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Implications of Alternative Policies on Nitrate Contamination of Groundwater

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This study estimates the cost effectiveness of alternative environmental policies for controlling nitrate contamination of groundwater in the Seymour aquifer region of Texas. Results from a biophysical simulation model are integrated with a farm-level optimization model. The study also compares the cost of bottled water, used as the lower-bound estimate of benefits of groundwater protection, with the least costly environmental policy. Results indicate that the least-cost policy alternative for the region is about $1 million either to farmers or to the local government and it is approximately three times the cost of bottled water.

Key words: aquifer, environmental policy, groundwater, nitrate, nonpoint pollution

Introduction

In the U.S., nitrate is the most widespread agriculturally related chemical found in groundwater samples (Swinton and Clark). Agriculture is perceived as a major contributor of nitrates, primarily from the use of nitrogen fertilizers (U.S. Environmental Protection Agency; Nielsen and Lee; Office of Technology Assessment). Over 97% of rural America's drinking water comes from groundwater and 40% of the population served by public water supplies uses groundwater (Nielsen and Lee). Nitrates in drinking water can cause methamoglobinemia, or "blue baby disease," but only at levels far above the EPA standard of 10 parts per million (Fan, Willhite, and Book). Nevertheless, the long-term health risk of consuming water with nitrates is not fully understood, and opinion research indicates that consumers, particularly those with children, are wary of risks (Winter, Seiber, and Nuckton). Pollution of groundwater is generally irreversible, and the diffuse nature of nitrate contamination makes it more difficult to control than many other forms of pollution. These factors have combined to make agricultural nitrate management an important policy issue.

Policymakers desire policies that will protect groundwater quality while minimizing adverse effects on farm income. Several policy alternatives for dealing with nonpoint source (NPS) pollution have been proposed. At the theoretical level, Griffin and Bromley focused on the choice between standards and incentives for agriculture. More recent theoretical literature focused on the properties of tax instruments based on ambient concentrations (e.g., Cabe and Herriges; Segerson 1988; Xepapadeas), liability rules (e.g., Segerson 1990; Wetzstein and Centner), and firm-specific taxes and standards based on emissions or inputs associated with emissions (e.g., Shortle and Dunn; Miltz, Braden, and Johnson; Shortle and

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Abler; Abler). Along with these theoretical developments, empirical research on NPS pollution has traditionally followed the "indirect approach" (Dosi and Moretto; Shortle and Dunn), which suggests basing policies on actual NPS emissions estimated by available biophysical simulation models.[1] These studies largely involved integrating available or simulated data with economic optimization models (e.g., Anderson, Opaluch, and Sullivan; Diebel et al.; Foltz et al.; Huang and Lantin; Johnson, Adams, and Perry; Kramer et al.; Mapp et al.; McSweeny and Shortle; Painter and Young; Taylor, Adams, and Miller; Wu, Mapp, and Bernardo).

The intent of our article is to contribute to the empirical literature by evaluating alternative policies for controlling nitrate contamination of groundwater in the Seymour aquifer region of Texas. The motivation for this article stems from the opportunity to improve and build upon the existing empirical studies. A number of issues have been somewhat ignored or not fully examined in past studies. First, an important consideration for the groundwater contamination problem is the need to model the movement of agricultural chemicals from the point of discharge to the point of entry into the aquifer. Studies to date have considered nutrient or pesticide percolation only to the bottom of the plant root zone. Second, risk behavior of farmers has received scant attention in the empirical literature of NPS pollution. Not incorporating risk aversion may lead to erroneous predictions about how farmers will respond to environmental policies. Third, in most studies, government farm program provisions have not been incorporated in the economic model. Finally, while numerous studies have focused on the costs of agricultural pollution reduction measures, no attempt has been made to compare the costs with a measure of benefits.

