ATTACHMENT B

THE BENEFITS OF REDUCING NITRATE CONTAMINATION IN PRIVATE DOMESTIC WELLS UNDER CAFO REGULATORY OPTIONS

U.S. Environmental Protection Agency
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CONTENTS

Chapter 5 Groundwater Valuation Studies

Chapter 6 Benefit Calculations

Appendices

EXHIBITS

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EXECUTIVE SUMMARY

Concentrated 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 to 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 untreated, 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.

The most recent U.S. Census data 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 concentration of nitrate in 9.7% of domestic wells in the U.S. exceeds 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 eight regulatory scenarios evaluated.

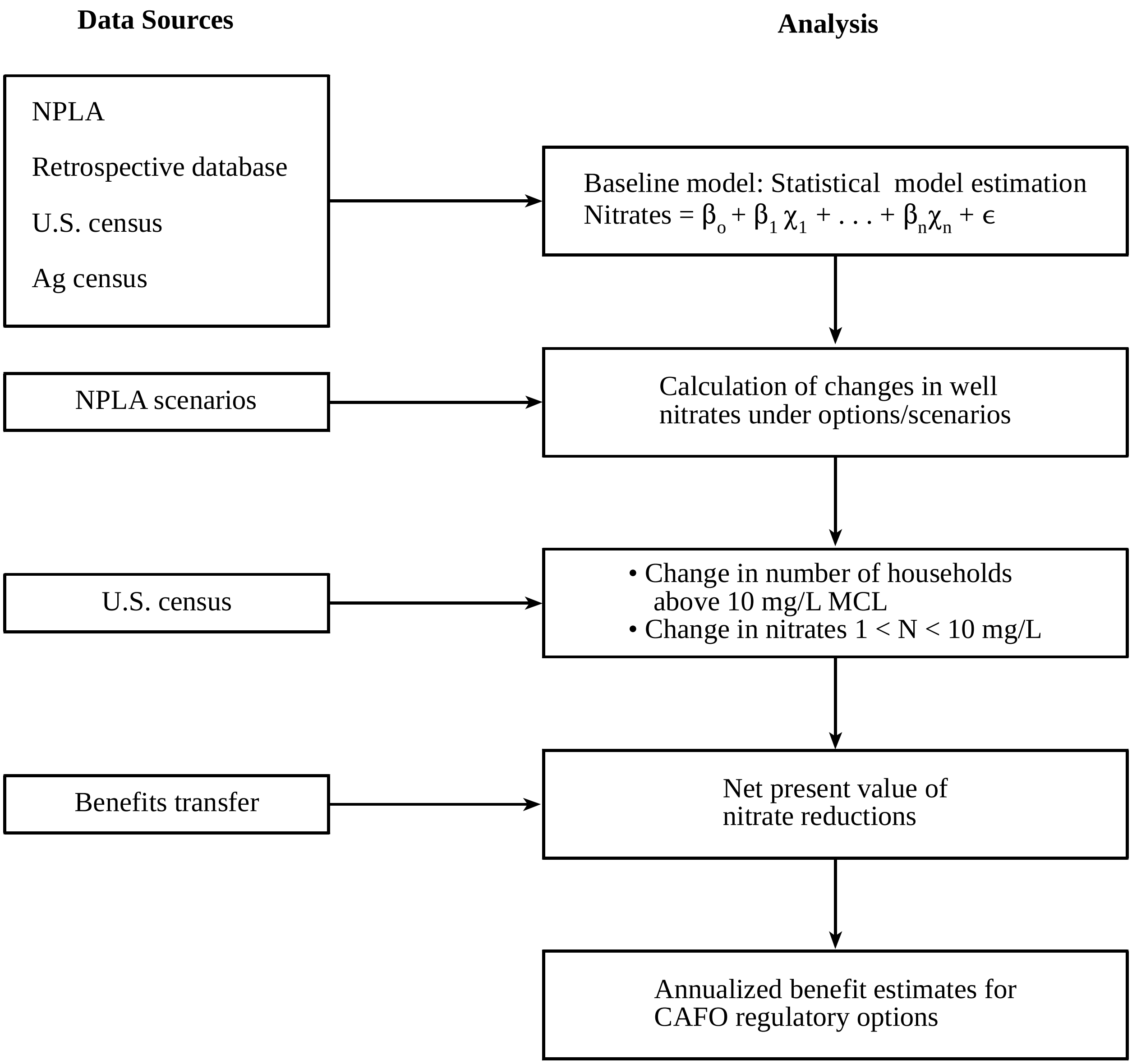
Exhibit S-1 provides an overview of the approach to estimating the benefits of well nitrate reductions. The analysis begins by developing a statistical model of the relationship between nitrate concentrations in private domestic wells and a number of variables found to affect nitrate levels, including nitrogen loadings from AFOs. 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 marginal 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 AFOs. The primary purpose of the model is to estimate the effects of nitrogen loadings from CAFOs on domestic well nitrate concentrations while *controlling for* other sources of nitrogen and well characteristics that could affect this relationship. Controlling for other sources of nitrogen in particular ensures that decreases in nitrogen loadings from CAFOs as a result of regulatory activities do not lead to overestimates of the resultant impact 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. The dependent variable, domestic well nitrate concentrations, was obtained from the USGS Retrospective Database. Data were compiled for 2,928 observations in 364 counties. The regression model includes variables characterizing nitrogen loadings from animal feeding operations [data obtained from the National Pollution Loadings 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).

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 eight 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 2,504 counties in the NPLA indicated as having CAFOs.

Exhibit S-2
Nitrogen Loadings from CAFOs: Mean, Total, and Percent Reduction from Baseline (2,504 counties)

Option/Scenario	Mean (pounds per county)	Total (pounds nitrogen)	Percent Reduction from Baseline
Baseline	609,553	1,526,322,559	not applicable
Option 1 — Scenario 1	317,572	795,201,054	47.9
Option 1 — Scenario 2/3	294,320	736,978,193	51.7
Option 1 — Scenario 4a	221,454	554,522,671	63.7
Option 1 — Scenario 4b	221,454	554,522,671	63.7
Option 2 — Scenario 1	280,802	703,130,686	53.9
Option 2 — Scenario 2/3*	254,556	637,410,305	58.2
Option 2 — scenario 4a*	174,807	437,718,632	71.3
Option 2 — Scenario 4b	174,807	437,718,632	71.3

* Proposed scenarios.
Source: Calculations based on NPLA (TetraTech, 2000).

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 150,000 and 166,000 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 1	152,204	961,741
Option 1 — Scenario 2/3	152,204	1,007,611
Option 1 — Scenario 4a	161,384	1,186,423
Option 1 — Scenario 4b	161,384	1,186,423
Option 2 — Scenario 1	161,384	1,103,166
Option 2 — Scenario 2/3*	161,384	1,159,907
Option 2 — Scenario 4a*	165,974	1,374,990
Option 2 — Scenario 4b	165,974	1,374,990

a. For wells at or below the MCL at baseline and above 1 mg/L.
* Proposed scenarios.

Incremental Changes below the MCL

Households currently served by wells with nitrate concentrations below the 10 mg/L level may also benefit from marginal 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. Marginal 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 marginal changes below the MCL for households that are above the MCL as 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. Approximately 600,000 households would benefit from these marginal reductions.

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 marginal changes in nitrate concentrations below the MCL.
- De Zoysa and (1995): values marginal 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 1999 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	1999$
Poe and Bishop	Annual WTP per household for reducing nitrates from above the MCL to the MCL	448.00
Average of Crutchfield et al. and De Zoysa	Annual WTP per mg/L between 10 mg/L and 1 mg/L	1.97

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 (1999$)

Scenario	Total WTP for Discrete Reduction to MCL	Total WTP for Marginal Changes below 10 mg/L	Total
Option 1 Scenario 1	68,187,392	1,894,630	70,082,022
Option 1 Scenario 2/3	68,187,392	1,984,994	70,172,386
Option 1 Scenario 4b	72,300,032	2,337,253	74,637,285
Option 1 Scenario 4a	72,300,032	2,337,253	74,637,285
Option 2 Scenario 1	72,300,032	2,173,237	74,473,269
Option 2 Scenario 2/3*	72,300,032	2,285,017	74,585,049
Option 2 Scenario 4a*	74,356,352	2,708,730	77,065,082
Option 2 Scenario 4b	74,356,352	2,708,730	77,065,082
* Proposed scenarios.			

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 2- Scenario 4a, using the 27 year timepath and a 3% discount rate, the present value of benefits would be $1,662.32 million. As shown in Exhibit S-6, a constant benefit flow of $38.4 million discounted at 3% would generate $1,662.32 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 1999$)

	3%	**5%**	**7%**
Scenario	**Annualized Value**	**Annualized Value**	**Annualized Value**
Option 1 Scenario 1	46.37	37.77	31.07
Option 1 Scenario 2/3	46.43	37.82	31.11
Option 1 Scenario 4b	49.39	40.23	33.09
Option 1 Scenario 4a	49.39	40.23	33.09
Option 2 Scenario 1	49.28	40.14	33.02
Option 2 Scenario 2/3*	49.35	40.20	33.07
Option 2 Scenario 4a*	50.99	41.54	34.17
Option 2 Scenario 4b	50.99	41.54	34.17

* Proposed scenarios.

