
PATTERSON, CITY OF	20,875	2
Poplar Comm Service Dist	2,200	4
Porterville Developmental Center	2,576	8
Porterville, City of	51,467	1
RAINBIRD VALLEY MUTUAL WATER COMPANY	188	2
RIPON, CITY OF	14,575	8
RODRIQUEZ LABOR CAMP	110	3
San Joaquin County-Raymus Village	1,086	1
SAN JOAQUIN ESTATES MUTUAL	220	6
SEVENTH STANDARD MUTUAL	110	3
SIERRA BREEZE MUTUAL WATER COMPANY	144	3
SIERRA MUTUAL WATER CO	39	2
SON SHINE PROPERTIES	400	1
SOULTS MUTUAL WATER CO.	100	13
STALLION SPRINGS CSD	4,300	2
Strathmore Public Util Dist	1,904	14
SUNNYSIDE CONVALESCENT HOSP	116	1
TEHACHAPI, CITY OF	7,218	11
Terra Bella Irrigation District – TBT	2,340	15
TOOLEVILLE WATER CO.	300	4
TRAVER WATER LLC	500	1
TRIPLE R MUTUAL WATER CO.	400	13
VALLEY VIEW ESTATES MUTUAL WATER CO	81	8
Vaughn WC INC	27,150	1
Wasco, City of	16,657	3
WATERTEK - GRANDVIEW GARDENS	350	7
WESTLAKE VILLAGE M H P	350	1
WHEELER FARMS HEADQUARTERS	25	15
WILSON ROAD WATER COMMUNITY	72	1

ZONNEVELD DAIRY	141	1
Total (93 Systems)	1,342,280	430

United States
Environmental Protection
Agency

The Benefits of Reducing Nitrate Contamination in Private Domestic Wells Under CAFO Regulatory Options

December 2002



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EPA-821-R-03-008

The Benefits of Reducing Nitrate Contamination in Private Domestic Wells Under CAFO Regulatory Options

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December 2002

ACKNOWLEDGMENTS AND DISCLAIMER

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CONTENTS

Appendices

EXHIBITS

ACKNOWLEDGMENTS AND DISCLAIMER

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ACRONYMS

AFOs	animal feeding operations
CAFOs	confined animal feeding operations
CARL	Colorado Association of Research Libraries
CES	constant elasticity of substitution
CPI	Consumer Price Index
CVM	contingent valuation method
CWS	community water supply
DOE	dichotomous choice question followed by an open-ended valuation question
ELGs	effluent limitations guidelines
EPA	U.S. Environmental Protection Agency
EVRI	Environmental Valuation Resource Inventory
GI	gastrointestional
IOE	information on current local government expenditures on public health and safety services followed by an open-ended valuation question
MCL	Maximum Contaminant Level
NAWQA	National Water Quality Assessment
NPDES	National Pollutant Discharge Elimination System
NPLA	National Pollutant Loading Analysis
pv	present value
VBSs	vegetated buffer strips
VSI	value of statistical illness
VSL	value of statistical life
WTP	willingness-to-pay
YN	yes/no
YNP	yes/no and protest votes

EXECUTIVE SUMMARY

Confined animal feeding operations (CAFOs) can contaminate aquifers and thus impose health risks and welfare losses on those who rely on groundwater for drinking water or other uses. Of particular concern are nitrogen and other animal waste-related contaminants (which come from manure and liquid wastes) that leach through soils and ultimately reach groundwater. Nitrogen loadings convert to elevated nitrate concentrations at household and public water system wells, and elevated nitrate levels in turn pose a risk to human health.

The federal health-based National Primary Drinking Water Standard for nitrate is 10 mg/L. This Maximum Contaminant Level (MCL) applies to all community water supply systems, but not to households that rely on private wells. As a result, households served by private wells are at risk of exposure to nitrate concentrations above 10 mg/L, which EPA considers unsafe for sensitive subpopulations (e.g., infants). Nitrate above concentrations of 10 mg/L can cause methemoglobinemia ("blue baby syndrome") in bottle-fed infants (National Research Council, 1997), which causes a blue-gray skin color, irritableness or lethargy, and potentially long-term developmental or neurological effects. Generally, once nitrate intake levels are reduced, symptoms abate. If the condition is not treated, however, methemoglobinemia can be fatal. No other health impacts are consistently attributed to elevated nitrate concentrations in drinking water; however, other health effects are suspected.

U.S. Census data (1990) show that approximately 13.5 million households located in counties with animal feeding operations (AFOs) are served by domestic wells. According to the nationwide USGS Retrospective Database (1996), the concentrations of nitrate in 9.45% of domestic wells in the United States exceed the 10 mg/L threshold. Thus, EPA estimates that approximately 1.3 million households in counties with AFOs are served by domestic wells with nitrate concentrations above 10 mg/L.

EPA's proposed revisions to the National Pollutant Discharge Elimination System (NPDES) regulation and effluent guidelines would affect the number and type of facilities subject to regulation as CAFOs, and would also introduce new requirements governing the land application of manure. As a result, EPA anticipates that its regulatory proposal will reduce nitrate levels in household wells. In light of clear empirical evidence from the economics literature that households are willing to pay to reduce nitrate concentrations in their water supplies — especially to reduce concentrations from above the MCL to below the MCL — the anticipated improvement in the quality of water drawn from private domestic wells represents a clear economic benefit. This report estimates these benefits for each of the 12 regulatory scenarios evaluated.

Exhibit S-1 provides an overview of the approach used to estimate the benefits of well nitrate reductions. The analysis begins by developing a statistical model of the relationship between nitrate concentrations in private domestic wells and a number of variables found to affect nitrate levels, including nitrogen loadings from CAFOs. It then applies this model, in combination with the projected change in nitrogen loadings from CAFOs under each regulatory scenario, to characterize the distribution of expected changes in well nitrate concentrations. Next, the analysis applies this distribution to the number of households served by private domestic wells to calculate (1) the increase in the number of households served by wells with nitrate concentrations that are below the MCL and (2) the incremental change in nitrate concentrations for households currently served by wells that are below the MCL. Finally, the analysis employs estimates of households' values for reducing well nitrate concentrations to develop a profile of the economic benefits of anticipated improvements in well water quality.

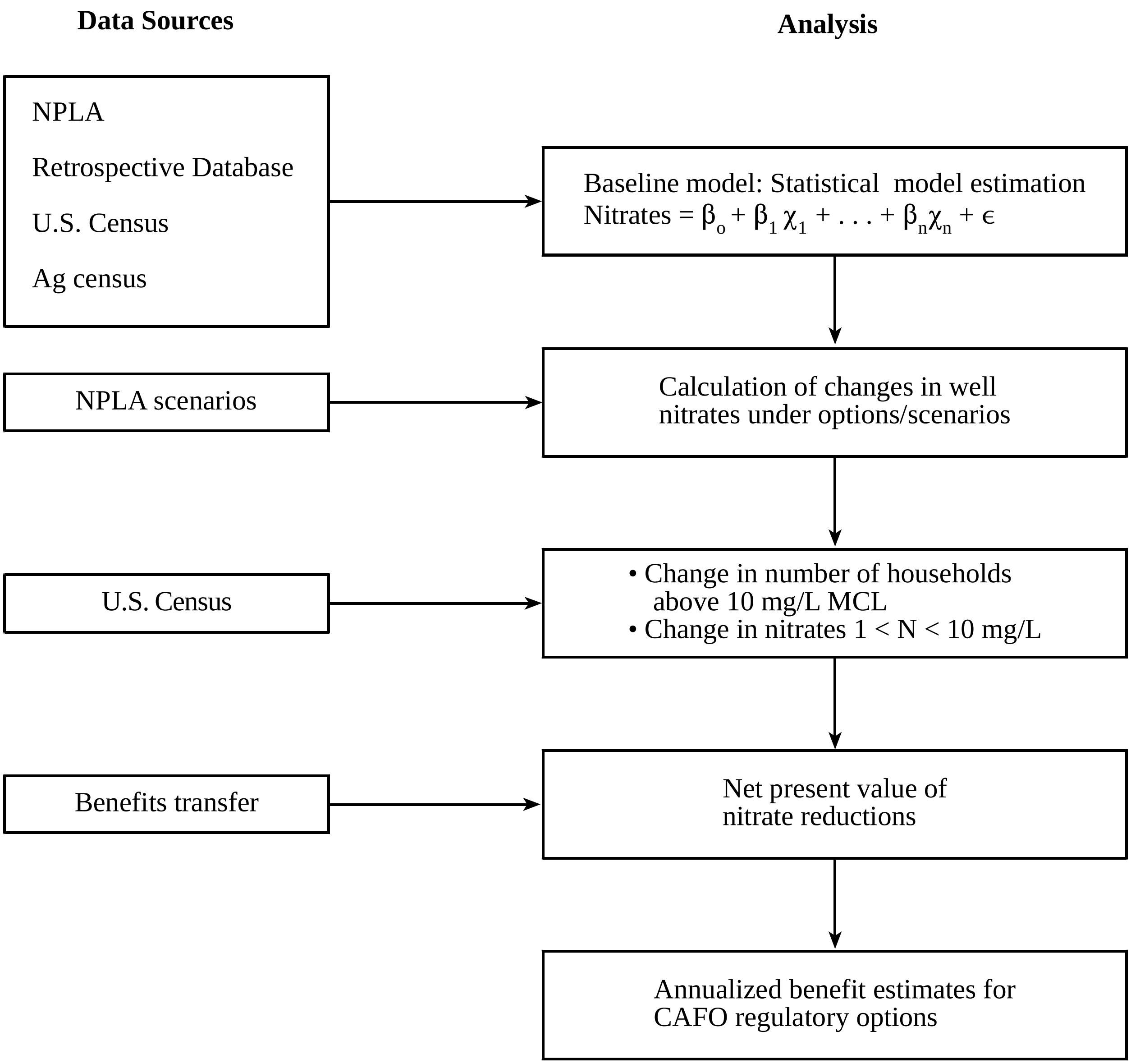
Regression Analysis: Baseline Model

The approach begins with the use of regression analysis to develop a model characterizing the empirical relationship between well nitrate concentrations and a number of variables that may affect nitrate levels, including nitrogen loadings from CAFOs. The primary purpose of the model is to estimate the effects of nitrogen loadings from CAFOs on domestic well nitrate concentrations. The model also accounts for other sources of nitrogen and well characteristics that could affect this relationship. Controlling for other sources of nitrogen in the model ensures that decreases in nitrogen loadings from CAFOs as a result of regulatory activities will not overestimate impacts on well nitrate concentrations.

The variables included in the model are based on a review of hydrogeological studies that have observed statistical relationships between groundwater nitrate concentrations and various other hydrogeological and land use factors. Data for the dependent variable, domestic well nitrate concentrations, were obtained from the USGS Retrospective Database. Data were compiled for 2,985 observations in 374 counties.[1] The regression model includes variables characterizing nitrogen loadings from animal feeding operations [data obtained from the National Pollutant Loading Analysis (NPLA)], agricultural fertilizers and atmospheric deposition (data obtained from the USGS Retrospective Database), and septic systems (data obtained from the 1990 U.S. Census). The model also includes variables describing well depth, soil type, and land use characteristics around the well (data obtained from the USGS Retrospective Database).

1. There are 678 counties with estimated nitrate loadings >0. Of these, 374 have one or more wells with enough data available to be included in the analysis.

Exhibit S-1
Overview of Analytic Approach



Calculation of Changes in Well Nitrates

After estimating the regression model using baseline loading information, the model was used to estimate expected values for well nitrate concentrations, both for baseline and for each of the 12 alternative regulatory scenarios. The calculation of expected values under each scenario employed data on AFO nitrogen loadings obtained from the NPLA; these loadings vary across the regulatory scenarios, reflecting different manure application rates, manure management practices, and other factors. To examine the impact of alternate regulatory scenarios on well nitrate concentrations, the AFO loadings variable is the only independent variable that changes value; the values for all other variables are held constant. Exhibit S-2 shows the reductions nationally in total nitrogen loadings from CAFOs under the different regulatory options/scenarios derived from the NPLA for the 2,637 counties in the NPLA indicated as having CAFOs.

Exhibit S-2
Nitrogen Loadings from CAFOs: Mean, Total, and Percent Reduction from Baseline (2,637 counties)

Option/Scenario	Mean (pounds per county)	Total (pounds nitrogen)	Percent Reduction from Baseline
Baseline	225,506	594,660,440	0%
Option 1 — Scenario 6	177,739	468,697,687	21%
Option 1 — Scenario 7	186,852	492,729,289	17%
Option 1 — Scenario 8	186,849	492,721,424	17%
Option 1 — Scenario 9	188,542	497,185,643	16%
Option 2/3 — Scenario 6	173,403	457,263,669	23%
Option 2/3 — Scenario 7	182,465	481,159,066	19%
Option 2/3 — Scenario 8	182,460	481,147,874	19%
Option 2/3 — Scenario 9	184,233	485,822,603	18%
Option 5 — Scenario 6	191,161	504,090,590	15%
Option 5 — Scenario 7	194,906	513,968,264	14%
Option 5 — Scenario 8	194,902	513,957,068	14%
Option 5 — Scenario 9	195,737	516,158,080	13%

Source: Calculations based on NPLA (TetraTech, 2002).

Discrete Changes from above the MCL to below the MCL

As noted above, under the baseline scenario, it is estimated that approximately 1.3 million households in counties with AFOs are served by domestic wells with nitrate concentrations above 10 mg/L. To estimate the impact of alternative CAFO standards on the number of wells that would exceed the nitrate MCL, the mean percentage reduction in nitrate concentrations predicted under each regulatory scenario was applied to the observed nitrate concentration values that the USGS Retrospective Database reports.

Based on the resulting values, the percentage reduction in the number of wells with nitrate concentrations exceeding 10 mg/L was calculated. These values were then applied to the baseline estimate of the number of households in counties with AFOs that are served by domestic wells with nitrate concentrations above 10 mg/L. Based on this analysis, it is estimated that the regulatory scenarios evaluated would bring between 106,000 and 149,000 households under the 10 mg/L nitrate threshold. Exhibit S-3 shows the number of households expected to have well nitrate concentrations reduced from above the MCL to below the MCL for each of the options/ scenarios.

Exhibit S-3
Expected Reductions in Number of Households with Well Nitrate Concentrations above 10 mg/L and in Total Nitrates under 10 mg/L

Scenario	Reduction in Number of Households above the MCL	Total Expected National Nitrate Reduction (mg/L)[a]
Option 1 — Scenario 6	148,705	854,326
Option 1 — Scenario 7	120,823	716,007
Option 1 — Scenario 8	120,823	716,007
Option 1 — Scenario 9	120,823	695,662
Option 2/3 — Scenario 6	148,705	927,730
Option 2/3 — Scenario 7	120,823	788,287
Option 2/3 — Scenario 8	120,823	788,305
Option 2/3 — Scenario 9	111,529	768,221
Option 5 — Scenario 6	144,058	836,895
Option 5 — Scenario 7	106,882	717,982
Option 5 — Scenario 8	106,882	717,995
Option 5 — Scenario 9	106,882	701,889
a. For wells at or below the MCL at baseline and above 1 mg/L.		

Incremental Changes below the MCL

Households currently served by wells with nitrate concentrations below the 10 mg/L level may also benefit from incremental reductions in nitrate concentrations. For purposes of this analysis, it is assumed that such incremental benefits would be realized only for wells with baseline nitrate concentrations between 1 and 10 mg/L; presumably, an individual would not benefit if nitrate concentrations were reduced to below background levels, which are assumed to be 1 mg/L. Incremental reductions in nitrate concentrations for wells that remain above the MCL are not calculated because we do not have reliable value estimates to apply to these changes. We also have not calculated values for incremental changes below the MCL for households that are above the MCL at baseline and below the MCL after new regulations. These values are potentially already captured by benefit estimates used in the benefits transfer for wells achieving safe levels. This analysis thus takes a conservative approach to benefits estimation.

For each regulatory scenario, the mean and median reduction in nitrate concentrations for wells with baseline values between 1 and 10 mg/L was estimated. The last column of Exhibit S-3 indicates the aggregate reduction in mg/L expected nationally for wells with nitrate levels below the MCL before new regulations. Between 5.3 and 5.8 million households would benefit from these incremental reductions depending on the option and scenario.

Valuation of Predicted Reductions in Well Nitrate Concentrations

The benefit valuation analysis relies on a benefits transfer approach to value predicted reductions in well nitrate concentrations. Three general steps were used to identify and apply values for benefits transfer. First, a literature search identified potentially applicable primary studies. Second, we evaluated the validity and reliability of the studies identified. Primary evaluation criteria included the applicability and quality of the original study, each evaluated on multiple criteria such as sample size, response rates, significance of findings in statistical analysis, etc. And, third, values for application to CAFO impacts were selected and adjusted. Through the review and evaluation of the relevant literature, three studies were selected to provide the primary values used for the benefit transfer:

- Poe and Bishop (1992): per household values for changes in well nitrate concentrations from above the MCL to below the MCL.

- Crutchfield et al. (1997): values incremental changes in nitrate concentrations below the MCL.

- De Zoysa (1995): values incremental changes in nitrate concentrations below the MCL.

The Consumer Price Index (CPI) was used to convert the annual mean household willingness-to-pay values obtained from these studies to 2001 dollars. Exhibit S-4 shows the point value estimates used for benefits transfer.

Exhibit S-4
Willingness-to-Pay Values Applied to Benefits Transfer

Study	Value	2001$
Poe and Bishop	Annual WTP per household for reducing nitrates from above the MCL to the MCL	$583.00
Average of Crutchfield et al. and De Zoysa	Annual WTP per mg/L between 10 mg/L and 1 mg/L	$2.09

Total Annual Benefits

Based on the benefit estimates from Exhibit S-4 and the changes in well nitrates under the potential regulatory options/scenarios indicated in Exhibit S-3, Exhibit S-5 indicates the estimated total annual (undiscounted) benefits. These values are then adjusted for the timing of the reductions in well nitrates and discounted over the time frame of the analysis.

Exhibit S-5
Undiscounted Annual Values under CAFO Regulatory Scenarios
(millions 2001$)

Scenario	Total WTP for Discrete Reduction to MCL	Total WTP for Incremental Changes below 10 mg/L	Total
Option 1 — Scenario 6	$86.70	$1.79	$88.48
Option 1 — Scenario 7	$70.44	$1.50	$71.94
Option 1 — Scenario 8	$70.44	$1.50	$71.94
Option 1 — Scenario 9	$70.44	$1.45	$71.89
Option 2/3 — Scenario 6	$86.70	$1.94	$88.63
Option 2/3 — Scenario 7	$70.44	$1.65	$72.09
Option 2/3 — Scenario 8	$70.44	$1.65	$72.09
Option 2/3 — Scenario 9	$65.02	$1.61	$66.63
Option 5 — Scenario 6	$83.99	$1.75	$85.74
Option 5 — Scenario 7	$62.31	$1.50	$63.81
Option 5 — Scenario 8	$62.31	$1.50	$63.81
Option 5 — Scenario 9	$62.31	$1.47	$63.78

Timing of Benefits

It is estimated that approximately 75% of affected wells would realize the new predicted nitrate levels within 20 years (Hall, 1996). Assuming that the number of wells achieving these levels increases linearly over time, this translates to approximately 3.7% of wells achieving new steady state conditions each year. This analysis assumes this rate, so that all affected wells reach the new levels in 27 years.

Discounting

Three discount rates are used to calculate the net present value of the benefits from reductions in domestic well nitrate levels: 3%, 5%, and 7%.

Annualized Benefit Estimates

Because the benefit flows are uneven over time, the annualized values are presented. The annualized present value represents the constant level of benefits that would yield the same discounted present value, using the same rate of discount, as the uneven flow of benefits. Exhibit S-6 presents the annualized benefit estimates for the total annual benefits shown in Exhibit S-5. For instance, for Option 5 Scenario 7, using the 27 year timepath and a 3% discount rate, the present value of benefits would be $1,458.4 million. A constant benefit flow of $43.75 million discounted at 3% shown in Exhibit S-6 for Option 5 Scenario 7 would generate $1,458.4 million in total present value of benefits, also discounted at 3%.

Exhibit S-6
Annualized Present Value of Option/Scenarios Using Different Rates of Discount (millions 2001$)

Scenario	3% Annualized Value	5% Annualized Value	7% Annualized Value
Option 1 — Scenario 6	$60.67	$49.20	$41.03
Option 1 — Scenario 7	$49.32	$40.00	$33.36
Option 1 — Scenario 8	$49.32	$40.00	$33.36
Option 1 — Scenario 9	$49.29	$39.98	$33.34
Option 2/3 — Scenario 6	$60.77	$49.29	$41.11
Option 2/3 — Scenario 7	$49.43	$40.08	$33.43
Option 2/3 — Scenario 8	$49.43	$40.08	$33.43
Option 2/3 — Scenario 9	$45.68	$37.05	$30.90
Option 5 — Scenario 6	$58.78	$47.67	$39.76
Option 5 — Scenario 7	$43.75	$35.48	$29.59
Option 5 — Scenario 8	$43.75	$35.48	$29.59
Option 5 — Scenario 9	$43.73	$35.46	$29.58

CHAPTER 1
INTRODUCTION AND OBJECTIVES

The U.S. Environmental Protection Agency (EPA) is revising and updating the two primary regulations that ensure that manure, wastewater, and other process waters generated by confined animal feeding operations (CAFOs) do not impair water quality. The proposed regulatory changes affect the existing National Pollutant Discharge Elimination System (NPDES) provisions that define and establish permit requirements for CAFOs, and the existing effluent limitations guidelines (ELGs) for feedlots, which establish the technology-based effluent discharge standard that applies to regulated CAFOs. The existing regulations were promulgated in the 1970s. EPA is revising the regulations to address changes in the animal industry sectors over the last 25 years and to clarify and improve implementation of CAFO requirements.

CAFOs can contaminate groundwater and thus cause health risks and welfare losses to people relying on groundwater for their potable supplies or for other uses. Of particular concern are nitrogen and other animal waste-related contaminants (which come from manure and liquid wastes) that leach through the soils and the unsaturated zone and ultimately reach groundwater. Nitrogen loadings convert to elevated nitrate concentrations at household and community system wells, and elevated nitrate levels in turn pose a risk to human health. The proposed regulation will generate benefits by reducing nitrate levels in household wells, and there is clear empirical evidence from the economics literature indicating that households are willing to pay to reduce nitrate concentrations in their water supplies.

The federal health-based National Primary Drinking Water Standard for nitrate is 10 mg/L, and this Maximum Contaminant Level (MCL) applies to all community water supply (CWS) systems. Households relying on private wells are not subject to the federal MCL for nitrate; however, levels above 10 mg/L are considered unsafe for sensitive subpopulations (e.g., infants). Nitrate above concentrations of 10 mg/L can cause methemoglobinemia ("blue baby syndrome") in bottle-fed infants (National Research Council, 1997), which causes a blue-gray skin color, irritableness or lethargy, and potentially long-term developmental or neurological effects. Generally, once nitrate intake levels are reduced, symptoms abate. If the condition is untreated, however, methemoglobinemia can be fatal. No other health impacts are consistently attributed to elevated nitrate concentrations in drinking water.

U.S. Census (1990) data show that there are currently approximately 13.5 million households with domestic wells located in counties with animal feedlot operations. CAFOs present a potential contaminant source to groundwater, particularly via nitrogen leached from manure. Manure from these operations is generally managed either by storing it in a waste lagoon, where waste has the potential to leak through the lining or overflow onto the surrounding ground and leach nitrogen into the groundwater, or by spreading it on surrounding farm fields, where, depending on the rate and timing of the applications, the soil hydrology, and precipitation, nitrate may leach into the groundwater. Nitrate is of particular concern because it leaches easily into groundwater, and is one of the most frequently found groundwater contaminants (Lichtenburg and Shapiro, 1997).

CAFOs are currently covered by existing effluent guidelines at 40 CFR Part 412 and permit regulations at 40 CFR Part 122. The effluent guidelines regulations, which require the largest CAFOs to achieve zero discharge of waste to surface waters except under extreme storm events, have not been sufficient to resolve water quality impairment from feedlot operations. Under the current permit regulations, a CAFO is a facility in one of the following three categories:

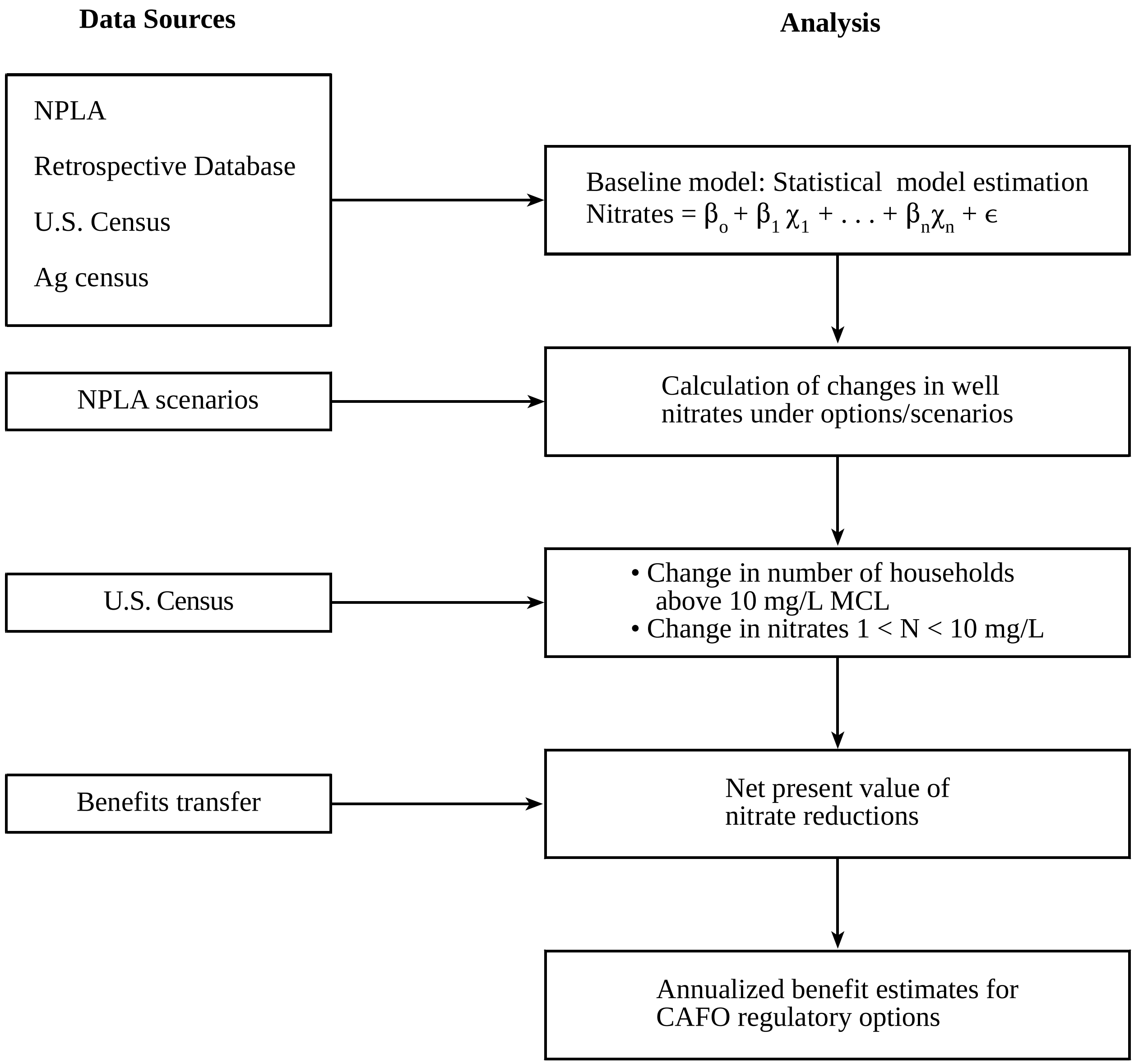
- more than 1,000 animal units confined at the facility
- 301-1,000 animal units confined and the facility also meets one of the specific criteria addressing the method of discharge [40 CFR Part 122 Appendix B]
- designated as a CAFO on a case-by-case basis if the NPDES-authorized permitting authority determines that it is a significant contributor of pollution to waters of the United States [40 CFR part 122.23(c)].