An additional motivation for this study arises from the urgency of studying an important environmental issue in Texas. Groundwater in the Seymour aquifer region contains elevated nitrate levels (Texas Water Commission; Neilsen and Lee). The aquifer is very shallow, and the soil is sandy and porous. The aquifer is the only source of groundwater for the area, and many residents do not have access to municipal water. Designated as a hydrologic unit area under the president's water quality initiative, the Seymour aquifer is one of 45 projects in 34 states that the U.S. Department of Agriculture implemented in 1991 to speed adoption of best management practices to improve water quality. The nitrate-nitrogen content of 62 water samples collected from the Seymour formation in 1962 varied from 5–41 ppm (parts per million) with 39 samples exceeding the EPA standard of 10 ppm (Ogilbee and Osborne). More recent studies (Kreitler; Harden and Associates; Aurelius) confirm that much of the water in the Seymour aquifer is well above the EPA drinking-water standard of 10 ppm.

Empirical Procedures and Data

Because of unavailable data on nitrate percolation and crop yield for alternative production practices, EPIC-WQ (Erosion Productivity Impact Calculator–Water Quality) simulation model was used to simulate crop yield and nitrate percolation for the study area. Data generated by EPIC-WQ were used to estimate yield and nitrate percolation response functions. The response functions were incorporated inside a risk-sensitive farm-level optimization model which was used to measure the economic and environmental impact of

[1] According to the proponents of this approach, while such biophysical models will never provide a perfect substitute for accurate monitoring of actual flows, "...they can serve as an important tool for diminishing the uncertainty about nonpoint loadings...furthermore, predictions obtained from such models offer an alternative to actual flows as a basis for the application of policy instruments" (Shortle and Dunn, p. 668).

alternative policies. Finally, the cost of bottled water, a lower-bound estimate of benefits of groundwater protection, was compared with the least costly environmental policy.

Application of EPIC-WQ

EPIC-WQ is composed of physically based components for processes of soil erosion, plant growth, weather, hydrology, nutrient cycling, tillage, soil temperature, and economics (Sharpley and Williams). The model has been tested extensively and used in a number of local, regional, and national studies to investigate the effects of weather, soil, and agricultural management practices on crop yields, soil erosion, and nutrient cycling (Cabelguenne et al.; Jones and O'Toole). EPIC-WQ, however, cannot model the geohydrology of an aquifer, and here the model is used only to simulate crop yield and nitrate percolation. Thus, while the concentration of percolated nitrate may not immediately translate into well water concentration, a reduction in concentration of percolated nitrate is expected to reduce the concentration of nitrate in well water, particularly in the long run.

EPIC-WQ's soil database maintains a two-meter soil data profile for different regions of the country. To modify the EPIC-WQ model by incorporating the vadose zone, thirty-six well logs were selected from a Texas Department of Water Resource's study on the Seymour aquifer (Harden and Associates). With the help of EPIC-WQ model developers (Williams; Benson), the vadose zone was divided into five layers, and these layers were added below the topsoil in the EPIC-WQ model. This enabled the EPIC-WQ model to simulate the percolation of nitrate through the vadose zone and into the aquifer.

EPIC-WQ was validated for irrigated cotton by using the data from a nitrogen-phosphorus fertility study on irrigated cotton conducted by the Texas Agricultural Experiment Station in Munday (Kramp). For dryland cotton and dryland wheat, the model was validated by using the crop yield data from the Natural Resource Conservation Service (Seymour Aquifer Hydrologic Unit Project). Results from the U.S. Geological Survey (USGS) well testing (two wells drilled in 1992) in the Gilliland/Truscott segment of the Seymour aquifer were used to validate EPIC-WQ for nitrate percolation to the aquifer. The Gilliland/Truscott segment is small, isolated from the main segment of the Seymour aquifer, and has no intensive agricultural production activities. By using the well logs and by simulating native pasture production for fifty years, nitrate leaching results were obtained from EPIC-WQ and compared with well testing results.[2]

Estimation of Response Functions

Data generated by EPIC-WQ were used to estimate response functions for crop yield and percolated nitrate for irrigated cotton, dryland cotton, and dryland wheat. The independent variables used in estimating these functions were preplant nitrogen fertilizer applications, postplant nitrogen fertilizer applications, irrigation applications during three specified periods, rainfall during the growing season, soil types, and tillage practices.