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 feedlot 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, and EPA is revising the regulations to address changes in the animal industry sectors over the last 25 years, 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 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 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 by either 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]

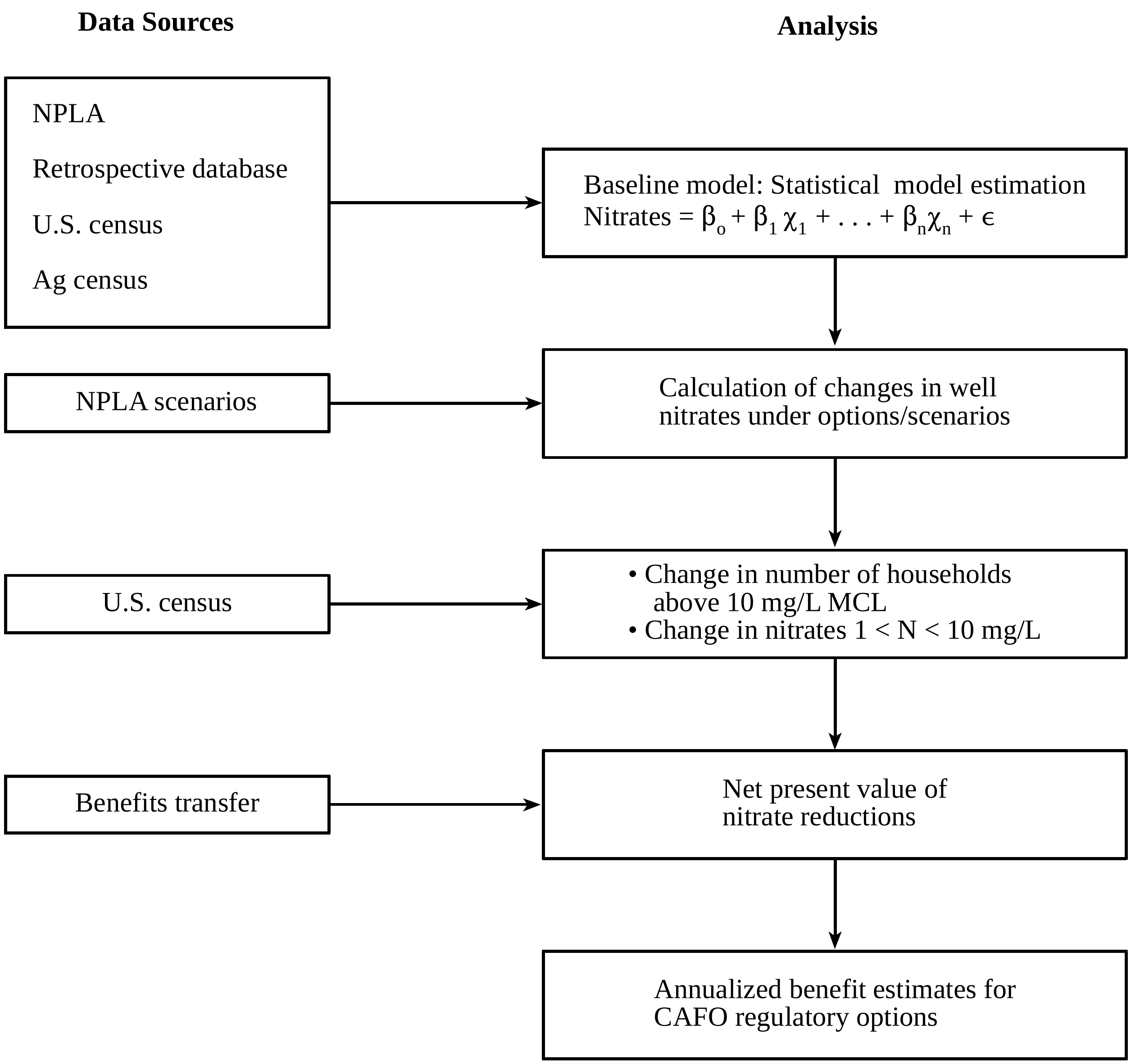
- designated as a CAFO on a case-by-case basis if the NPDES-authorized permitting authority determines that it is a significant contributor of pollution to waters of the United States [40 CFR part 122.23(c)].

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 a 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, as well as 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, as well as 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 data sources used to model this and 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; Richards et al., 1996; Clawges and Vowinkel, 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 Groundwater Quality, 1998). Farms that applied manure as fertilizer tended to have higher nitrate concentrations in groundwater as well (Rausch, 1992; Swistock et al., 1993; Richards et al., 1996; Clawges and Vowinkel, 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, 1988; Ritter and Chirnside, 1990; Kross et al., 1993; Spalding and Exner, 1993; Swistock et al., 1993; Lichtenberg and Shapiro, 1997; Ham et al., 1998; North Carolina Division of Water Quality, 1998; Sparco, 1995). 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 a positive correlation 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 in the vicinity of 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; Richards et al., 1996; Clawges and Vowinkel, 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 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 were available for our analysis on the location of animal feedlots, cropland, and septic systems.

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 following 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, 2000), 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 approximately 10,000 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 wells, in mg/L
- nitrogen inputs from atmospheric, manure, and fertilizer loadings
- water use of the well (e.g., irrigation, domestic)
- depth to water in the well
- land use in the vicinity of 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.

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 concentrations are reported by well. The Retrospective database was the limiting data source for this analysis because it included 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 were at or below the detection limit. 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 (0.05 mg/L) to the detection limit.

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). We used data from counties with 10 or more observations because these provide information on countywide conditions that could correspond better with the countywide data used for loadings, septic systems, and other variables. The final dataset used in the analysis included 138 counties and 2,504 wells.

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. 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 Pollutants Loadings Analysis

The National Pollutants Loadings Analysis (NPLA; Tetra Tech, 2000) provided data on 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 (hereby termed Sample Farms). These Sample Farms were developed from county, regional, and national data sources, including the 1997 Census of Agriculture data.

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 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 nine scenarios (baseline and two options each with four scenarios) evaluated in this analysis are based on different combinations of two factors: limits for land application of manure and variations on how many facilities will be subject to the regulation. All 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) or total phosphate applied (Option 2). The nitrogen and phosphate content of the manure and subsequent manure application rates under these two options are based on the type of animal operation. Under both 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 options for the number of affected facilities determine how many facilities that have 300-1,000 animal units (AUs) or 500-1,000 AUs will be defined as CAFOs under the regulation, and therefore will be subject to the nitrogen-based or phosphate-based limits. An animal unit is defined as one beef cow, and other animal types are defined based on their size relative to a beef cow (e.g., 9.09 swine = 1 AU, 88.5 turkeys = 1 AU). Facilities with fewer than 300 (or 500) AUs, are not subject to the regulation, and therefore are not included in the baseline analysis. All facilities with more than 1,000 AUs are considered CAFOs and therefore subject to nitrogen-based or phosphate-base limits. At baseline, some operations greater than 300 (or 500) AUs are regulated and therefore produce varying nitrogen loadings.

Similarly, all 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.

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% w/ >1000 AUs, plus AFOs w/ >300 AU that meet certain requirements	>300 AU
Option 1 — Scenario 1	Nitrogen	Baseline scenario plus qualifying dry poultry and immature swine and heifer operations	>300 AU
Option 1 — Scenario 2/3	Nitrogen	New NPDES conditions for identifying CAFOs between 300-1,000 AUs, plus qualifying dry poultry and immature swine and heifer operations	>300 AU
Option 1 — Scenario 4a	Nitrogen	All AFOs w/ >500 AUs, plus qualifying dry poultry and immature swine and heifer operations	>500 AU
Option 1 — Scenario 4b	Nitrogen	All AFOs w/ >300 AUs, plus qualifying dry poultry and immature swine and heifer operations	>300 AU
Option 2 — Scenario 1	Phosphate	Baseline scenario plus qualifying dry poultry and immature swine and heifer operations	>300 AU
Option 2 — Scenario 2/3*	Phosphate	New NPDES conditions for identifying CAFOs between 300 and 1,000 AUs, plus qualifying dry poultry and immature swine and heifer operations	>300 AU
Option 2 — Scenario 4a*	Phosphate	100% w/ >500 AUs, plus qualifying dry poultry and immature swine and heifer operations	>500 AU
Option 2 — Scenario 4b	Phosphate	All AFOs w/ >300 AUs, plus qualifying dry poultry and immature swine and heifer operations	>300 AU

* Proposed scenarios.

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 assumed 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:

Total Loadings for one Type of Operation (AnimalX, SizeY) in a county =
(% of facilities regulated * Scenario loadings-regulated * Number of facilities) (2-1)
+ [(1 - % of facilities regulated) * Baseline loadings-unregulated * Number of facilities].

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 to get total county loadings.

CHAPTER 3
MODELING WELL NITRATE CONCENTRATIONS

A statistical model of the relationship between nitrogen loadings and well nitrate concentrations was developed to analyze the impact 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 impact 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:

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

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, we feel that the USGS Retrospective database, at 9.7% of domestic wells 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, 1991 (as cited by Giraldez and Fox, 1995)	Ontario	Domestic farm	13
Andres 1991 (as cited in Sparco 1993)	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, WI	Rural	16
Retrospective Database (USGS), 1996	National	Domestic	9.7
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, 1985	Upper Conestoga River Basin	Rural	40+
Vitosh, 1985 (cited in Walker and Hoehn, 1990)	Southern Michigan	Rural	34

Actual nitrate concentrations in groundwater reported in the USGS 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, any nitrate concentration below 0.05 mg/L was automatically set to 0.05 mg/L. Approximately 15% of the observations were at or below the detection limit.[1]

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 loadings parameter, which is the key component of the model for CAFO loadings analysis.

The intercept (β_0) will capture ambient nitrate levels in the absence of human influences from septic systems, AFOs larger than 300 animal units, and alternative nitrogen sources. As we do not have loadings data for AFOs smaller than 300 AUs, these 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 USGS 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 5,310 ft. For observations used in the regression analysis, the maximum well depth was 1,996 ft and the mean depth was 169 ft.

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.

Alternate N Source: Alternate nitrogen sources include fertilizer and atmospheric deposition and are measured in pounds applied annually per acre.

Loadings Ratios and Scenarios: The loadings ratio was calculated using surface nitrogen loadings, in annual pounds per county, divided by the total acreage of the county. We used total county acres to create a consistent unit across all counties, assuming in general that once nitrate leached into the groundwater it would be dispersed in a volume of groundwater proportional to the county size. Loadings were reported for the eight regulatory scenarios and the preregulatory baseline. Preregulatory baseline loadings were used to estimate the statistical models.

Exhibit 3-2 lists summary statistics for the dependent and independent variables for the 2,928 observations used in the regressions described below.

Exhibit 3-2
Summary Statistics
(nobs = 2928)

Variable	Mean	Std Dev	Minimum	Maximum
Nitrate Concentration	3.585	6.552	0.050	84.300
Ag Dummy	0.775	0.418	0.000	1.000
Soil Group	2.418	0.658	1.000	4.000
Well Depth	169.191	133.468	1.000	1,996.000
Septic Ratio	0.029	0.028	0.000	0.151
Alternate N Source	28.890	18.981	0.869	99.631
Loadings Ratio	6.626	14.022	0.003	63.354

3.2 THE STATISTICAL MODEL

EPA used regression analysis to estimate the statistical model described in Equation 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.[2] 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. All of the explanatory variables are significant and of the expected sign. This implies 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 estimate on the three sources of nitrogen (septic systems, fertlizers and atmospheric, and animal feeding operations) indicate that each does contribute to well nitrate concentrations. The model can thus be used to help understand how changes in the independent variables (e.g., nitrogen loadings, well depth, land use) will affect the expected level of nitrate at the well. We can therefore use the model to examine how changing nitrogen loadings from CAFOs will affect nitrate concentrations in domestic drinking water wells.

