Economic and Water Quality Impacts of Reducing Nitrogen and Pesticide Use in Agriculture

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A multiyear regional risk programming model was used in evaluating the impacts of different environmental policies on cropping systems, input use, nonpoint source pollution, farm income, and risk. A direct expected utility maximizing problem (DEMP) objective with a Von Neuman Morgenstern utility function was used in deriving optimal cropping systems. A biophysical simulation model provided input for the optimization. Three types of policies—taxing, regulating the aggregate, and regulating the per acre level—were studied for two farm inputs—nitrogen and atrazine. It was observed that policies had varied and multiple cross-effects on pollutant loads, farm income, and risk. This information is crucial in developing successful policies toward improving water quality. If an appropriate input policy is chosen, both targeted and nontargeted pollutant loads can be managed. The three policies varied in their effects on pollutant loads and involved tradeoffs in water quality and economic attributes.

Agriculture remains a major source of nonpoint source (NPS) pollution. Erwin (1988) estimated that NPS pollution deteriorates water quality in 64% of rivers and 57% of lakes in the United States. Annual loss through sediment alone has been estimated to be 2.2 billion dollars in this country (Clark, Haverkamp, and Chapman 1985). Chemical contaminants like fertilizer and pesticide residues from agricultural lands reduce the quality of surface and ground water at both on-site and off-site locations. Protecting water quality from these externalities is an important public policy issue. Policies to protect water quality can be classified as either broad-based or micro-targeted, according to their level of administration. Broad-based policies include Pigouvian taxes (Baumol and Oates 1988), per acre and aggregate input use regulation (Mapp et al. 1994), restricting per acre and aggregate emission levels, permit trading (Baumol and Oates 1988), and land retirement (Young and Osborn 1990). The micro-targeted policies concentrate on specific soils, cropping systems, irrigation systems, or locations within a watershed (Mapp et al. 1994; Braden et al. 1989; Lovejoy, Lee, and Beasley 1985). These policies often involve varying economic and environmental implications that need careful analysis in addressing the problem of water quality deterioration. For example, direct policies such as taxes on fertilizer are easier to implement and enforce within existing input markets. However, less information exists on their economic and water quality implications in a risk programming framework.

Policies like taxing and regulating input use, in addition to reducing input use, have spillover effects on other forms of agricultural nonpoint source pollutant loads, income, and risk levels. In other words, those policies that are directed toward a particular input (e.g., nitrogen) have a direct effect on the targeted pollutant,[1] an indirect effect on nontargeted contaminants (phosphorus, sediments, etc.), and an indirect effect on economic attributes. The economic variables that are affected by such policies include income and financial risk. Disaggregating effects on mean and variance of income can provide better information on policy implications. The spillover effects on other nonpoint source pollutants are not clear in the current literature. Policies resulting from an assumption of less

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[1] We define targeted pollutants with respect to a particular policy as representing the chemical derivatives of the input directly affected by the policy instrument; that is, in the case of restricting nitrogen use, targeted pollutants are organic nitrogen, nitrates in surface and subsurface water, and nitrogen pollutants in percolate leaving the two-meter soil profile. All other nonnitrogenous pollutants are termed nontargeted pollutants under this policy instrument.

or nonexistent spillover effects can result in suboptimal levels of emission reduction or ineffective economic policies. Studying the cross-effects of policies aimed at reducing agricultural NPS pollution can provide better information for local, state, and federal agencies charged with improving surface water.

In the formulating of appropriate water quality policies, information deficiency is identifiable at both the policy and methodological levels. On the policy side, the implications of policies on nontargeted pollutants and farm risk have not always been considered. Interpretation of information on the nature of simultaneous adjustments in cropping sequences, input use levels, substitution in cropping systems, profits, and risk facing the producers requires a holistic framework. On the modeling side, the use of a direct expected utility risk programming framework to optimize cropping systems over time can capture producer behavior, which can lead to better estimation of policy implications on producer risk and water quality parameters. This is because of the model's lower reliance on assumptions underlying the distribution of stochastic variables. Also at the modeling level, an understanding of the cross-effects of water quality policies is important for successful policy design. This study attempts to fill some of these voids in the water quality literature.