[2] EPIC-WQ simulation of native pasture scenario for two wells were 6.2 ppm and 5.7 ppm compared with actual well tests of 9.3 ppm and 8.4 ppm, respectively. This suggests that the EPIC-WQ generated value may be slightly lower than actual, but the relationship of one value to another is in the right direction. Furthermore, since the USGS results were obtained by single well tests only (instead of repeated testing of the same site), the discrepancy between the well tests and simulated values does not necessarily reflect any weakness in EPIC-WQ.

The crop yield response functions were estimated with ordinary least squares (OLS). Functional forms considered were required to allow a positive crop yield for a zero input level and to allow the first derivatives to be unrestricted in sign and value. These criteria resulted in the preliminary selection of four functional forms: quadratic, three-halves, square root, and cubic. Using the highest adjusted R^2 and the lowest Schwarz criterion, the quadratic form was chosen for irrigated cotton and dryland wheat, and the three-halves form was chosen for dryland cotton.

Nitrate percolation values obtained from EPIC-WQ composed a *censored* sample since percolation values were often zero, and positive percolation values were observed only under certain climatic events and input levels. Ordinary least squares with censored data produce biased and inconsistent parameter estimates (Maddala; Amemiya). A sample selection model introduced by Heckman was used to estimate the response functions for percolated nitrate. Unlike the tobit model, the Heckman model is better suited for situations where the probability of occurrence of nitrate percolation and the amount of percolation is not intimately related.[3]

The main criterion used for choosing the appropriate functional forms for nitrate percolation functions was the ability to include nitrate leaching even when no production inputs were used. This choice criterion is particularly important for the Seymour aquifer area, where some nitrate contamination may have originated from natural soil nitrates (Harris). Allowing an intercept for the nitrate percolation function resulted in the initial selection of six functional forms: linear, semi-log, cubic, quadratic, square root, and three-halves. The linear form showed a poor fit, suggesting that percolation of nitrate is a nonlinear process. The final selection was made from semi-log, quadratic, cubic, square root, and three-halves. Using the highest adjusted R^2 and the lowest Schwarz criterion, the semi-log (log of dependent variable) form was selected for all three crops.

Risk-Sensitive Farm-Level Modeling

Agricultural production is generally risky, and many farmers exhibit risk-averse behavior, sacrificing income for reduced risk (Lin, Dean, and Moore). Lambert showed that excluding risk gives misleading costs and benefits of regulatory policies. A risk-sensitive farm-level model was developed for a representative 1,500-acre farm to determine the expected-profit–maximizing farm plan with and without environmental policies.[4] The optimization model was static and was based on expected-value–standard deviation utility:

$$MEANINCOME - \phi\left[\sum_k P_k\left\{(DEV_k^+)^2 + (DEV_k^-)^2\right\}\right]^{1/2}.$$

The above objective function maximizes expected net income minus a risk-aversion coefficient (ϕ) times the standard deviation of income.[5] The risk-aversion coefficient (ϕ)

[3] In an agricultural setting influenced by stochastic weather, it is not hard to imagine a scenario where a particular soil or tillage practice might have a lower probability of percolation, but might also have greater average percolation when percolation actually occurs.

[4] The representative farm size was selected after consulting with local extension economists and officials at the Seymour aquifer hydrologic unit project (Bevers; Lamberth).

[5] This expected-value–standard deviation formulation can be considered as a generalized form of expected-value–variance (E-V) and MOTAD (minimization of total absolute deviation) model. By dropping the square root exponent, the model becomes an E-V model. Similarly, by dropping the square root exponent and by not squaring the deviation terms, it becomes a MOTAD model.

can be interpreted as the number of standard errors one wishes to discount income and thus ranges between 0 and 2.5 in most circumstances. The term DEV_k refers to positive and negative deviation of net income from mean net income under the kth state of nature. P_k is the probability of observing any state of nature k (which is one divided by the total number of states of nature). The model maximizes the objective function subject to various site-specific resource constraints, farm program provisions, and environmental policies. These environmental policies include a performance standard, a performance tax, a performance subsidy, an input tax (nitrogen), an input subsidy (nitrogen), and some selected design standards (split application of fertilizer, adoption of minimum tillage, and adoption of both split fertilizer application and minimum tillage).