2. We refer to the gamma model 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 distribution.

Exhibit 3-3
Gamma Regression Results
Nobs = 2928

Variable	Parameter Estimate	Std. Err.	Asymptotic t-statistic[b]
Intercept	1.492	0.151	9.891
Ag Dummy	0.691	0.066	10.452
Soil Group	-0.335	0.043	-7.725
Well Depth (per 1/100 ft)[a]	-0.106	0.015	-7.178
Septic Ratio	2.623	1.102	2.380
Alt N Source (1/1,000 lbs N/acres in county)[a]	20.258	1.628	12.444
Loadings Ratio (lbs N/acres in county)	0.010	0.002	5.037
Alpha	0.498	0.011	46.368

a. The raw data was scaled by a factor of 100 for well depth and 1000 for Alt N Source in order for the GAUSS program to converge to a solution.
b. All parameter estimates are significant at or below the 1% level.
Mean log-likelihood = 1.854.

3.3 FITTED VALUES AND SCENARIO MODELING

After estimating the gamma model using the baseline loading information, expected values for y_i were calculated using observed baseline loadings and loadings from the eight regulatory scenarios, from 2,928 observations. As described above, the eight regulatory scenarios are based on different manure application rates, manure management practices, and monitoring requirements. Loadings for the eight 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 concentrations y_i from the eight scenario loadings were compared with the expected y_i using the baseline loadings. We 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 eight different scenarios by the expected values from the baseline loadings. These percentage differences were then applied to the actual nitrate concentrations, the observed y_i, to calculate well nitrate concentrations under the various scenarios. The expected percentage changes in nitrate concentration for each scenario are summarized in Exhibit 3-4.

Exhibit 3-4
Characteristics of Benefits Analysis Scenarios

Regulatory Scenario	Loadings (nitrogen, in lb/yr)		Nitrate (mg/L), Predicted by Gamma Model	
	Mean % Reduction	Median % Reduction	Mean % Reduction	Median % Reduction
Baseline	0	0	0	0
Option 1 — Scenario 1	34.3	33.3	3.0	0.8
Option 1 — Scenario 2/3	39.4	40.8	3.2	0.8
Option 1 — Scenario 4a	53.1	47.5	3.7	0.8
Option 1 — Scenario 4b	53.1	47.5	3.7	0.8
Option 2 — Scenario 1	26.8	39.4	3.5	0.8
Option 2 — Scenario 2/3*	31.5	48.0	3.7	0.8
Option 2 — Scenario 4a*	41.0	61.0	4.3	0.9
Option 2 — Scenario 4b	41.0	61.0	4.3	0.9

* Proposed scenarios.

As indicated in the literature surveyed, although nitrogen loadings from CAFOs are significant contributors to elevated 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 were used to estimate nitrate concentrations.

We checked the ability of the model to estimate low-end 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 (setting ag_dummy and all human nitrogen sources to zero), the model predicts an ambient nitrate concentration of 0.829 mg/L on nonagricultural lands. Using the same approach, the predicted value on agricultural land is 1.657 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 around 1.0 mg/L. Therefore we feel that the model's intercept of 0.829 mg/L on non-agricultural land is a reasonable estimate of nitrate concentrations in the absence of the pollution from the human nitrogen sources accounted for in the model.

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

3.4 DISCRETE CHANGES FROM ABOVE THE MCL TO BELOW THE MCL

Census data show that currently approximately 13.5 million households have domestic wells in counties with animal feedlot operations. Of the 3,078 counties in the NPLA, 2,504 counties are identified in the NPLA as having AFOs. Based on the USGS Retrospective data, 9.74% of these wells in the U.S. currently exceed 10 mg/L, or roughly 1.3 million households. Applying the percentage reductions, between 161,000 and 166,000 households are expected to be brought under 10 mg/L. Results are displayed in Exhibit 3-5.

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 1	152,204
Option 1 — Scenario 2/3	152,204
Option 1 — Scenario 4a	161,384
Option 1 — Scenario 4b	161,384
Option 2 — Scenario 1	161,384
Option 2 — Scenario 2/3*	161,384
Option 2 — Scenario 4a*	165,974
Option 2 — Scenario 4b	165,974
* Proposed scenarios.	

3.5 INCREMENTAL CHANGES BELOW THE MCL

Many households on wells with nitrate concentrations below the MCL at baseline may also gain benefits from marginal changes in nitrate concentrations below the 10 mg/L level and above the natural level, which we assume here to be 1 mg/L (see discussion in Section 3.3). We thus assume that these incremental benefits are gained only for wells beginning with concentrations between 1 and 10 mg/L. We did not calculate values for marginal changes where well concentrations remain above the MCL because we do not have reliable value estimates for changes in marginal nitrate concentrations above the MCL.

For households that start above the MCL preregulation and move below the MCL post-regulation, we also did not calculate values for marginal changes below the MCL. Based on the available valuation literature (see Chapter 5) we did not have reliable estimates for valuing marginal changes below the MCL in addition to valuing changes down 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. Data for all wells are included for comparison. Approximately 600,000 households will benefit from these marginal 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)	Total Expected National Nitrate Reduction (mg/L)
Option 1 — Scenario 1	0.16	0.12	961,741
Option 1 — Scenario 2/3	0.16	0.12	1,007,611
Option 1 — Scenario 4a	0.19	0.15	1,186,423
Option 1 — Scenario 4b	0.19	0.15	1,186,423
Option 2 — Scenario 1	0.18	0.14	1,103,166
Option 2 — Scenario 2/3*	0.19	0.15	1,159,907
Option 2 — Scenario 4a*	0.22	0.18	1,374,990
Option 2 — Scenario 4b	0.22	0.18	1,374,990
* 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 at the ground surface and realization of the benefits as lower nitrate concentrations in water withdrawn from wells. 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 characteristically 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 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, while 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, we must estimate some representative response time of wells. More specifically, how long after implementation will the benefit of improved nitrate concentrations 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 at 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 we will assume the conservative estimate of the concentration curve is linear, resulting in an assumption of a “clean” aquifer in approximately 27 years after implementation of improved CAFO waste management.

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 whether to 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 price, transfer a function, calculate a metafunction, or calibrate a preference. Crutchfield et al. (1997) discuss transferring an average price 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 price 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 as well as 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 benefit 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 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 price approach is most feasible 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 a potentially regulated CAFO is 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 intend to 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. Through the process of evaluating the literature on valuation of nitrate contamination in wells some studies were eliminated that were of poorer overall quality or for which only limited information was available.

We identified 11 such studies through an extensive search of relevant 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 willingness to pay (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 would be paid to a local water agency for the installation and maintenance of the filter. 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 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 per year. Crutchfield et al. compared annual per household WTP estimates from their study to three others (including Jordan and Elnagheeb, 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/hh/yr is not unreasonably higher than the $412-$484/hh/yr values discussed in Poe and Bishop below. The difference in values between the two programs is likely to be representative of values for marginal 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 $1.92 annually (per mg/L in 1997$).

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 positive 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 marginal 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, 1998

Using a CVM survey of 1,000 residents in two counties in southeastern Pennsylvania (with a 68% response rate), Delavan (1998) 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 such as 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% for 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 significant and positively correlated with respondents' willingness to pay. He also found that males were more likely to pay more for groundwater protection.

Evaluation: Delavan thoroughly designed and 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 significant 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) contingent valuation (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

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.

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.

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 $1623/yr (1987$).

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 comprises primarily use values.

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 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 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/hh/yr NINT, $961.16/hh/yr WINT, $244.32/hh/yr NIWT, and $526.63/hh/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 $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 statements 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 marginal 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. As a 25% reduction from 14.5 mg/L would reduce nitrate levels to very near the MCL, this reduction could be considered to be 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 have reviewed where respondents had empirical information on the nitrate levels in their own well. Although the stage two sample size is not large (271), the quality of the data is likely to be higher than for larger samples using less well

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.

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

developed surveys. The surveyed population (rural Wisconsin) is likely representative of individuals facing potential well nitrate contamination from CAFOs.

Two value estimates from Poe and Bishop are evaluated to be the most applicable for benefits transfer. First is the mean WTP of $484/hh/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 contamination. Second is the $412/hh/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. A VSI 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 profarm 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 marginal 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 marginal 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 marginal cost of providing water after nitrate contamination, and is the same as the precontamination function plus the additional marginal cost of removing nitrates.

The marginal cost of nitrate removal was estimated from a sample of costs for nitrate removal generated from the engineering model. The marginal 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, values from WTP 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 marginal value estimates from the damage model, an average household with $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 $65/year (1983$) value estimate from Walker and Hoehn thus could represent a lower bound estimate of use values. The value estimate here 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 at $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 of the reports or papers 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 uni-dimensional 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 is 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 are applying benefit estimates. Since we currently do not have national sociodemographic information specific to the population on private wells, focusing the transfer on studies conducted with more rural populations will help account for potential income differences between rural and urban populations.

Exhibit 5-1
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
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), published in *Food Safety*.
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 marginal costs of water production.

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 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 of 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 than 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 methodology
- 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 consider 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 place 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 it is felt that 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 note studies that do not involve a random sample (e.g., Sparco, 1995) in order to minimize potential sample selection bias (see below on response rates).

Respondents

For contingent valuation surveys, we feel that 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 attempt 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.

Payment Vehicle

There are numerous types of payment vehicle that 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 rank 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 rate 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 rate 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 will 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]

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.

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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.