This study uses a multiyear risk programming model and a biophysical simulator to investigate the economic and environmental implications of taxing and regulating farm inputs directed toward reducing nonpoint source pollution from agricultural lands. Specifically, we study the effect of selected environmental policies on crop production decisions, input use, water quality, soil erosion, producer income, and farm risk in the White River Basin in Central Indiana. A direct expected utility maximization formulation was used to model producer behavior. The economic and environmental impacts of taxing nitrogen and pesticide use and restricting their use at the aggregate (entire watershed) and per acre levels were analyzed. Responses of targeted and nontargeted pollutants, farm income, and risk (variance in income) to various policy instruments were assessed at varying policy intensities.

Background Literature

Water quality studies can be classified based on (1) the level of application (watershed level or farm level), (2) the level of targeting (broad-based or micro-targeting), (3) the incorporation of risk (stochastic or deterministic), and (4) the method of estimating pollutant loading (fixed delivery ratio or variable estimation). Several approaches have been used to study the impact of agricultural practices on water quality. In particular, process simulation models have been used extensively to assess policy impacts on water quality. For example, Lovejoy, Lee, and Beasley (1985) used ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulator) to evaluate the cost-effectiveness of several micro-targeting policies to reduce sediment yields in an Indiana watershed. Likewise, Braden et al. (1989) used the SEDEC (SEDiment EConomics) model to evaluate optimal spatial management of sediment in an Illinois watershed. Phillips et al. (1993) used the EPIC (Erosion Productivity Impact Calculator) model to study the responses of soil erosion and exports to several tillage and crop rotation practices in Illinois. Finally, Kozloff, Taff, and Wang (1992) used the AGNPS (Agricultural NonPoint Source pollution) model to study the effectiveness of targeting schemes with respect to budget outlays and sediment yields.

Other studies have combined simulation models with mathematical programming routines to address water quality issues. For example, Mapp et al. (1994) used EPIC-PST along with a mathematical programming model to study the impact of limiting nitrogen use at various levels on the Central High Plains. Teague, Bernardo, and Mapp (1995) also used EPIC-PST along with a farm-level Target-MOTAD model to evaluate income and environmental tradeoffs. Taylor, Adams, and Miller (1992), used linear programming along with a biophysical simulator to study the effect of economic incentives to offset nonpoint source pollution. None of the above studies used the Direct Expected Utility Maximizing (DEMP) framework of risk analysis. This study estimates economic and water quality implications of policy using a risk programming framework to evaluate alternative policies related to nitrogen and pesticide use in agriculture.

The Model

Farmers in the watershed are assumed to maximize expected utility of wealth under a stochastic environment defined in equation (1):

$$\max_{x_s} EU(W_s), \tag{1}$$

where U(.) is a Von Neuman Morgenstern utility function, which is concave, continuous, and twice differentiable. W_s is the wealth in state s ($s = 1, \ldots, q$) and $EU(.)$ is the expected utility over states. Wealth in state s is determined by equation (2), where $R(\mathbf{x}_s)$ is net return in state s, $\mathbf{x}_s$ is a vector of activities in state s, and W_0 is the initial wealth.

$$(2) \qquad W_s = W_0 + R(\mathbf{x}_s) \; \forall \; s = \ldots q.$$

The resources allocated to activities are limited to an endowment **B** as an inequality constraint in equation (3):

$$(3) \qquad \mathbf{A}\,\mathbf{x} \leq \mathbf{B}$$

where, **A** is the technology matrix. The income in each state is net profit from production activities calculated as in equation(4):

$$(4) \qquad R(\mathbf{x}_s) = \sum_{\mathbf{x}_s} (c_s\,\mathbf{x}_s - r_s p_s),$$

where, c_s represents the product prices and r_s is the per unit cost in inputs of practice (p) under state s. Biophysical and production processes involved in the system are represented by equation (5), where M_s represents the production environment facing the farmer (weather, soil conditions, etc.), and $\vartheta(.)$ is the relationship involved in activity $\mathbf{x}_s$:

$$(5) \qquad \mathbf{x}_s = \vartheta\,(p_s, M_s, \mathbf{B}).$$

The nonpoint source pollution loading (N) is generated according to equation (6), where $\varphi(.)$ is an emission function for pollutants by farm activities:

$$(6) \qquad N_s = \varphi\,(p_s, M_s).$$

The farm decision-making framework (represented by equations [1] through [6]) is used to study the impacts of selected nonpoint source pollution policies. For the sake of clarity, this framework is disaggregated into two subsystems: the economic decision-making of producers (equations [1] to [4] and nonnegativity constraints) and the biophysical processes (equations [5] and [6]).