Results and Discussion

Table 1 reports the mean net income (revenues net of variable cost only), standard deviation of net income, and average per acre nitrate percolation (in ppm) for the baseline and alternative policy scenarios. Farmers are discouraged from planting crops other than the base crop because every acre of base planted in another crop reduces the size of their deficiency payment. With additional constraints, such as limits on percolation of nitrate, the choice of alternative farm plans is narrower. Thus, the model solution was sensitive only at the two extremes of risk behavior considered (risk neutrality and high risk aversion). The results are listed under two categories: risk neutral (RAC = 0) and risk averse (RAC = 2.5).

Farm-Level Implications

The base case, which represents conventional producer behavior in the Seymour aquifer area, provides a benchmark against which the effects of alternative policies can be evaluated. The baseline results replicate the conventional management practices, acreage allocation between crops, and farmer profit for the area (Texas Agricultural Statistics Service; Texas Agricultural Extension Service). The management practices include preplant application of fertilizer, conventional tillage, and irrigation during three specified periods. Other management practices or options available to farmers that were incorporated in the model include split application of fertilizer, minimum tillage, various combinations of crop mix, and allocation of crops to alternative soil.

The primary changes in management practice observed due to a performance standard were a decrease in fertilizer rate applied for all crops, a switch to minimum tillage for irrigated cotton, and a split application of fertilizer for wheat. Irrigation levels did not show any major change under the performance standard, as lower irrigation levels substantially reduced yield and had a negligible effect on the concentration of percolated nitrate. The split fertilizer application in wheat offset some yield decline, although costs for split application were slightly higher. The net income losses under a performance standard were \$6,300 and \$5,200 for the risk-neutral and risk-averse case, respectively.

Percolation can also be targeted with a "pricing" instrument such as a performance tax. The performance taxes that would keep the concentration of percolated nitrate below 10 ppm were estimated as \$0.83 per ppm/acre (risk neutral) and \$0.80 per ppm/acre (risk averse). The optimal farm plan under the performance tax (risk neutral case) was similar to the farm-plan under the performance standard, since an appropriate performance tax imposes the same abatement cost as a standard. The result thus confirms the theoretical proposition

Table 1. Farm-Level Results for Baseline and Alternative Environmental Policies

	Risk Neutral			Risk Averse		
Scenario	Net Income ($)[a]	Standard Deviation ($)[a]	Concentration of Nitrate Percolated into the Aquifer (ppm)	Net Income ($)[a]	Standard Deviation ($)[a]	Concentration of Nitrate Percolated into the Aquifer (ppm)
Baseline	154,600	24,000	15.56	148,400	21,400	14.68
Performance standard	148,300	23,700	9.91	143,200	22,200	9.76
Performance tax	137,200	24,300	9.91	133,100	22,100	9.76
Performance subsidy	155,000	23,600	9.96	149,200	23,000	9.88
Input tax (nitrogen)	124,900	27,300	9.95	111,500	18,800	9.79
Input subsidy (nitrogen)	159,000	23,400	9.95	156,700	21,800	9.83
Split fertilizer application	149,400	23,400	13.23	143,300	21,500	12.84
Minimum tillage	153,700	24,000	13.75	147,600	21,100	13.16
Split fertilizer and minimum tillage	148,700	23,900	12.26	142,500	21,800	11.81

[a]Rounded to the nearest hundred.

that when farms are identical, a performance standard is a quantity dual of a performance tax. Net income is substantially reduced under the performance tax (11.25% and 10.3% for risk-neutral and risk-averse cases, respectively) because of tax payments paid on every unit of percolated nitrate and abatement costs.