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	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

<table>
<tr><th colspan="2" rowspan="2"></th><th colspan="3">Applicability of Study to CAFOs</th></tr>
<tr><th>High</th><th>Medium</th><th>Low</th></tr>
<tr><th rowspan="3">Quality of Study</th><th>High</th><td>Poe and Bishop, 1992</td><td></td><td>Crutchfield et al., 1997
Edwards, 1988</td></tr>
<tr><th>Medium</th><td>Jordan and Elnagheeb, 1993</td><td>De Zoysa, 1995
Delavan, 1998</td><td></td></tr>
<tr><th>Low</th><td>Giraldez, 1995</td><td>Hurley et al., 1999
Sparco, 1993
Walker and Hoehn, 1990</td><td>Wattage, 1993</td></tr>
</table>

5.5 VALUES FOR BENEFITS TRANSFER TO CAFOS

We applied the Consumer Price Index (CPI) to convert the annual mean household willingness-to-pay values obtained from these studies to 1999 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	107.6
1987	109.6
1988	113.6
1989	118.3
1990	124.0
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

Source: U.S. Bureau of Labor Statistics, 2000.
ftp://ftp.bls.gov/pub/special.requests/cpi/cpiai.txt, Accessed 10/03/00.

Exhibit 5-5 shows summary mean per household annual WTP in 1999 dollars for several of the studies discussed above. Not all values are shown for all reports.

Based on this summary, WTP values for reducing nitrate contamination to safe levels fall into a range between $50 and $2,400 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' $2,400/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.

Exhibit 5-5
Mean Annual WTP per Household

Study Reference	Year of Analysis	Mean Household WTP in 1999 dollars
Crutchfield et al., 1997	1994	$658.80 to reduce nitrates to safe level $21.72 to reduce from 10 mg/L to 0 mg/L ($2.17 per mg/L)
De Zoysa, 1995	1994	$59.33 (lower bound mean) $1.78 per mg/L (using 3% discount rate)
Delavan, 1998	1996	$188.64 IOE w/o protest bidders (see Section 5.2.3)
Edwards, 1988	1987	$2380.21 increase probability of supply from 0.0 to 1.0
Jordan and Elnagheeb, 1993	1991	$192.79 (private wells)
Poe and Bishop, 1992	1991	$412.00 (25% reduction in nitrates to safe level) $484.00 (households with 100% probability of future contamination)
Sparco, 1993	1993	$142.46 per mg/L

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. Crutchfield's estimate for WTP per mg/L under the MCL provides a lower bound estimate that we can conservatively use in the benefits transfer.

Crutchfield et al.'s value estimate for reducing nitrates to safe levels are derived from a more diverse sample than Poe and Bishop. Crutchfield et al.'s WTP estimate is $658.80/hh/yr (1999$). As indicated in Exhibit 5-2 though, we ranked Crutchfield et al.'s 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 Crutchfield et al.'s 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 (to below the MCL). The average of these two estimates it $448.00 per household per year.

We use the average of De Zoysa and Crutchfield et al. for changes in marginal 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 $52.78 per mg/L change in nitrate concentrations for marginal changes below the 10 mg/L MCL. Since the valuation question was posed as a one-time special tax, we annualize this to be $52.78 per mg/L. Using a 3% discount rate, this translates into an annual WTP of $1.61 per mg/L in 1994$ or $1.76 in 1999$.

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, 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 marginal 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, $2.17 per mg/L per household per year, 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 $1.97 per household per year per mg/L. 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	1999$
Poe and Bishop	Annual WTP	$448.00
Average of Crutchfield et al. and De Zoysa	Annual WTP per mg/L between 10 mg/L and 1 mg/L	$1.97

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 marginal reductions between 1 mg/L and 10 mg/L for households that were below the MCL prior to regulatory changes. The last column shows total annual national benefits.

Exhibit 6-1
Undiscounted Annual Values under CAFO Regulatory Scenarios (1999$)

Scenario	Total WTP for Discrete Reduction to MCL	Total WTP for Marginal Changes below 10 mg/L	Total
Option 1 Scenario 1	68,187,392	1,894,630	70,082,022
Option 1 Scenario 2/3	68,187,392	1,984,994	70,172,386
Option 1 Scenario 4b	72,300,032	2,337,253	74,637,285
Option 1 Scenario 4a	72,300,032	2,337,253	74,637,285
Option 2 Scenario 1	72,300,032	2,173,237	74,473,269
Option 2 Scenario 2/3*	72,300,032	2,285,017	74,585,049
Option 2 Scenario 4a*	74,356,352	2,708,730	77,065,082
Option 2 Scenario 4b	74,356,352	2,708,730	77,065,082

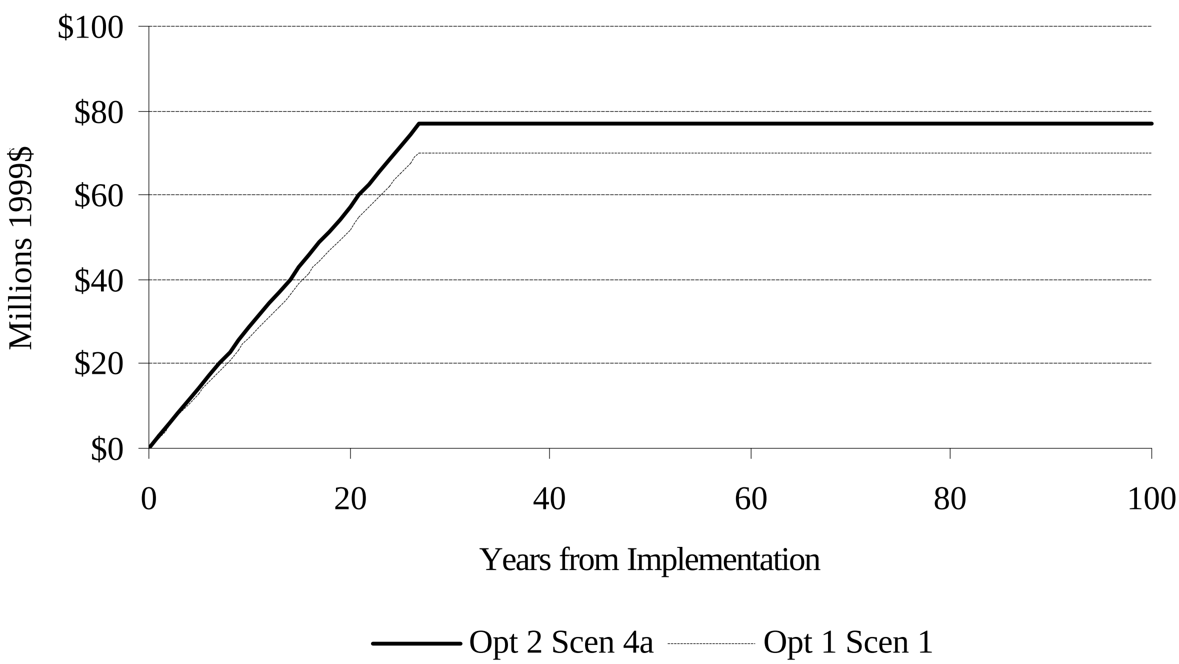
* Proposed scenarios.

6.2 DISCOUNTING AND AGGREGATING TO NET 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 through the 100th year the benefits are equal to the total benefits when of all 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 Scenario 4a and the lower line shows the timepath of benefits for Option 1 Scenario 1, which produces the lowest benefits. The benefit flow from all the options/scenarios falls within these bounds.

A 100-year time frame was used for calculating the timepath of benefits because the majority benefits will be included in this time period if any positive rate of interest is applied for calculating present values. For instance, extending the analysis over a 250 year time frame, an additional 150 years, generates only about 8% additional benefits using a 3% rate of discount, 1.35% additional benefits using a 5% rate of discount, and 0.25% additional benefits using a 7% rate of discount.

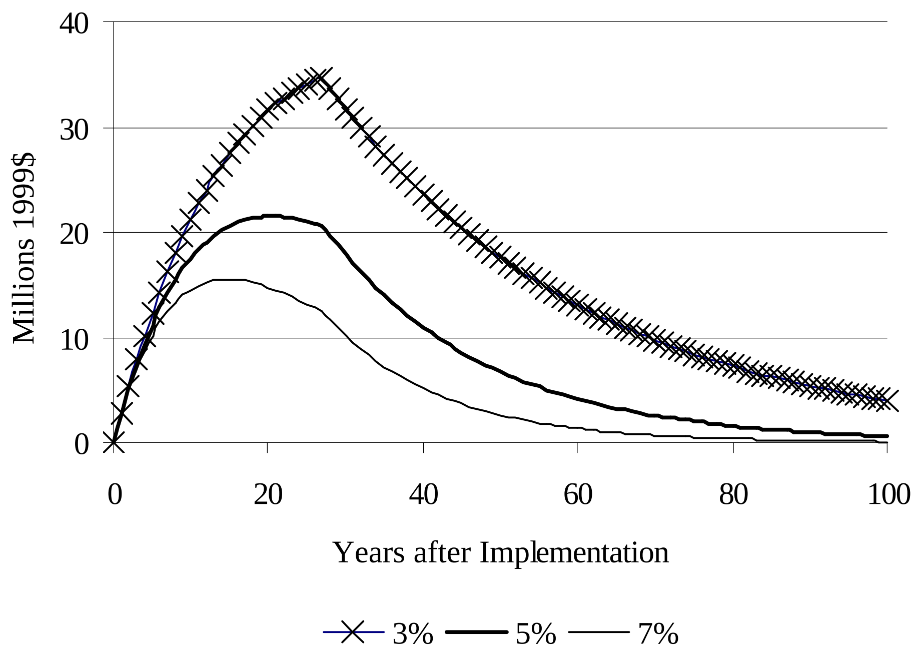
Exhibit 6-2
Timepath of Undiscounted Benefit Flows



6.3 DISCOUNTED BENEFITS

Exhibit 6-3 shows the timepath of discounted benefits for Option 2-Scenario 4a using a 3%, 5%, and 7% rate of discount.[1] 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.

Exhibit 6-3
Discounted Value of Annual Benefits Using 3%, 5%, and 7% Discount Rates
Option 2 Scenario 4a



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 the various options/scenarios using three different rates of discount: 3%, 5%, and 7%. Note that these numbers are presented in millions of 1994$ so the discounted present value for Option 2 Scenario

1. The graph of the discounted value of annual benefits for Option 2 Scenario 2/3 would look virtually identical to that shown in Exhibit 6-3 and thus we only show the graph for one of the proposed options, Option 2 Scenario 4a.