This report estimates benefits for national reductions in nitrate concentrations in private domestic wells achieved by changing regulations for effluents from CAFOs. Benefits achieved via this regulation for public and surface water systems are considered elsewhere in this regulatory analysis. The proposed regulatory options include different criteria for the definition of a CAFO, therefore changing the number of operations that will have to comply with the proposed regulations. They also include requirements for the quantity and rate of land application of manure, as well as water quality reporting. The current regulations address only controls at the feedlot; land application of manure is not addressed. This analysis evaluates the potential benefits from eight regulatory scenarios.

1.1 OVERVIEW OF BENEFIT ASSESSMENT METHOD

The assessment of benefits of well nitrate reductions from CAFO regulations followed the multistep process outlined in Exhibit 1-1.

Exhibit 1-1
Analysis Plan and Data Sources



To estimate the benefits achieved by reducing nitrogen loadings from animal manure and thus improving groundwater quality, we first established baseline water quality under current loadings and current regulations using available data on nitrate concentrations in individual wells. These data, described further in Chapter 2, were obtained from a national database of groundwater quality. We then used these baseline data for nitrate concentrations and data on current nitrogen loadings by county to model the relationship between nitrate concentrations and nitrogen loadings. Our model also included significant explanatory variables such as well depth and soil hydrological characteristics that were identified from a literature survey. We then applied the parameter estimates generated from this model to projected loadings under each regulatory scenario to estimate changes in nitrate concentrations in the wells for each regulatory option.

From these data we established the percentage of wells above the MCL (10 mg/L) under each scenario, and the nitrate reduction for wells that were already below 10 mg/L at baseline. We then extrapolated these values to the total number of household units on private wells in the country to estimate the number of households that would have nitrate concentrations reduced from above the MCL to below the MCL, and how many households that were already below the MCL at baseline and would have further water quality improvements under the regulatory scenarios.

After reviewing studies that estimated household-level monetary benefits of improving water quality through reduced nitrate concentrations, we established a range of values for both reducing nitrate from above the MCL to below the MCL and reducing nitrate concentrations in wells that were already below the MCL at baseline. Using benefits transfer methods, we then estimated the total monetary benefits that could be achieved under each scenario, based on the number of households brought from above the MCL to below the MCL and the number of households that achieved water quality improvements below the MCL.

Monetary benefits were estimated annually over a 100 year time period to capture the time path until well nitrates would achieve a steady state following implementation of each regulatory option. We assumed that it would take 27 years to achieve the steady state. Discounting was applied to determine net present values, and these were then annualized to derive a benefit estimate to be used in comparison to annualized cost estimates. Sensitivity analysis was performed to examine how annualized benefit estimates change using different discount rates, years until clean, and per household benefit values.

1.2 REPORT STRUCTURE

Chapter 2 discusses the choice of variables to include in modeling the relationship between loadings from CAFOs and well nitrate concentrations, and data sources used in the analysis. This chapter also includes information on the methods used to calculate loadings for each scenario and descriptions of each scenario.

Chapter 3 discusses the model of the relationship between nitrogen loadings and well nitrate concentrations. Statistical analyses and parameter estimates from analyses based on this model, assuming a gamma distribution, are included. Chapter 3 also discusses the results from running the parameter estimates through each of the regulatory scenarios with different loadings and the subsequent changes in well nitrate concentrations.

Chapter 4 discusses the benefits transfer method in detail.

Chapter 5 discusses the groundwater valuation studies used in this analysis, including a ranking of their relevance to this study, the various methods that each used to estimate benefits, and their respective values for reducing groundwater contamination.

Chapter 6 provides a summary of benefit estimates using the different assumptions regarding which approach to apply for extrapolating from the model to the population, the time until a new steady state is achieved, and the discount rate used. Omissions, biases, and uncertainties in the analysis are discussed here.

References are provided for both the nitrate modeling and benefits analysis portions of this report.

The appendices include information on nitrogen loading data sets, details of the statistical analyses of the nitrogen-nitrate relationship, and tables summarizing the literature used in the benefits transfer analysis.

CHAPTER 2
LOADINGS AND WELL NITRATE CONCENTRATIONS

This chapter identifies variables affecting nitrate contamination in wells that can be used to model the relationship between nitrogen loadings and nitrate concentrations in wells. We then review the sources of data used to model this to the regulatory scenarios to be used for benefits analysis.

2.1 RELATIONSHIP BETWEEN NITROGEN LOADINGS AND WELL NITRATE CONCENTRATIONS

We selected the variables to include in the model used to predict nitrate concentrations in groundwater under different regulatory scenarios based on our review of hydrogeological studies that have observed statistical relationships between groundwater nitrate concentrations and various other hydrogeological and land use factors. Although the groundwater monitoring and modeling studies reviewed for this report covered different geographic areas and focused on varying nitrogen sources (septic systems, agricultural fertilizers, animal feedlots), certain variables were significant across many of the studies. These studies were generally regional or local in scope, and obtained their data by sampling the wells directly.

2.1.1 Included Variables

Nitrogen application rates, whether from agricultural fertilizers, animal wastes, or private septic systems, were the most consistent and significant factor affecting nitrate levels in wells across the studies reviewed for this analysis (Rausch, 1992; Spalding and Exner, 1993; Clawges and Vowinkel, 1996; Richards et al., 1996; Lichtenberg and Shapiro, 1997; Lindsey, 1997; Burrow, 1998; CDC, 1998; Letson et al., 1998; Nolan et al., 1998; Kerr-Upal et al., 1999).

Nitrate is found in groundwater because of surface applications of two forms of the nutrient nitrogen: nitrate and amine groups (of which nitrogen is a component). Generally nitrogen from fertilizer is already in the nitrate form, which leaches more readily into the soil. Nitrogen from manure and septic systems generally occurs as large organic molecules called amine groups. Once in the soil, these large molecules convert to nitrate and ammonia as microbes break down the organic matter. The ammonia then volatizes as a gas into the atmosphere, and the nitrate leaches through the soil and potentially into groundwater. This process takes a few hours to a few weeks, depending on the soil conditions (M. Hall, CH2M Hill, pers. comm, Sept. 15, 2000).

Studies that investigated the effects of animal manure production on groundwater nitrate concentrations found manure to be positively correlated with groundwater nitrate. Animal waste lagoons were associated with elevated groundwater nitrate concentrations, particularly as the distance to the water table decreased (Miller et al., 1976; Ritter and Chirnside, 1990; North Carolina Division of Water Quality, 1998). Farms that applied manure as fertilizer tended to have higher nitrate concentrations in groundwater as well (Rausch, 1992; Swistock et al., 1993; Clawges and Vowinkel, 1996; Richards et al., 1996; Lindsey, 1997; Letson et al., 1998; Kerr-Upal et al., 1999).

Several studies focused on agricultural practices such as type of crop and crop rotations, and how they may be correlated with nitrate concentrations in groundwater. Swistock et al. (1993), Stuart et al. (1995), and Lichtenberg and Shapiro (1997) found corn production to be associated with higher nitrate levels because corn demands higher fertilizer input and extensive irrigation, which increases the rate at which nitrate leaches to the groundwater. Spalding and Exner (1993) found that groundwater beneath any row-cropped, irrigated area tended to have higher nitrate levels. Rausch (1992) found that tillage practices, which change the amount of organic matter in the root zone, and planting nitrogen-fixing legumes as a part of the crop rotation cycle decreased the quantity of nitrate available for leaching and were associated with lower levels of nitrate in groundwater.

The proximity of septic systems to wells was found to be a small, but significant, contributing factor to elevated nitrate concentrations in groundwater in several studies (Carleton, 1996; Richards et al., 1996; CDC, 1998; Nolan et al., 1998).

Well depth was also frequently found to be a significant factor, inversely related to nitrate concentrations in wells, regardless of nitrate source (Detroy et al., 1988; Ritter and Chirnside, 1990; Kross et al., 1993; Spalding and Exner, 1993; Swistock et al., 1993; Sparco, 1995; Lichtenberg and Shapiro, 1997; Ham et al., 1998; North Carolina Division of Water Quality, 1998). Swistock et al. (1993) found that wells deeper than 100 ft tended to have significantly lower nitrate concentrations, and Kross et al. (1993) found that wells deeper than 45 ft generally had much lower nitrate concentrations.

A number of studies identified at least one geological characteristic as a significant factor affecting nitrate concentrations. Two studies found unconfined aquifers to be associated with elevated nitrate in groundwater (Lichtenberg and Shapiro, 1997; Lindsey, 1997). Other studies found higher nitrate levels associated with more permeable, well-drained soils (Ritter and Chirnside, 1990; Spalding and Exner, 1993; Sparco, 1995; Burrow, 1998; Chen, 1998; Ham et al., 1998; Nolan et al., 1998; Kerr-Upal et al., 1999). Several studies explored the possibility of using DRASTIC, an index intended to reflect the groundwater pollution potential of a region. DRASTIC incorporates several hydrogeological factors: drainage, aquifer recharge rate, aquifer media, soil media, topography, impact of the vadose zone, and hydraulic conductivity of the aquifer. All found positive correlations between county-level DRASTIC scores and groundwater nitrate concentrations, but none were statistically significant. All agreed that DRASTIC scores are not reliable predictors of groundwater nitrate levels (U.S. EPA, 1990; Rausch, 1992; Richards et al., 1996). We included DRASTIC scores in some early regression analyses, but they did not strengthen the results and were thus dropped from further analysis.

Different types of land use near wells are also associated with higher groundwater nitrate. Several studies found agricultural land use in general to be associated with higher groundwater nitrate than other land uses (Rausch, 1992; Spalding and Exner, 1993; Swistock et al., 1993; Mueller et al., 1995; Sparco, 1995; Carleton, 1996; Clawges and Vowinkel, 1996; Richards et al., 1996; Nolan et al., 1998). Results from Carleton's study, for example, suggest that nitrate concentrations in West Windsor Township in New Jersey have decreased as residential use has replaced agriculture.

2.1.2 Omitted Variables

Because of incomplete or unreliable national data, we did not include all the significant variables identified in these studies. First, well construction and age were cited as significant variables in several studies (Spalding and Exner, 1993; Swistock et al., 1993; Richards et al., 1996; Burrow, 1998; CDC, 1998). In general, older wells were more vulnerable to nitrate contamination because the casing could be cracked, allowing surface contaminants to enter the groundwater. Different construction materials and methods also affected how easily nitrate or other pollutants could reach the groundwater supply via direct contamination at the wellhead. This variable, however, is often unreliable because it is generally obtained by surveying well owners and relying on their subjective assessment of how and when the well was constructed. No reliable data on well construction were available nationally.

Second, the distance from a pollutant source to well was significantly correlated with groundwater nitrate in several studies (Rausch, 1992; Swistock et al., 1993; CDC, 1998; Ham et al., 1998; North Carolina Division of Groundwater Quality, 1998). Although spatial data were available for well locations, no spatial data on the location of animal feedlots, cropland, and septic systems were available for our analysis.

Two studies in the literature surveyed (Sparco, 1995; Lichtenberg and Shapiro, 1997) developed models to predict nitrate concentrations in groundwater, based on the variables described above. These models were not used in the final analysis because they incorporated either spatial or temporal data that are not available for a national level assessment. In addition, as discussed below, our analysis indicates that a gamma distribution more closely matches the distribution of nitrate concentrations than the linear and lognormal distributions assumed in the other models. Aside from these differences, the final model used similar variables and assumptions regarding land use and hydrogeology.

2.2 DATA SOURCES

The independent variables for the analysis were chosen based on the preceding literature review to identify variables that have significant impact on nitrate concentration in groundwater. Data availability also dictated which variables were included in the model. The data for this analysis were obtained primarily from three sources: the USGS Retrospective Database, the National Pollutant Loading Analysis (TetraTech, 2002),[1] and the 1990 U.S. Census. Appendix A provides additional detail on how these data sets were combined and some additional summary statistics.

2.2.1 USGS Retrospective Database

The Retrospective Database contains water quality and land use data from 10,426 wells sampled from 725 counties in 38 states. The data were gathered between 1969 and 1992. Data relevant to this analysis were:

- nitrate concentrations in the well, in mg/L
- water use of the well (e.g., irrigation, domestic)
- nitrogen inputs from manure and fertilizer loadings
- atmospheric nitrogen deposition
- depth to water in the well
- soil hydrologic group, a measure that includes runoff potential, soil permeability, depth to water table, depth to an impervious layer, water capacity, and shrink-swell potential
- number of septic systems per acre near the well
- land use near the well
- region of the United States the well is located in.

Within any given county, the reported nitrogen loadings data used in the data analysis are the same (nitrogen loading data vary between counties but not within counties). These data were obtained from other published data sources (U.S. Census, U.S. Census of Agriculture, and U.S. EPA fertilizer sales data) that report at a county level. Water use, well depth, and nitrate

1. The National Pollutant Loading Analysis (NPLA) (TetraTech, 2002) comprises three Excel spreadsheets provided by EPA. These are described in Section 2.2.3.

concentrations are reported by well. The Retrospective Database was the limiting data source for this analysis because it includes only 725 counties out of approximately 3,100 counties in the United States. Implicit in our use of these data and in our analysis is the assumption that the Retrospective Database is representative of private domestic wells nationwide. Potential biases related to this assumption are discussed in Chapter 6.

In the Retrospective Database, approximately 18% of the reported nitrate concentrations in domestic wells were at or below the detection limit (0.05 mg/L). Because this database is a compilation of several databases, these nondetects are reported in several ways: at the detection limit, at half the detection limit, and at zero. To standardize our data we set all values reported at or below the detection limit to the detection limit. In addition, because this analysis is concerned only with the benefits gained from reducing nitrate contamination in domestic wells, we eliminated wells with nondomestic uses (stock, irrigation, urban, and unknown).

2.2.2 1990 U.S. Census

We obtained the total number of household units on wells nationwide and the number of household units using septic systems in each county in the United States from the 1990 U.S. Census.[2] The number of households on septic systems in each county, divided by the total acres in the county, provided an estimate of septic system density for the analysis.

2.2.3 National Pollutant Loadings Analysis

The National Pollutant Loading Analysis (NPLA; Tetra Tech, 2002) provided estimates of leached nitrate from animal feedlot operations under different regulatory options. The NPLA developed a national estimate of pollutant load reductions expected from meeting the requirements of revised animal feeding operation effluent guidelines.

The estimate is based on loadings for the current effluent guidelines (preregulation baseline) and after the implementation of revised effluent guidelines (postregulation modeling scenarios). The national estimate of nutrient, pathogen, and metal loadings is based on conditions identified on a broad range of sample farms. These farm conditions consisted of animal groupings of various size classes, current management practices and animal waste management systems, and regionally based physiographic information regarding the soil, rainfall, hydrology, crop rotation, and other factors for a given region of the country. Model farms were developed from county, regional, and national data sources, including the 1997 Census of Agriculture data.

2. County level source of water data did not appear to be available from the 2000 U.S. Census.

Total nitrate leached to groundwater was based on the size and type of operations in the area and subsequent manure produced, crop nutrient removal rates, and the GLEAMS model. GLEAMS can be used to evaluate the effects of various agricultural management practices on the movement of pollutants to water sources, using hydrology, erosion, and biochemical processes to evaluate pollutant transport.

Along with the NPLA, the U.S. EPA also provided the estimated number of facilities of each size in each county and the percentage of facilities that would be subject to regulation in each state. We assume this percentage to be constant for all counties within that state. In general, all “large” operations will be subject to regulations, and varying percentages of “medium” operations will be regulated. These data included loadings from beef, dairy, veal, swine, layer, broiler, and turkey operations.

Details on how these data were combined to estimate total nitrogen loadings in each county are provided in Section 2.3.

2.3 REGULATORY SCENARIOS USED FOR BENEFITS ANALYSIS

The regulatory scenarios evaluated in this analysis are based on different combinations of two factors: limits for land application of manure (options), and variations on how many facilities will be subject to the regulation (scenarios). EPA analyzed nitrate loadings under 12 option-scenario combinations plus the baseline conditions, for a total of 12 regulatory scenarios. All regulatory scenarios will entail common criteria, which include best management practices in the feedlots (stormwater diversions, lagoon/pond depth markers, periodic inspections, record keeping); mortality handling requirements; nutrient management planning and record keeping (soil and manure sampling requirements); and prohibition of manure application within 100 ft of surface water, tile drain inlets, and sinkholes.

The land application options are based on either total nitrogen applied (Option 1), total phosphate applied (Option 2), or total phosphate applied plus covered lagoons (Option 5).[3] The nitrogen and phosphate content of the manure and subsequent manure application rates under these options are based on the type of animal operation. Under all three options, manure will be land-applied at allowable manure application rates, providing adequate nutrients for crop uptake, runoff, and leaching.

The percentage of affected facilities differs according to the size of the facility and state. The scenarios for the number of affected facilities determine how many small, medium, and large facilities will be defined as CAFOs under the regulation, and thus become subject to the

3. Option 3 is similar to Option 2 but also requires liners for lagoons. Since the leached nitrate loadings are the same for our analysis under Options 2 and 3, these are reported simply as Option 2/3 throughout this report.

nitrogen-based or phosphate-based limits. The size categories are based on the number of animals at the facility and vary by animal type (see Exhibit A-1 in Appendix A). For example, a large beef farm is defined as any farm with more than 1,000 head of cattle, and a medium beef farm is defined as those farms with 300 to 1,000 head of cattle. In comparison, by definition, a swine farm is large if it has more than 2,500 pigs, and a turkey farm is large if it has more than 55,000 turkeys. Small facilities are not subject to the regulation, and therefore are not included in the baseline analysis. All large facilities are considered CAFOs and therefore subject to nitrogen-based or phosphate-base limits. At baseline, some medium-sized operations are regulated and therefore produce varying nitrogen loadings.

Similarly, most dry poultry operations were assumed to produce unregulated loadings at baseline. Under the regulatory scenarios, however, some of these operations will be regulated and produce reduced loadings.

Exhibit 2-1 summarizes the key nutrients, percentage of facilities that will be regulated, and how a CAFO will be defined, based on animal type and size, for each scenario.

In the NPLA, animal operations are divided into two general categories: those currently with controls at the feedlot and those currently without controls at the feedlot. Those currently with controls are assumed to be in complete compliance with existing regulations. Operations with controls are modeled to have different loadings than operations without controls. Different loadings data are provided in the NPLA for operations with and without controls.

Loadings for the scenarios, including baseline, are calculated based on the assumption that facilities with controls produce one amount of loadings and facilities without controls produce loadings equivalent to baseline. For all scenarios, including baseline, the regulated percentage of operations will produce "regulated loadings," and the remaining percentage will produce "baseline loadings." The equation for calculating total loadings for one category of facility (e.g., medium beef) in one county is:

$$\begin{aligned}&\text{Total Loadings for Type of Operation (AnimalX, SizeY) in a county} = \\ &\ (\text{\% of facilities regulated} * \text{Scenario loadings-regulated} * \text{Number of facilities}) + \\ &\ [(1 - \text{\% of facilities regulated}) * \text{Baseline loadings-unregulated} * \text{Number of facilities}].\end{aligned} \quad (2\text{-}1)$$

This equation generates the total loadings for operations of each animal type and size in each county. The loadings are then summed across all operations (all animal types and relevant facility sizes) to get total county loadings.

Exhibit 2-1
Characteristics of Benefits Analysis Scenarios

Regulatory Scenario	Key Nutrient	Percentage of Facilities Regulated	Size of Facility Subject to Regulation
Baseline	Manure application not regulated	100% of large AFOs, plus medium AFOs that meet certain requirements	All large, some medium
Option 1 — Scenario 6	Nitrogen	100% of large and medium AFOs	All large, all medium
Option 1 — Scenario 7	Nitrogen	100% of large AFOs, plus medium AFOs that meet certain requirements	All large, some medium
Option 1 — Scenario 8[a]	Nitrogen	New NPDES conditions for identifying medium-sized CAFOs, plus qualifying dry poultry and immature swine and heifer operations	All large, some medium
Option 1 — Scenario 9[b]	Nitrogen	100% of large AFOs, all medium AFOs regulated under current rules	All large, no medium
Option 2/3 — Scenario 6	Phosphate	100% of large and medium AFOs	All large, all medium
Option 2/3 — Scenario 7	Phosphate	100% of large AFOs, plus medium AFOs that meet certain requirements	All large, some medium
Option 2/3 — Scenario 8[a]	Phosphate	New NPDES conditions for identifying medium-sized CAFOs, plus qualifying dry poultry and immature swine and heifer operations	All large, some medium
Option 2/3 — Scenario 9[b]	Phosphate	100% of large AFOs, all medium AFOs regulated under current rules	All large, no medium
Option 5 — Scenario 6	Phosphate	100% of large and medium AFOs	All large, all medium
Option 5 — Scenario 7	Phosphate	100% of large AFOs, plus medium AFOs that meet certain requirements	All large, some medium
Option 5 — Scenario 8[a]	Phosphate	New NPDES conditions for identifying medium-sized CAFOs, plus qualifying dry poultry and immature swine and heifer operations	All large, some medium
Option 5 — Scenario 9[b]	Phosphate	100% of large AFOs, all medium AFOs regulated under current rules	All large, no medium

a. The benefits reported in later chapters for Scenario 8 represent the estimated benefits of regulating all large and some medium facilities that meet new NPDES conditions. The difference between Scenario 8 and Scenario 9 represents the increase in estimated benefits attributable to new regulations on the identified medium facilities, given that all large facilities are regulated.
b. The benefits reported in later chapters for Scenario 9 represent the benefits attributable to new regulations on all large facilities while adding no new regulations to medium facilities.

CHAPTER 3
MODELING WELL NITRATE CONCENTRATIONS

A statistical model of the relationship between nitrogen loadings and well nitrate concentrations was developed to analyze the effect of different regulatory options. An alternative to a statistical model would be representative hydrogeological models, to examine how changes in nitrogen loadings would translate into well nitrate concentrations. This approach was considered infeasible because of time and budgetary constraints as well as the likely limitation on data needed to generalize such models to the national level. As described below, though, the statistical model attempts to capture the effects of several variables that would also be used in a hydrogeological model, such as well depth, soil type, and land use.

The statistical modeling approach uses existing data to estimate the relationship between sources of nitrogen and well nitrate concentrations. This approach allows us to control for non-CAFO sources of nitrogen, including septic systems, fertilizers, and natural (background) levels of nitrate.

3.1 MODEL VARIABLES

Analysis of the relationship between loadings and well nitrate concentrations is based on the following linear model:

$$\begin{aligned}\text{Nitrate (mg/L)} = ß_0 + ß_1 \text{ ag dummy} + ß_2 \text{ soil group} + ß_3 \text{ well depth} \\ + ß_4 \text{ septic ratio} + ß_5 \text{ atmospheric nitrogen} + ß_6 \text{ loadings ratio} + ß_7 \text{ regional} \\ \text{dummy variables.}\end{aligned} \tag{3-1}$$

Dependent Variable

Nitrate concentration is the dependent variable in this model, expressed in mg/L.

The percentage of drinking water wells with nitrate concentrations greater than 10 mg/L varies widely, depending on well, hydrologic, and pollutant characteristics. Exhibit 3-1 summarizes the widely varying percentages found in different studies. Given this wide range of values, EPA determined that the USGS Retrospective Database, which estimates that 9.45% of domestic wells have nitrate levels above 10 mg/L, contains a reasonable representation of affected wells in the United States.

Exhibit 3-1
Percentage of Wells Exceeding the MCL

Study	Location	Type of Well	% Exceeding 10 mg/L
Agriculture Canada, 1993 (as cited by Giraldez and Fox, 1995)	Ontario, Canada	Domestic farm	13
Andres 1991 (as cited in Sparco, 1995)	Sussex County, Delaware	Rural	23
CDC, 1998	Illinois, Iowa, Missouri, Kansas, Nebraska, Wisconsin, Minnesota, S. Dakota, N. Dakota	Domestic	13.4
Chen, 1998	Nemaha Natural Resources District, Nebraska	Rural	10
Kross et al., 1993	Iowa	Rural	18
National Water Quality Assessment (NAWQA) Database, USGS, 1998	National	All	16.2
Poe and Bishop, 1999	Portage County, Wisconsin	Rural	16
Retrospective Database, USGS, 1996	National	Domestic	8.9[a]
Richards et al., 1996	Ohio, Indiana, W. Virginia, Kentucky	Rural	3.4
Spalding and Exner, 1993	Iowa, Nebraska, Kansas, Texas, N. Carolina, Ohio	Rural	20, 20, 20, 8.2, 3.2, 2.7, respectively
Swistock et al., 1993	Pennsylvania	Private	9
U.S. EPA, 1990	National	Rural domestic	2.4
USGS, 1986	Upper Conestoga River Basin	Rural	40+
Vitosh, 1985 (cited in Walker and Hoehn, 1990)	Southern Michigan	Rural	34

a. 8.9% of all domestic wells in the Retrospective Database exceed 10 mg/L. From all domestic wells in this database, of the wells with enough data in order to be included in our analysis, 9.45% of the wells exceeded the 10 mg/L. As discussed further in Section 3.4, we use this 9.45% as the baseline percent of wells above the MCL for our analysis.

Actual nitrate concentrations in groundwater reported in the Retrospective Database, which were used to scale predicted values, ranged from 0 mg/L to 84.3 mg/L. Nitrate concentrations below the detection limit were reported in one of three ways: at the detection limit (0.05 mg/L), at half the detection limit, or at zero. To account for this variability, EPA set any nitrate concentration below 0.05 mg/L to 0.05 mg/L. Approximately 18% of the observations were at or below the detection limit.[1]

The intercept (β_0) will capture ambient nitrate levels in the absence of human influences from septic systems, medium and large AFOs, and atmospheric nitrogen deposition. Loadings estimates for small-sized AFOs were not available. Thus, they are implicitly included in the intercept term.

Independent Variables

The independent variables used to explain nitrate concentrations in well water are classified into two groups: well and land characteristics, and nitrogen inputs. All data are from the Retrospective Database unless otherwise noted.

Well and Land Characteristics

Ag Dummy: This is a dummy variable for agricultural land use. The ag dummy variable was set to 1 when the land use in the vicinity of the well was agricultural. For all other land uses (the remaining categories were woods, range, urban, and other), the dummy was set to zero.

Soil Group: Soil group is a classification system that integrates several hydrological variables, including runoff potential, permeability, depth to water table, depth to an impervious layer, water capacity, and shrink-swell potential. Lower numbers have the greatest permeability and water transmission rates, and are therefore more susceptible to surface pollutants (Mueller et al., 1995).

Well Depth: Well depths in the Retrospective Database ranged from 1 ft to 1,996 ft. For observations used in the regression analysis, the maximum well depth was 1,996 ft and the mean depth was 170 ft.