A performance subsidy can provide the same incentive as a performance tax when the opportunity cost is the same for a subsidy as it is for a tax. The decision maker can pay taxes by polluting more or can receive a subsidy by polluting less. The policies would differ, however, with respect to distribution of income and political acceptability. Performance subsidies of $0.83 (risk-neutral case) and $0.80 (risk-averse case) per ppm of reduced nitrate percolation were used starting at the baseline percolation level and effective down to 10 ppm. The performance subsidy resulted in the same farm plan as a performance tax for the risk-neutral case. This result supports the theoretical proposition that a performance tax provides the same incentive as a performance subsidy.[6] For the risk-averse case, however,

[6] Some authors have shown that this symmetric effect of tax and subsidy for the risk-neutral case is valid only for a static analysis but not for a dynamic analysis (e.g., Kim et al.).

irrigation was used more intensively on irrigated acres and less fertilizer was used on dryland wheat. The subsidy let the farmer cover the extra cost of additional irrigation applications. Consequently, the net abatement costs of tax and subsidy policies were not identical under risk aversion. As shown by Baumol and Oates, this asymmetric income effect between tax and subsidy occurs because taxes reduce after-tax profits, whereas the subsidy increases postsubsidy profits.

Input taxes are particularly attractive for a situation when actual emissions are unobservable or too costly to monitor. A nitrogen fertilizer tax was used in the model by increasing the price of nitrogen fertilizer so that percolated nitrate into the aquifer was reduced to the EPA standard of 10 ppm. The loss of profit under a nitrogen tax was substantial, since a farmer was forced to make drastic reduction in fertilizer use. For a risk-neutral and a risk-averse farmer, the tax on nitrogen was estimated at $0.60/lb. and $0.72/lb. The tax was approximately 200% to 250% of the purchase price of nitrogen, and it reduced fertilizer use by 30% in irrigated cotton, 29% in dryland cotton, and 54% in dryland wheat. The net income loss was approximately 20% and 25% for the risk-neutral and risk-averse cases, respectively. These results imply that a single input tax would be very costly to producers.

An input subsidy to farmers for using lower fertilizer compared with a baseline is the mirror image of an input tax. Farmers face a similar penalty at the margin for nitrogen use. An input subsidy was used in the model in which a farmer received a subsidy of $0.60/lb. and $0.72/lb. (risk-neutral and risk-averse cases, respectively) for every pound of nitrogen used below the conventional level. Under risk neutrality, the farm mean income was $4,400 higher than the baseline with the same management plan as the nitrogen tax policy. The farm plan under the risk-averse case, however, was slightly different because of the income effect of the subsidy combined with risk behavior.

Design standards specify what actions must or must not be taken by landowners. Three design standards were selected to evaluate the effect on farm income and water quality. These included splitting applications of nitrogen fertilizer, adopting minimum tillage, and, finally, simultaneously adopting minimum tillage and splitting applications of nitrogen fertilizer. With a split fertilizer application, the mean net income was approximately $5,000 lower than the base scenario (for both risk-neutral and risk-averse cases) because of the extra cost of a split fertilizer application and because of postplant nitrogen's lower effect on yield. The income loss under minimum tillage was approximately $1,000, as the slight yield loss under minimum tillage was offset by the reduced cost of tillage. The effect of combined split application of fertilizer and minimum tillage was an income loss of approximately $6,000 for both risk behavior scenarios. In all three cases of design standards, the concentration of nitrate percolated into the aquifer exceeds 10 ppm by approximately 20% to 40%.

Implications for Nonparticipation in Farm Programs

With the emphasis on reducing the federal budget, future government farm program supports are expected to decline. Therefore, the economic and environmental implications of a nonparticipating farmer were considered. Approximately 90–95% of farms in the study area participate in farm programs. Most farmers have established base acres in cotton and wheat through past production history. The effect of nonparticipation was evaluated under a separate scenario in which no farm program constraints were used in the optimization model.