4a using a 3% rate of discount is roughly $1.66 billion. Using a 7% rate of discount this falls to $522 million.

Exhibit 6-4
Total Present Value of Option/Scenarios Using Different Rates of Discount
(millions 1999$)

Scenario	3% Present Value	5% Present Value	7% Present Value
Option 1 Scenario 1	1,511.69	787.51	474.43
Option 1 Scenario 2/3	1,513.64	788.52	475.04
Option 1 Scenario 4b	1,609.95	838.69	505.27
Option 1 Scenario 4a	1,609.95	838.69	505.27
Option 2 Scenario 1	1,606.41	836.85	504.16
Option 2 Scenario 2/3*	1,608.82	838.11	504.92
Option 2 Scenario 4a*	1,662.32	865.98	521.70
Option 2 Scenario 4b	1,662.32	865.98	521.70

* Proposed scenarios.

6.4 ANNUALIZED DISCOUNTED BENEFIT ESTIMATES

Because the benefit flow is uneven over time, it is useful to annualize the present value. 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 6-5 presents the annualized benefit estimates for the total present value benefits shown in Exhibit 6-4. For instance, for Option 2 Scenario 4a, a constant benefit flow of $50.99 million discounted at 3% over 100 years would generate $1,662.32 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 hade been realized at the well) and then the regulations would revert to current (2000) 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 (2000). Exhibit 6-6 shows the maximum undiscounted annual value, the net present value, and the annualized value for this scenario using a 3% rate of discount.

Exhibit 6-5
Annualized Present Value of Option/Scenarios Using Different Rates of Discount
(millions 1999$)

Scenario	3% Annualized Value	5% Annualized Value	7% Annualized Value
Option 1 Scenario 1	46.37	37.77	31.07
Option 1 Scenario 2/3	46.43	37.82	31.11
Option 1 Scenario 4b	49.39	40.23	33.09
Option 1 Scenario 4a	49.39	40.23	33.09
Option 2 Scenario 1	49.28	40.14	33.02
Option 2 Scenario 2/3*	49.35	40.20	33.07
Option 2 Scenario 4a*	50.99	41.54	34.17
Option 2 Scenario 4b	50.99	41.54	34.17

* Proposed scenarios.

Exhibit 6-6
Benefits under Alternative Scenario of Regulatory Discontinuation
in 27 Year (3% rate of discount)
(millions 1999$)

Scenario	Maximum Undiscounted Annual Value	Net Present Value	Annualized Value
Option 1 Scenario 1	70.08	897.98	32.56
Option 1 Scenario 2/3	70.17	899.13	32.60
Option 1 Scenario 4b	74.64	956.34	34.68
Option 1 Scenario 4a	74.64	956.34	34.68
Option 2 Scenario 1	74.47	954.24	34.60
Option 2 Scenario 2/3*	74.59	955.67	34.65
Option 2 Scenario 4a*	77.07	987.45	35.81
Option 2 Scenario 4b	77.07	987.45	35.81

* Proposed scenarios.

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 6-7
Annualized Benefits under Alternative Scenario of Regulatory Discontinuation in 27 Year (3%, 5%, and 7% rate of discount)
(millions 1999$)

Scenario	3%	5%	7%
Option 1 Scenario 1	32.56	29.87	26.75
Option 1 Scenario 2/3	32.60	29.91	26.79
Option 1 Scenario 4b	34.68	31.81	28.49
Option 1 Scenario 4a	34.68	31.81	28.49
Option 2 Scenario 1	34.60	31.74	28.43
Option 2 Scenario 2/3*	34.65	31.79	28.47
Option 2 Scenario 4a*	35.81	32.84	29.42
Option 2 Scenario 4b	35.81	32.84	29.42

* Proposed scenarios.

6.6 SENSITIVITY ANALYSIS

6.6.1 Ranges of Value Estimates

As shown in Exhibit 5-5, Delavan (1996) reported a willingness to pay of $188.64 (see Section 5.2.3) and Jordan and Elnagheeb (1991) reported a willingness to pay of $192.79 per household per year (1999$). Using an approximation of $190.00 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 (1987) reported WTP of $2,380.21 as an upper bound value for household benefits for achieving the MCL.

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 18.5% and 33.0% reduction in estimated annualize benefits respectively.

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	$448.00	$190.00		$2,380.21	
Value for Changes below 10 mg/L	$1.97	$1.97		$1.97	
Scenario	**Annualized Value (1999$)**	**Annualized Value (1999$)**	**Percent Change in Annualized Value**	**Annualized Value (1999$)**	**Percent Change in Annualized Value**
Option 1 Scenario 1	46.37	20.39	-56.0%	240.97	+419.6%
Option 1 Scenario 2/3	46.43	20.45	-56.0%	241.03	+419.1%
Option 1 Scenario 4b	49.39	21.84	-55.8%	255.72	+417.8%
Option 1 Scenario 4a	49.39	21.84	-55.8%	255.72	+417.8%
Option 2 Scenario 1	49.28	21.73	-55.9%	255.61	+418.7%
Option 2 Scenario 2/3	49.35	21.80	-55.8%	255.68	+418.1%
Option 2 Scenario 4a*	50.99	22.66	-55.6%	263.19	+416.1%
Option 2 Scenario 4b*	50.99	22.66	-55.6%	263.19	+416.1%
* Proposed scenarios.					

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 10.4%. Spreading out time until steady state is achieved decreases the present annualized value by 26.1%.

6.6.4 Benefits for Changes under the 10 mg/L

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 adds 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	$448.00	$448.00		$448.00	
Value for Changes below 10 mg/L	$1.97	$1.97		$1.97	
Scenario	**Annualized Value (millions 1999$)**	**Annualized Value (millions 1999$)**	**Percent Change in Annualized Value**	**Annualized Value (millions 1999$)**	**Percent Change in Annualized Value**
Option 1 Scenario 1	46.37	37.77	-18.5%	31.07	-33.0%
Option 1 Scenario 2/3	46.43	37.82	-18.5%	31.11	-33.0%
Option 1 Scenario 4b	49.39	40.23	-18.5%	33.09	-33.0%
Option 1 Scenario 4a	49.39	40.23	-18.5%	33.09	-33.0%
Option 2 Scenario 1	49.28	40.14	-18.5%	33.02	-33.0%
Option 2 Scenario 2/3*	49.35	40.20	-18.5%	33.07	-33.0%
Option 2 Scenario 4a*	50.99	41.54	-18.5%	34.17	-33.0%
Option 2 Scenario 4b	50.99	41.54	-18.5%	34.17	-33.0%

* Proposed scenarios.

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 ~$147 (1999$) is seen when initial nitrate levels are close to 10 mg/L. Below 10 mg/L the per mg WTP falls to about $100 (1999$) per mg when the initial level is 4 mg/L. Sparco (1995) also estimated WTP for marginal changes in nitrate concentrations of $142.46 per mg/L (1999$). 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 marginal changes below the MCL.

Exhibit 6-10
Sensitivity to Changes in Time until Steady State (20 and 50 years)

Discount Rate	**3%**	**3%**		**3%**	
Years to Steady State	27	20		50	
Value for Crossing 10 mg/L	$448.00	$448.00		$448.00	
Value for Changes below 10 mg/L	$1.97	$1.97		$1.97	
Scenario	**Annualized Value (1999$)**	**Annualized Value (1999$)**	**Percent Change in Annualized Value**	**Annualized Value (1999$)**	**Percent Change in Annualized Value**
Option 1 Scenario 1	46.37	51.18	10.4%	34.25	-26.1%
Option 1 Scenario 2/3	46.43	51.24	10.4%	34.30	-26.1%
Option 1 Scenario 4b	49.39	54.50	10.4%	36.48	-26.1%
Option 1 Scenario 4a	49.39	54.50	10.4%	36.48	-26.1%
Option 2 Scenario 1	49.28	54.38	10.4%	36.40	-26.1%
Option 2 Scenario 2/3*	49.35	54.47	10.4%	36.45	-26.1%
Option 2 Scenario 4a*	50.99	56.28	10.4%	37.67	-26.1%
Option 2 Scenario 4b	50.99	56.28	10.4%	37.67	-26.1%
* Proposed scenarios.					

6.7 OMISSIONS, BIASES, AND UNCERTAINTIES

OBU with the Data and Statistical Analysis

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 Net Benefit" in Exhibit 6-12 indicates how the benefit estimate is influenced due to 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 wells sampled in the USGS Retrospective database are focussed more on areas with nitrate problems. 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.

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	$448.00	$448.00		$448.00	
Value for Changes below 10 mg/L	$1.97	$0.00		$100.00	
Scenario	**Annualized Value (1999$)**	**Annualized Value (1999$)**	**Percent Change in Annualized Value**	**Annualized Value (1999$)**	**Percent Change in Annualized Value**
Option 1 Scenario 1	46.37	45.12	-2.7%	108.76	134.5%
Option 1 Scenario 2/3	46.43	45.12	-2.8%	111.79	140.8%
Option 1 Scenario 4b	49.39	47.84	-3.1%	126.34	155.8%
Option 1 Scenario 4a	49.39	47.84	-3.1%	126.34	155.8%
Option 2 Scenario 1	49.28	47.84	-2.9%	120.84	145.2%
Option 2 Scenario 2/3*	49.35	47.84	-3.1%	124.59	152.5%
Option 2 Scenario 4a*	50.99	49.20	-3.5%	140.18	174.9%
Option 2 Scenario 4b	50.99	49.20	-3.5%	140.18	174.9%

* Proposed scenarios.

Data availability limited the variables included in this statistical analysis for the OLS and Gamma models. 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.

This analysis assumes constant nitrate concentrations and loadings over time, 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 across a 20 year period, and nitrates may take decades to reach the groundwater.

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

Variable	Likely Impact on Net 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 275 counties nationwide.
Well location selection	Positive	Wells sampled in the USGS Retrospective database may not be random. Samples may be focused on 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 0-300AU	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 USGS 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 USGS Retrospective database, EPA assumes that 9.74 % 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.
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.

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

Variable	Likely Impact on Net Benefit[a]	Comment
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 as EPA does not have reliable value estimates for changes marginal changes above the MCL.
Exclusion of values for marginal 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 as 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.

REFERENCES

Agriculture Canada. 1992. Ontario Farm Groundwater Quality Survey. Agriculture Canada, Ottawa.