1. Alternative treatment of observations below the detection limit were evaluated using the gamma model described below. These alternatives included setting nondetects equal to 0.001 mg/L and setting all nitrate levels below 1 mg/L equal to 1 mg/L. These alternative specifications had little impact on the model overall, and almost no impact on the estimated loadings parameter, which is the key component of the model for CAFO loadings analysis.

Regional Dummies: EPA defined five regions for use in this analysis: Central, Mid-Atlantic, Midwest, Pacific, and Southwest.[2] A regional dummy was created for each region (equal to one if the well is in the region, equal to zero otherwise), to help account for regional differences not captured by the other independent variables included in the model. The Midwest dummy was used as the basis variable and was not included in the model. Thus the estimated parameters for each of the other dummies indicate how nitrate levels in that region compare to nitrate levels in the Midwest.

Nitrogen Inputs

Septic Ratio: The septic ratio is equal to the number of housing units using septic systems per acre in the county. The number of septic systems was obtained from the 1990 U.S. Census. County size (in acres) was taken from the 1992 Census of Agriculture.

Atmospheric Nitrogen: Estimated atmospheric nitrogen deposition in the area near each well is included in the Retrospective Database. The values used in the regression ranged from 0.54 to 8.92 pounds per acre.

Loadings Ratio: EPA calculated total nitrate loadings in each county as the total estimated leached nitrogen from AFOs (both from manure application and from a variety or sources at or near the AFO production areas), and from the application of fertilizers. EPA divided this total by total county acres to create a consistent unit across all counties. The assumption is that, in general, once nitrate leaches into the groundwater it is dispersed in a volume of groundwater proportional to the county size. EPA obtained estimates of leached nitrogen from manure and fertilizer loadings from the NPLA (TetraTech, 2002).

Exhibit 3-2 lists summary statistics for the dependent and independent variables for the 2,985 observations used in the regressions described below.

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2. The composition of the five regions is:

Midwest: ND, SD, MN, WI, IA, IL, MI, IN, MO, NE, KS
Mid-Atlantic: ME, NH, VT, NY, MA, RI, CT, OH, NJ, PA, DE, MD, VA, WV, KY, TN, NC
Pacific: CA, OR, WA, AK, HA
Central: ID, MT, WY, NV, UT, CO, AZ, NM, TX, OK
South: AR, LA, MS, AL, GA, SC, FL.

Exhibit 3-2
Summary Statistics

Variable	N	Mean	Standard Deviation	Minimum	Maximum
Nitrate Concentrations	2,985	3.57	6.51	0.05	84.30
Loadings Ratio	2,985	2.02	4.16	0.00	18.35
Atmospheric Nitrogen	2,985	5.07	1.87	0.54	8.92
Well Depth	2,985	170.07	136.11	1.00	1,996.00
Soil Group	2,985	2.42	0.66	1.00	4.00
Septic Ratio	2,985	0.03	0.03	0.00	0.15
Ag Dummy	2,985	0.78	0.42	0.00	1.00
Central Region Dummy	2,985	0.06	0.25	0.00	1.00
Mid-Atlantic Region Dummy	2,985	0.39	0.49	0.00	1.00
Pacific Region Dummy	2,985	0.12	0.33	0.00	1.00
South Region Dummy	2,985	0.07	0.26	0.00	1.00

3.2 THE STATISTICAL MODEL

EPA used regression analysis to estimate the statistical model described in Equation 3-1 using the data sources discussed in Section 2.2. EPA evaluated several different statistical models and chose a "gamma model" because it best fit the data.[3] The gamma model and the other statistical models EPA tested are discussed in detail in Appendix B.

Exhibit 3-3 provides the output of the gamma regression model. Most of the explanatory variables are significant. The exceptions are atmospheric nitrogen and the septic ratio. In addition, all have the expected sign. This implies that the model produces intuitive results and that the independent variables do help explain the variation in the nitrate levels. In particular the regression results indicate that wells on agricultural land (ag dummy) have a higher well nitrate concentrations. Wells located under less permeable soils (soil group) and deeper wells (well depth) have lower well nitrate concentrations. The positive parameter estimates for the three sources of nitrogen (septic systems, atmospheric deposition, and animal feeding operations) indicate that each source positively contributes to well nitrate concentrations. The model can thus be used to help understand how changes in the independent variables (e.g., nitrogen loadings,

3. The term gamma model is used because the chosen regression is based on a gamma distribution, rather than the normal distribution (as is used in ordinary least squares regression), or another type of distribution.

Exhibit 3-3
Gamma Regression Results

Variable	Parameter Estimate	Standard Error	Asymptotic T-Statistic	Significance
Intercept	2.201	0.194	11.352	0.000
Loadings Ratio	0.046	0.007	6.543	0.000
Atmospheric Nitrogen	0.032	0.028	1.144	0.253
Well Depth[a]	-0.171	0.012	-13.782	0.000
Soil Group	-0.384	0.044	-8.660	0.000
Septic Ratio	1.618	1.728	0.936	0.349
Ag Dummy	0.686	0.064	10.663	0.000
Central Region Dummy	-0.076	0.160	-0.475	0.635
Mid-Atlantic Region Dummy	-0.165	0.098	-1.691	0.091
Pacific Region Dummy	0.812	0.117	6.918	0.000
South Region Dummy	-0.907	0.127	-7.170	0.000
Alpha	0.497	0.010	50.639	0.000
Mean log-likelihood = -1.85646.				
N = 2,985.				
a. In the model, well depth is scaled to units of hundreds of feet.				

well depth, land use) affect the expected level of nitrate at the well. Therefore EPA used this model as the basis of its analysis of how reducing nitrogen loadings from CAFOs will affect nitrate concentrations in domestic drinking water wells.

3.3 FITTED VALUES AND SCENARIO MODELING

After estimating the gamma model using the baseline loading information, expected values for well nitrate concentrations were calculated using baseline loadings from 2,985 observations and loadings from the 12 regulatory scenarios. As described above, the 12 regulatory scenarios are based on different manure application rates, manure management practices, and monitoring requirements. Loadings for the 12 regulatory scenarios were input into the model to estimate well nitrate concentrations under these scenarios. In the analysis, the loadings ratio is the only variable that changes across scenarios.

Expected well nitrate concentrations under the 12 loadings scenarios were compared with the expected well nitrate concentrations using the baseline loadings. EPA used the changes projected from the model to calculate percentage differences in expected well nitrate concentrations under the different regulatory options and scenarios. These were calculated by dividing the difference from baseline for the expected values from the 12 different scenarios by the expected values from the baseline loadings. These percentage differences were then applied to the actual nitrate concentrations, the observed well nitrate concentrations from the Retrospective Database, to calculate well nitrate concentrations under the various scenarios. The expected percentage changes in nitrate concentration for each scenario are summarized in the last two columns of Exhibit 3-4.

Exhibit 3-4
Characteristics of Benefits Analysis Scenarios

Regulatory Scenario	Change in Nitrate Loadings from CAFOs (calculated across all counties in the loadings dataset)[b]		Nitrate (mg/L), Predicted by Gamma Model	
	Mean % Reduction	Median % Reduction	Mean % Reduction	Median % Reduction
Option 1 — Scenario 6	21.05%	13.34%	2.23%	0.40%
Option 1 — Scenario 7	16.00%	8.42%	1.89%	0.31%
Option 1 — Scenario 8	16.02%	8.42%	1.89%	0.31%
Option 1 — Scenario 9	14.46%	5.71%	1.83%	0.18%
Option 2/3 — Scenario 6	22.46%	15.24%	2.41%	0.43%
Option 2/3 — Scenario 7[a]	17.38%	9.74%	2.07%	0.35%
Option 2/3 — Scenario 8[a]	17.40%	9.74%	2.07%	0.35%
Option 2/3 — Scenario 9	15.77%	7.32%	2.02%	0.22%
Option 5 — Scenario 6	14.12%	8.65%	2.16%	0.32%
Option 5 — Scenario 7	11.52%	5.18%	1.88%	0.28%
Option 5 — Scenario 8	11.54%	5.18%	1.88%	0.28%
Option 5 — Scenario 9	10.35%	3.38%	1.84%	0.16%

a. Proposed scenarios.
b. Includes loadings from fertilizer application.

As indicated in the literature surveyed, although nitrogen loadings from CAFOs are significant contributors to nitrate concentrations in wells, they are not the only important factor. Therefore an analysis that does not incorporate these other factors, and assumes that the relationship between nitrate concentrations and nitrogen loadings is directly proportional, will overestimate the potential changes in nitrate concentrations due to decreased loadings. Exhibit 3-4 summarizes changes in nitrate concentrations as predicted by the gamma model, compared with percentage changes that would be assumed if only changes in loadings (shown in the second and third columns of Exhibit 3-4) were used to estimate nitrate concentrations.

EPA tested the ability of the gamma model to estimate small nitrate concentrations by comparing the model's intercept with the natural, or ambient, level of nitrate in groundwater in the United States.[4] Using the mean values for soil group and well depth and setting all other variables to zero (i.e., setting the ag dummy and all human nitrogen sources to zero), the model predicts an ambient nitrate concentration in the Midwest region of 1.32 mg/L on nonagricultural lands. Using the same approach, the predicted value on agricultural land is 2.63 mg/L. Several studies report natural nitrate levels ranging between 2 and 3 mg/L (Poe and Bishop, 1992; Kross et al., 1993; Poe, 1998), although one study suggests that 3 mg/L may be too high, given the high number of wells with nitrate levels below the detection limit in many groundwater monitoring studies (Spalding and Exner, 1993). Giraldez and Fox (1995) report that natural nitrate concentration in groundwater is generally about 1.0 mg/L. Therefore the model's estimate of 1.32 mg/L on non-agricultural land seems to be a reasonable estimate of nitrate concentrations in the absence of the pollution from septic systems, atmospheric deposition, and AFOs.

3.4 DISCRETE CHANGES FROM ABOVE THE MCL TO BELOW THE MCL

Census data show that approximately 13.5 million households in the United States use domestic wells and are located in counties with animal feedlot operations.[5] Based on the USGS Retrospective data, 9.45% of these wells currently exceed 10 mg/L.[6] This is roughly 1.3 million domestic wells. Applying the percentage reductions, between 107,000 and 149,000 households that are above the MCL at baseline are expected to be brought under 10 mg/L. Results are displayed in Exhibit 3-5.

3.5 INCREMENTAL CHANGES BELOW THE MCL

Many households on wells with nitrate concentrations below the MCL at baseline may also gain benefits from incremental changes in nitrate concentrations below the 10 mg/L level and above the natural level, which is assumed to be 1 mg/L (see discussion in Section 3.3). Thus EPA

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4. Technically, the intercept term includes ambient levels of nitrates as well as those induced by loadings from AFOs with less than 300 AUs since these are not included in the loadings data.

5. The NPLA data indicates that 2,637 counties in the United States have AFOs.

6. Thus 9.45% of the wells in the Retrospective Database that had enough information to be included in the gamma model (see discussion in Section 3.1) were found to have nitrate concentrations above the MCL.

Exhibit 3-5
Expected Reductions in Number of Households with Well Nitrate Concentrations above 10 mg/L

Regulatory Scenario	Reduction Using Expected Percentage Change
Option 1 — Scenario 6	148,705
Option 1 — Scenario 7	120,823
Option 1 — Scenario 8	120,823
Option 1 — Scenario 9	120,823
Option 2/3 — Scenario 6	148,705
Option 2/3 — Scenario 7[a]	120,823
Option 2/3 — Scenario 8[a]	120,823
Option 2/3 — Scenario 9	111,529
Option 5 — Scenario 6	144,058
Option 5 — Scenario 7	106,882
Option 5 — Scenario 8	106,882
Option 5 — Scenario 9	106,882

a. Proposed scenarios.

assumed that these incremental benefits are gained only for wells beginning with concentrations between 1 and 10 mg/L. EPA did not calculate values for incremental changes where well concentrations remain above the MCL because reliable value estimates do not exist for changes in incremental nitrate concentrations above the MCL.

For households that start above the MCL preregulation and move below the MCL post-regulation, EPA also did not calculate values for incremental changes below the MCL. Based on the available valuation literature (see Chapter 5) there are no reliable estimates for valuing incremental changes below the MCL in addition to valuing changes reductions to the MCL; thus counting both values could double count some portion of the benefits for these households. Exhibit 3-6 shows the average reduction in nitrate concentrations for wells between 1 and 10 mg/L at baseline, for each of the scenarios. Approximately 5.77 million households will benefit from these incremental reductions.

Exhibit 3-6
Mean and Median Reductions in Nitrate Concentrations for Wells with Concentrations between 1 and 10 mg/L at Baseline

Scenario	Mean Reduction in [N] (mg/L)	Median Reduction in [N] (mg/L)	Households Benefitting from Incremental Nitrate Reductions	Total Expected National Nitrate Reduction (mg/L)
Option 1 — Scenario 6	0.14	0.02	5,785,564	854,326
Option 1 — Scenario 7	0.12	0.02	5,729,800	716,007
Option 1 — Scenario 8	0.12	0.02	5,729,800	716,007
Option 1 — Scenario 9	0.11	0.02	5,543,918	695,662
Option 2/3 — Scenario 6	0.15	0.03	5,813,446	927,730
Option 2/3 — Scenario 7[a]	0.13	0.02	5,771,623	788,287
Option 2/3 — Scenario 8[a]	0.13	0.02	5,771,623	788,305
Option 2/3 — Scenario 9	0.13	0.02	5,595,036	768,221
Option 5 — Scenario 6	0.14	0.02	5,427,742	836,895
Option 5 — Scenario 7	0.12	0.02	5,399,860	717,982
Option 5 — Scenario 8	0.12	0.02	5,399,860	717,995
Option 5 — Scenario 9	0.11	0.01	5,292,978	701,889

a. Proposed scenarios.

3.6 TIMELINE FOLLOWING SCENARIO IMPLEMENTATION

Once new animal waste management practices are implemented, a time lag will exist between implementation of these practices and realization of lower nitrate concentrations in wells and the benefits from these reductions. The length of this time lag may be highly variable for any given well and depends on a number of site-specific variables. The following is a brief description of some of the more important variables affecting the time lag in response.

Depth to the saturated groundwater at the location where waste is applied affects the length of time required for lower concentration (assuming improved waste management at the surface) water to reach the groundwater. A considerable amount of water is stored in the unsaturated soil zone beneath agricultural areas. When new “fresh” water leaches below the zone of plant rooting (root zone), it replaces the uppermost water in this unsaturated storage, and “pushes” some of the lower water into the saturated groundwater where it can move laterally toward surrounding wells.

In many cases, relatively little change occurs in the nitrate concentration of the water between the bottom of the root zone and the top of the saturated groundwater. While the progression of the freshwater is not uniform because of faster flow along paths of preferential flow, generally the fresh water must replace all the stored water in the unsaturated zone before an improvement is seen in the groundwater immediately beneath the site of application.

In agricultural areas of the United States, depths to groundwater may vary from a few feet to over 100 ft. While some selected regions may have shallow or deep groundwater, these depths do not vary clearly according to regional patterns, since they are determined as much by landscape position and geology as by climate. Shallow groundwater is found in riparian areas and river valleys of the arid West as well as on the Atlantic coastal plain.

The amount of excess water and properties of the soil or rock in this unsaturated zone also affect the length of time required for the fresh water to reach the groundwater. A coarse-textured material such as a sandy soil may only hold 1 inch of water for each foot of soil. In this case, 1 ft of excess water infiltrating (a reasonable amount for a humid climate or a moderate irrigation in a semi-arid climate) would move the "front" of cleaner water an average of 12 ft downward. However, less coarse media such as a soil with moderate clay content may easily hold an average of 3 inches of water per foot of soil, so the same excess water infiltration will move the leading edge of the cleaner water only 4 ft downward.

Other factors that influence how quickly the nitrate concentration at a well responds to improved surface management are the amount of groundwater present, the distance between the well and the point of waste application, and the velocity and direction of regional groundwater flow. In a highly conductive aquifer with a steep groundwater gradient, the water may move a mile or more in a year. In other cases, 10 ft or 20 ft in a year is more realistic. In addition to how fast the groundwater flows, the amount of "older" water in the aquifer from which a well is drawing will affect how quickly the response to improved management is reflected in a well. If the well is drawing from 100 vertical ft of an aquifer, the upper levels of the aquifer may have nitrate concentrations reflecting relatively recent management on nearby lands, and the lower levels of the aquifer still reflect poor management from prior years. Other local factors such as pumping of other wells and other sources of aquifer inflow such as leakage from nearby reservoirs or water exchange with rivers combine to make the question of lag in well water response time highly variable and site specific.

To estimate the value of improved groundwater quality from implementation of new CAFO waste regulations, an estimate of the response time of an average well is needed. More specifically, a realistic estimate is needed of how much time it will take after regulatory implementation for the benefit of improved nitrate concentrations to be realized at the wellhead.

In sandy soils in central Kansas, Townsend et al. (1996) observed a response in the top layers of the shallow groundwater, approximately 30 feet below the ground surface, in the first year after implementation of improved surface management. The concentrations in this uppermost layer continued to improve and had dropped from near 25 mg/L to near 5 mg/L in six years. However, nitrate concentrations approximately 20 ft lower in the aquifer continued to increase during the same period.

Simulations by Hall (1996) of nitrate concentrations in the alluvial aquifer along the South Platte River in northeastern Colorado suggest that significant improvements in nitrate concentrations in the aquifer were realized as soon as a few years after implementation of improved management practices. However, in these simulations, reductions in concentrations continued for more than 50 years, with relatively rapid improvements in the first 15 years and a decreasing rate of improvement in later years as the simulated concentrations in the aquifer approached a new steady state. The new steady state was somewhat reflective of the leaching concentrations under the improved management scenario.

The South Platte alluvial setting is a highly conductive aquifer with modest regional groundwater gradients. The saturated groundwater at both the Kansas and Colorado sites is also somewhat shallow. The response times in these cases are likely to be more rapid than for the United States as a whole. Considering the range of aquifer depths and characteristics that might be expected, we have assumed that 75% of the reduction in nitrate concentrations at the well heads will be realized in 20 years. The drop in nitrate concentration is likely to be nonlinear, with more rapid declines in early years. The shape of the concentration curve through time is unknown, however, and the additional decline in concentration in later years becomes increasingly small. Without better information, for this analysis EPA has made the conservative assumption that the concentration curve is linear, resulting in an estimated period of 27 years for improved CAFO waste management to improve an aquifer to its new equilibrium (i.e., “clean”) status.

CHAPTER 4
VALUATION: BENEFITS TRANSFER

Several approaches could be used to estimate the benefits from changes in well nitrate concentrations. The first issue to address is whether to obtain primary data on potential benefits or use existing data. Given limited time and budget constraints, collecting primary data for a nationwide sample is not feasible. We thus decided to apply a benefits transfer approach to existing studies of household values for reduced well nitrate contamination.

"Benefits transfer" refers to the "application of existing valuation point estimates or valuation function estimates and data that were developed in one context to value a similar resource and/or service affected by the discharge of concern" [59 FR 1183]. In other words, benefits transfer entails applying empirical results obtained from a primary research effort conducted at one site and set of circumstances to another (similar) site and set of circumstances. In this manner, existing research findings from a "study site" can be used as an expeditious means of drawing inferences regarding the magnitude of benefits or damages associated with a change in resource conditions at a "policy site."

4.1 BENEFITS TRANSFER METHODS

There are four ways to transfer benefits: transfer an average value, transfer a function, calculate a metafunction, or calibrate a preference. Crutchfield et al. (1997) discuss transferring an average value and transferring a function, preferring transferring a function if data are available on the sociodemographic characteristics of the original study and the policy site. Walsh et al. (1992) develop what is essentially a meta-analysis of outdoor recreation demand studies for use in benefits transfer analysis, and Boyle et al. (1994) present preliminary results of a meta-analysis of groundwater valuation studies. Smith et al. (1999) discusses the preference calibration approach. These four approaches are ordered in terms of increasing data requirements, increasing costs of implementation, and increasing sophistication of the value estimates provided.

4.1.1 Transfer an Average Value

Transferring an average value has been the most common approach to benefits transfer. It entails subjective evaluation on the part of the researcher to evaluate the validity and reliability of the original studies and to make reasonable assumptions in transforming the original values into those to be used in the new application. Transferring an average value can in a sense be a qualitative meta-analysis. Adjustments are often made based on the characteristics of the original scenarios and the new scenario and on sociodemographic characteristics of the affected population (e.g., income). Primary evaluation criteria would include:

- the relevance of the commodity being valued in the original studies to the policy options being considered for CAFOs

- the quality (robustness) of the original study, evaluated on multiple criteria such as sample size, response rates, and significance of findings in statistical analysis.

Much of the summary analysis of existing studies necessary for the average value method is also necessary for the next three approaches. At a minimum, the initial work required for an average value approach provides an initial assessment of the quality and availability of data that could be used in the other approaches.

4.1.2 Transfer a Function

Transferring a function from a specific study is generally more limited than using average values from a number of different studies. Our evaluation of nitrate related groundwater valuation studies does not reveal any one study that would be best suited for this approach. The primary limitation in transferring a function is the fact that none of the studies involves a national sample of values for reducing nitrate contamination. The applicability of a single local or regional study to a national benefits assessment requires careful consideration of the likely representativeness of the original study. Loomis (1992) further examined the benefit transfer function approach and empirically tested for the transferability of a function between states. Loomis' findings suggest that benefit functions are not always directly transferable between states. This suggests that, whatever method is adopted, spatially distinct benefit estimates should be examined for consistency when transferring benefit estimates.

4.1.3 Calculate a Metafunction

Meta-analysis is a set of statistical procedures used to assess results across independent studies that address a related set of research questions. It is a method for combining the effect sizes from several studies; it is essentially an analysis of analyses (Wolf, 1986). A metafunction is the end product of a meta-analysis in which the marginal effects of study or scenario characteristics on willingness to pay are estimated. Such a function could potentially be used in a new policy situation by inputting the relevant scenario characteristics for the policy analysis to derive the relevant value estimate.

As discussed in Chapter 5, we identified 11 studies that derive values for reducing nitrates in groundwater. Our examination of the 11 nitrate valuation studies suggests that a meta-analysis of these was not reasonable for the current benefits transfer. There is considerable difference in the basic nature of many of the studies, which limits the number that would be usable in a meta-analysis. There are significant differences in the commodities being valued (e.g., certain current cleanup versus potential future cleanup of a portion of contaminated waters) and the types of values being elicited (e.g., use values versus total values versus option values).

4.1.4 Calibrate a Preference

Preference calibration is a relatively new approach to benefits transfer analysis that builds on existing methods and attempts to develop a utility-theoretic approach to benefits transfer (Smith et al., 1999). Rather than deriving a transfer function, this approach attempts to derive a model of preferences based on results from prior studies. This method may prevent errors in the other approaches that may bias value estimates either up or down. Preference calibration requires several steps:

1. Specification of a preference ordering that dictates how a "representative" individual makes decisions (such as a constant elasticity of substitution, CES).

2. Identification of relationships, axioms, and assumptions (such as utility maximizing behavior, demand is obtainable using Roy's identity, or a choke price exists) necessary so that the preference parameters are identified.

3. Derivation of a closed-form solution for a willingness-to-pay (WTP) function (e.g., compensating variation) and addition of supplemental data to identify the unknown parameters. Using data on consumer surplus values associated with marginal and/or incremental change in environmental quality to be valued by the benefits transfer and other information on variables such as income, rent, or travel costs for the representative individual, the implied values of the parameters are backed out of the WTP function.

4. With the identified and estimated parameters, the WTP function is now estimated and any set of environmental variables can be input to generate other Hicksian consumer surplus estimates.

Smith et al. (1999) do not claim the new approach necessarily results in smaller error. In fact, the authors state, ". . . the measure from preference calibration is simply a more complex set of numerical calculations." The advantage of preference calibration is that it is based on utility-theoretic behavioral theory. Preference calibration is expected to rely on a much larger set of assumption, axioms, economic relationships, and possible supplemental data than either the unit value approach or meta-analysis. The data requirements for preference calibration and the additional assumptions required to choose any one particular functional form may outweigh the benefits of using a more theory-based approach.

4.2 CHOICE OF METHODS

The average value approach is the most feasible one for analysis of potential benefits under the proposed regulatory options. In part this choice is made because of the difference between the benefits transfer approach used here and those generally discussed in the literature. Most literature discusses the transfer of benefits from a specific study situation to another specific policy situation. Adjustments are then made based on differences between the “study site” and the “policy site.” In the case of benefits of CAFO regulations, the “policy site” is all counties in which potentially regulated CAFOs are located. Given limited resources, it is not feasible to identify individual county characteristics in a manner that would allow the use of a transfer function. In particular, we do not have information on income or other sociodemographic characteristics of those individuals living in any given county who obtain their water from a private well, as opposed to sociodemographic characteristics of the general population of the county. In part to control for this, we use benefit estimates from studies that focus on private well users in situations likely to be similar to that around CAFO locations. In this manner, the original studies are more likely to already have captured sociodemographic characteristics of the “policy situation” population.

As noted above, we do not believe that there is sufficient information in the studies considered below to use a transfer function or to develop a meta-analysis that would provide information significantly better than that gained from the average price approach because of the limited number of studies and the significant methodological differences between them. The same scarcity of information and limited resources preclude the use of the preference calibration approach.

CHAPTER 5
GROUNDWATER VALUATION STUDIES

5.1 LITERATURE SEARCH AND REVIEW

The objective of the literature search was to identify studies that had developed or elicited values for changes in groundwater quality. A number of studies deal with groundwater contamination not related to nitrates. We limit the discussion here to those that focused on values for reductions in or prevention of increases in nitrate contamination for drinking water wells. Our evaluation of the literature led us to eliminate some studies that were of poorer overall quality or for which only limited information was available.

We identified 11 relevant studies through an extensive search of literature using databases, listservers, and the bibliographies of similar studies that addressed groundwater valuation. The databases searched for this study were the Colorado Association of Research Libraries (CARL), which includes the holdings of several university libraries in Colorado and the West, and the Environmental Valuation Resource Inventory (EVRI), a database compiled by Environment Canada that includes empirical studies on the economic value of environmental benefits and human health effects. Messages were sent to the ResEcon listserver, which includes approximately 700 individuals in the field of natural resource and environmental economics, soliciting suggestions for articles pertaining to groundwater valuation and nitrate contamination. Finally, several references cited in the studies that we identified using the databases and listserver were used as well.

5.2. OVERVIEW OF GROUNDWATER NITRATE VALUATION STUDIES

The following is a brief overview of the 11 studies we evaluated for inclusion in the benefits transfer. Some of the information about these studies came from more than one report or paper based on the study. Where relevant, we identified the most recent information about each study from available literature. Summary information on these studies is presented in Appendix C.

5.2.1 Crutchfield et al., 1997

Crutchfield et al. (1997), Crutchfield et al. (1995), and Crutchfield and Cooper (1997) evaluated the potential benefits of reducing or eliminating nitrates in drinking water by estimating average WTP for safer drinking water. They received survey responses from 819 people in rural and nonrural areas in four regions of the United States (Indiana, Nebraska, Pennsylvania, Washington). Using the contingent valuation method (CVM) with valuation questions in a dichotomous choice format, respondents were asked what their willingness to pay would be to have the nitrate levels in their drinking water a) reduced to “safe levels,” and b) completely eliminated. Respondents were told that this would be accomplished using a filter installed at their tap, and the cost for the installation and maintenance of the filter would be paid to a local water agency. Respondents were also asked sociodemographic characteristics such as income, age, education, and whether they currently use treated or bottled water. Crutchfield et al. used a bivariate probit estimation for responses to the dichotomous choice questions. Across all regions, the calculated willingness to pay per household to reduce nitrates to safe levels ranged from $45.42/month to $60.76/month, with a mean of $52.89. The willingness to pay to remove nitrates from drinking water ranged from $48.26/month to $65.11/month, with a mean of $54.50. Besides income and program cost, Crutchfield et al. found two variables to be significantly related to a respondent’s willingness to pay: “years lived in ZIP code” was positively correlated and “age of respondent” was negatively correlated.