The estimated net income for a risk-neutral and risk-averse nonparticipating farmer was approximately $5,000 and $7,500 lower than the participating risk-neutral and risk-averse farmer, respectively. These income losses would have been larger if the nonparticipating farmer did not have the option of switching from wheat to cotton. Furthermore, a nonparticipating farmer is not required to set aside land and can use all available farm acres. Without base acre requirements, the risk-averse nonparticipating farmer used more irrigated cotton acreage because of lower variability of returns. Fertilization rates were the same as for the participating scenario. Wheat was not chosen because it is less profitable than cotton. The average per acre percolation of nitrate was higher than for the participating case, as the set aside acres are cultivated.

Regional Implications and Comparison of Policy Options

To provide a basis for comparing costs of alternative policies at a regional level, the per acre and farm-level results are extrapolated to the Seymour aquifer region. Of the 274,500 acres in the Seymour aquifer area, approximately 218,200 acres are used for crop production. This figure was derived by subtracting pasture and rangeland acres (25,000), CRP acres (26,300), and other acres (5,000) not used for crop production (Seymour Aquifer HUA Project Annual Report). The Seymour aquifer area is relatively homogeneous with a predominantly sandy soil and similar production practices. Thus, although farms are not truly identical, this aggregation provides some insight into the cost of controlling nitrate percolation for the whole region.

The regional cost estimates are presented in table 2. These costs include net government expenses and revenues as well as private producer welfare changes. These estimates do not include transaction costs such as administration costs, information costs, and enforcement costs. To review the distribution of farmer and government costs or revenues, we first consider a performance standard and a performance tax. When farms are identical (as assumed here), a performance standard is generally viewed as the quantity dual of a performance tax. At equal control levels (10 ppm), both strategies showed identical abatement costs ($976,000 for risk neutral and $820,000 for risk averse), but under a performance tax, the reduction in farmers' net returns was higher ($2,722,000 and $2,391,000 for the two cases) due to an additional $1,746,000 and $1,571,000 tax payment for the risk-neutral and risk-averse case, respectively. Although a performance tax targets the problem more directly, costs of monitoring and enforcement of a performance-based tax would be high. Likewise, while targeting would be relatively easy with a performance standard, enforcing a performance standard depends on farmers' perceptions regarding the links between their behavior and achieving the standard. Because of stochastic weather, difficulty of monitoring, and the confounding role of neighboring polluters, that perceived link may be weak (Braden and Segerson). Thus, targeting and enforcement issues aside, the only noteworthy difference between these policies is the added tax payments under a performance tax. This finding may lend support to the Buchanan and Tullock hypothesis that farmers may prefer a standard to a tax because of the added burden of a tax. However, by judiciously using the redistributed tax revenues, performance taxes can be used to significantly improve environmental quality.

A performance tax scheme can have political support if it is implemented as a subsidy to reduce percolated nitrate from a base level. Some studies show that political acceptability is more important than tax revenues (e.g., Oehmke and Yao). Without political support, a strongly noncooperative farm community could raise the transaction costs so high that any policy to reduce agricultural nonpoint source pollution would be blocked.[7] The enforcement

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[7] Such an idea might encourage farmers to form "water quality cooperatives" where the group as a whole would work toward achieving the standard in order to be eligible for a package of benefits (Braden and Segerson).

Table 2. Regional Implications of Alternative Environmental Policies

	Risk Neutral			Risk Averse		
Policy Instrument	Change in FarmNet Income ($/thsd.)	Change in Farm Net Income (%)	Total Tax Revenue or Govt. Subsidy Payment ($/thsd.)	Change in Farm Net Income ($/thsd.)	Change in Farm Net Income (%)	Total Tax Revenue or Govt. Subsidy Payment ($/thsd.)
Performance standard	(976)	(4.0)	0	(820)	(3.5)	0
Performance tax	(2,722)	(11.2)	1,746	(2,391)	(10.3)	1,571
Performance subsidy	71	0.25	(1,047)	123	0.54	(943)
Input tax (nitrogen)	(4,659)	(19.2)	3,270	(5,794)	(24.8)	4,345
Input subsidy (nitrogen)	702	2.9	(2,091)	1,300	5.6	(2,832)
Split fertilizer application	(816)	(3.3)	0	(801)	(3.5)	0
Minimum tillage	(141)	(0.6)	0	(126)	(0.53)	0
Split fertilizer and minimum tillage	(920)	(4.0)	0	(926)	(3.8)	0