Andres, A.S. 1991. Results of the Coastal Sussex County, Delaware Groundwater-Water Quality Survey. Report of investigations no. 49. Delaware Geological Survey, Newark, DE.

Bishop, R., P.A. Champ, T.C. Brown, and D.W. McCollum. 1997. "Measuring the Non-Use Values: Theory and Empirical Application." In *Determining the Value of Non-Marketed Goods: Economic, Psychological, and Policy Relevant Aspects of Contingent Valuation Methods*. R.J. Kopp, W.W. Pommerehne, and N. Schwarz (eds.), Kluwer Academic Publishers, pp. 59-81.

Boyle, K., G. Poe, and J. Bergstrom. 1994. "What do we know about groundwater values? Preliminary Implications from a Meta-Analysis of Contingent-Valuation Studies." American Journal of Agricultural Economics, December 1994, 1055-1064.

Burrow, K.R. 1998. Occurrence of Nitrate and Pesticides in Groundwater beneath Three Agricultural Land-Use Settings in the Eastern San Joaquin Valley, California, 1993-1995. U.S. Geological Survey, Denver, CO.

Carleton, G.B. 1996. Nitrate in Groundwater and Surface Water in a Residential Subdivision, West Mercer Township, New Jersey. U.S. Geological Survey, Denver, CO.

Carson, R.T. 1997. "Contingent Valuation Surveys and Tests of Insensitivity to Scope." In *Determining the Value of Non-Marketed Goods: Economic, Psychological, and Policy Relevant Aspects of Contingent Valuation Methods*. R.J. Kopp, W.W. Pommerehne, and N. Schwarz (eds.), Kluwer Academic Publishers, pp. 127-164.

CDC. 1998. A Survey of the Quality of Water Drawn from Domestic Wells in Nine Midwest States. NCEH 97-0265. National Center for Environmental Health, Centers for Disease Control and Analysis, Atlanta, GA.

Chen, A.H. 1998. Nitrate Concentrations in Groundwater in the Paleovalley Alluvial Aquifers of the Nemaha Natural Resources District, Nebraska. U.S. Geological Survey, Lincoln, NE.

Chestnut, L.G., D. Mills, and A. Alberini. 1997. Monetary Valuation of Human Mortality Risks in Cost-Benefit Analyses of Environmental Programs: Background Paper and Bibliography.

Prepared for U.S. Environmental Protection Agency, Office of Policy Planning and Evaluation by Hagler Bailly Services, Boulder, CO.

Clawges, R.M. and E.F. Vowinkel. 1996. "Variables Indicating Nitrate Contamination in Bedrock Aquifers, Newark Basin, New Jersey." *Water Resources Bulletin* 32: 1055-1066.

Crutchfield, S.R. and J. Cooper. 1997. "Valuing Risk Reduction: The Example of Nitrates in Drinking Water." *Food Review* January-April, 38-41.

Crutchfield, S.R., J.C. Cooper, and D. Hellerstein. 1997. Benefits of Safer Drinking Water: The Value of Nitrate Reduction. Agricultural Economic Report 752. U.S. Department of Agriculture, Economic Research Service, Food and Consumer Economics Division.

Crutchfield, S.R., P.M. Feather, and D.R. Hellerstein. 1995. The Benefits of Protecting Rural Water Quality, An Empirical Analysis. U.S. Department of Agriculture, Economic Research Service, Agricultural Economic Report Number 701.

De Zoysa, A.D.N. 1995. A Benefit Evaluation of Programs to Enhance Groundwater Quality, Surface Water Quality and Wetland Habitat in Northwest Ohio. Dissertation, The Ohio State University, Columbus.

Delavan, W. 1997. Valuing the Benefits of Groundwater Protection from Nitrate Contamination in Southeastern Pennsylvania. Master's Thesis, Department of Agricultural Economics and Rural Sociology, Penn State University, State College.

Desvousges, W.H., M.C. Naughton, and G.P. Parsons. 1992. "Benefit Transfer: Conceptual Problems in Estimating Water Quality Benefits Using Existing Studies." *Water Resources Research* 28(3):675-683.

Detroy, M.G., P.K.B. Hunt, and M.A. Holub. 1988. Groundwater Quality Monitoring Program in Iowa: Nitrate and Pesticides in Shallow Aquifers. U.S. Geological Survey, Denver, CO.

Dillman, D. 2000. Reduction of Coverage and Sampling Errors. *In* Mail and Internet Surveys. John Wiley and Sons, New York.

Edwards, S.F. 1988. "Option Prices for Groundwater Protection." *Journal of Environmental Economics and Management* 15:475-487.

Fischhoff, B. and L. Furby. 1988. "Measuring Values: A conceptual framework for interpreting transactions with special reference to contingent-valuation of visibility. Journal of Risk and Uncertainty, 1:147-184.

Freeman, A.M. 1993. Valuing the Longevity of Health. *In* The Measurement of Environmental and Resource Values. Resources for the Future, Washington, D.C.

Giraldez, C. and G. Fox. 1995. "An Economic Analysis of Groundwater Contamination from Agricultural Nitrate Emissions in Southern Ontario." *Canadian Journal of Agricultural Economics* 43:387-402.

Hall, M. 1996. Simulation of Nitrates in a Regional Subsurface System: Linking Surface Management with Ground Water Quality. PhD Thesis, Colorado State University, Fort Collins.

Ham, J.M., L.N. Reddi, C.W. Rice, and J.P. Murphy. 1998. Earthen Lagoons for Containing Animal Waste: Review of Factors Affecting Seepage Losses and Groundwater Quality. Department of Agronomy, Kansas State University, Manhattan.

Hanley, N. 1989. Problems in valuing environmental improvements resulting from agricultural change: The case of nitrate pollution. *In* Economic Aspects of Environmental Regulation in Agriculture, A. Dubgaard and A. Hjortshoj Nielsen, eds. Wissenschaftsverlag: Vauk Kiel KG.

Hurley, T.M., D. Otto, and J. Holtkamp. 1999. "Valuation of Water Quality in Livestock Regions: An Application to Rural Watersheds in Iowa." *Journal of Agricultural and Applied Economics* 31:177-184.

Jordan, J.L. and A.H. Elnagheeb. 1993. "Willingness to Pay for Improvements in Drinking Water Quality." *Water Resources Research* 29:237-245.

Kalton, G. 1983. Introduction to Survey Sampling. Sage University Paper 35. Newbury Park, CA.

Kerr-Upal, M., T. Van Seters, and M. Stone. 1999. "Assessing the Risk of Groundwater Nitrate Contamination in the Region of Waterloo, Ontario. *Canadian Water Resources Journal* 24: 225-233.

Kross, B.C., G.R. Hallberg, and D.R. Bruner. 1993. "The Nitrate Contamination of Private Well Water in Iowa." *American Journal of Public Health* 83: 270-272.

Letson, D., N. Gollehon, and C. Mose. 1998. "Confined Animal Production and Groundwater Protection." *Review of Agricultural Economics* 20: 348-364.

Lichtenberg, E. and L.K. Shapiro. 1997. "Agriculture and Nitrate Concentrations in Maryland Community Water System Wells." *Journal of Environmental Quality* 26:145-153.

Lindsey, B.D. 1997. Nitrate in Groundwater and Streambase Flow in the Lower Susquehanna River Basin, Pennsylvania and Maryland. U.S. Geological Survey, Denver, CO.

Loomis, J. 1992. "The Evolution of a More Rigorous Approach to Benefit Transfer."Water Resources Research, 28(3):701-705.

Miller, M.H., J.B. Robinson, and D.W. Gallagher. 1976. "Accumulation of Nutrients on Soil Beneath Hog Manure Lagoons." *Journal of Environmental Quality* 5: 279-288.

Mitchell, R. and R. Carson, 1989. Using Surveys to Value Public Goods: The Contingent-Valuation Method. Resources for the Future, Washington, D.C.

Morrison, M.D., R.K. Blamey, and J.W. Bennett. 2000. Minimising Payment Vehicle Bias in Contingent Valuation Studies. *Environmental & Resource Economics* 16(4):407-422.

Mueller, D.K., P.A. Hamilton, D.R. Helsel, K.J. Hitt, and B.C. Ruddy. 1995. Nutrients in Ground Water and Surface Water of the United States — An Analysis of Data through 1992. USGS Water-Resources Investigations Report 95-4031, U.S. Geological Survey.

National Research Council. 1997. *Valuing Ground Water: Economic Concepts and Approaches*. National Academy Press, Washington, DC.

Nolan, B.T., B.C. Ruddy, K.J. Hitt, and D.R. Helsel. 1998. "A National Look at Nitrate Contamination of Groundwaters." *Water Conditioning and Purification* 39:76-79.

North Carolina Division of Water Quality, Groundwater Section. 1998. Impact of Animal Waste Lagoons on Ground Water Quality. Draft Report. North Carolina Department of Environment and Natural Resources, Raleigh.

Poe, G.L. 1998. "Valuation of Groundwater Quality Using a Contingent Valuation-Damage Function Approach." *Water Resources Research* 34(12):3627-3633.

Poe, G.L. 1993. Information, Risk Perceptions, and Contingent Value: The Case of Nitrates in Groundwater. PhD Dissertation, University of Wisconsin-Madison.

Poe, G.L. and R.C. Bishop. 1999. "Valuing the Incremental Benefits of Groundwater Protection When Exposure Levels Are Known."*Environmental and Resource Economics* 13:341-367.

Poe, G. and R.C. Bishop. 1992. Measuring the Benefits of Groundwater Protection from Agricultural Contamination: Results from a Two-Stage Contingent Valuation Study. Staff Paper Series, University of Wisconsin-Madison.

Randall, A. and D. DeZoysa. 1996. Groundwater, Surface Water, and Wetlands Valuation for Benefits Transfer: A Progress Report. W-133 Benefits and Costs Transfer in Natural Resource Planning, Ninth Interim Report. Department of Economics, Iowa State University, Ames.

Rausch, J.N. 1992. Sources of Ohio Farm Water Well Nitrate-Nitrogen Contamination and the Willingness to Pay for Remediation. Masters Thesis, available from Ohio State University, Columbus.

Richards, R.P., D.B. Baker, and L.K. Wallrabenstein. 1996. "Well Water Quality, Well Vulnerability, and Agricultural Contamination in the Midwestern United States." *Journal of Environmental Quality* 25:389-402.