Evaluation: An important advantage of the Crutchfield et al. valuation approach is that they surveyed individuals in four different areas of the country, thus providing value estimates more representative of national values. The annual WTP to reduce nitrates to the safe level ($52.89/month x 12 months) is $634.68. Crutchfield et al. compared annual per household WTP estimates from their study to three others (including Jordan and Elnagheeb, 1993, described below). Values for reducing nitrates to either safe levels or to zero are higher in Crutchfield et al. than the other three studies. Crutchfield et al.’s estimate of $634.68/household/yr is not unreasonably higher than the $412-$484/household/yr values discussed in Poe and Bishop (1992) below. The difference in values between the two programs is likely to be representative of values for incremental reductions in nitrates in drinking water. The difference between reducing nitrates to zero and reducing nitrates to safe levels is $1.61 per month. For a change between the MCL of 10 mg/L and 0 mg/L, this represents a per mg/L monthly WTP of $0.16, which is $1.92 per mg/L annually (in 1997$).

5.2.2 De Zoysa, 1995

De Zoysa (1995) and Randall and De Zoysa (1996) discuss a contingent valuation study designed to estimate the benefits from three environmental services in the Maumee River basin in northwestern Ohio, including stabilization and reduction of nitrate levels. Rural and urban areas in the river basin were sampled and one out-of-basin urban area was sampled, with 427 returned questionnaires. Using a dichotomous choice format, a portion of the respondents were asked whether they would pay different amounts, via a one time special tax, to reduce nitrate contamination from fertilizer applied to fields. Under the hypothetical scenarios, nitrate concentrations would be reduced from the current range of 0.5-3.0 mg/L to a range of 0.5-1.0 mg/L. Individuals were also asked questions regarding sociodemographic characteristics, preferences for priorities for public spending, and how they used the resource in question (e.g., how many trips they had taken to the area). From these responses, Randall and De Zoysa formed two datasets: one that included only yes/no (YN) responses, and another that included yes/no and protest votes (YNP). The multivariate analysis was conducted using a probit model; income, the level of priority placed on groundwater protection, and the interest in increasing government spending on education, healthcare, and vocational training all were positively and significantly correlated with willingness to pay to improve groundwater quality.

Randall and De Zoysa reported various WTP estimates using median and lower bound mean estimates for groundwater, surface water, and wetlands programs or combinations of these programs. For this analysis, we examine "stand-alone" WTP estimates for groundwater programs that would reduce nitrates in groundwater. Median WTP for groundwater ranged from $71.03 for the YN responses to $20.80 for YNP responses. Lower bound mean WTP for groundwater ranged from $88.49 for the YN responses to $52.78 for YNP responses. Randall and De Zoysa expressed a preference for the YNP models because they felt there was no strong reason to assume that the protest responders had nonzero values. They also stated that for policy purposes the mean values are the appropriate measure for which the "lower bound mean" provides a lower bound estimate.

Evaluation: The reduction in groundwater nitrate levels is from a range of 0.5 to 3.0 mg/L to a range of 0.5 to 1.0 mg/L. Taking range means, the reduction in nitrates is from 1.75 mg/L to 0.75 mg/L, or a reduction of 1.0 mg/L. Using the lower bound mean values from the YNP model, this represents a WTP of $52.78 per mg/L change in nitrate concentrations for incremental changes below the 10 mg/L MCL. Since the valuation question was posed as a one-time special tax, we can annualize the $52.78 per mg/L, which represents a net present value (since the program would continue indefinitely). Using a 3% discount rate, this translates into an annual WTP of $1.61 per mg/L ($2.69 using a 5% discount rate and $3.76 using a 7% discount rate).

5.2.3 Delavan, 1997

Using a CVM survey of 1,000 residents in two counties in southeastern Pennsylvania (with a 68% response rate), Delavan (1997) estimated willingness to pay to improve groundwater quality (in 10 years, 75% of wells would meet the MCL). Delavan used CVM with two survey formats: one presented a dichotomous choice question followed by an open-ended valuation question (DOE), and the other presented information on current local government expenditures on public health and safety services followed by an open-ended valuation question (IOE). Subjects were also asked questions on their duration of residence, the current quality/safety of their water, and their prior knowledge of water quality issues. Respondents were told that they would be assessed a special tax annually for 10 years to increase the percentage of wells satisfying the MCL from 50% to 75% in their area. Tobit analysis was used to model the relationship between explanatory variables and open-ended WTP, and a logit model was used to model protest bidders. Mean annual WTP was $44.78 for the DOE surveys and $29.26 for the IOE surveys with protest bidders, and $67.85 and $47.16, respectively, without protest bids. Delavan found that at household incomes above $50,000, respondents' concern for their own safety as it relates to drinking water, the priority respondents feel that government should place on protecting groundwater, and respondents' perception of safety with and without the program were all significantly and positively correlated with respondents' willingness to pay. He also found that males were more likely to pay more for groundwater protection.

Evaluation: Delavan designed and thoroughly pretested the survey instrument and received a reasonably strong response rates (68%) from a reasonably large sample (889). He tested and controlled for protest bids and examined numerous hypotheses regarding respondents' attitudes and values with respect to groundwater nitrate pollution. Although 40% of the respondents are on private wells, regression analysis does not indicate a significant difference in WTP between private well users and other water users.

Delavan elicited annual WTP for 10 years for a program to reduce the percentage of wells not meeting the MCL from 50% to 25% (increase safe wells from 50% to 75%). Assuming individuals perceive this as their own chance of having a well above the MCL and assuming a "linear in probabilities" utility function, the value for going from unsafe to safe for an individual household with certainty will be four times that of going from 50% to 75% certainty. Based on these assumptions, annual WTP each year for 10 years from the IOE group without protests will be $188.64. Annualizing this from a 10 year payment to a payment in perpetuity yields annual WTP per household for reducing nitrates from unsafe to safe of $48.89, $74.22, and $94.96, respectively, for 3%, 5%, and 7% discount rates. Given the assumptions made to translate the Delavan values into annual WTP estimates, we do not consider these estimates as reliable as others that value WTP in a manner more consistent with those needed for benefits transfer to CAFOs.

5.2.4 Edwards, 1988

Edwards (1988) conducted a contingent valuation study of household willingness to pay to prevent uncertain future nitrate contamination of groundwater on Cape Cod, Massachusetts. The 785 respondents (585 provided useable responses), 89% of whom used a public water system, were renters and both resident and nonresident property owners. The groundwater supply was currently assumed to be safe, but fertilizer and sewage posed a potential problem because Cape Cod relies on a sole source aquifer and measured nitrate levels had been increasing. Edwards used dichotomous choice questions to estimate how much people would pay, using four payment vehicles: (1) an annual bond to be paid in perpetuity, (2) a voluntary contribution, (3) water bills, and (4) an unspecified payment mechanism. No significant difference was found between the different payment vehicles. Edwards used a logit model to generate parameter estimates. Edwards reported a WTP of $1,623 per household per year, for a management plan that would increase the probability of supply from 0.0 to 1.0. Respondents' income, interest in ensuring safe groundwater for future generations, and probability of how long they will live on Cape Cod were all significantly and positively correlated with their willingness to pay for groundwater protection.

Evaluation: Using the logit model with mean sociodemographic characteristic values, an annual WTP for a certain water supply is calculated as $1,623 per year (1987$). This value is higher than those found in other studies reviewed here, for several possible reasons. Edwards specifically valued option price and option values, which may include risk premiums that some of the other studies may not include. The unique characteristics of Cape Cod involving a sole source aquifer suggest that WTP values will be higher there than in other locations with alternative water resources. If nonuse values are a large component of Edwards' value estimate because of the uniqueness of Cape Cod, then his value estimate will be higher than those for less unique locations more typical of counties with CAFOs. The high mean income of the sample ($55,000 in 1987$) is likely to lead to higher WTP estimates compared to other (lower mean income) rural water users nationwide. Thus value estimates from Edwards probably represent an upper bound if they are to be used in benefits transfer.

5.2.5 Giraldez and Fox, 1995

Giraldez and Fox (1995) conducted a cost-benefit analysis of controlling groundwater pollution from agricultural use of nitrogen fertilizer in the village of Hensall (population 1,155 in 1986), in southwestern Ontario. Nitrate concentrations in two wells in the village had recently exceeded 10 mg/L. These wells are sources for a public water distribution system that apparently does not treat the water prior to delivery. Based on willingness-to-pay values from other studies, Giraldez and Fox used three approaches to estimating values for reducing nitrates: (1) value of human life as present value of lifetime average earning, (2) value of statistical life (VSL) based on wage-risk premiums, and (3) CVM. Based on values from CVM studies by Hanley (1989) and Edwards (1988), Giraldez and Fox aggregated a cost of nitrate contamination for the entire village of Hensall to range between about $30,000[1] and $700,000 per year, depending whether bequest and option values are included in the calculation. Based on a lifetime earnings approach, annual costs ranged from $693 to $6,289 for the entire village. Using VSL estimates, Giraldez and Fox estimated an annual benefit range of $984 to $111,639 for the village for reducing mortality related to nitrate contamination. Potential mortality from nitrates is in infants only. The authors concluded that because substantial uncertainty in both the benefits and costs calculations, they could not decisively indicate whether the health benefits of reduced nitrate concentrations justified the cost of changing local agricultural practices.

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1. All dollars from Giraldez and Fox as reported in Canadian dollars. It is unclear what year Giraldez and Fox are reporting dollar values for.

Evaluation: This study is primarily a benefits transfer study, which limits its use for the current valuation exercise because we can simply use the primary studies if they are relevant. Giraldez and Fox attempted to use two non-CVM approaches for deriving value estimates. It is generally believed that the use of value of lifetime earnings is not an appropriate measure of welfare impacts involving mortality risks (Freeman, 1993). It also seems unlikely that VSL estimates from wage-risk studies can be directly applied to infant mortality risks. The value estimates providing secondary value information from Hanley (1989) and Edwards (1988) imply values between $72.73/year and $1,696.97/year (presumably in 1995$ Canadian), although as discussed above Edwards provides a mean WTP of $1,623/year (1987$).

5.2.6 Hurley et al., 1999

Hurley et al. (1999) used data from a contingent valuation study in Clark and Adams counties in Iowa to determine rural residents' willingness to pay to delay, by 10, 15, and 20 years, nitrate contamination of their water supply from large animal confinement facilities. Baseline water quality was not specified, although several highly publicized spills from these types of facilities had occurred recently, and both counties rely heavily on surface water supplies for drinking water. The authors mailed 1,000 surveys to a random sample of residents, of which 332 were completed thoroughly and returned. Apparently 26% of respondents (about 85 total) were on private groundwater wells (not municipal or rural water supply). It also appears that there could be significant scenario rejection in this survey because less than 50% of respondents stated any WTP for any delay in nitrate contamination and less than 10% stated WTP for 10 or 20 year delays in nitrate contamination.

An ordered probit specification, with thresholds adjusted for possible anchoring, was used to analyze the results. The results showed that higher education, income, and expected length of time to remain in the community were positively and significantly correlated with willingness-to-pay values. Male respondents were significantly less inclined to pay for water protection than females. Based on analysis of these referendum questions, the willingness to pay ranged from $118.13 (for a 10 year delay) to $190.75 (for a 20 year delay) per year for a household with sample mean characteristics.

Evaluation: A low overall response rate (33%), a small sample of private well users (85), and potentially high scenario rejection bring results from this study into question for use in benefits transfer. Some aspects of the scenario are unclear, such as what payment mechanism is used in the valuation scenario. WTP in this study was elicited for delays in nitrate contamination, and this does not translate directly into WTP for reducing current nitrates in private wells. Furthermore, this study does not distinguish clearly between groundwater and surface water nitrate contamination. We thus feel we cannot reliably translate values from this study to groundwater contamination from CAFOs without making significant assumptions to derive per household annual WTP estimates for current benefits.

5.2.7 Jordan and Elnagheeb, 1993

Jordan and Elnagheeb (1993) conducted a contingent valuation study of residents' willingness to pay for improvements in drinking water quality, using data from a statewide survey of a random sample of 567 Georgia residents. Of the 199 complete responses received, 78% of respondents were on public water systems and 22% (40 subjects) used private water systems. Water quality was rated as "poor" by 27% of public users and 13% of private users. Respondents on private wells were told to imagine that nitrate levels currently exceeded safety standards and those on public supply were told to imagine that nitrate levels were increasing (from an unspecified baseline to an unspecified endpoint). Nitrate impacts were indicated as being due to nearby agricultural activities. Respondents were asked how much they would be willing to pay (circling one of seven values between $0 and $100) to "avoid the risk of increasing nitrate in [their] drinking water." Public and private water users were given two separate scenarios to value: private wells users were told they would be provided installation and maintenance of filtering equipment and public system users were told that the water supplier would guarantee safe drinking water. The cost for these services would be paid monthly, in perpetuity, through the water bill for public users and a fee for private users.

Jordan and Elnagheeb used both OLS and maximum likelihood functions to generate parameter estimates for their WTP model. The mean WTP for public and private water users, respectively, was $128.20/household per year and $157.61/household per year (1993$). The median was $69.89/household per year for public users and $93.95/household per year for private users. Respondents' income, years of education, and degree of uncertainty regarding their water quality were positively and significantly correlated with the amount they were willing to pay. Females and respondents who lived on farms were willing to pay more to avoid increases in nitrate in their drinking water.

Evaluation: Jordan and Elnagheeb had a low overall response rate (35%) and a small sample of individuals on private wells (38 after rejecting outliers). The scenario is unclear because it specifies nitrate levels currently somewhere above safe levels. The survey appears to be vague on actual health impacts and specifies nitrate reduction to safe levels with little clarification of what this means. Nitrate control is at the point of use for private wells and thus values are primarily use values (no action is indicated to prevent aquifer contamination). Jordan and Elnagheeb did not report the number or percentage of zero bids, and thus it is difficult to evaluate potential scenario rejection. The best point estimate for private well owners' WTP for reducing nitrate contamination to safe levels is $157.61/household per year (1994$), which is primarily a use value.

5.2.8 Poe and Bishop, 1992

Poe and Bishop (1992, 1999), and Poe (1993, 1998) conducted a contingent valuation study in rural Portage County, Wisconsin, to estimate conditional incremental benefits of reducing nitrate levels in household wells. The area had extensive nitrate problems, and previous research suggested that 18% of private wells in the area exceeded the MCL. Two WTP valuation scenarios are discussed in the various Poe and Bishop papers: WTP for a program to keep all wells in Portage County at or below the MCL and WTP for a program to reduce well nitrates in all wells by 25%. Sources of nitrates identified in the information materials included “septic; tanks, farm, lawn, and garden fertilizers; livestock holding areas; and abandoned wells.” In particular, Poe and Bishop were interested in how providing respondents with information on their own well nitrate concentrations was related to willingness to pay for nitrate reductions.

The survey thus comprised two stages. In the first stage, individuals were asked to submit water samples from their tap and to complete an initial questionnaire. In the second stage, the individuals were provided with their nitrate test results, general information about nitrates, and a graphical depiction of their exposure levels relative to natural levels and the MCL, and they were asked to complete contingent valuation questions. Poe and Bishop found no sample selection bias between the first and second survey stages. Poe addressed potential nonlinearities by allowing for a nonlinear WTP function where the degree of convexity or concavity is estimated based on the data.

A total of 271 completed Stage 2 responses were received. In general, Poe and Bishop found that respondents’ knowledge of their water quality and awareness of the health effects of nitrates to be positively and significantly correlated with willingness to pay. The various Poe and Bishop papers report different WTP values for different types of analysis and for different portions of the data set.

In their 1992 working paper, Poe and Bishop (1992) report a mean ex post WTP of $257.10 per household per year for a program to keep all wells in Portage County at or below the MCL. Poe (1993) reports per household per year mean WTP values for a program to keep all wells in Portage County at or below the MCL for different information levels and depending on whether the individual had a prior test of actual well nitrate levels. These mean reported WTP values are $199.73/household/yr NINT, $961.16/household/yr WINT, $244.32/household/yr NIWT, and $526.63/household/yr WIWT (where NINT, WINT, NIWT, and WIWT mean “no information-no test,” “with information-no test,” “no information-with test,” and “with information-with test,” respectively).[2] Poe then calculates a mean WTP for prevention of well nitrates above the MCL of

2. Values reported here from Table VII.2.3.2 from Poe (1993) for the mean and median values based on 1,000 draws using a Duffield and Paterson Simulation method for estimating mean WTP values.

$484 per household per year for households with a 100% probability of future contamination.[3] In terms of policy uses, it could be argued that the $484 value estimate represents the best informed and most relevant value statement from respondents and thus should be used for benefits transfer.

Poe (1993) also calculates an imputed WTP for a 1 mg/L reduction (or increase) in nitrates as a function of initial nitrate levels. A maximum per mg WTP of ~$120 is seen when initial nitrate levels are close to 10 mg/L. Above 10 mg/L the per mg WTP falls off to zero at about 22 mg/L. Below 10 mg/L the per mg WTP falls to about $90 per mg when the initial level is 4 mg/L. While this is an order of magnitude greater than Crutchfield et al. (1997) or De Zoysa (1995), it is more in line with WTP values derived by Sparco (1995) for incremental changes in nitrate concentrations of $123.56 per mg/L.

Poe (1998) reports WTP for the program to keep all wells in Portage County at or below the MCL as a function of the individuals' observed well nitrate concentrations. Estimated WTP values varied, as expected, by the results of the respondent's nitrate test. Those with a nitrate level of 2 mg/L would pay $84.07/year, whereas a respondent with 40 mg/L of nitrate would be willing to pay $515.59/year to keep nitrate levels below the MCL.

Poe and Bishop (1999) also estimated a nonlinear WTP function, including both single-power and cubic formulations. They report WTP for the program to reduce well nitrates in all wells by 25%. Using the cubic function, Poe and Bishop show that incremental benefits increase between 2 mg/L and 14.5 mg/L and then fall to zero at about 22.5 mg/L. Since a 25% reduction from 14.5 mg/L would reduce nitrate levels to very near the MCL, this reduction could be considered a WTP to reduce nitrates to safe levels. The estimated WTP for a 25% reduction from 14.5 mg/L is reported by Poe and Bishop as $412 per year per household.

Evaluation: Overall, the high quality of the Poe and Bishop study suggests that benefit estimates from this work are likely to be reliable and valid. The Poe and Bishop work is based on a well developed theoretical model of respondents' willingness to pay (e.g., Poe and Bishop is one of the only studies to empirically assess potential nonlinearities in the WTP function). Survey development, implementation, and analysis meet or exceed standards for CVM studies at the time of the study. Poe and Bishop is also the only work we reviewed where respondents had empirical information on the nitrate levels in their own wells. Although the stage 2 sample size is not large (271), the quality of the data is likely to be higher than for larger samples using less well developed surveys. The surveyed population (rural Wisconsin) is most likely representative of individuals facing potential well nitrate contamination from CAFOs.

Two value estimates from Poe and Bishop are the most applicable for benefits transfer. The first is the mean WTP of $484/household/yr from the scenario of a program to keep all wells in Portage County at or below the MCL for a household with a 100% probability of future

3. $347-$655 95% confidence interval (Table VII.2.4.2; Poe, 1993).

contamination. The second is the $412/household/yr incremental value for a program to reduce well nitrates by 25% for a well with a current nitrate concentration of 14.5 mg/L.

5.2.9 Sparco, 1995

Sparco (1995) used conjoint analysis to estimate the benefits of reduced groundwater contaminant concentrations and subsequent risks of illness in Sussex County, Delaware. The county is predominantly agricultural, and nitrate is a common pollutant in the groundwater. A survey of private wells (Andres, 1991) found nitrate levels at or above 10 mg/L in 23% of the county's wells, and 50% of households rely on their own drilled or dug wells for water. Respondents were surveyed at public gatherings such as state fairs, and were asked to rate preferences over four cards, including different attribute levels of willingness to pay, nitrate levels, atrazine levels, fecal coliform, and illness characteristics, as well as "attitudinal" questions regarding the respondent's opinion on government intervention, agriculture, and the environment.

Respondents were told that the contamination originated from agricultural activities. Sparco used an ordered probit regression to analyze the responses. The total number of respondents was not specified. The mean annual WTP (calculated from the ordered probit model) to reduce nitrate contamination by 1 mg/L was $123.56. Calculated WTP values for 1 in 10,000 reductions in one-week illness now or gastrointestinal (GI) cancer in 20 years of $129.58 and $370.72, respectively, imply extremely large "value of statistical illness" (VSI) estimates. VSIs of nearly $13 million for one week of illness now and of $37 million for GI cancer in 20 years seem implausible compared to common value of statistical life estimates between $5 and $10 million (Chestnut et al., 1997). A pro-environmental attitude was significant and negatively correlated with WTP for nitrate reduction, and antigovernment intervention, and pro-farm viewpoints were significant and correlated with WTP. While the signs of all three principal components appear to be unexpected in the regression model, Sparco suggested that the signs of these three factors indicate that survey respondents are supportive of farming in the county and believe that the government should adopt a laissez-faire approach toward environmental regulation.

Evaluation: The methods and analysis used in this study are good and predate current methods in stated preference analysis using conjoint methods. Several issues, though, suggest limits to the reliability and validity of value Sparco's estimates for use in benefits transfer. The sample is nonrandom and the final sample size and response rates are unspecified. The apparently incorrect signs on attitudinal variables from the principal components analysis raise questions about the model estimates. The experimental design had a significant effect on preference statements, and it is unclear how this factors into value calculations. Sparco did not separate values between private well users and municipal or community system users. And, as stated above, the value estimates for illness characteristics seem implausibly high, casting some doubt on the reliability of value estimates for incremental changes in nitrate concentrations of $123.56 per mg/L.

5.2.10 Walker and Hoehn, 1990

Using information obtained primarily from an engineering model of the costs of water purification technology, Walker and Hoehn (1990) developed a model of economic damages of nitrate contamination in rural Michigan. The area has a history of elevated nitrate concentrations, with a study reporting 34% of rural drinking water wells exceeding the MCL for nitrates (Vitosh, 1985). Over 95% of the rural residential water supply comes from groundwater. The authors calculated net economic damages as the sum of producer and consumer surplus. The model requires three components: a residential water demand function, a precontamination supply function, and a post-contamination supply function. The demand function was assumed to be linear, based on the quantity of water used per household, the average water price, household income, rainfall, and the number of persons in the household. The precontamination supply function is the incremental cost of providing water before contamination occurs, and is simply a linear relationship with the initial price of water. The post-contamination function is the incremental cost of providing water after nitrate contamination, and is the same as the precontamination function plus the additional incremental cost of removing nitrates.

The incremental cost of nitrate removal was estimated from a sample of costs for nitrate removal generated from the engineering model. The incremental costs are thus entirely determined by the parameters of the engineering model. Based on these three functions, Walker and Hoehn estimated that total damages from nitrate contamination range from $40 to $330/household per year, depending on the treatment location, household water consumption, the price of water, the damages and benefits per household, household income, the level of nitrate contamination, and an estimate of annual costs for point-of-use nitrate removal.

Evaluation: This study deals with public water supply cost savings as a measure of benefits from reducing or avoiding nitrate contamination. Although it is not directly transferable to private wells, WTP values to prevent nitrates in public water systems may indicate use values for prevention of nitrates in private wells for comparable uses of drinking water. Based on incremental value estimates from the damage model, an average household with a $15,000 income in a community of 500 households would be willing to pay $65/yr (1983$) for prevention of nitrate contamination. Since the Walker and Hoehn model incorporates economies of scale to estimate per household damages, the value per household in a 500 household community is lower than that of a one household community (e.g., a private well). A $65/year estimate from Walker and Hoehn thus could represent a lower bound estimate of use values. The estimate represents an avoided cost measure of welfare change based on the parameterization of the engineering model. Because the validity of this model cannot be judged based on the information provided, it is not possible to determine the validity of this avoided cost measure.

5.2.11 Wattage, 1993

Wattage (1993) conducted a contingent valuation survey to elicit WTP for improved water quality in the predominantly rural Bear Creek watershed in central Iowa. The purpose of the survey was to determine values for vegetated buffer strips (VBSs) in terms of benefits for groundwater protection. A single survey instrument was used to reach farmers and the general public and asked different questions of each group. The survey involved multiple valuation questions for several different "commodities" involving impacts to surface and groundwater from agriculture. The valuation scenarios were not fully specified: there was no explanation of a payment vehicle or of a program for achieving groundwater protection and cleanup. Based on the discussion in the report, it is apparent that VBSs are the program that will provide improved groundwater conditions. In the survey instrument, though, discussion and questions about VBSs come after the valuation questions.

The 346 respondents were farmers, absentee owners, and town residents. Fifty percent of respondents were on private wells; over 90% of respondents relied on groundwater for drinking water supplies. Groundwater quality was of major concern to many of the respondents: only 16% ranked water quality as suitable for human drinking purposes. Using both open-ended WTP questions and dichotomous choice formats, respondents were asked how much they would be willing to pay for programs to reduce contamination of groundwater and surface water supplies. Wattage estimated a mean monthly WTP of $80, using both probit and logit models, finding that the different models had little impact on the final estimation results. Wattage also used an integration method to generate a conditional WTP estimate from the logit model of $49 per month per household. The year of analysis is uncertain. Wattage found that income was positively correlated with an individual's WTP, and the respondent's perception of current groundwater quality and the distance from the respondent's land to the potentially polluted creek were negatively correlated with an individual's WTP.

Evaluation: Given problems in scenario presentation, it seems likely that there is significant misunderstanding of the scenario or potential scenario rejection. This position is supported by the fact that only 32% of respondents strongly agreed that VBS could be effective in reducing contamination from runoff. Given the information in the report and based on the survey instrument, it is not possible to determine exactly what commodity is being valued or whether this represents WTP for moving from unsafe to safe drinking water (since it is unclear what initial conditions are). Since the endpoint is safe water and the baseline may also be safe water, average value statements would be an underestimate for cleaning up unsafe water.

Using the value estimate from integration under the logit curve and the sample means for sociodemographic characteristics yields a conditional WTP of $49 per month per household. This translates to an annual WTP of $588, which is larger than the cost of point-of-use controls of $330. While the larger WTP may represent additional consideration of nonuse values such as protection of aquifers, these values most likely represent an upwardly biased estimate of values for protection of groundwater from nitrate contamination.