Note: These figures do not include transaction costs.

problems may also be reduced under a subsidy policy by making eligibility for certain benefits contingent on meeting the standard. The estimated abatement cost of a performance subsidy was the same as a performance tax and performance standard, since it provided the same economic incentive to control nitrate percolation at the margin. The abatement cost was recovered through the performance subsidy received, and the net gain was $71,000 to farmers. The subsidy outlay (government costs) was not as high as the tax payment made by the farmers because the subsidy was paid only on the units of percolation reduced (from the base level). The total government subsidy outlay for a performance subsidy policy was estimated to be $1,047,000 and $943,000 for the risk-neutral and risk-averse scenarios.

Since accurate measurement of percolated nitrate is virtually impossible, ambient-based policies discussed above are impractical in an agricultural setting. These estimates, nevertheless, provide important information for policymaking by indicating the costs of such policies. In addition, since optimization model solutions under a performance standard generate management practices which reach the target of 10 ppm, these management practices can be used as a guide to set design standards.

For a nitrogen tax targeted to reduce nitrate percolation to 10 ppm, the estimated farmer net income loss was considerably higher than for all other policies evaluated ($4,659,000 and $5,794,000 for risk neutral and risk averse, respectively). Consequently, government revenues were also relatively large at $3,270,000 and $4,345,000. This high nitrogen tax is also consistent with other studies, which indicate the inelastic demand for fertilizer (e.g., England; Dubgaard). For the same reason, a relatively large nitrogen subsidy was needed to provide an appropriate economic incentive for reducing nitrate percolation. For the risk-neutral and risk-averse scenarios, the net gains in farmer net returns under a nitrogen subsidy policy were $702,000 and $1,300,000, with corresponding government costs of $2,091,000 and $2,832,000.

In addition to high government costs and a large reduction in farmer profit, there are other advantages and disadvantages of a nitrogen subsidy and nitrogen tax policy. A uniform nitrogen tax would be strongly opposed by the agricultural sector because it distorts the optimal mix of production inputs and imposes unnecessary financial burdens on farmers who use nitrogen efficiently. A uniform nitrogen tax focuses exclusively on how much nitrogen is used instead of when, where, and how it is used. To avoid the tax, farmers may attempt to purchase nitrogen from outside the taxed region. However, enforcement and administration of a nitrogen tax would be relatively easy since the tax could be implemented through fertilizer dealers. The taxing authority would be able to avoid dealing directly with each farmer, which would greatly reduce the costs of tax collection and monitoring. Revenues from a nitrogen tax could be used to fund nitrogen reduction education and technical assistance programs, as is done in Iowa, or to compensate those whose welfare has decreased. One option would be to use tax rebates for those who adopt and follow an efficient nitrogen management plan.[8] Unlike a nitrogen tax, political attractiveness is an important advantage of nitrogen subsidy. However, once established, it is extremely difficult to revise or abandon a subsidy policy. Another well-known disadvantage of a subsidy policy is slippage, which has plagued agricultural programs in the past.

The preceding analysis compares the cost of controlling nitrate percolation in terms of farmers' net income loss (or gain) and government revenue (or outlay). Although the net income losses under the three design standards were lower than under other policies, design standards are not included in the comparison because of their inability to reach the target of nitrate percolation (10 ppm). This conclusion does not necessarily imply that policymakers should not consider design standards. Along with ease of enforcement, design standards can be successfully used to minimize NPS pollution once the correlation with water quality has been established for a specific site.