Ritter, W.F. and A.E.M. Chirnside. 1990. "Impact of Animal and Waste Lagoons on Groundwater Quality." *Biological Wastes* 34:39-54.

Smith, V.K., G. Van Houtven, and S. Pattanayak. 1999. "Benefit Transfer as Preference Calibration," Resources for the Future discussion paper 99-36. http://www.rff.org/disc_papers/PDF_files/9936.pdf. Accessed 9/1/00.

Spalding, R.F. and M.E. Exner. 1993. "Occurrence of Nitrate in Groundwater-A Review." *Journal of Environmental Quality* 22:392-402.

Sparco, J. 1995. "Marginal Valuation of Health-Related Attributes of Ground Water Using Conjoint Analysis." PhD Dissertation, University of Delaware, Newark.

Sparco, J. 1995. "Agricultural Land Uses and Ground-Water Contamination: A Spatial Regression Analysis." PhD Dissertation, University of Delaware, Newark.

Stuart, M.A., F.J. Rich, and G.A. Bishop. 1995. "Survey of Nitrate Contamination in Shallow Domestic Drinking Water Wells of the Inner Coastal Plain of Georgia." *Journal of the National Water Well Association* 33:284.

Swistock, B.R., W.E. Sharpe, and P.D. Robillard. 1993. "A Survey of Lead, Nitrate, and Radon Contamination of Individual Water Systems in Pennsylvania." *Journal of Environmental Health* 55:6-12.

Tetra Tech. 2000. Development of Pollutant Loading Reductions from the Implementation of Nutrient Management and Best Management Practices. Draft report prepared for U.S. Environmental Protection Agency, Office of Science and Technology by Tetra Tech, Fairfax, VA.

Townsend, M.A., R.O. Sleezer, and S.A. Macko. 1996. Effects of Agricultural Practices and Vadose Zone Stratigraphy on Nitrate Concentration in Ground Water in Kansas, USA. *Water Science Technology* 33:4:219-226.

U.S. Bureau of Labor Statistics. 2000. Consumer Price Index. ftp://ftp.bls.gov/pub/special.requests/cpi/cpiai.txt. Accessed 10/03/00.

U.S. EPA. 1990. National Survey of Pesticides in Drinking Water Wells. U.S. Environmental Protection Agency.

USGS. 1986. Occurrence of Nitrate and Herbicides in Groundwater in the Upper Conestoga River Basin, Pennsylvania. Water Resources Investigations Report 85-4202, U.S. Geological Survey.

USGS. 1996. Retrospective Database for Nutrients in Groundwater and Surface Water. U.S. Geological Survey. http://water.usgs.gov/nawqa/nutrients/datasets/retrodata.html. Accessed 6/1/00.

USGS. 1998. National Water Quality Assessment Database. U.S. Geological Survey. http://water.usgs.gov/nawqa/nutrients/datasets/cycle91/. Accessed 6/1/00.

Walker, D.R. and J.P. Hoehn. 1990. "The Economic Damages of Groundwater Contamination in Small Rural Communities: An Application to Nitrates." *Northcentral Journal of Agricultural Economics* 12:47-56.

Walsh, R., D. Johnson, and J. McKean. 1992. "Benefit Transfer of Outdoor Recreation Demand Studies, 1968-1988." Water Resources Research 28(3):707-713.

Wattage, P.M. 1993. Measuring the Benefits of Water Resource Protection from Agricultural Contamination: Results from a Contingent Valuation Study. Dissertation, Iowa State University, Ames.

Wolf, F.M. 1986. Meta-Analysis: Quantitative Methods for Research Synthesis. Sage University Paper 59. Sage Publications, Newbury Park, CA.

APPENDIX A
NITROGEN SOURCES AND WELL DATA

Several individual data sets 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 included data on the total nitrogen loadings for each county under each scenario, and was created by combining information from three different data sets 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 data set was provided as a csv file that was imported into an Excel spreadsheet. The file contains information on modeled leached and surface nitrogen and phosphorous loadings for baseline and for the regulatory options. The data set has 196 rows for the facility types and 30 columns for the facility type identifiers and the loadings for each type of facility in each region, for baseline and for the two regulatory options. Baseline data includes different loadings for facilities that are currently regulated and unregulated. In addition to baseline data, the Option 1 data is based on regulations according to nitrogen application amounts, and the Option 2 data is based on regulations defined by phosphorous application amounts. The data set also contains information on other options that are not being analyzed at this time.

Facility: The facility data set was provided as a csv file that was imported into an Excel spreadsheet. This data set identifies the average number of facilities by animal type and size for 3078 counties (including some counties that have 0 facilities). The data set identifies animal types of beef, veal, broilers, dairy, two types of swine, wet layers, dry layers, turkey, and heifers. Size categories include M1a, M1b, M1, M2, L1, and L2. The data set has 3078 rows (one for each county) and 45 columns, including identifier columns for the counties and number of facilities for the different animal type and facility size.

State Percent: The state percent data set was provided as a csv file that was imported into an Excel spreadsheet. The data set identifies the percentage of each type and size of facility that will be regulated under each scenario (including baseline) for each state. Dry poultry operations are excluded from the baseline.

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 2504 counties with AFOs. Data from 368 of these 2504 counties are combined with data from the USGS retrospective database (described below) for estimation of the Gamma model. An issue is whether the counties used for the Gamma modeling are different in some manner from those (2136) not used for estimating the nitrogen-nitrate relationship. Exhibit A-1 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 counties. In addition the included counties are somewhat smaller (28% smaller) and have a roughly 10-12% larger population, median income, and number of housing units. The included counties also have a smaller portion of their population living in rural areas.

Septic Ratio: Two other datasets were used to derive data on potential nitrogen loadings from septic systems. This data is not in the form of loadings in terms of pounds of nitrogen per acre but in terms of the number density of households on septic systems in county. This provides a proxy measure for the contribution of septic systems to well nitrate concentrations. The ratio of septic systems in a county to the total number of acres in the county was based on the 1990 U.S. Census and the 1997 Census of Agriculture. 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 (also called the Retrospective database) contains water quality and land use data from approximately 10,426 well samples from 725 counties in 38 states. The data were gathered between 1969 and 1992.

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.

Exhibit A-1
Comparison of Mean Loadings and Sociodemographics for Counties in the Loadings Database Used in the Gamma Modeling
(nobs for not in retro = 2136; nobs for in retro = 368)

Variable	Mean (not used in the Gamma model)	Mean (used in the Gamma model)	Percent Difference	Z (Wilcoxon rank test)
Baseline Nitrogen Loadings	469,715	1,421,224	203%	8.70
Option 1 — Scenario 1	257,615	665,583	158%	8.06
Option 1 — Scenario 2/3	239,250	613,966	157%	8.19
Option 1 — Scenario 4a	177,713	475,348	167%	9.38
Option 1 — Scenario 4b	177,713	475,348	167%	9.38
Option 2 — Scenario 1	228,387	585,046	156%	8.49
Option 2 — Scenario 2/3*	207,687	526,603	154%	8.68
Option 2 — Scenario 4a*	140,848	371,924	164%	10.80
Option 2 — Scenario 4b	140,848	371,924	164%	10.80
Acres	723,368	520,831	-28%	-2.72
Loadings per Acre (baseline)	1.20	3.09	157%	8.70
Population[a]	76,341	85,477	12%	3.65
Median Household Income[a]	23,908	26,349	10%	7.63
Housing Units[a]	31,200	34,434	10%	3.23
Percent of Population Rural[a]	64.2%	59.5%	-7%	-2.96

* Proposed scenarios.
a. nobs = 2110 for “not in the retrospective database” and 363 for “in the retrospective database.”

Pollution input information includes: atmospheric nitrogen input, fertilizer nitrogen input in tons 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-2 provides summary statistics on the observations from the USGS Retrospective Dataset for all observations in the dataset. This includes all water use types. Only a subset of these observations (2504 observations) were usable for the analysis described in Chapter 3 due to 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 percent exceed the MCL of 10 mg/L.

Exhibit A-3 shows the distribution of well water use for observations in the USGS dataset. As the benefits transfer exercise is focussed on domestic water well use we limited analysis to wells listed as domestic. This amounts to roughly 31% of the observations from the USGS Retrospective dataset.

Exhibit A-4 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 and atmospheric nitrogen inputs 8.85 tons/sq. mile exceeds that from manure of 6.03 tons/sq. mile. On average therefore, 59% of nitrogen loadings come from non-manure sources. 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 other words, not recognizing these other sources of nitrogen inputs to groundwater would likely overstate the impact (and thus benefits) of reducing nitrogen inputs from CAFOs. The contribution from CAFOs on average is even a smaller percent as loadings from septic systems are not included here. As discussed above, we only have a proxy measure for septic systems in terms of the density of septic systems in a county.

It should also be noted that there are considerably fewer observations in the data set on phosphate concentrations in well water samples (about 69% of the observations do not include total phosphate measurements. This may thus make it difficult to reliably model the relationship between phosphorous loadings and well phosphate concentrations (an avenue we have not explored at this time).