5.3 EVALUATING STUDIES FOR BENEFITS TRANSFER

5.3.1 Purpose of Rating Studies Based on Quality and Applicability

The purpose of this work is to identify estimates of the benefits from changes in well nitrate concentrations that are applicable for this benefit estimation for potential CAFO regulations. Desvousges et al. (1992) developed five criteria that they used to guide the selection of studies used in their application of the technique to a surface water quality issue. In essence, their five criteria are that the studies to be transferred (1) be based on adequate data, sound economic method, and correct empirical technique (i.e., "pass scientific muster"); (2) evaluate a change in water quality similar to that expected at the policy site; (3) contain regression results that describe willingness to pay as a function of socioeconomic characteristics; (4) have a study site that is similar to the policy site (in terms of site characteristics and populations); and (5) have a study site with a similar market as the policy site. NOAA condenses the five Desvousges et al. criteria into three considerations: (1) comparability of the users and of the resources and/or services being valued and the changes resulting from the discharge of concern, (2) comparability of the change in quality or quantity of resources and/or services, and (3) the quality of the studies being used for transfer [59 FR 1183].

In a general sense, items (2), (4), and (5) of Desvousges et al. and items (1) and (2) of NOAA are concerned with the *applicability* of an original study to a policy site. Items (1) and (3) of Desvousges et al. and item (3) of NOAA are concerned with the *quality* of the original study. To assess original studies for use in the benefits transfer for benefits assessments from CAFO regulations, we assess the *applicability* and the *quality* of the original studies on several criteria.

The 11 studies summarized in Appendix C represent a diverse range of valuation exercises. To the extent feasible, information was obtained or derived from each report or paper for 28 categories of information used to characterize the studies. While this is largely a qualitative assessment, the purpose of the following discussion is to make this assessment as transparent as possible. Because applicability to CAFOs and quality of the value estimates are distinct concepts, we want to rate these characteristics of the studies separately. Overall, the goal of the rating process is to identify studies that elicit high-quality values (reliable and valid) and which are most applicable to the benefits assessment. There are three steps in undertaking the rating process:

1. identify study characteristics upon which to judge applicability and quality
2. assign scores to the studies based on these characteristics

3. assign weights to these scores for aggregating scores into unidimensional measures of applicability and quality.

We assigned scores according to the criteria discussed below and identified in Exhibit 5-1. For this rating schema, the weighting on the various characteristics related to quality or applicability is simple so that the effect of changing the weighting scheme will be transparent.

5.3.2 Criteria for Ranking Based on Applicability

The first criterion for ranking the groundwater valuation studies is applicability. Applicability refers to the relationship between values elicited in the groundwater valuation studies and benefit estimates necessary for application to the analysis of CAFO regulatory options. Values necessary for benefit analysis of CAFO regulatory options primarily involve potential health risks related to elevated nitrate levels in drinking water. While CAFOs may introduce other contaminants into drinking water, nitrate contamination is a primary focus of regulatory options. Criteria for evaluation of study applicability include characteristics of the original studies such as:

- location (urban, rural, etc.)
- water supply/groundwater use (percent on wells)
- contaminants (scenario involves nitrate contamination of groundwater)
- source of contaminants (scenario involves conditions similar to those relevant for CAFOs)
- value estimates are for the correct theoretical construct (e.g., total WTP for reducing groundwater contamination from nitrates).

Location

In general, urban residents are not on private groundwater wells and thus have less experience with potential groundwater contamination. A higher applicability rating was given to studies that are primarily rural than to those with urban/rural or purely urban samples. Concentrating on rural populations is also more likely to be similar to the population of individuals on private wells to which we apply benefit estimates. Since we do not have national sociodemographic information specific to the population on private wells, focusing the transfer on studies conducted with more rural populations helps account for potential income differences between rural and urban populations.

Water Supply

Studies received a higher score if more than 50% of the respondents indicated that they were currently using groundwater for their primary water supply. Again, the policy population is

Exhibit 5-1
Scoring Matrix for Groundwater Valuation Studies

Scoring Criteria	Scoring	Applic-able	Quality	Crutchfield et al.	Delavan	De Zoysa	Edwards	Giraldez and Fox[a]	Hurley et al.	Jordan and Elnagheeb	Poe and Bishop	Sparco	Walker and Hoehn	Wattage
Location (urban, rural, etc.)	Rural = 2; Rural/urban = 1; Urban/other = 0	✓		1	1	1	1	2	1	1	2	1	2	2
HH H_2O Supply/GW Use	> = 50% on wells = 1; <50% = 0	✓		0	0	0	0	1	0	1[b]	1	1	0	0
Contaminants	Nitrates = 2; nitrates + other = 1; Not nitrates = 0	✓		2	2	2	2	2	2	2	2	1	2	0
Source of Contaminants	CAFOs/Agr = 2; Mixed sources w/ag = 1; Not specified = 0	✓		0	1	2	0	2	2	2	2	1	2	1
Values Estimated	WTP = 1; Other = 0	✓		1	1	1	1	0	1	1	1	1	0	1
Published/Peer Reviewed?	Peer rvw. = 2; Dissert. = 1; Other = 0		✓	2[c]	0	1	2	2	2	2	2	1	2	1
Type of Study	Primary data = 1; Other = 0		✓	1	1	1	1	0	1	1	1	1	0	1
Survey Implement	Mail/in person = 1; Other = 0		✓	0	1	1	1	0	1	1	1	0	0	1
Respondents	>1000 = 2;500-1000 = 1; <500 = 0		✓	2[d]	1	2	1	0	0[e]	0	0	0	0	0
Response Rate	>70% = 2;40%-70% = 1; <40% = 0		✓	1	0	0	2	0	0[f]	0	2	0	0	0
Groundwater Baseline	Specified = 1; Not specified = 0		✓	0	1	1	1	1	0	1	1	0	1	0
Change in Groundwater Scenario	Defined change = 1; Undefined or vague = 0		✓	1	1	1	1	1	1	0	1	1	1	0
Credibility of Scenario Change	Assessed credibility = 1; Didn’t asses = 0		✓	1	1	1	1	0	0	0	1	0	0	0

Exhibit 5-1 (cont.)
Scoring Matrix for Groundwater Valuation Studies

Scoring Criteria	Scoring	Applic-able	Quality	Crutchfield et al.	Delavan	De Zoysa	Edwards	Giraldez[a]	Hurley et al.	Jordan and Elnagheeb	Poe and Bishop	Sparco	Walker and Hoehn	Wattage
Valuation Methodology	Valid = 1; Questionable = 0		✓	1	1	1	1	1[g]	0	1	1	1	0	0
Payment Vehicle	Specified = 1; Not specified = 0		✓	1	1	1	1	0	0	1	1	0	0	0
Duration of Payment Vehicle	Continuous = 2; One time = 1; Other = 0		✓	2	2	1	2	0	2	2	2	2	0	2
Analysis	Advanced = 1; Other = 0		✓	1	1	1	1	0	1	1	1	1	1	1
Significant Explanatory Variables	Validity indicated = 1; Other = 0		✓	1	1	1	1	1	1	1	1	1	0[h]	1
				Crutchfield et al.	**Delavan**	**De Zoysa**	**Edwards**	**Giraldez**	**Hurley et al.**	**Jordan and Elnagheeb**	**Poe and Bishop**	**Sparco**	**Walker and Hoehn**	**Wattage**
			Total Applicability	4	5	6	4	7	6	7	8	5	6	4
			Total Quality	14	12	13	16	6	9	11	15	8	5	7

a. Benefits transfer study and thus many categories are not applicable.
b. Using analysis for private wells only.
c. Crutchfield and Cooper, 1997.
d. Based on indication of 819 usable responses and ~50% response rate.
e. Only 85 private well users in the analysis.
f. 44.7% returned: 33.2% usable.
g. Valid for benefits transfer.
h. Significant explanatory variables in Walker and Hoehn are entirely the result of generating data using an engineering model of incremental costs of water production.

individuals on private wells, and thus studies of this population are more applicable for benefits transfer.

Nitrate Contamination of Groundwater

We considered primarily valuation studies that present a scenario of nitrates as a source of contamination in groundwater. Nitrate contamination scenarios are more likely to present individuals with impacts and risks that are similar to those necessary for the valuation of CAFO control benefits. While some studies indicated other contaminants in addition to nitrates, we placed higher weight on values identified as specifically associated with nitrates. While other scenarios will also elicit values for reducing risks of drinking contaminated groundwater, they may involve health risks different from those from nitrate contamination.

Relationship of Valuation Scenario to CAFOs

Some of these studies consider sources other than CAFOs or agricultural sources. While values for reduced health risks from groundwater contamination may be elicited in other studies, it seems likely that studies specifically considering scenarios similar to CAFOs or agricultural contamination will be more amenable to benefits transfer. In addition, CAFO-type contamination sources and their regulation may involve decisions and impacts that are different from other contamination sources such as air deposition or contamination from septic systems.

Valuation Scenario

While most of the studies elicit total values for reduced contamination, some are designed to elicit option values. While these are theoretically valid values, we need to further consider their applicability to the regulatory options under consideration. In particular we rated studies as to whether they elicited willingness to pay as the appropriate theoretical construct applicable for policy analysis. In addition, studies directly eliciting values for reducing nitrate contamination in individuals' own wells are more directly transferred to the current policy scenario than studies valuing prevention of future possible contamination (e.g., Edwards, 1988) or the probability of contamination in a group of wells (e.g., Delavan, 1998).

5.3.3 Criteria for Ranking Based on Study Quality

Analysis of study quality is based on evaluation of the validity and reliability of the value estimates derived in the groundwater valuation studies. This is primarily a qualitative exercise examining multiple facets of the studies under consideration. Based on suggested criteria as to what contributes to a valid and reliable stated preference valuation study, we identified characteristics of these studies that indicate reliability and validity (Bishop et al.,1997). Criteria for evaluation of study quality include:

- published/peer reviewed
- type of study (design/method)
- survey implementation
- respondents: number and well usage
- response rate
- groundwater baseline
- change in groundwater scenario
- credibility of scenario change
- valuation method
- payment vehicle
- duration of payment vehicle
- analysis (method of empirical estimation)
- significant explanatory variables.

Peer Reviewed

Peer reviewed publications may provide more reliable and defensible value estimates than nonreviewed reports. To this end we also considered PhD dissertations to be more reliable than master's theses because they have generally undergone more rigorous review and meet a higher standard than master's theses or general staff publications. While we do not mean to say that master's work or staff publications cannot be of as high or higher quality than peer reviewed work, there is more evidence that peer reviewed work has met an accepted professional standard.

Type of Study

We placed a higher rating on studies that elicit empirical values from actual households as opposed to being theoretical modeling exercises. Some of the studies are primarily theoretical exercises that do not elicit primary data from households (e.g., Walker and Hoehn, 1990). As such, these studies may not provide information on values directly transferable for the benefits assessment.

Survey Implementation

Survey implementation is defined here as the method of conducting the survey. In general telephone surveys are less likely to generate reliable data in CVM surveys because of the abbreviated nature of telephone surveys. While some researchers favor in-person surveys, mail surveys have been shown to generate reliable responses (Dillman, 2000). In our evaluation of study quality, we also noted studies that did not involve a random sample (e.g., Sparco, 1995) to minimize potential sample selection bias (see below on response rates).

Respondents

For contingent valuation surveys, it is important that a sufficient sample size has been used to ensure representativeness of the value estimates. While there is no clear-cut rule for assessing adequate sample size in CVM studies, statistical methods used in sampling design can indicate sample sizes necessary to obtain estimates of population parameters. For instance, with a population size of 1 million, a sample size of 1,066 is needed to estimate a 95% confidence interval with a ±3% sampling error (Dillman, 2000, see also Kalton, 1983). When evaluating the number of respondents, we also attempted to identify those respondents on private wells because many studies elicit values from other water users (e.g., municipal).

Response Rate

Higher response rates are used as an indication of the representativeness of the value estimates and as an indication of overall study quality. Because of potential sample selection and nonresponse biases (Mitchell and Carson, 1989), response rates above 70% are considered good for CVM surveys, while those below 40% are rated as poor for evaluating these studies.

Groundwater Baseline

A full definition of the commodity being valued includes identifying baseline conditions. The survey instrument must either specify baseline conditions or elicit individuals' perceptions of baseline conditions (Fischhoff and Furby, 1988). In our evaluation of study quality, we identified studies where baseline is actually defined or elicited in the survey instrument as opposed to only mentioned in the study report. Not specifying baseline in the survey leaves the commodity inadequately defined.

Change in Groundwater Scenario

Scenario development is essential in CVM studies to ensure that individuals understand the valuation exercise and that the values elicited are for the commodity being studied (Fischhoff and Furby, 1988). Several aspects of the study design fall under the concept of scenario development, including identifying baseline groundwater conditions, identifying changes in groundwater conditions as discussed above, specifying the source of contamination, assessing the credibility of the scenario, and using a realistic payment vehicle. This study quality criterion evaluates whether the change in groundwater quality is specified, because if it is not, we cannot determine exactly what commodity is being valued.

Credibility of Scenario Change

To elicit a valid value statement from individuals, the proposed program or commodity change must be credible to respondents. Credibility depends on how the program is described to individuals and the perceived likelihood of whether or not such a program would ever be provided or would even be possible to provide. A not credible scenario is likely to induce scenario rejection and misstatements of actual values. Studies were scored depending on whether or not they had assessed the credibility of the scenario to respondents (e.g., attempted to identify scenario rejection).

Valuation Method

The method for estimating the value of a commodity has to be appropriate for the value being estimated. As part of implementing correct valuation methods, the appropriate population needs to be sampled, the correct type of value (e.g. WTA or WTP) elicited from that population, and the appropriate method applied for deriving the value. For instance, if values are elicited from non-groundwater users for cleaning up drinking water that comes from groundwater, these values likely are to be different than values that groundwater users would have. Additionally, engineering cost models cannot be used to derive individuals' WTP values because such models are based on different theoretical values (i.e., costs, not welfare values).

Payment Vehicle

Numerous types of payment vehicle can be proposed in a CVM survey. CVM researchers generally feel that the payment vehicle should be well defined and plausibly related to the commodity being valued (Morrison et al., 2000). The payment vehicle should be assessed for adequacy in pretests or in quantitative analysis (Carson, 1997) as in Edwards (1988). We ranked studies lower if they do not specify a payment vehicle.

Duration of Payment Vehicle

Similar to the requirement that the payment vehicle be commensurate with the commodity, the duration of the payment should be reasonably related to the duration of the commodity or program providing the commodity being valued. Since most groundwater nitrate control programs and benefits are continuous, we rated studies with continuous (e.g., Poe and Bishop, 1992) or multiyear (e.g., Delavan, 1998) payment vehicles, e.g., monthly water bills, higher than those with one-time payments (e.g., De Zoysa, 1995). Likewise, we rated lower those studies that do not appear to specify the payment vehicle duration, because this indicates inadequate commodity definition.

Methods of Analysis

Statistical analysis includes appropriate econometric methods (e.g., probit or logit models rather than ordinary least squares for qualitative choice surveys or tobit for truncated at zero, open-ended WTP questions) and adequate reporting on the results of statistical analysis. In general, all of the studies present reasonably high quality analysis where applicable.

Significant Explanatory Variables

Economic theory suggests that willingness to pay is related to certain sociodemographic characteristics; for example, it is generally positively related to income. Other relationships are expected, although not based on microeconomic theory. For instance, rural residents are expected to be willing to pay more for clean groundwater from private wells than urban dwellers who rely on public water supplies. *Ceteris paribus*, individuals who use private wells are expected to be willing to pay more than those on public supplies, even in rural areas. Perceptions of water quality also can be expected to be related to WTP for reducing nitrates in drinking water. For several studies the likelihood that an individual would live in an area in the future was positively correlated with WTP for safe drinking water.

5.3.4 Scoring Matrix

Most of the screening information items presented in Appendix C were used for these assessments. Characteristics summarized in Appendix C but not used for the assessment were year of analysis, place, who was asked, actual groundwater baseline condition, number of survey versions, and the values actually estimated.[4] Based on these characteristics and scoring criteria, Exhibit 5-1 presents the scoring matrix for the 11 nitrate valuation studies evaluated. The "scoring" column indicates the scoring method for evaluating the various studies using the criteria discussed above for applicability and quality. Several of the criteria apply only to primary data collection (e.g., contingent valuation surveys) such as survey implementation, respondents, response rate, credibility of scenario change, valuation methodology, and payment vehicle. Studies that are not based on primary data collection thus score low on these criteria and are not likely to be included in the benefits transfer assessment. Checkmarks in the applicability and quality columns indicate which scores were summed to aggregate the study characteristics to the unidimensional applicability and quality scores at the bottom of the exhibit.

The scoring was undertaken without weighting the various characteristics for importance in determining applicability or quality of study. A weighting scheme was derived to provide more reliable assessment.[5]

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4. "Place" does play a role in that the Edwards study is not weighted highly in the benefits transfer in part because of the unique location of the study. It involved a sole source aquifer in a unique location (Cape Cod) where mean income of respondents is most likely higher than would be expected at typical rural sites where CAFO impacts are expected.

5. The weighting scheme was based on collaborative professional judgment with EPA and consultant economists.

5.4 RANKING OF NITRATE VALUATION STUDIES

Using the scoring from Exhibit 5-1, we sorted the studies into high, medium, and low categories based on their applicability and reliability for use in CAFO analysis. Our results are shown in Exhibit 5-2. It must be emphasized that these scorings and rankings are not intended as judgments of the studies *except* for purposes of their use in benefits assessments for CAFO regulatory options. Many aspects of these studies that explore important theoretical or methodological issues are not as applicable for the benefits assessment and thus may receive low weights. Possible applicability scores range from 0 to 8. Studies scoring from 0 to 4 were rated as low, 5 and 6 as medium, and 7 and 8 as high. Possible quality scores range from 0 to 17. Studies scoring from 0 to 9 were rated as low, 10 to 13 as medium, and 14 and above as high. Exhibit 5-2 summarizes the scoring and rating according to this criterion.

Exhibit 5-2
Ranking of Studies Based on Scoring Exercise

Study	Total Applicability	Total Quality	Total Applicability	Total Quality
Crutchfield et al.	4	14	low	high
De Zoysa	6	12	medium	medium
Delavan	5	12	medium	medium
Edwards	4	16	low	high
Giraldez and Fox	7	6	high	low
Hurley et al.	6	9	medium	low
Jordan and Elnagheeb	7	11	high	medium
Poe and Bishop	8	15	high	high
Sparco	5	8	medium	low
Walker and Hoehn	6	5	medium	low
Wattage	4	7	low	low

Based on the scoring and qualitative rankings, Exhibit 5-3 indicates where these studies fall across the two dimensions of applicability to CAFOs and quality of studies.

Exhibit 5-3
Groundwater Valuation Applicability and Quality Matrix

		Applicability of Study to CAFOs		
		High	**Medium**	**Low**
Quality of Study	**High**	Poe and Bishop, 1992		Crutchfield et al., 1997 Edwards, 1988
	Medium	Jordan and Elnagheeb, 1993	De Zoysa, 1995 Delavan, 1998	
	Low	Giraldez and Fox, 1995	Hurley et al., 1999 Sparco, 1995 Walker and Hoehn, 1990	Wattage, 1993

5.5 VALUES FOR BENEFITS TRANSFER TO CAFOS

We applied the CPI to convert the annual mean household willingness-to-pay values obtained from these studies to 2001 dollars. Exhibit 5-4 shows the CPI values used for these conversions.

Exhibit 5-4
Consumer Price Index — All Urban Consumers — (CP - U)
U.S. City Average — All Items (1982-1984 = 100)

Year	Annual
1983	99.6
1984	103.9
1986	109.6
1987	113.6
1988	118.6
1989	124.0
1990	130.7
1991	136.2
1992	140.3
1993	144.5
1994	148.2
1995	152.4
1996	156.9
1997	160.5
1998	163.0
1999	166.6
2000	172.2
2001	177.1
Source: U.S. Bureau of Labor Statistics, 2000.	

Exhibit 5-5 shows summary mean per household annual WTP in 2001 dollars for several of the studies discussed above. Not all values are shown for all reports.

Exhibit 5-5
Mean Annual WTP per Household

Study Reference	Year of Analysis	Mean Annual Household WTP in 2001 dollars
Crutchfield et al., 1997	1994	\$758.40 to reduce nitrates to safe level \$22.90 to reduce from 10 mg/L to 0 mg/L (\$2.29 per mg/L)
De Zoysa, 1995	1994	\$63.07 (lower bound mean) \$1.89 per mg/L (using 3% discount rate)
Delavan, 1997	1996	\$212.92 IOE w/o protest bidders (see Section 5.2.3)
Edwards, 1988	1987	\$2,530.22 to increase probability of supply from 0.0 to 1.0
Jordan and Elnagheeb, 1993	1991	\$204.94 (private wells)
Poe and Bishop, 1992	1991	\$535 (25% reduction in nitrates to safe level) \$629 (households with 100% probability of future contamination)
Sparco, 1995	1993	\$151.44 per mg/L

Based on this summary, WTP values for reducing nitrate contamination to safe levels fall into a range between \$60 and \$2,500 a year. The exact interpretation of the commodity varies for these studies, as discussed above in the study evaluations. For reasons outlined there, we feel Edwards' \$2,500/year represents a high estimate not directly applicable to the conditions of CAFO counties nationwide. Also as discussed above, the Delavan and De Zoysa values represent either lower bound estimates or value estimates that are not reliably translated into those necessary for CAFO benefit transfer assessment. Jordan and Elnagheeb's small sample, unclear scenario, and potential scenario rejection make their value estimate less reliable than Poe and Bishop, but may provide a lower bound value for nitrate reductions. Poe and Bishop's work represents the most rigorous analysis and provides the only value estimates based on respondents knowing their actual well nitrate levels.

For estimates of the per mg/L values for nitrate reductions, Sparco's value estimates appear to be implausibly high, especially relative to the values for potentially larger total mg/L reductions from unsafe to safe levels. The Crutchfield et al. estimate for WTP per mg/L under the MCL provides a lower bound estimate that we can conservatively use in the benefits transfer.

The Crutchfield et al. value estimate for reducing nitrates to safe levels are derived from a more diverse sample than Poe and Bishop. The Crutchfield et al. WTP estimate is $758.40/household/yr (2001$). As indicated in Exhibit 5-2, though, we ranked the Crutchfield et al. study as being of low applicability for benefits transfer to CAFOs primarily because they did not specify the source of the nitrate contamination in their scenario and less than 50% of their respondents were on private wells. We thus consider the Crutchfield et al. values as a possible upper bound for application for this benefits transfer. We thus rely primarily on the average of Poe and Bishop's two WTP estimates as reliable estimates of WTP for reducing nitrates to safe levels (from above the MCL to below the MCL). The average of these two estimates is $583.00 per household per year.

We use the average of De Zoysa and Crutchfield et al. for changes in incremental nitrate concentrations below the MCL. The values from Poe and Bishop are expressed as willingness to pay per year as long as the individual lives in the county, and thus can be directly translated to the policy scenarios.

In De Zoysa's study, the reduction in groundwater nitrate levels is from a range of 0.5 to 3.0 mg/L to a range of 0.5 to 1.0 mg/L. Taking range means, the reduction in nitrates is thus from 1.75 mg/L to 0.75 mg/L, or a reduction of 1.0 mg/L. Using the annual lower bound mean values, this represents a WTP of $63.07 per mg/L (in 2001$) change in nitrate concentrations for incremental changes below the 10 mg/L MCL. Using a 3% discount rate, this translates into an annual WTP of $1.89.

Crutchfield et al. report monthly willingness-to-pay values for reducing nitrates, and thus we adjust their values to an annual WTP per mg/L. They report values for reducing nitrates from above the MCL to the MCL and from above the MCL to zero. The difference between these two values is taken as the value of reducing nitrate concentrations from the MCL of 10 mg/L to 0 mg/L. Using the monthly willingness-to-pay values reported in Crutchfield et al., we calculated a per-year per-mg/L value for incremental changes in nitrate concentrations below 10 mg/L. This adjustment assumes a "linear" value per mg/L between 10 mg/L and 0 mg/L, indicating no threshold effects. The resulting value, $2.29 per mg/L per household per year (in 2001$), is applied to changes in well nitrate concentrations between 10 mg/L and 1 mg/L, assuming that there is a natural, or ambient, background level of 1 mg/L of nitrates in groundwater.

For purposes of benefits transfer we use an average of the values from the De Zoysa and Crutchfield et al. of $2.09 per household per year per mg/L (in 2001$). Exhibit 5-6 shows the point value estimates used for benefits transfer.

Exhibit 5-6
Willingness-to-Pay Values Applied to Benefits Transfer

Study	Value	2001$
Poe and Bishop	Annual WTP	$583.00
Average of Crutchfield et al. and De Zoysa	Annual WTP per mg/L between 10 mg/L and 1 mg/L	$2.09

CHAPTER 6
BENEFIT CALCULATIONS

6.1 TOTAL ANNUAL VALUES

Exhibit 6-1 shows the undiscounted annual benefit estimates when all the effects of reduced nitrogen loadings have been achieved at the well. The second column shows the benefits derived from reductions in the number of households above the MCL, and the third column shows benefits from incremental reductions between 1 mg/L and 10 mg/L for households that were below the MCL before regulatory changes. The last column shows total annual national undiscounted benefits.

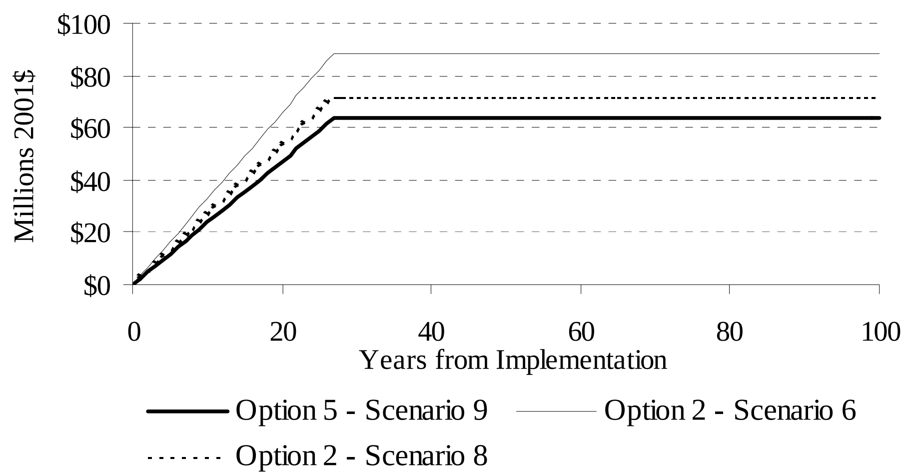
Exhibit 6-1
Undiscounted Annual Values under CAFO Regulatory Scenarios (2001$)

Scenario	Total WTP for Discrete Reduction to MCL	Total WTP for Incremental Changes below 10 mg/L	Total
Option 1 — Scenario 6	$86,695,000	$1,786,000	$88,481,000
Option 1 — Scenario 7	$70,440,000	$1,496,000	$71,936,000
Option 1 — Scenario 8	$70,440,000	$1,496,000	$71,936,000
Option 1 — Scenario 9	$70,440,000	$1,454,000	$71,894,000
Option 2/3 — Scenario 6	$86,695,000	$1,939,000	$88,634,000
Option 2/3 — Scenario 7	$70,440,000	$1,648,000	$72,087,000
Option 2/3 — Scenario 8	$70,440,000	$1,648,000	$72,087,000
Option 2/3 — Scenario 9	$65,021,000	$1,606,000	$66,627,000
Option 5 — Scenario 6	$83,986,000	$1,749,000	$85,735,000
Option 5 — Scenario 7	$62,312,000	$1,501,000	$63,813,000
Option 5 — Scenario 8	$62,312,000	$1,501,000	$63,813,000
Option 5 — Scenario 9	$62,312,000	$1,467,000	$63,779,000

6.2 DISCOUNTING AND AGGREGATING TO PRESENT VALUES

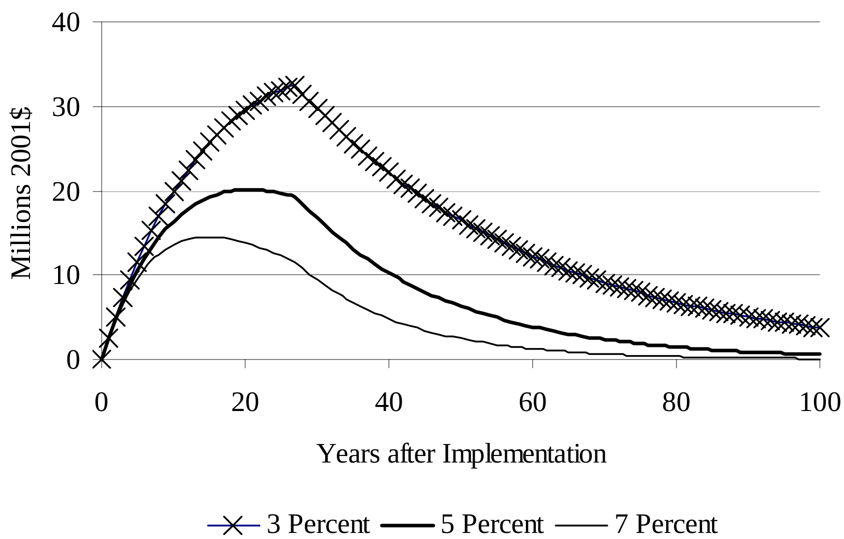
Exhibit 6-2 shows the timepath of undiscounted benefits under the primary assumptions used in the benefits assessment. As discussed in Section 3.6, we assume that impacts from nitrogen reductions will be translated into reduced well nitrate concentrations in a linear manner over 27 years. Benefits thus increase from the year of implementation until the 27th year when all the effects of reduced nitrogen loadings have been achieved at the well. From the 27th year onward the benefits are equal to the total benefits when all of the effects of reduced nitrogen loadings have been achieved at the well, as shown in Exhibit 6-2. The top line in Exhibit 6-2 shows the timepath of benefits for the Option 2/3 — Scenario 6, the lower line shows the timepath of benefits for Option 5 — Scenario 9, which produces the lowest benefits, and Option 2/3 — Scenario 8 falls within these bounds.