Evaluating the Benefits and Costs of Pollution Control

An efficient strategy of managing water quality must account for both the costs of reducing the pollutants and the benefits of increased quality (Ribaudo). Since markets provide insufficient information about the value of groundwater quality, techniques such as the avoidance cost method (ACM) have been developed to measure lower-bound estimates of benefits. From a public decision-making standpoint, the benefit of groundwater protection can be viewed as damage avoided from groundwater contamination. The avoidance cost method, for example, can generate lower-bound estimates of benefits by using expenditure on bottled water as a proxy for consumer willingness-to-pay to avoid nitrate exposure. One

[8] With a proof of fertilizer purchase and a report of crop acreage confirmed by the county Farm Services Agency office, a farmer could apply for a tax rebate if the quantity of nitrogen used did not exceed a given amount.

key assumption underlying this approach is that averting actions perfectly substitute for pollution reductions.

The cost of bottled water for the region was estimated to be $352,000/year.[9] The least-cost policy alternatives for the region were about $1 million either to farmers or to the local government. This cost is approximately two and one-half to three times the cost of bottled water. When comparing the cost of bottled water to the reduction in farmer net returns, or to the government's cost of a subsidy policy, the least costly way to provide "safe" household water is bottled water. However, assuming pollution of the aquifer is attributable to agricultural practices, there is an opportunity for the "polluters" to compensate those impacted by the externality. For example, a small tax on nitrogen may raise enough funds to compensate the users of bottled water. Assuming 40 lbs./acre of nitrogen fertilizer is used on average, a $0.04 tax/lb. on nitrogen fertilizer would generate approximately $349,000/year, which is the approximate consumer expenditure on bottled water.[10] Although this policy does not exhibit the purity of instruments such as a performance tax or a tax on multiple inputs, administration and enforcement costs are expected to be much lower. However, issues such as farmers' purchase of nitrogen outside the taxed region, nonresidents taking advantage of the subsidized bottled water, and the mechanism for taxing and distributing the tax dollars to residents must be given serious consideration.

The avoidance cost method ignores aspects such as ecological damages (plants and wildlife affected in the ecosystem) or nonuse values. Furthermore, in estimating the potential averting costs, it was assumed that individuals in the area who do not have access to municipal water would use bottled water. This assumption ignores issues such as accessibility to bottled water, availability of information about health risk, and presence of children in the household. Despite these limits, a measure of potential averting costs is an important part of the information needed for policymaking.

Conclusions

This study has investigated alternative policies to attain a target for nitrate percolation for a representative farm in the Seymour aquifer region of Texas. A performance standard reduced farm profit by approximately $1 million, while a performance tax reduced farmer profit by approximately $2.7 million and added $1.7 million to government revenues. A performance subsidy resulted in a government outlay of approximately $1 million. A nitrogen tax or subsidy involved relatively large costs to either the farmer or the government. The overall implication is that there is approximately a $1 million annual cost to either the farm or the local government to meet the EPA standard of 10 ppm. This comparison is made in terms of farmers' net income loss and achieving the environmental target of 10 ppm.

The results of this research are not meant to be a recommendation. In a farm-level study, the predicted effects of a policy are conditional on the characteristics of the specific site modeled. While such efforts are useful, particularly for a relatively small and homogeneous area such as the Seymour aquifer, predictions of aggregate effects must be made with caution. It is also worthwhile to reemphasize that since transaction cost and political viability

[9] This figure was derived by collecting information on the number of households that do not have access to municipal water, expected average consumption of bottled water by a household, and price of bottled water (Moore; Fitzgerald; Scholz). The information was provided by chamber of commerce, north-central Texas municipal authority, and Jacobs Well, a water bottling company.

[10] (218,200 acres)(40 lbs.)($0.04) = $349,120.

may dramatically change the ranking of policy instruments in terms of cost effectiveness, there is a need for caution in recommending the cost advantages of a particular policy. A final note is in order regarding the use and interpretation of avoidance cost estimates. Avoidance costs are not a substitute for willingness-to-pay estimates, and such substitution may lead to misinterpretation of results. Thus, whenever possible, information about the full benefits should be obtained through other valuation methods and compared with avoidance costs.

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