Exhibit A-2
USGS Retrospective Dataset 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-3
Distribution of Well Water Use in USGS Dataset

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-4
USGS Retrospective Dataset 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

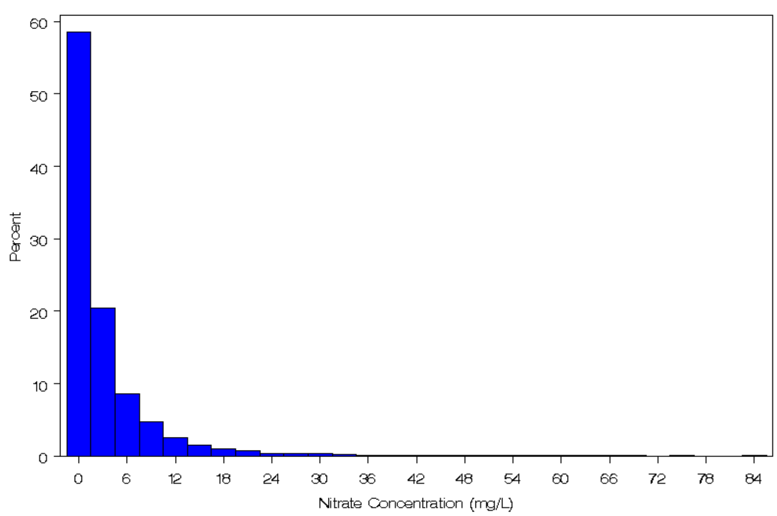
APPENDIX B
STATISTICAL MODELS

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:

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

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 = 4.84) and a thick tail (kurtosis = 36.95) (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:

$$f(y) = \frac{q^{a}}{\Gamma(a)} \exp(-q y) y^{a-1}$$

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 $q_i = \exp(-bx_i)$. For this distribution, $E(y_i) = a / q_i = a \exp(b x_i)$. Maximum likelihood methods are used to estimate the parameters. The log likelihood function is:

$$\log L(y_i | x_i; a, b) = \sum_i \left[a \log q_i - \log \Gamma(a) - q_i y_i + (a-1)\log(y_i) \right]$$

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. A likelihood ratio test of the difference between the gamma model with no independent variables and the model as shown in Exhibit B-2 yielded a χ^2 statistic of 556.9 with 6 degrees of freedom, which is significant at the 1% level. From the gamma model, expected values can be calculated using:

$$E(y_i) = a / q_i = a \exp(bx_i) \, .$$

Exhibit B-2
Gamma Regression Results (Nobs = 2928)

Variable	Parameter Estimate	Std. Err.	Asymptotic t-statistic[b]
Intercept	1.492	0.151	9.891
Ag Dummy	0.691	0.066	10.452
Soil Group	-0.335	0.043	-7.725
Well Depth (per 1/100 ft)[a]	-0.106	0.015	-7.178
Septic Ratio	2.623	1.102	2.380
Alt N Source (1/1,000 lbs N/acres in county)[a]	20.258	1.628	12.444
Loadings Ratio (lbs N/acres in county)	0.010	0.002	5.037
Alpha	0.498	0.011	46.368

a. The raw data was scaled by a factor of 100 for well depth and 1000 for Alt N Source.
b. All parameter estimates are significant at or below the 1% level.
Mean log-likelihood = 1.854.

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 1,285.1, so that the null hypothesis that α = 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.

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 and Tobit models were estimated using SAS Version 7.

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
Nobs = 2,928

Variable	Parameter Estimate	Standard Error	T Value
Intercept	2.888	0.594	4.860[a]
Ag Dummy	1.479	0.288	5.140[a]
Soil Group	-0.970	0.180	-5.400[a]
Well Depth	-0.004	0.001	-4.810[a]
Septic Ratio	4.601	4.813	0.960
Alt N Source	0.074	0.007	11.280[a]
Loadings Ratio	0.047	0.010	5.180[a]

a. Indicates significant at the 0.01% level.
F Value = 60.35; Adjusted R^2 = 0.109.

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 not significant. As this is the primary policy variable, other models were estimated. It must also 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

As well nitrates at or below the detection limit were reported in a number of ways, non-detects were set to 0.05 mg/L. 519 of the 2928 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
Nobs = 2,928

Variable	Estimate	Standard Error	Chi-Square	Pr > ChiSq
Intercept	2.838	0.699	16.468	<.001
Ag Dummy	2.377	0.344	47.849	<.001
Soil Group	-1.879	0.214	77.442	<.001
Well Depth	-0.391	0.100	15.202	<.001
Septic Ratio	12.836	5.569	5.313	0.021
Alt N Source	84.408	7.634	122.250	<.001
Loadings Ratio	0.052	0.010	25.019	<.001
Log likelihood -8487.49.				

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 and weighted Tobit models purely to explore the data and the relationships between dependent and independent variables as well as potential misspecifications of the error term.

Weighted Tobit

As observations on several of the variables are at the county level, it was felt that there may be a relationship between county size and the quality of the observations. For instance, as soil group is a county-wide measure of the soil types in a given county, using this measure for a large county may be more likely to misrepresent the soil types near a given well than is using soil group in a smaller county. It was thus felt observations on individual well conditions using county data were more likely to contain error for large counties than for smaller counties. We thus repeated the

Tobit model estimation from Exhibit B-4, weighting observations by the inverse of the acres in the county. The inverse of acres in the county was multiplied by 100,000 so that the largest weight was close to one (0.978) and the smallest weight was 0.019 (mean weight = 0.266). Exhibit B-5 present the results from this weighted Tobit analysis.

Exhibit B-5
Weighted Tobit Regression Results
(weighted by inverse of acres in county * 100,000)
Nobs = 2,928

Variable	Estimate	Standard Error	Chi-Square	Pr > ChiSq
Intercept	3.937	1.341	8.620	0.003
Ag Dummy	2.057	0.596	11.920	0.001
Soil Group	-2.373	0.411	33.332	<.0001
Well Depth	-0.510	0.204	6.241	0.013
Septic Ratio	24.390	9.834	6.151	0.013
Alt N Source	82.571	14.756	31.312	<.0001
Loadings Ratio	0.038	0.020	3.619	0.057
Scale	6.524	0.189		
Log likelihood -2,157.71.				

As determined by the probabilities of the χ^2s, except for the coefficient on the septic ratio, the weighted Tobit in general produced poorer results than the unweighted Tobit model. Likewise a weighted Tobit regression using the inverse of nitrates as the weight did not produce better results. We also explored regression analysis in the OLS and Tobit model using some non-linearities such as the inverse of well depth. These did not produce significantly better models and were not pursued.

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 as 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:

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

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

$$\log L(y_i | x_i; b) = \sum_i [\log q_i - q_i y_i]$$

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-6 presents the results of estimating the exponential model using GAUSS.

Exhibit B-6
Exponential Regression Results
Nobs = 2,928

Variable	Parameter Estimate	Standard Error	Asymptotic T Value
Intercept	0.795	0.105	7.543[b]
Ag Dummy	0.691	0.047	14.814[b]
Soil Group	-0.335	0.031	-10.949[b]
Well Depth (per 1/100 ft)	-0.106	0.010	-10.174[b]
Septic Ratio	2.623	0.777	3.374[b]
Alt N Source (1/1,000 lbs N/acres in county)[a]	20.258	1.149	17.637[b]
Loadings Ratio (lbs N/acres in county)	0.010	0.002	7.140[b]

a. The raw data was scaled by a factor of 100 for well depth and 1000 for Alt N Source in order for the GAUSS program to converge to a solution.
b. Indicates significant at the 0.01% level. Mean log-likelihood -2.0733.

All of the coefficients are significant at the 1% level and of the expected sign. The coefficients are identical to those estimated in the gamma model except for the alpha coefficient which is implicitly restricted to one in the exponential model (Exhibit B-2). Note that the data for well depth and alternative nitrogen sources was scaled in order for GAUSS to converge to a solution. To compare the parameter estimate on Alt N Source to Loadings Ratio it is necessary to divide the Alt N Source parameter estimate by 1,000.

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 is 0.823 mg/L for non-agricultural land and 2.238 mg/L for wells on agricultural land. On average across all land types, expected well nitrate concentrations are 1.407 mg/L according to the exponential model. These values are within the range of natural or ambient nitrate concentrations as reported in Section 3.3.

APPENDIX C
SUMMARY OF GROUNDWATER VALUATION OF NITRATE CONTAMINATION LITERATURE

Study Reference	Crutchfield et al.	Delavan	de Zoysa	Edwards	Giraldez
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, Massachusetts	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. Crutchfield et al. indicate that there were 819 usable responses and about a 50% response rate.
c. 84 of the 147 versions included the GW valuation scenario. These were randomly distributed proportionally to the 1000 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	Crutchfield et al.	Delavan	de Zoysa	Edwards	Giraldez
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% GW apparently from a public water supply distributing untreated well water.

Study Reference	Crutchfield et al.	Delavan	de Zoysa	Edwards	Giraldez
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 IOD: $29.26 w/protest bidders DOE: $67.85 w/o protest bidders IOD: $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 disagregating value community value estimate: $412.50 per HH ($72.73/yr to $1696.97/yr)
24. Mean Annual Household WTP in 1999 Dollars	$59.46/month (reduced N to MCL); $61.27 (no N): $1.81 difference.	DOE: $47.55 w/protest bidders IOD: $31.07 w/protest bidders DOE: $72.04 w/o protest bidders IOD: $50.08 w/o protest bidders	$59.33 lower bound mean	$2380.21 for a management plan to increase the probability of supply from 0.0 to 1.0	Based on disagregating value community value estimate:$450.94 per HH ($79.51/yr to $1,855.09/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	Crutchfield et al.	Delavan	de Zoysa	Edwards	Giraldez
25. Median Annual Household WTP in Study year dollars	n.a.	DOE: $5 w/protest bidders IOD: $0 w/protest bidders DOE: $50 w/o protest bidders IOD: $25 w/o protest bidders	$20.80 median (1994$ from YNP model)	n.a.	n.a.
26. Median Annual Household WTP in 1999 Dollars	n.a.	DOE: $5.31 w/protest bidders IOD: $0 w/protest bidders DOE: $53.09 w/o protest bidders IOD: $26.55 w/o protest bidders	$30.50 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

Study Reference	Hurley et al.	Jordan and Elnagheeb	Poe and Bishop	Sparco
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
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 1999 Dollars	Not specified	Public:$156.81/hh/yr private: $192.79/hh/yr[j] (primarily use)	$244.31 NINT $1,175.69 WINT $298.85 NIWT $644.17 WIWT	$142.46 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 1999 Dollars	$118.13 (10 year delay) to $190.75 (20 year delay) for household with mean socio-economic characteristics	Public: $85.49/hh/yr private: $114.92/hh/yr	$237.85 NINT $1,043.95 WINT $296.72 NIWT $621.31WIWT	n.a.
27. Range	n.a.	$128.20 — $157.61/hh/yr	$199.73-$961.16/hh/yr	n.a.

Study Reference	Hurley et al.	Jordan and Elnagheeb	Poe and Bishop	Sparco
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

Study Reference	Walker and Hoehn	Wattage
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
23. Mean Annual Household WTP in Study Year Dollars	n.a.	$80/month
24. Mean Annual Household WTP in 1999 Dollars	n.a.	$95.00/month
25. Median Annual Household WTP in Study Year Dollars	n.a.	Not specified
26. Median Annual Household WTP in 1999 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.		