Exhibit 6-2
Timepath of Undiscounted Benefit Flows



In calculating present values we use an infinite time horizon. Exhibits 6-2 and 6-3 show the timepath for undiscounted or discounted present to 100 years for illustrative purposes only. Benefits received in the distant future (e.g., 100 years plus) are only a small percentage of total benefits even at the lowest discount rate used in this analysis (3%).

Exhibit 6-3
Discounted Value of Annual Benefits Using 3%, 5%, and 7% Discount Rates
Option 2/3 — Scenario 8



6.3 DISCOUNTED BENEFITS

Exhibit 6-3 shows the timepath of discounted benefits for Option 2/3 — Scenario 8 using a 3%, 5%, and 7% rate of discount. As can be seen, the present value of benefits increases over time as the number of wells achieving the steady state following regulation increases and then decreases from the maximum toward zero benefits because of the discounting of the future benefits.

The total present value of any given scenario/option will be the area under the curve using the given rate of discount. Exhibit 6-4 shows the total discounted present value for all scenarios using three different rates of discount: 3%, 5%, and 7%. Note that these numbers are presented in millions of 2001$, so the discounted present value for Option 2/3 — Scenario 8 using a 3% rate of discount is roughly $1,648 million. Using a 7% rate of discount, this falls to $478 million.

Exhibit 6-4
Total Present Value of Option/Scenarios Using Different Rates of Discount
(millions 2001$)

	3%	**5%**	**7%**
Scenario	**Present Value**	**Present Value**	**Present Value**
Option 1 — Scenario 6	$2,022.19	$984.01	$586.20
Option 1 — Scenario 7	$1,644.07	$800.02	$476.59
Option 1 — Scenario 8	$1,644.07	$800.02	$476.59
Option 1 — Scenario 9	$1,643.10	$799.54	$476.31
Option 2/3 — Scenario 6	$2,025.69	$985.72	$587.22
Option 2/3 — Scenario 7	$1,647.52	$801.70	$477.59
Option 2/3 — Scenario 8	$1,647.53	$801.70	$477.59
Option 2/3 — Scenario 9	$1,522.73	$740.97	$441.42
Option 5 — Scenario 6	$1,959.44	$953.47	$568.01
Option 5 — Scenario 7	$1,458.41	$709.67	$422.77
Option 5 — Scenario 8	$1,458.41	$709.67	$422.77
Option 5 — Scenario 9	$1,457.64	$709.30	$422.55

6.4 ANNUALIZED DISCOUNTED BENEFIT ESTIMATES

In addition to calculating the present value of estimated benefits, EPA developed an estimate of the annualized benefits attributable to the regulatory scenarios analyzed; these annualized values reflect the constant flow of benefits over time that would generate the associated present value.

The constant annual benefit A that, over a period of n years, equals the estimated present value (PV) of benefits is determined by:

$$A = PV(r) / \{1 - [1 / (1 + r)^n]\} ,$$

where r represents the annual discount rate. As n approaches infinity, this equation simplifies to:

$$A = PV * r .$$

EPA uses this equation to calculate the annualized benefits reported in this analysis. Exhibit 6-5 presents the annualized benefit estimates for the total present value benefits shown in Exhibit 6-4. For instance, for Option 2/3 — Scenario 8, a constant benefit flow of $49.43 million

Exhibit 6-5
Annualized Present Value of Option/Scenarios Using Different Rates of Discount (millions 2001$)

	3%	5%	7%
Scenario	**Annualized Value**	**Annualized Value**	**Annualized Value**
Option 1 — Scenario 6	$60.67	$49.20	$41.03
Option 1 — Scenario 7	$49.32	$40.00	$33.36
Option 1 — Scenario 8	$49.32	$40.00	$33.36
Option 1 — Scenario 9	$49.29	$39.98	$33.34
Option 2/3 — Scenario 6	$60.77	$49.29	$41.11
Option 2/3 — Scenario 7	$49.43	$40.08	$33.43
Option 2/3 — Scenario 8	$49.43	$40.08	$33.43
Option 2/3 — Scenario 9	$45.68	$37.05	$30.90
Option 5 — Scenario 6	$58.78	$47.67	$39.76
Option 5 — Scenario 7	$43.75	$35.48	$29.59
Option 5 — Scenario 8	$43.75	$35.48	$29.59
Option 5 — Scenario 9	$43.73	$35.46	$29.58

discounted at 3% would generate $1,648 million in total present value of benefits, also discounted at 3%.

6.5 ALTERNATIVE SPECIFICATION OF TIMEPATH: DISCONTINUATION OF NEW REGULATIONS IN 27TH YEAR

A potential alternative timepath specification involves the analysis of a regulatory regime where the proposed regulatory scenario would be in place for 27 years (until all reductions in nitrates had been realized at the well) and then the regulations would revert to current (2002) regulations. Under this scenario there would be an increase in benefits from the year of implementation until the 27th year, and then a decrease in benefits until the 54th year when conditions are assumed to have returned to current (2002). Exhibit 6-6 shows the maximum undiscounted annual value, the present value, and the annualized value for these scenarios using a 3% rate of discount.

Under this alternative specification of the timepath for regulations, Exhibit 6-7 shows the annualized benefits for the various options/scenarios using the three discount rates (3%, 5%, and 7%).

Exhibit 6-6
Benefits under Alternative Scenario of Regulatory Discontinuation in 27 Year (3% rate of discount)
(millions 2001$)

Scenario	Maximum Undiscounted Annual Value	Present Value	Annualized Value
Option 1 — Scenario 6	$88.48	$1,133.72	$41.11
Option 1 — Scenario 7	$71.94	$921.73	$33.42
Option 1 — Scenario 8	$71.94	$921.73	$33.42
Option 1 — Scenario 9	$71.89	$921.19	$33.40
Option 2/3 — Scenario 6	$88.63	$1,135.69	$41.18
Option 2/3 — Scenario 7	$72.09	$923.67	$33.49
Option 2/3 — Scenario 8	$72.09	$923.67	$33.49
Option 2/3 — Scenario 9	$66.63	$853.70	$30.96
Option 5 — Scenario 6	$85.74	$1,098.54	$39.83
Option 5 — Scenario 7	$63.81	$817.65	$29.65
Option 5 — Scenario 8	$63.81	$817.65	$29.65
Option 5 — Scenario 9	$63.78	$817.21	$29.63

6.6 SENSITIVITY ANALYSIS

6.6.1 Ranges of Value Estimates

As shown in Exhibit 5-5, Delavan (1997) reported a willingness to pay of $212.92 (see Section 5.2.3) and Jordan and Elnagheeb (1991) reported a willingness to pay of $204.94 per household per year (2001$). Using an approximation of $209 per household per year, Exhibit 6-8 shows how the annualized benefit estimates would change using this lower value for benefits to households achieving the MCL. Alternatively, Exhibit 6-8 also uses Edwards (1988) reported WTP of $2,530.22 (2001$) as an upper bound value for household benefits for achieving the MCL.

Exhibit 6-7
Annualized Benefits under Alternative Scenario of Regulatory Discontinuation in 27 Year (3%, 5%, and 7% rate of discount)
(millions 2001$)

Scenario	3%	5%	7%
Option 1 — Scenario 6	$41.11	$37.71	$33.78
Option 1 — Scenario 7	$33.42	$30.66	$27.46
Option 1 — Scenario 8	$33.42	$30.66	$27.46
Option 1 — Scenario 9	$33.40	$30.64	$27.45
Option 2/3 — Scenario 6	$41.18	$37.78	$33.84
Option 2/3 — Scenario 7	$33.49	$30.72	$27.52
Option 2/3 — Scenario 8	$33.49	$30.72	$27.52
Option 2/3 — Scenario 9	$30.96	$28.40	$25.43
Option 5 — Scenario 6	$39.83	$36.54	$32.73
Option 5 — Scenario 7	$29.65	$27.20	$24.36
Option 5 — Scenario 8	$29.65	$27.20	$24.36
Option 5 — Scenario 9	$29.63	$27.18	$24.35

6.6.2 Discount Rate

As shown in Exhibit 6-9, compared to the basic parameters used in the analysis, increasing the discount rate from 3% to 5% and 7% leads to a 18.9% and 32.4% reduction in estimated annualized benefits, respectively.

6.6.3 Time Line until Steady State is Achieved

As shown in Exhibit 6-10, comparing 27 years to 20 years until steady state is achieved increases the present annualized value by 9.8%. Spreading out time until steady state is achieved to 50 years decreases the present annualized value by 15.8%.

Exhibit 6-8
Change in Value for Crossing 10 mg/L

Discount Rate	3%	3%		3%	
Years to Steady State	27	27		27	
Value for Crossing 10 mg/L	$583.00	$209.00		$2,530.22	
Value for Changes below 10 mg/L	$2.09	$2.09		$2.09	
Scenario	**Annualized Value (2001$)**	**Annualized Value (2001$)**	**Percent Change in Annualized Value**	**Annualized Value (2001$)**	**Percent Change in Annualized Value**
Option 1 — Scenario 6	$60.67	$22.53	-62.9%	$259.20	327.3%
Option 1 — Scenario 7	$49.32	$18.34	-62.8%	$210.63	327.1%
Option 1 — Scenario 8	$49.32	$18.34	-62.8%	$210.63	327.1%
Option 1 — Scenario 9	$49.29	$18.31	-62.9%	$210.60	327.2%
Option 2/3 — Scenario 6	$60.77	$22.64	-62.7%	$259.31	326.7%
Option 2/3 — Scenario 7	$49.43	$18.44	-62.7%	$210.73	326.4%
Option 2/3 — Scenario 8	$49.43	$18.44	-62.7%	$210.73	326.4%
Option 2/3 — Scenario 9	$45.68	$17.08	-62.6%	$194.58	326.0%
Option 5 — Scenario 6	$58.78	$21.84	-62.8%	$251.11	327.2%
Option 5 — Scenario 7	$43.75	$16.34	-62.6%	$186.45	326.1%
Option 5 — Scenario 8	$43.75	$16.34	-62.6%	$186.45	326.1%
Option 5 — Scenario 9	$43.73	$16.32	-62.7%	$186.43	326.3%

6.6.4 Benefits for Changes under the 10 mg/L MCL

Counting only the value for reductions from above the MCL to below the MCL does not have a significant impact on the total annualized benefit estimate. As shown in Exhibit 6-11, reductions of nitrate concentrations below the 10 mg/L MCL and above the 1 mg/L "background" level add less than 5% to the estimated benefits.

Exhibit 6-9
Sensitivity to Changes in Discount Rate

Discount Rate	**3%**	**5%**		**7%**	
Years to Steady State	27	27		27	
Value for Crossing 10 mg/L	$583.00	$583.00		$583.00	
Value for Changes below 10 mg/L	$2.09	$2.09		$2.09	
Scenario	**Annualized Value (millions 2001$)**	**Annualized Value (millions 2001$)**	**Percent Change in Annualized Value**	**Annualized Value (millions 2001$)**	**Percent Change in Annualized Value**
Option 1 — Scenario 6	$60.67	$49.20	-18.9%	$41.03	-32.4%
Option 1 — Scenario 7	$49.32	$40.00	-18.9%	$33.36	-32.4%
Option 1 — Scenario 8	$49.32	$40.00	-18.9%	$33.36	-32.4%
Option 1 — Scenario 9	$49.29	$39.98	-18.9%	$33.34	-32.4%
Option 2/3 — Scenario 6	$60.77	$49.29	-18.9%	$41.11	-32.4%
Option 2/3 - Scenario 7	$49.43	$40.08	-18.9%	$33.43	-32.4%
Option 2/3 — Scenario 8	$49.43	$40.08	-18.9%	$33.43	-32.4%
Option 2/3 — Scenario 9	$45.68	$37.05	-18.9%	$30.90	-32.4%
Option 5 — Scenario 6	$58.78	$47.67	-18.9%	$39.76	-32.4%
Option 5 — Scenario 7	$43.75	$35.48	-18.9%	$29.59	-32.4%
Option 5 — Scenario 8	$43.75	$35.48	-18.9%	$29.59	-32.4%
Option 5 — Scenario 9	$43.73	$35.46	-18.9%	$29.58	-32.4%

The per mg/L value used for changes below the MCL came from the Crutchfield et al. and De Zoysa reports. As discussed in Chapter 5, Poe (1993) calculates an imputed WTP for a 1 mg/L reduction (or increase) in nitrates as a function of initial nitrate levels. A maximum per mg WTP of ~$147 (2001$) is seen when initial nitrate levels are close to 10 mg/L. Below 10 mg/L the per mg WTP falls to about $100 (2001$) per mg when the initial level is 4 mg/L. Sparco (1995) also estimated WTP for incremental changes in nitrate concentrations of $142.46 per mg/L (2001$). Using a conservative lower bound for these estimates of $100 per mg/L WTP value, the right-hand side of Exhibit 6-11 shows how much benefit estimate would increase using these value per mg/L estimates for incremental changes below the MCL.

Exhibit 6-10 Sensitivity to Changes in Time until Steady State (20 and 50 years) (all in 2001$)					
Discount Rate	**3%**	**3%**		**3%**	
Years to Steady State	27	20		50	
Value for Crossing 10 mg/L	$583.00	$583.00		$583.00	
Value for Changes below 10 mg/L	$2.09	$2.09		$2.09	
Scenario	**Annualized Value (2001$)**	**Annualized Value (2001$)**	**Percent Change in Annualized Value**	**Annualized Value (2001$)**	**Percent Change in Annualized Value**
Option 1 — Scenario 6	$60.67	$66.60	9.8%	$51.08	-15.8%
Option 1 — Scenario 7	$49.32	$54.15	9.8%	$41.53	-15.8%
Option 1 — Scenario 8	$49.32	$54.15	9.8%	$41.53	-15.8%
Option 1 — Scenario 9	$49.29	$54.11	9.8%	$41.51	-15.8%
Option 2/3 — Scenario 6	$60.77	$66.71	9.8%	$51.17	-15.8%
Option 2/3 — Scenario 7	$49.43	$54.26	9.8%	$41.62	-15.8%
Option 2/3 — Scenario 8	$49.43	$54.26	9.8%	$41.62	-15.8%
Option 2/3 — Scenario 9	$45.68	$50.15	9.8%	$38.47	-15.8%
Option 5 — Scenario 6	$58.78	$64.53	9.8%	$49.50	-15.8%
Option 5 — Scenario 7	$43.75	$48.03	9.8%	$36.84	-15.8%
Option 5 — Scenario 8	$43.75	$48.03	9.8%	$36.84	-15.8%
Option 5 — Scenario 9	$43.73	$48.01	9.8%	$36.82	-15.8%

6.7 OMISSIONS, BIASES, AND UNCERTAINTIES

Omissions, biases, and uncertainties are inherent in any analysis relying on several different data sources, particularly those that were not created specifically for that analysis. Exhibit 6-12 summarizes the omissions, biases, and uncertainties for this analysis. The column labeled “likely impact on benefit” indicates how the benefit estimate is influenced by the omission, bias, or uncertainty indicated for that row. For instance, in the row on “well location selection,” the benefit estimates discussed above may be positively biased (higher than true value) if the sampled wells in the Retrospective Database are mainly in areas with nitrate problems.

Exhibit 6-11
Sensitivity to Benefits from Changes below the MCL

Discount Rate	3%	3%		3%	
Years to Steady State	27	27		27	
Value for Crossing 10 mg/L	$583.00	$583.00		$583.00	
Value for Changes below 10 mg/L	$2.09	$0.00		$100.00	
Scenario	**Annualized Value (2001$)**	**Annualized Value (2001$)**	**Percent Change in Annualized Value**	**Annualized Value (2001$)**	**Percent Change in Annualized Value**
Option 1 — Scenario 6	$60.67	$59.44	-2.0%	$118.02	94.5%
Option 1 — Scenario 7	$49.32	$48.30	-2.1%	$97.39	97.5%
Option 1 — Scenario 8	$49.32	$48.30	-2.1%	$97.39	97.5%
Option 1 — Scenario 9	$49.29	$48.30	-2.0%	$95.99	94.7%
Option 2/3 — Scenario 6	$60.77	$59.44	-2.2%	$123.05	102.5%
Option 2/3 — Scenario 7	$49.43	$48.30	-2.3%	$102.34	107.1%
Option 2/3 — Scenario 8	$49.43	$48.30	-2.3%	$102.35	107.1%
Option 2/3 — Scenario 9	$45.68	$44.58	-2.4%	$97.25	112.9%
Option 5 — Scenario 6	$58.78	$57.58	-2.0%	$114.96	95.6%
Option 5 — Scenario 7	$43.75	$42.72	-2.4%	$91.95	110.2%
Option 5 — Scenario 8	$43.75	$42.72	-2.4%	$91.95	110.2%
Option 5 — Scenario 9	$43.73	$42.72	-2.3%	$90.85	107.7%

Alternatively, the benefit estimates discussed above may understate true values if, as indicated in the row on "per household value for reducing well nitrates to the MCL," the benefit estimates from Poe and Bishop are lower bound estimate of true values.

Data availability limited the variables included in this statistical analysis for the gamma model. Several variables, such as well construction and well age, proximity of wells to a pollutant source, and aquifer volume, composition and flow direction, were not included in this analysis even though they were significant factors in other studies.

Exhibit 6-12
Omissions, Biases, and Uncertainties in the Nitrate Loadings Analysis

Variable	Likely Impact on Benefit[a]	Comment
Well, land, and nitrate data		
Geographic coverage	Unknown	Date availability limited the well samples used in the statistical modeling to those from approximately 374 counties nationwide.
Well location selection	Positive	Wells sampled in the Retrospective Database may not be random. Samples may come from areas with problems with nitrate.
Year of sample	Unknown	Samples taken over 23 years. Land use and other factors influencing nitrate concentrations in the vicinity of the well may have changed over time.
Nitrate loadings from AFOs with 0-300 AU	Positive	Data for the smallest AFOs were not included in this analysis because they will not be affected by the proposed regulations. This may subsequently underestimate total loadings, resulting in an overestimate of the impact of nitrogen loadings on well nitrate concentrations.
Loadings estimates across counties in the NPLA loadings dataset	Positive	Average loadings estimates for counties included in the Retrospective Database are greater than in non-USGS counties. Estimated nitrate reductions in non-USGS counties may thus be overstated.
Percent of wells above 10 mg/L	Unknown	Based on the Retrospective Database, EPA assumes that 9.45% of wells currently exceed the MCL. If the true national percent is lower (higher) our analysis overstates (understates) benefits.
Sampling methods	Unknown	Data set compiled from data collected by independent state programs, whose individual methods for measuring nitrate may differ.
Model variables		
Well construction and age	Unknown	No reliable data available nationally.
Spatial data	Unknown	No national data available on the distance from well to pollutant source.

Exhibit 6-12 (cont.)
Omissions, Biases, and Uncertainties in the Nitrate Loadings Analysis

Variable	Likely Impact on Benefit[a]	Comment
Benefit calculations		
Per household value for reducing well nitrates to the MCL	Negative	The Poe and Bishop values generally appear to be a lower bound estimate of households' WTP for reducing nitrates to the MCL.
Years until wells achieve steady state.	Negative	The analysis assumes a linear path over 27 years until reduced nitrogen loadings would result in most wells achieving reduced nitrate concentrations. A large portion of wells (especially shallower wells) may achieve this on a much faster time path.
Exclusion of values for changes for wells still above the MCL after new regulations	Negative	Changes in nitrate concentrations for wells that are still above the MCL after new regulations are not valued because EPA does not have reliable value estimates for changes incremental changes above the MCL.
Exclusion of values for incremental changes for wells above the MCL before new regulations but below the MCL after new regulations	Negative	Changes in nitrate concentrations for wells that were above the MCL before new regulations, but below after new regulations, are not calculated since such values may be captured in benefit estimates used to value changes from above the MCL to below the MCL nitrate concentrations.

a. "Positive" impact implies that estimated benefits may be overstated; "negative" means that estimated benefits may be understated if the bias, omission, or uncertainty is not corrected for in the benefit estimate calculation.

This analysis assumes constant nitrate concentrations and loadings over time (including the past when data in the Retrospective Database were collected), omitting the potentially significant time lag associated with nitrate transport through soil and into the aquifer. This may be a significant source of error, considering that the loadings data are based on current conditions, the nitrate concentrations were sampled over a 20 year period, and nitrates may take decades to reach the groundwater.

With respect to the issue of loadings estimates across counties in the NPLA loadings dataset, there may be a potential bias due to selection of wells sampled for nitrate testing. Counties that had wells included in the gamma model dataset have different characteristics than counties not included. This may be because wells that are more likely to have higher nitrates because of conditions in their surrounding area are more likely to be tested. We attempted to explore this issue with the sample selection model discussed in Appendix B. Overall, our results suggest little impact due to potential sample selection.

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APPENDIX A
NITROGEN SOURCES AND WELL DATA

Several individual datasets were combined to create the county level loadings data used to model the relationship between nitrogen loadings and nitrate concentrations in private wells. The final loadings dataset includes estimates of the total nitrogen loadings for each county under each scenario, and was created by combining information from three different datasets provided by EPA. These separate datasets contained information on the number of facilities in each county, the percentage of these facilities that would be regulated under various scenarios, and the loadings for each type of facility in each region of the country. These individual data files were the loadings, facility, and state percent data files.

Loadings: The loadings dataset was provided as an Excel spreadsheet with multiple worksheets. The file contains information on modeled surface and leached nitrogen and phosphorous loadings from a variety of sources, including on-site and off-site manure application, fertilizer application, and loadings generated at farm production areas. A total of 250 facility types are included in the dataset. The farm types are defined as the combination of 10 animal types, 5 facility size categories, and 5 regions. Loadings were estimated for baseline conditions and for four regulatory options. Option 1 regulates loadings by setting limits on nitrogen application amounts. Option 2 regulates loadings by setting limits on phosphorous application amounts. Option 3 is similar to Option 2, but also requires liners for lagoons.[1] Finally, Option 5 is similar to Option 2 but also requires that lagoons be covered.

Facility: The facility dataset was provided as an Excel spreadsheet. This dataset identifies the average number of facilities by animal type and size for 2,637 counties (including some counties that have no facilities). The dataset identifies animal types of beef, veal, broilers, dairy, two types of swine, wet layers, dry layers, turkey, and heifers. Facilities are ranked as small, medium, or large based on the definitions in Exhibit A-1. The dataset has 2,637 observations (one for each county) and 55 variables, including identifier columns for the counties and number of facilities for the different animal type and facility size.

State Percent: The state percent dataset was provided as an Excel spreadsheet. The dataset identifies the percentage of each facility type that will be regulated under each scenario (including baseline) for each state.

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1. Option 3 is similar to Option 2 but also requires liners for lagoons. As the leached nitrate loadings are the same for our analysis under Options 2 and 3 these are reported simply as Option 2/3 throughout this report.

Exhibit A-1
Summary of Size Category Definitions for All Animal Types

Sector	Large	Medium[a]	Small[b]
Mature Dairy Cattle	More than 700	200-700	Less than 200
Veal Calves	More than 1,000	300-1,000	Less than 300
Cattle or Cow/Calf Pairs	More than 1,000	300-1,000	Less than 300
Heifer	More than 1,000	300-1,000	Less than 300
Swine (weighing over 25 kilograms)	More than 2,500	750-2,500	Less than 750
Swine (weighing less than 25 kilograms)	More than 10,000	3,000-10,000	Less than 3,000
Horses[c]	More than 500	150-500	Less than 150
Sheep or Lambs[c]	More than 10,000	3,000-10,000	Less than 3,000
Turkeys	More than 55,000	16,500-55,000	Less than 16,500
Chickens (wet manure systems)	More than 30,000	9,000-30,000	Less than 9,000
Chickens Other than Laying Hens (dry manure systems)	More than 125,000	30,000-125,000	Less than 30,000
Laying Hens (dry manure systems)	More than 82,000	25,000-82,000	Less than 25,000
Ducks (dry operations)[c]	More than 40,000	12,000-40,000	Less than 12,000
Ducks (wet operations)[c]	More than 5,000	1,500-5,000	Less than 1,500

a. Must also meet one of two criteria to be defined as a CAFO.
b. Must be designated by the permitting authority.
c. Not included in final analysis.

Output — County Level Total Nitrogen Loadings Dataset: The output of combining these datasets is the nitrogen loadings for each county for each of the options/scenarios for the 2,637 counties with AFOs. Data from 678 of these 2,637 counties are combined with data from the USGS Retrospective Database (described below) for estimation of the gamma model.[2] An issue is whether the counties used for the gamma modeling are different in some manner from those (1,959) not used for estimating the nitrogen-nitrate relationship. Exhibit A-2 shows mean values for the average loadings and various sociodemographic data from these two groups of counties. The "percent difference" column indicates how much larger (or smaller for negative values) the mean values are for counties used in the gamma modeling compared to counties not used in the gamma model. The Z score from a Wilcoxon rank test show whether the differences are statistically significant. All of the Z scores are significant at the 1% level. In general the average county nitrogen loadings in the gamma model counties are higher than the excluded

2. Of these 678 counties, 374 have at least one well with enough data to be included in the analysis.

Exhibit A-2
Comparison of Mean Loadings and Sociodemographics for Counties in the Loadings Database Used in the Gamma Modeling
(for counties in model, n = 374; for counties not in model, n = 2,263)

Variable	Mean (counties included in the gamma model)	Mean (counties not included in the gamma model)	Percent Difference (from counties not included)	Z (Wilcoxon rank test)
Baseline nitrogen loadings	410,019	195,013	-52%	9.66
Option 1 — Scenario 6	299,521	157,612	-47%	10.72
Option 1 — Scenario 7	319,964	164,853	-48%	10.61
Option 1 — Scenario 8	319,959	164,851	-48%	10.61
Option 1 — Scenario 9	323,081	166,307	-49%	10.59
Option 2/3 — Scenario 6	289,843	154,159	-47%	10.71
Option 2/3 — Scenario 7	311,410	161,154	-48%	10.63
Option 2/3 — Scenario 8[a]	311,402	161,150	-48%	10.63
Option 2/3 — Scenario 9	314,788	162,657	-48%	10.63
Option 5 — Scenario 6	315,212	170,659	-46%	10.81
Option 5 — Scenario 7	327,810	172,942	-47%	10.67
Option 5 — Scenario 8	327,802	172,938	-47%	10.67
Option 5 — Scenario 9	329,605	173,613	-47%	10.66
Loadings per Acre (baseline)	0.99	0.51	-48%	9.76
Acres	528,201	732,337	39%	-2.68
Population[b]	83,296	74,900	-10%	3.67
Population Density[b]	0.13	0.11	-11%	4.36
Percent of County Land in Farms[b]	0.62	0.53	-14%	5.05
Median Household Income[b]	26,196	23,761	-9%	7.71
Housing Units[b]	33,451	30,766	-8%	3.17

a. Proposed scenario.
b. nobs = 2,257 for “not in the retrospective database” and 374 for “in the retrospective database.”

counties. In addition the included counties are somewhat smaller (30% smaller) and have a roughly 10-30% larger population, higher median income, and greater number of housing units. The included counties also have a larger portion of their land area in farms.

Septic Ratio: EPA calculated the number of septic systems per acre in each county using data from the 1990 U.S. Census and the 1997 Census of Agriculture. This provides a proxy measure for the contribution of septic systems to well nitrate concentrations. The number of household units on septic systems for each county was reported in the U.S. Census, and the total acres per county was reported in the Census of Agriculture.

The USGS Retrospective Database: As discussed in Section 2.2.1, the USGS Retrospective Database contains water quality and land use data from 10,426 well samples from 725 counties in 38 states. The data were gathered between 1969 and 1992.[3]

The dataset provides information on well location, well characteristics, pollution inputs, and well water sample. Each observation provides well location information, including FIPS code, town, state FIPS code, county FIPS code, study unit, well identification number, and latitude and longitude. Well characteristics include water use (e.g., domestic, stock, public, or irrigation), well depth in feet, depth to water in feet, geographic region, soil hydrologic group, lithological description of the aquifer, land use category (e.g., agricultural, woods, or urban), population density in people per square kilometer, the ratio of pasture to cropland, and the ratio of woodland to cropland.

Pollution input information includes atmospheric nitrogen input, fertilizer nitrogen input in tons sold per square mile, fertilizer plus atmospheric nitrogen inputs in tons per square mile, fertilizer plus atmospheric nitrogen inputs in pounds per acre, manure nitrogen input in tons per square mile, and the sum of nitrogen inputs. Well water sample information includes ammonia as nitrogen in mg/L, nitrate as nitrogen in mg/L, total phosphate in mg/L, and orthophosphate as phosphorous in mg/L, and the year of the sample.

Exhibit A-3 provides summary statistics on the observations from the USGS Retrospective Database for all observations in the dataset. This includes all water use types. Only a subset of these observations (2,985 observations) were usable for the analysis described in Chapter 3 because of missing data. The mean well nitrate concentration is 2.89 mg/L, ranging from no nitrates to 125.64 mg/L. Of the 10,426 observations in the retrospective dataset, 19.8% are at or below 1.0 mg/L and 7.4% exceed the MCL of 10 mg/L.

3. Of these 725 counties, 374 are also estimated to have nitrate loadings >0.

Exhibit A-3
USGS Retrospective Database Summary Data
(all water use types)

Variable	N	Missing	Mean	Std Dev	Min.	Max.
Well Depth (feet)	9141	1285	282.728	400.204	1.000	5310.000
Soil Hydrologic Group	10419	7	2.549	0.729	1.000	4.000
Pop. Density (people per km square)	10426	0	131.958	427.002	0.000	13516.670
Atmospheric Nitrogen Input	10426	0	1.355	0.598	0.172	2.910
Fertilizer Nitrogen Input (tons/mi sq)	10426	0	5.958	6.210	0.000	30.010
Fertilizer plus Atmospheric Nitrogen Input (tons/mi sq)	10426	0	7.313	6.374	0.208	31.882
Fertilizer plus Atmospheric Nitrogen Input (lbs/acre)	10426	0	22.853	19.920	0.650	99.631
Manure Nitrogen Input (tons/mi sq)	10426	0	4.086	5.614	0.000	34.502
Sum of Nitrogen Inputs	10426	0	11.400	9.887	0.219	50.048
Ratio of Pasture to Cropland	9981	445	5.502	19.374	0.006	147.991
Ratio of Woodland to Cropland	9772	654	0.500	1.391	0.000	14.880
Year of Sample	9289	1137	1982.509	5.629	1969.000	1992.000
Nitrate as Nitrogen (mg/L)	10426	0	2.886	5.958	0.000	125.640
Total Phosphate (mg/L)	3336	7090	0.069	0.263	0.000	7.500

Exhibit A-4 shows the distribution of well water use for observations in the USGS dataset. Because the benefits transfer exercise is focused on domestic water well use, we limited analysis to wells listed as domestic, which make up roughly 31% of the observations from the Retrospective Database.

Exhibit A-4
Distribution of Well Water Use in Retrospective Database

Water Use	Frequency	Percent
Domestic	3226	30.94
Irrigation	838	8.04
Public	1088	10.44
Stock	209	2.00
Unknown	5065	48.58

Exhibit A-5 presents the summary information for only those wells listed as being for domestic use. Of particular interest for the modeling described in Chapter 3 is the observation that the average total of fertilizer sales and atmospheric nitrogen inputs (8.85 tons/square mile) exceeds that from manure of 6.03 tons/square mile. This suggests that in understanding the potential benefits of controlling nitrogen inputs to groundwater from CAFOs it is important to control for non-CAFO nitrogen sources. In the analysis for this rule, EPA estimated leached nitrogen from the application of fertilizer under each regulatory scenario. As discussed above, EPA used a proxy measure, density of septic systems in a county, to control for nitrogen loadings from septic systems.

Exhibit A-5
USGS Retrospective Database Summary Data
(domestic water use only)

Variable	N	Missing	Mean	Std Dev	Min.	Max.
Well Depth (feet)	3068	158	169.320	135.569	1.000	1996.000
Soil Hydrologic Group	3225	1	2.425	0.654	1.000	4.000
Pop. Density (people per km square)	3226	0	47.071	136.469	0.045	2321.628
Atmospheric Nitrogen Input	3226	0	1.627	0.595	0.172	2.855
Fertilizer Nitrogen Input (tons/mi sq)	3226	0	7.224	5.992	0.000	30.010
Fertilizer plus Atmospheric Nitrogen Input (tons/mi sq)	3226	0	8.851	5.999	0.215	31.882
Fertilizer plus Atmospheric Nitrogen Input (lbs/acre)	3226	0	27.658	18.746	0.672	99.631
Manure Nitrogen Input (tons/mi sq)	3226	0	6.033	7.271	0.000	34.502
Sum of Nitrogen Inputs	3226	0	14.884	10.343	0.219	44.114
Ratio of Pasture to Cropland	3143	83	0.945	2.432	0.012	24.597
Ratio of Woodland to Cropland	3117	109	0.234	0.437	0.000	6.227
Year of Sample	2789	437	1983.068	5.542	1969.000	1991.000
Nitrate as Nitrogen (mg/L)	3226	0	3.548	6.406	0.000	84.300
Total Phosphate (mg/L)	1006	2220	0.068	0.291	0.000	6.400

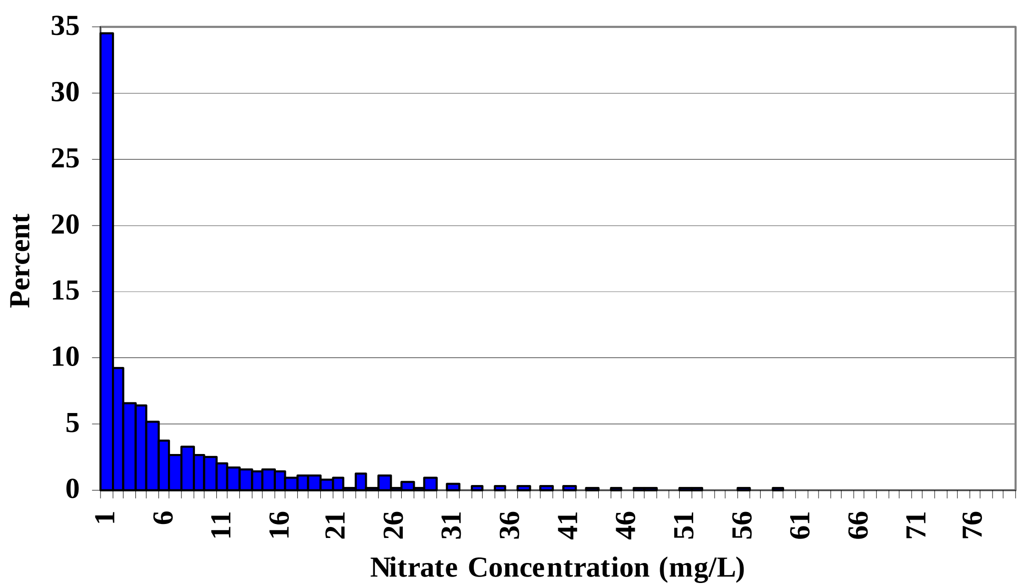
APPENDIX B
STATISTICAL MODELS

As described in Section 3.2, the statistical analysis of the relationship between loadings and well nitrate concentrations is based on the following linear model:

$$\text{Nitrate (mg/L)} = ß_0 + ß_1 \text{ ag dummy} + ß_2 \text{ soil group} + ß_3 \text{ well depth}$$
$$+ ß_4 \text{ septic ratio} + ß_5 \text{ atmospheric N} + ß_6 \text{ loadings ratio}$$

Well nitrate concentrations are the dependent variable in the analysis. Summary statistics on the distribution of observed values for well nitrates indicate a nonnegative distribution with a rightward skew (skew = 4.85) and a thick tail (kurtosis = 37.15) (see Exhibit B-1).

Exhibit B-1
Nitrate Distribution: Observed Values



The gamma and exponential distributions both allow for fitting of nonnegative, right skewed distributions (no observations are assumed to be censored in the exponential or gamma models). The gamma distribution has the density function:

$$f(y) = \frac{\theta^{\alpha}}{\Gamma(\alpha)} \exp(-\theta y) y^{\alpha-1} .$$

We used the gamma distribution instead of the more commonly used exponential distribution since it is more general that the exponential model (includes the exponential specification as a special case).[1] The gamma distribution allows for the density function to be more flexible and allows for more curvature in the distribution. To model the relationship between the nitrate levels (y) and the independent variables, let $\theta_i = \exp(-\beta x_i)$. For this distribution, $E(y_i) = \alpha / \theta_i = \alpha \exp(\beta x_i)$. Maximum likelihood methods are used to estimate the parameters. The log likelihood function is:

$$\log L(y_i | x_i; \alpha, \beta) = \sum_i [\alpha \log \theta_i - \log \Gamma(\alpha) - \theta_i y_i + (\alpha - 1) \log(y_i)].$$

This log likelihood was maximized using GAUSS software.[2] Estimation results are displayed in Exhibit B-2. All of the parameter estimates are significant at the 1% level and are of the expected sign. From the gamma model, expected values can be calculated using:

$$E(y_i) = \alpha / \theta_i = \alpha \exp(\beta x_i).$$

EPA tested the ability of the gamma model to estimate small nitrate concentrations by comparing the model's intercept with the natural, or ambient, level of nitrate in groundwater in the United States.[3] Using the mean values for soil group and well depth and setting all other variables to zero (i.e., setting the ag dummy and all human nitrogen sources to zero), the model predicts an

1. A likelihood ratio test of the difference between the exponential model (where α is restricted to equal 1) and the gamma model (where alpha is estimated) yielded a χ^2 statistic of 659.98, so that the null hypothesis that $\alpha = 1$ is rejected at any level of significance (the 1% tail of the $\chi^2_{(1)}$ distribution is at 6.63).

2. A range of starting values were used in the GAUSS program to examine the sensitivity of results to starting values. For all starting values for which the program converged, virtually the identical parameter estimates were obtained.

3. Technically, the intercept term includes ambient levels of nitrates as well as those induced by loadings from AFOs with less than 300 AUs since these are not included in the loadings data.

Exhibit B-2
Gamma Regression Results

Variable	Parameter Estimate	Standard Error	Asymptotic T-Statistic	Significance
Intercept	2.201	0.194	11.352	0.000
Loadings Ratio	0.046	0.007	6.543	0.000
Atmospheric Nitrogen	0.032	0.028	1.144	0.253
Well Depth[a]	-0.171	0.012	-13.782	0.000
Soil Group	-0.384	0.044	-8.660	0.000
Septic Ratio	1.618	1.728	0.936	0.349
Ag Dummy	0.686	0.064	10.663	0.000
Central Region Dummy	-0.076	0.160	-0.475	0.635
Mid-Atlantic Region Dummy	-0.165	0.098	-1.691	0.091
Pacific Region Dummy	0.812	0.117	6.918	0.000
South Region Dummy	-0.907	0.127	-7.170	0.000
Alpha	0.497	0.010	50.639	0.000
Mean log-likelihood = -1.85646.				
N = 2,985.				
a. In the model, well depth is scaled to units of hundreds of feet.				

ambient nitrate concentration in the Midwest region of 1.32 mg/L on nonagricultural lands. Using the same approach, the predicted value on agricultural land is 2.63 mg/L. Several studies report natural nitrate levels ranging between 2 and 3 mg/L (Poe and Bishop, 1992; Kross et al., 1993; Poe, 1998), although one study suggests that 3 mg/L may be too high, given the high number of wells with nitrate levels below the detection limit in many groundwater monitoring studies (Spalding and Exner, 1993). Giraldez and Fox (1995) report that natural nitrate concentration in groundwater is generally about 1.0 mg/L. Therefore the model's estimate of 1.32 mg/L on non-agricultural land seems to be a reasonable estimate of nitrate concentrations in the absence of the pollution from septic systems, atmospheric deposition, and AFOs.

Other Models

In addition to the gamma model described above, several other model types were explored for this analysis. Given the nature of nitrate contaminations, a nonnegative distribution is preferred. The OLS and Tobit models discussed here were estimated to allow us to explore whether these simpler models would suffice for purposes of modeling the nitrate-nitrogen relationship. The OLS, Tobit, and Selection-Trunctation models were estimated using GAUSS Version 4.0.

Ordinary Least Squares (OLS)

OLS was used initially to model the loadings-well nitrate relationship to explore how well the data could explain this relationship. Estimation results are displayed in Exhibit B-3.

Exhibit B-3
OLS Regression Results

Variable	Parameter Estimate	Standard Error	T Value
Intercept	4.907	0.768	6.391
Loadings Ratio	0.197	0.031	6.374[a]
Atmospheric Nitrogen	0.176	0.117	1.511
Well Depth[b]	-0.625	0.086	-7.278[a]
Soil Group	-1.234	0.184	-6.722[a]
Septic Ratio	-2.768	6.786	-0.408
Ag Dummy	1.709	0.289	5.910[a]
Central Region Dummy	0.048	0.688	0.070
Mid-Atlantic Region Dummy	-0.262	0.394	-0.666
Pacific Region Dummy	3.292	0.496	6.637[a]
South Region Dummy	-1.865	0.535	-3.485[a]
F value = 31.946; Adjusted R^2 = 0.094.			
N = 2,985.			
a. Indicates significant at the 1% level. b. In the model, well depth is scaled to units of hundreds of feet.			

The results indicate that there are significant relationships between the dependent and most independent variables. The signs are all of the expected direction. The coefficient on Loadings Ratio is significant at the 1% level. It must be emphasized that there are a priori reasons to prefer a distribution that does not allow for negative values in the dependent variable (well nitrate concentrations), and thus the OLS and Tobit models were purely exploratory models.

Tobit

Since well nitrates at or below the detection limit were reported in a number of ways, nondetects were set to 0.05 mg/L; 522 of the 2,985 observations had nitrate values reported at the detection limit. Treating this as a censoring of the distribution, we used a Tobit model to estimate the parameter coefficients. Exhibit B-4 reports the Tobit model estimates.

Exhibit B-4
Tobit Regression Results

Variable	Estimate	Standard Error	Chi-Square	Pr > ChiSq
Intercept	5.200	0.941	5.529	0.000
Loadings Ratio	0.216	0.036	5.981	0.000
Atmospheric Nitrogen	0.222	0.140	1.588	0.112
Well Depth[a]	-0.731	0.102	-7.186	0.000
Soil Group	-1.618	0.233	-6.950	0.000
Septic Ratio	4.184	8.959	0.467	0.641
Ag Dummy	2.227	0.371	6.011	0.000
Central Region Dummy	0.382	0.803	0.476	0.634
Mid-Atlantic Region Dummy	-0.543	0.548	-0.992	0.321
Pacific Region Dummy	3.447	0.563	6.129	0.000
South Region Dummy	-4.149	0.735	-5.646	0.000
Sigma	1.914	0.014	132.804	0.000
Log likelihood: -1.90865.				
N = 2,985.				
a. In the model, well depth is scaled to units of hundreds of feet.				

As seen in Exhibit B-4, the Tobit model produced generally strong results with significant coefficient estimates of the correct sign. While the Tobit model is used for modeling observations on non-negative values, in this case with observations truncated at nitrate concentrations below the detection limit, using the model to fit expected values could still predict negative nitrate concentrations. We thus used the Tobit model purely to explore the data and the relationships between dependent and independent variables as well as potential misspecifications of the error term.

Exponential

As with the OLS model, the Tobit model may not be appropriate to use to explore the physical relationship between nitrogen loadings and well nitrate concentrations because the Tobit model assumes a censoring of true values at zero, and true nitrate concentrations are non-negative. We thus explored the use of the exponential and gamma models as nonnegative distributions. Assuming the y_i follow the exponential distribution, the density function is:

$$f(y) = \theta \exp(-\theta y)$$

Letting $\theta_i = \exp(-\beta x_i)$, the expected value of y_i is $E(y_i) = 1/\theta_i = \exp(\beta x_i)$. Maximum likelihood methods are used to estimate the parameters. The log-likelihood function is:

$$\log L(y_i | x_i; \beta) = \sum_i [\log \theta_i - \theta_i y_i]$$

The only difference between the exponential and gamma models is that α is set to 1 for the exponential model. In the more general gamma model, α is estimated. As discussed above, α was found to be significantly different from 1 and thus we felt the gamma model represented a better model to use for scenario analysis for CAFOs. Exhibit B-5 presents the results of estimating the exponential model using GAUSS.

Exhibit B-5
Exponential Regression Results

Variable	Parameter Estimate	Standard Error	Asymptotic T Value
Intercept	1.502	0.141	10.641[a]
Loadings Ratio	0.046	0.005	9.192[a]
Atmospheric Nitrogen	0.032	0.020	1.571
Well Depth[a]	-0.171	0.009	-18.415[a]
Soil Group	-0.384	0.030	-12.723[a]
Septic Ratio	1.616	1.219	1.326
Ag Dummy	0.686	0.049	13.995[a]
Central Region Dummy	-0.076	0.116	-0.652
Mid-Atlantic Region Dummy	-0.165	0.072	-2.293[a]
Pacific Region Dummy	0.812	0.087	9.304[a]
South Region Dummy	-0.907	0.093	-9.812[a]
Mean log-likelihood = -2.07756.			
N = 2,985.			

a. In the model, well depth is scaled to units of hundreds of feet.
b. Indicates significant at the 0.01% level.

Most of the coefficients are significant at the 1% level. The exceptions are atmospheric nitrogen deposition, the septic ratio, and the Central Region dummy. All the variables have the expected sign. The coefficients are nearly identical to those estimated in the gamma model except for the alpha coefficient, which is implicitly restricted to 1 in the exponential model (Exhibit B-2), and the septic ratio coefficient, which differs by 0.001. Note that the data for well depth was scaled in order for GAUSS to converge to a solution.

Using the parameter estimates from the exponential model, we can calculate expected ambient nitrate levels. Using mean values for well depth and soil type, and setting all anthropocentric nitrogen sources equal to zero, the expected ambient well nitrate concentration in the Midwest is 1.32 mg/L for non-agricultural land and 2.62 mg/L for wells on agricultural land. These values are within the range of natural or ambient nitrate concentrations as reported in Section 3.3.

Selection Model

Some of the models above perform relatively well in predicting nitrogen concentration. Some are statistically stronger than others. They all share a common weakness, though, in that the data used in the model may not be an unbiased sampling of wells in the United States. Of the more than 3,000 counties in the United States, only 374 were represented in the final database created for this analysis. Geographically, these counties tend to be concentrated in the Midwest and Middle Atlantic states (with a very large percentage of the wells sampled located in Nebraska and Pennsylvania). The results presented Exhibit A-2 in Appendix A indicate that not only are the counties used in the model geographically different than counties not used included in the model, but also the characteristics of these counties appear to be quite different. In particular, it appears that the wells included in the Retrospective Database tend to be located in counties with higher than average levels of nitrogen loadings from manure and fertilizer.

To explore correcting for this potential problem EPA developed a selection model that includes components which aim to capture the effects of sample selection bias. Ultimately, this model generates estimates of nitrate concentration that are quite similar to the estimates from the gamma model. Thus, EPA used the gamma model, which was used as the primary model for the proposed rule, as the primary model for the final benefit analysis as well. The results of the sample selection model are included here for comparison. The details of the sample selection model follow.

Let y_{ij}^{*} be the theoretical nitrate concentration of the *i*-th well in county *j*, for $j = 1, \ldots, J\,(\sim 300)$ and $i = 1, \ldots, n_j \left(\sum n_j = 2985\right)$. Concentration is modeled as a function of well characteristics and county characteristics, both observed and unobserved:

$$y_{ij}^{*} = \beta' x_{ij} + z_j' \gamma + u_j + \varepsilon_{ij}, \qquad \text{(B-1)}$$

where the z_j are the observed covariates common to all wells in county *j*, the x_{ij} are the observed characteristics of the particular well, and the random unobserved factors are $u_j \sim N\left(0, \sigma_u^2\right)$ and $\varepsilon_{ij} \sim N\left(0, \sigma_\varepsilon^2\right)$, assumed mutually independent and independent of one another. That is,

$$E\left(u_j \varepsilon_{ij}\right) = 0 \quad \forall i, j$$

and

$$\begin{aligned} E(u_j u_{j'}) &= 0 \\ E(\varepsilon_{ij} \varepsilon_{ij'}) &= 0 \quad \forall j \neq j'. \end{aligned} \qquad \text{(B-2)}$$

Suppose nitrate concentration data for a well exists only if the county-specific component, $z_j'\gamma + u_j$, is sufficiently high. Furthermore, we can measure concentration only down to 0.05 per mg/L. The selection model then is

$$y_{ij} = \begin{cases} y_{ij}^* & \text{if } y_{ij}^* > .05 \text{ and } u_j > -z_j'\gamma \left(\text{regime I}\right) \\ .05 & \text{if } y_{ij}^* \le .05 \text{ and } u_j > -z_j'\gamma \left(\text{regime II}\right), \end{cases} \tag{B-3}$$

and there is information on y_{ij} only if $u_j > -z_j'\gamma$. This is a censored/truncated regression model (censored by the detection limit; truncated by the county rule). Let $w_{ij} = u_j + \varepsilon_{ij}$. The likelihood is

$$\begin{aligned} L(y_{ij}) &= \prod_{\mathrm{I}} f(y_{ij}, u | y_{ij} > .05, u_j > -z_j'\gamma) \times P(y_{ij} > .05, u_j > -z_j'\gamma) \\ &\times \prod_{\mathrm{II}} P[y_{ij} \le .05, u_j > -z_j'\gamma], \end{aligned} \tag{B-4}$$

which can be written

$$L\left(y_{ij}\right) = \prod_{\mathrm{I}} \int_{-\gamma' z}^{\infty} f\left(y_{ij}, u\right) du \cdot \prod_{\mathrm{II}} P\left[w_{ij} < .05 - \left(\beta x_{ij} + z_j'\gamma\right), u_j > -z_j'\gamma\right]. \tag{B-5}$$

In Equation B-5 $f\left(\cdot,\cdot\right)$, is the bivariate normal density, and the bivariate probability is from the normal distribution:

$$\begin{pmatrix} w_{ij} \\ u_j \end{pmatrix} \sim N\left[\begin{pmatrix} 0 \\ 0 \end{pmatrix}, \begin{pmatrix} \sigma_u^2 + \sigma_\varepsilon^2 & \sigma_u^2 \\ \sigma_u^2 & \sigma_u^2 \end{pmatrix}\right]. \tag{B-6}$$

The contribution to the likelihood in the first data regime can be written

$$\frac{1}{\sigma_w} \phi\left[\frac{y_{ij} - \beta' x_{ij} - z_j'\gamma}{\sigma_w}\right] \cdot \Phi\left[\frac{\gamma' z + \rho \frac{\sigma_u}{\sigma_w}\left(y_{ij} - \beta' x_{ij} - z_j'\gamma\right)}{\sigma_w \sqrt{1-\rho^2}}\right], \tag{B-7}$$

where $\sigma_w^2 = \sigma_u^2 + \sigma_\varepsilon^2$. For the second data regime ($y_{ij}^* < .05$ and $u_j > -z_j'\gamma$), we have

$$P\left[w_{ij} < .05 - \left(\beta' x_{ij} + z_j'\gamma\right), u_j > -z_j'\gamma\right] = \tag{B-8}$$
$$P\left[w_{ij} < .05 - \left(\beta' x_{ij} + z_j'\gamma\right), -u_j < z_j'\gamma\right] =$$
$$P\left[\frac{w_{ij}}{\sqrt{\sigma_u^2 + \sigma_\varepsilon^2}} < \frac{.05 - \left(\beta' x_{ij} + z_j'\gamma\right)}{\sqrt{\sigma_u^2 + \sigma_\varepsilon^2}}, \frac{-u_j}{\sigma_u} < \frac{z_j'\gamma}{\sigma_u}\right] =$$
$$\Phi\left[\frac{.05 - \left(\beta' x_{ij} + z_j'\gamma\right)}{\sqrt{\sigma_u^2 + \sigma_\varepsilon^2}}, \frac{z_j'\gamma}{\sigma_u}; -\rho\right],$$

where Φ is the cdf of the cumulative distribution function for the standardized bivariate normal random vector with correlation

$$\rho = \frac{\sigma_u}{\sqrt{\sigma_u^2 + \sigma_\varepsilon^2}}. \tag{B-9}$$

The results from the estimation of the sample selection model are presented in Exhibit B-6. The results are very similar to the Tobit model reported in Exhibit B-4. All the significant parameters have intuitive signs. The only variable in the model with a counterintuitive sign is atmospheric nitrogen deposition, which has a negative but not statistically significant sign. Most of the other parameters are significant at the 1% level of significance; the septic ratio and two of the regional dummies are the exceptions.

Estimated Benefits

Exhibit B-7 presents the benefits of four scenarios (Scenarios 6 and 8, combined with Options 2 and 2), as estimated using the gamma, Tobit, and Selection models. The estimates in the table range from a low of $66.88 million (Option1-Scenario8 in the Selection model), to a high of $94.07 million (Option2/3-Scenario6 in the Tobit model). The largest spread is found for Option1-Scenario6, where the difference between the minimum and maximum estimates from the three models is $8.13 million.

It is interesting to note that the relative magnitudes of the total benefits vary by model, option, and scenario. For Option 1, the gamma model produces the largest benefits estimates. For proposed option, Option 2/3, the gamma model produces estimated benefits that are smaller than the selection model and only slightly larger than the Tobit model. It is not entirely clear why the Selection model should produce benefits estimates that are larger than the gamma model. Since the gamma model produces the most conservative estimate of benefits for the proposed rule, and because it does not allow the prediction of negative concentrations, it is the preferred model for the groundwater analysis.

Exhibit B-6
Sample Selection Model Regression Results

Variable	Parameter Estimate	Standard Error	Asymptotic T-Statistic	Significance
Intercept	7.540	1.056	7.143	0.000
Loadings Ratio	0.218	0.040	5.472	0.000
Atmospheric Nitrogen	-0.116	0.141	-0.819	0.413
Well Depth[a]	-0.716	0.105	-6.811	0.000
Soil Group	-2.317	0.221	-10.474	0.000
Septic Ratio	1.100	10.653	0.103	0.918
Ag Dummy	2.220	0.495	4.488	0.000
Central Region Dummy	0.052	1.108	0.047	0.963
Mid-Atlantic Region Dummy	0.385	0.593	0.650	0.516
Pacific Region Dummy	3.580	0.609	5.877	0.000
South Region Dummy	-4.465	0.895	-4.991	0.000
σ^2_e	3.863	0.012	320.014	0.000
Mean log-likelihood = -2.89556.				
N = 2,985.				
a. In the model, well depth is scaled to units of hundreds of feet.				

Exhibit B-7
Estimated Benefits from Various Models

Regulatory Scenario	Model	Expected Reductions in Number of Households with Well Nitrate Concentrations above 10 mg/L	Total Expected National Nitrate Reduction (mg/L) for Wells with Concentrations between 1 and 10 mg/L at Baseline	Undiscounted Annual Benefits under CAFO Regulatory Scenarios (Millions 2001$)
Option 1 — Scenario 6	Gamma	148,705	854,326	88.48
Option 1 — Scenario 6	Tobit	134,764	851,575	80.35
Option 1 — Scenario 6	Selection	144,058	1,085,613	86.25
Option 1 — Scenario 8	Gamma	120,823	716,007	71.94
Option 1 — Scenario 8	Tobit	116,176	695,699	69.18
Option 1 — Scenario 8	Selection	111,529	891,660	66.88
Option 2/3 — Scenario 6	Gamma	148,705	927,730	88.63
Option 2/3 — Scenario 6	Tobit	157,999	935,138	94.07
Option 2/3 — Scenario 6	Selection	148,705	1,190,218	89.18
Option 2/3 — Scenario 8[a]	Gamma	120,823	788,305	72.09
Option 2/3 — Scenario 8[a]	Tobit	120,823	775,856	72.06
Option 2/3 — Scenario 8[a]	Selection	130,117	992,566	77.93
a. Proposed scenario.				

APPENDIX C
SUMMARY OF GROUNDWATER VALUATION OF NITRATE CONTAMINATION LITERATURE

Study Reference	Crutch field et al.	Delavan	de Zoysa	Edwards	Giraldez and Fox
1. Published/Peer Reviewed?	USDA ERS Report	Master's thesis	PhD dissertation	Journ. of Environmental Economics and Management	Canadian Journ. of Agricultural Economics
2. Year of Analysis	1994	1996	1994	1987	1995
3. Place	IN, Central NE, PA, WA	Southeastern PA (parts of Lebanon and Lancaster counties)	Maumee River Basin, northwest Ohio	Cape Cod, MA	Hensall, southwestern Ontario
4. Type of Study	Survey eliciting WTP for improved water quality	Survey eliciting WTP for improved water quality	Survey eliciting WTP for improved water quality	Survey eliciting WTP to prevent contamination of aquifer	Metadata (lifetime earnings, wage risk studies, & CVM)
5. Survey Implement	Telephone	Mail	Mail	Mail	n.a.[a] (n.a. = not applicable)
6. Respondents	1600?[b]	1000 mailed	1050[c]	1000 mailed	n.a.
7. Response Rate	50% (819 usable responses)	68.6%	51% overall	78.5% (58.5% analyzable)	n.a.
8. Location (urban, rural, etc.)[d]	Unspecified	75% of respondents live in borough or city; 6.3% involved with farming	Urban, suburban, and rural	Primarily rural	n.a.
9. Who Was Asked?	Residents	Residents	Urban and rural residents in the drainage, urban residents outside the drainage	Households listed in phone book (renters, resident and nonresident property owners)	n.a.

a. "n.a." indicates that either the information was not available or was not relevant to this study.
b. Crutch field et al. indicate that there were 819 usable responses and about a 50% response rate.
c. Of the 147 versions, 84 included the groundwater valuation scenario. These were randomly distributed proportionally to the 1,000 person sample.
d. Poe and Bishop (1992) define rural as census tracts that do not have municipally provided water. Although the definition of rural in most other studies is not clarified, we interpret rural, as used in these studies, to mean areas with nonmunicipal water supply for domestic use.

Study Reference	Crutch field et al.	Delavan	de Zoysa	Edwards	Giraldez and Fox
10. Household Water Supply/Groundwater Use	Municipal?: IN 73%; NE 69%; PA 53%; WA 74%	40% private wells, 60% public sources (incl bottled water)	Not specified	89% public 11% private wells	n.a.[e]
11. Actual Groundwater Baseline Condition	Unknown — between 17% and 53% for the four regions had heard about N contamination	Perceived GW quality is 71 on a scale of 0-100, w/0 as not safe and 100 as definitely safe	0.5 to 3.0 mg/L with some higher	Assumed that current water quality is safe	King St Well > 10mg/L York St Well high also
12. Groundwater Baseline Scenarios	None given	50% of private wells meet 10mg/L MCL	Typical N concentrations range from 0.5-3 mg/L, although some are much higher	Safe (state and county systematically monitor nitrate levels) — respondents were told to assume no health risks	n.a.
13. Change in Groundwater Scenario	If tap water has 50% greater N levels than EPA's MCL, how much to reduce to min. safety standards; how much to completely eliminate	In 10 years, 75% of private wells will meet MCL	Reduce levels to 0.5-1 mg/L	Prevent uncertain nitrate contamination of Cape Cod's sole source aquifer	n.a.
14. Credibility of Scenario Change	Not reported — several questions were asked that could be used to identify scenario rejecters	Checked for scenario rejection and also the scenario was very specific	Reduce N contamination from fertilizer applied to farm fields	Although vague, respondents were told to suppose the program was possible	n.a.
15. Contaminants	Nitrates	Nitrates	Nitrates	Nitrates	Nitrates
16. Source of Contaminants	Not specified	Fertilizer, septic, manure	Agricultural fertilizer	Fertilizer and sewage (primarily sewage)	n.a.
17. Types of Values Estimated	Primarily use values (commodity is a point-of-use filter)	Total	Total	Option price (use value)	Total value benefits transfer from CVM

e. 100% groundwater apparently from a public water supply distributing untreated well water.

Study Reference	Crutch field et al.	Delavan	de Zoysa	Edwards	Giraldez and Fox
18. Valuation Methodology	Dichotomous choice	Dichotomous choice open ended (DOE); informed open ended (IOE)	Dichotomous choice followed by open-ended	Dichotomous choice	(1) Loss of lifetime earnings; (2) value of statistical life; (3) total value benefits transfer from CVM
19. Payment Vehicle	Payment for local water agency for filter installation and maintenance	Special tax	Special tax	Fur versions: (1) bond, (2) water bill, (3) voluntary contribution, and (4) unspecified	n.a.
20. Duration of Payment Vehicle	Monthly, in perpetuity	Annually for 10 years	One time	Annually (in perpetuity)	n.a.
21. # of Survey Versions	Not specified, but multiple	2[f]	147	10	n.a.
22. Analysis	Bivariate probit	Tobit model[g]	Probit model	logit	n.a.
23. Mean Annual Household WTP in Study Year Dollars	$52.89/month (reduced N to MCL); $54.50 (no N): $1.61 difference.	DOE: $44.78 w/protest bidders IOE: $29.26 w/protest bidders DOE: $67.85 w/o protest bidders IOE: $47.16 w/o protest bidders	$52.78 lower bound mean (1994$ from YNP model)	$1623 for a management plan to increase the probability of supply from 0.0 to 1.0	Based on disaggregating value community value estimate: $412.50 per HH ($72.73/yr to $1696.97/yr)
24. Mean Annual Household WTP in 2001 Dollars	$63.20/month (reduced N to MCL); $65.13 (no N): $1.92 difference.	DOE: $50.55 w/protest bidders IOE: $33.03 w/protest bidders DOE: $76.59 w/o protest bidders IOE: $53.23 w/o protest bidders	$63.07 lower bound mean	$2,530.22 for a management plan to increase the probability of supply from 0.0 to 1.0	Based on disaggregating value community value estimate:$479.36 per HH ($84.52/yr to $1,972.00/yr)

f. Two “types” of survey (DOE and IOE). The DOE had eight versions differing only in the bid amount.
g. Also used a logit model to examine protest bids.

Study Reference	Crutch field et al.	Delavan	de Zoysa	Edwards	Giraldez and Fox
25. Median Annual Household WTP in Study year dollars	n.a.	DOE: $5 w/protest bidders IOE: $0 w/protest bidders DOE: $50 w/o protest bidders IOE: $25 w/o protest bidders	$20.80 median (1994$ from YNP model)	n.a.	n.a.
26. Median Annual Household WTP in 2001 Dollars	n.a.	DOE: $5.98 w/protest bidders IOE: $0 w/protest bidders DOE: $59.75 w/o protest bidders IOE: $29.88 w/o protest bidders	$23.48 median (from YNP model)	n.a.	n.a.
27. Range	$45.42-$60.76/month	$29.26-$67.85	Not specified	n.a.	CVM: $29,938 -$669,487 per year for entire village (higher estimate includes option prices as well) lifetime earnings/wage risk: $693-$30,855
28. Significant Explanatory Variables	Bid value (-) income (+) years lived in ZIP code (+) age (-)	-Income (+) -perceptions of increased safety (+) -age (-) -concern for drinking water safety (+) -high priority placed on spending for drinking water protection (+)	-Income (+) -high priority for groundwater (+) -increase gov't spending on education, healthcare, and vocational training (+)	Bequest motivation (+) income effect (+) probability of future supply (+) probability of future demand (+)	n.a.

Study Reference	Hurley et al.	Jordan and Elnagheeb	Poe and Bishop	Sparco
1. Published/Peer Reviewed?	Journal of Agricultural and Applied Economics	Water Resources Research	Environmental and Resource Economics	PhD dissertation
2. Year of Analysis	Apparently 1993	1991	1991	1993
3. Place	Clarke and Adams counties, IA	Georgia (statewide sample)	Portage County, WI	Sussex County, DE
4. Type of Study	Survey eliciting WTP for delaying water quality deterioration	Survey eliciting WTP for improved water quality	Survey eliciting WTP for improved water quality	Survey eliciting WTP for improved water quality
5. Survey Implement	Mail	Mail	Mail	booth at public gathering
6. Respondents	1000 (500 to each county)	567 mailed	480 mailed	3 occasions (# of respondents not specified) (not a random sample)
7. Response Rate	33.2%[h]	35%	77.9% (ex-ante) 83% (ex post) 64.4% (2nd stage)	Not specified
8. Location (urban, rural, etc.)	Rural — possibly some urban/rural municipalities	Unspecified mix of community sizes	Rural	Predominantly rural
9. Who Was Asked?	Residents	Residents	Residents not hooked up to municipal water supply	Passersby
10. Household Water Supply/Groundwater Use	75% use municipal or rural water	78% public 22% private wells	100% on private wells	61.9% of respondents use individual wells; remainder use municipal or community water systems
11. Actual Groundwater Baseline Condition	Not specified	50% of wells contain nitrates — did not specify % exceeding the MCL — 27% of public users rated water quality poor, 13% of private well users rated water quality poor	18% of wells had nitrate contamination exceeding EPA safety level - 16% of water tested > MCL	N concentrations >10 mg/L in 23% of samples(cited Andres 1991)

h. Doesn’t indicate bad addresses; 44.7% returned of which 332 had usable data.

Study Reference	**Hurley et al.**	**Jordan and Elnagheeb**	**Poe and Bishop**	**Sparco**
12. Groundwater Baseline Scenarios	Presumable currently safe	Not specified as individuals' own water conditions — baseline indicated as average conditions over all of GA (no individual probability of >MCL specified)	An increase in the number of wells in Portage County with nitrate contamination	Not specified
13. Change in Groundwater Scenario	Delay N contamination in drinking water for 10, 15, and 20 years, assuming existing facilities would result in contamination beyond legal limits w/in 5 years.	Private wells: water supplier provides new equipment, fee includes installation and maintenance public: water supplier guarantees safe drinking water for private wells — specified N >MCL, for public water specified N increasing (not indicated whether or not safe)	Groundwater protection program to reduce nitrates by 25% or to keep nitrate levels below the MCL for all wells in Portage County	WTP for a 1 part per million decr. in N contamination
14. Credibility of Scenario Change	Not assessed? No significant difference in WTP over 10 to 20 years. High percent of zero WTPs.	Examination of zero bidders did not indicate any significant scenario rejection	Although vague, respondents were told to suppose the program was possible- the survey was thoroughly pretested	Not assessed
15. Contaminants	Nitrates (from AFOs)	Nitrates	Nitrates	Nitrates, fecal coliform, atrazine
16. Source of Contaminants	CAFOs (mostly hog)	Agricultural activities (fertilizers)	Agricultural activities and other sources discussed in the survey	Agricultural activities (primarily poultry manure from AFOs)
17. Types of Values Estimated	Total value	Total value (primarily use as nitrate controls are at well head not reductions in N in the aquifer)	Total value — option price (use value)	Marginal value
18. Valuation Methodology	Referendum (dichotomous choice)	Close-ended payment card ("checklist")	Dichotomous choice, referendum format	Conjoint analysis (contingent rating)
19. Payment Vehicle	Not specified?	Water bill for public users costs for equipment to clean nitrates from water for private wells	Higher taxes, lower profits, increased costs and prices	Not specified

Study Reference	Hurley et al.	Jordan and Elnagheeb	Poe and Bishop	Sparco
20. Duration of Payment Vehicle	Annually	Monthly (in perpetuity)	Annually, for as long as respondent lives in the county	Annually, in perpetuity
21. # of Survey Versions	Not specified	One	2	8
22. Analysis	Ordered probit	Ordered probit	Logit	Ordered probit
23. Mean Annual Household WTP in Study Year Dollars	Not specified	Public: $128.20/hh/yr private: $157.61/hh/yr[i] (primarily use)	$199.73/hh/yr NINT[k] $961.16/hh/yr WINT $244.32/hh/yr NIWT $526.63/hh/yr WIWT	$123.56 per mg/L reduction in nitrates
24. Mean Annual Household WTP in 2001 Dollars	Not specified	Public:$166.70/hh/yr private: $204.94/hh/yr[j] (primarily use)	$259.71 NINT $1,249.79 WINT $317.69 NIWT $684.77 WIWT	$151.44 per mg/L reduction in nitrates
25. Median Annual Household WTP in Study Year Dollars	$118.13 (10 year delay) to $190.75 (20 year delay) for household with mean socio-economic characteristics	Public: $69.89/hh/yr private: $93.95/hh/yr	$194.45/hh/yr NINT $853.46/hh/yr WINT $242.58/hh/yr NIWT $507.94/hh/yr WIWT	n.a.
26. Median Annual Household WTP in 2001 Dollars	$118.13 (10 year delay) to $190.75 (20 year delay) for household with mean socio-economic characteristics	Public: $90.88/hh/yr private: $122.16/hh/yr	$252.84 NINT $1,109.75 WINT $315.43 NIWT $660.47 WIWT	n.a.
27. Range	n.a.	$128.20 — $157.61/hh/yr	$199.73-$961.16/hh/yr	n.a.
28. Significant Explanatory Variables	Education (+) likelihood that respondent will remain in area longer than 5 yrs (+) income (+)	Income (+)[l] gender (F+) black (+) education (+) uncertainty (+) live on farm (+)	Knowledge (+) quiz score (+)	Pro-environment attitude (-) cost (-) health risks (-) anti-government intervention (+) pro-farm viewpoints (+)

i. Using unconditional mean values from maximum likelihood estimates after rejecting outliers.
j. Using unconditional mean values from maximum likelihood estimates after rejecting outliers.
k. NINT, WINT, NIWT, WIWT = No information-no test; with information-no test; no information-with test; with information-with test respectively.
l. Significant variables from maximum likelihood on private wells excluding outliers.

Study Reference	Walker and Hoehn	Wattage
1. Published/ Peer Reviewed?	Northcentral Journal of Agricultural Economics	PhD dissertation
2. Year of Analysis	1983	1992?
3. Place	Rural MI	Bear Creek watershed, central IA
4. Type of Study	Model for estimating N values based on marginal cost of public treatment	Survey eliciting WTP for improved water quality
5. Survey Implement	n.a.	mail
6. Respondents	n.a.	345
7. Response Rate	n.a.	40%
8. Location (urban, rural, etc.)	Rural	Predominantly rural
9. Who Was Asked?	n.a.	Farmers, absentee owners, town residents
10. Household Water Supply/ Groundwater Use	>95% rural supply from GW	50% private wells 43% municipal (also GW) 93% GW
11. Actual Groundwater Baseline Condition	34% of 191 wells >10 mg/L	Perceived: 16% ranked water quality as suitable for human drinking purposes
12. Groundwater Baseline Scenarios	Modeled specific scenarios	Individuals' perceived water quality
13. Change in Groundwater Scenario	Modeled specific scenarios	Installing vegetative buffer strips (VBSs) to reduce overland flow of contaminated water into GW & SW supplies
14. Credibility of Scenario Change	n.a.	32% of respondents strongly agree that VBS could control N in the root zone- possibly significant scenario rejection
15. Contaminants	Nitrates	Nitrates, pesticides; sediments
16. Source of Contaminants	Agricultural activities	All runoff sources including: fertilizers, manure, illegal wastes, gasoline
17. Types of Values Estimated	Damages (producer + consumer surplus) (use values only)	Total value
18. Valuation Methodology	n.a.	Dichotomous choice and open-ended. WTP and WTA for various scenarios
19. Payment Vehicle	n.a.	Not specified
20. Duration of Payment Vehicle	n.a.	Monthly, as long as live in watershed
21. # of Survey Versions	n.a.	not specified
22. Analysis	Welfare theory	4 analyses: OLS, linear probability model, probit, logit

Study Reference	Walker and Hoehn	Wattage
23. Mean Annual Household WTP in Study Year Dollars	n.a.	$80/month
24. Mean Annual Household WTP in 2001 Dollars	n.a.	$100.98/month
25. Median Annual Household WTP in Study Year Dollars	n.a.	Not specified
26. Median Annual Household WTP in 2001 Dollars	n.a.	Not specified
27. Range	$40-330/household/yr[m]	Not specified
28. Significant Explanatory Variables	-Treatment location (point of use vs. centralized) -water consumption -price of water -damages and benefits per household -household income -nitrate contamination	Income (+) distance from creek to land (+) present GW quality (-)
m. $330/yr is based on annual cost of point-of-use nitrate removal.		

APPENDIX D
ASSESSMENT OF DATA USED TO ESTIMATE BENEFITS

The majority of the data EPA used to estimate the environmental and economic benefits associated with the effluent guideline limitations for CAFOs are from existing sources. As defined in the Office of Water 2002 Quality Management Plan (USEPA, 2002), existing (or secondary) data are data that were not directly generated by EPA to support the decision at hand. Existing data were used to model:

1. reductions in leached nitrogen loadings resulting from new phosphorus-based and nitrogen-based manure application regulations which would apply to all large AFOs as well as any medium AFOs identified under new NPDES conditions,

2. the reduction in nitrate concentrations in private drinking water wells, as a result of the new regulations, and

3. the value of the reductions.

In keeping with the graded approach to quality management embodied in the quality management plan, EPA must assess the quality of existing data relative to their intended use. The procedures EPA used to assess existing data for use in estimating the benefits associated with effluent guideline limitations for CAFOs varied with the specific type of data. In general, EPA's assessment included:

- reviewing a description of the existing data that explains how the data were collected or produced (e.g., who collected the data, what data were collected; why were the data originally collected; when were the data collected; how were they collected; are the data part of a long-term collection effort, or was this a one-time effort; who else uses the data; what level of review by others have the data undergone?)

- specifying the intended use of the existing data relative to the CAFO final rule

- developing a rationale for accepting data from this source, either as a set of acceptance criteria, or as a narrative discussion

- describing any known limitations with the data and their impact on EPA's use of the data.

Brief descriptions of the data and their limitations are presented in Chapters 3 and 5 and Appendices A and C, as each data source is introduced. In addition, Section 6.7 presents a detailed accounting of known omissions, biases, and uncertainties in the analysis of the benefits of reduced nitrate in private drinking water wells attributable to new CAFO regulations.

In searching for existing data sources and determining their acceptability, EPA generally used a hierarchical approach designed to identify and utilize data with the broadest representation of the industry sector of interest. EPA began by searching for national-level data from surveys and studies by USDA and other federal agencies. When survey or study data did not exist, EPA considered other types of data from federal agencies.

Where national data did not exist, as the second tier, EPA searched for data from land grant universities. Such data are often local or regional in nature. EPA assessed the representativeness of the data relative to a national scale before deciding to use the data. When such data came from published sources, EPA gave greater consideration to publications in peer-reviewed professional journals compared to trade publications that do not have a formal review process.

The third tier was data supplied by industry. Prior to proposal, EPA requested data from a variety of industry sources, including trade associations and large producers. The level of review applied to data supplied by industry depended on the level of supporting detail that was provided. For example, if the industry supplied background information regarding how the data were collected, such as the number of respondents and the total number of potential respondents, EPA reviewed the results, compared them to data from other potential sources to determine their suitably for use in this rulemaking. If the data provided by industry originated from an identifiable non-industry source (e.g., a state government agency), EPA reviewed the original source before determining the acceptability of the data. In a limited number of instances, EPA conducted site visits to substantiate information supplied by industry. In contrast, data supplied by industry without any background information were given much less weight and generally were not used by EPA. Further, some data that were supplied by industry prior to the proposal were included in the proposal for comment. In the absence of any negative comments, such data were relied on to a greater extent than data submitted by industry during the comment period itself.

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The Benefits of Protecting Rural Water Quality

An Empirical Analysis

Stephen R. Crutchfield
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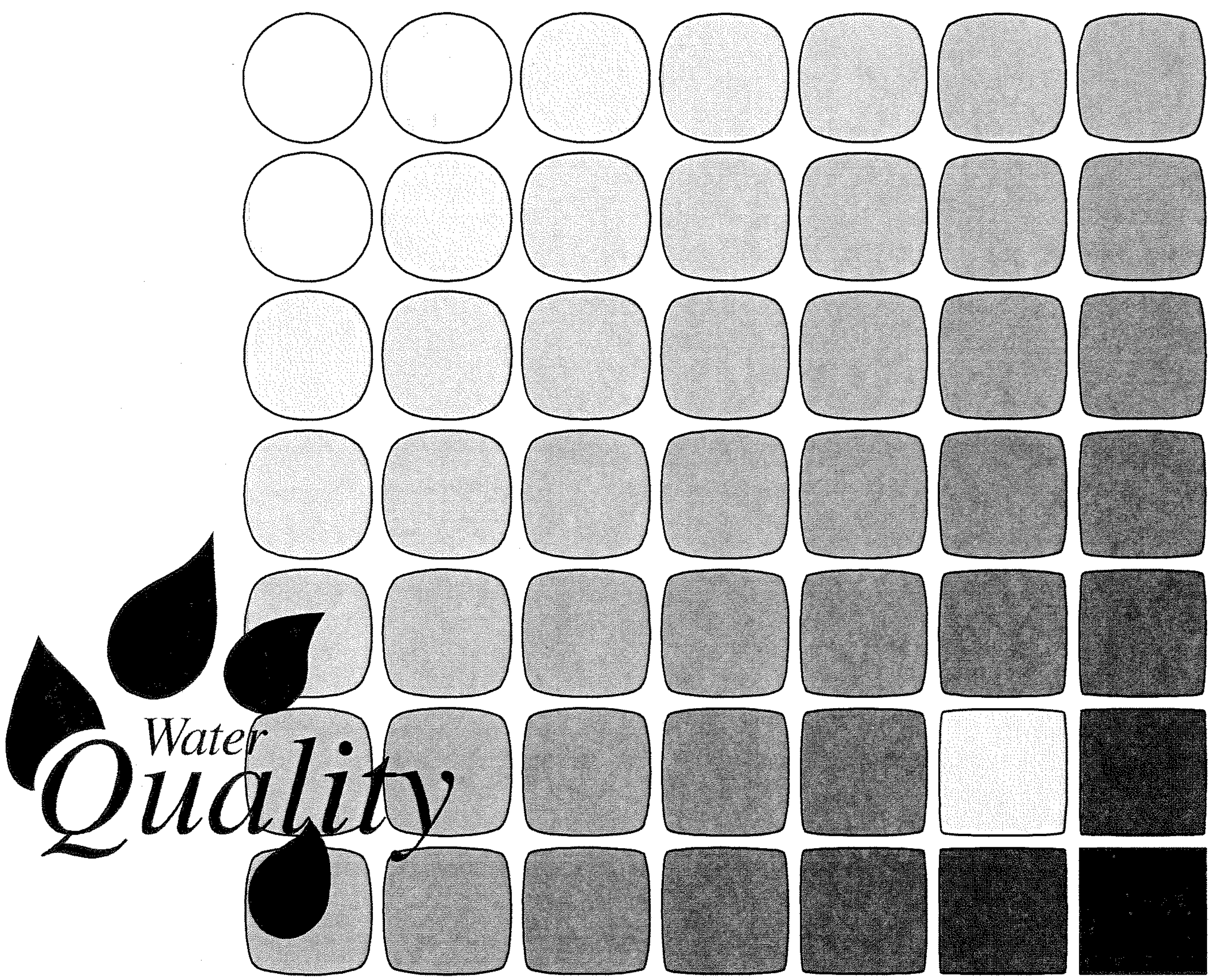

The Benefits of Protecting Rural Water Quality: An Empirical Analysis. By Stephen R. Crutchfield, Peter M. Feather, and Daniel R. Hellerstein. Natural Resources and Environment Division, Economic Research Service, U.S. Department of Agriculture. Agricultural Economic Report No. 701

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

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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.

The Benefits of Protecting Rural Water Quality
An Empirical Analysis

Stephen R. Crutchfield
Peter M. Feather
Daniel R. Hellerstein

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) (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 socio-economic 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

		Area Studies region				
	Willingness to pay	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)

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

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:

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

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:

$$CS_i = POP_i{*}E{*}CS_i \quad \text{(A3)}$$

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.