Methodology

To represent the two subsystems in equations (1) to (6) of the conceptual model, we use a nonlinear mathematical programming formulation combined with a simulation model. A DEMP (Direct Expected Utility Maximization Problem) formulation was used to represent equation (1). This formulation has fewer restrictions on the form of utility function and the assumptions regarding the distribution of the random variables (Lambert and McCarl 1985) compared with the Expected Income-Variance (E-V) representation. The DEMP formulation is of the form:

$$\underset{x_s}{MAX} = \sum_{s=1}^{q} \Pi_s \, U(W_s),$$

where Π_s is the probability of occurrence of the s^{th} state. A negative exponential utility function

$$(W_s) = -\,e^{-\rho W} \text{ for } \rho > 0$$

was used in the study, because it represents constant absolute risk aversion (CARA) over changes in wealth. The term ρ is the risk aversion parameter.

The activities in the model included crop sequences (continuous corn, corn-soybean, corn-alfalfa, and corn-soybean-wheat), input-use levels (low input use, medium input use, and high input use), and tillage (conventional tillage, minimum tillage, and no tillage). A biophysical simulator, the Erosion Productivity Impact Calculator (EPIC/WQ[2]) (Williams et al. 1989), was used to simulate the crop yields and other biophysical processes in equations (5) and (6). Validation of the yields for each cropping systems (involving corn, soybean, wheat, and alfalfa crops) was done for various input levels and planting dates. Calibration and validation of the results of the EPIC model follow the procedures used by Foltz et al. (1995). The predicted yields were validated by comparing them with the actual yields of the region using a regression analysis. The simulated yields were found to be reasonable in predicting mean crop yields. The simulated yields were further validated using expert opinion of agronomists (Foltz et al. 1995).

To capture stochastic effects of weather and time, a set of twenty-five ten-year simulations was performed for each cropping system to develop empirical crop yield distribution by cropping system, input level, and tillage practice. Apart from the yield data of each activity, nonpoint source pollutants, including sediment, fertilizer and pesticide loadings in runoff water, and chemical contaminants attached to sediments, were estimated with EPIC/WQ. Simulation of cropping systems was repeated for alternate placement of crops in each sequence, that is, if the sequence had corn and soybean, the simulation was done for a sequence starting with corn and one starting with soybean

[2] EPIC/WQ is an extension of EPIC that includes water quality information. This model simulates the effects of soil, climate, conservation practices, and crop rotations on soil erosion and other pollutants including pesticides. This model was chosen for its capability to simulate multiyear/multicrop rotations, ability to provide estimates of environmental flows, computational efficiency, user convenience, and accessibility to model builders.

for each weather seed. A set of constraints on per acre input use was added to the mathematical model to perform policy simulation. Restriction on aggregate loading of pollutants was also included by using a set of equations constraining the total pollutant loads leaving the system. Background information on land use and cropped area for counties in the watershed was obtained from the 1992 agricultural census, and the watershed was delineated using GIS (Geographical Information Systems) with GRASS 4.1.[3]

The revenues for each ten-year crop sequence were calculated using historical price data published by USDA in *Agricultural Prices*. The net margins were discounted using the "All crop price index (prices received)" for Indiana, with 1980 as the base. A complete mathematical system representing the problem was programmed using the GAMS (General Algebraic Modeling System) language developed by Brooke, Kendrik, and Meeraus (1988). The risk aversion coefficient was parameterized to test its influence on crop plans, but the policy analysis was conducted using a single risk aversion parameter.

The mathematical model integrates both crop and water quality information into a direct expected utility maximization model. This model was used to derive baseline crop production activities and corresponding pollutant loads. The results of the base run were validated for the watershed using data from *Indiana Agricultural Statistics* (USDA and Purdue University 1992, various volumes) aggregated over counties, and an area study in the region conducted by USDA/ERS/Resources and Technology Division, (1993). To accomplish this validation, the base plans were disaggregated according to different crops, input use, and tillage. These were then area-weighted to derive allocation under each in the optimal plan. These area allocations were used in comparison to evaluate the validity of the base plan for policy analysis. The base plan performed well in representing the agricultural scenario of the region and was comparable to area studies and regional agricultural statistics. The deviation in percentage allocation in base plan compared with area statistics and statistics was calculated within 5% for the area under individual crops.

Mean income, variance in income, and pollution loading were calculated using accounting equations on the optimal variables. The impact of policies on agricultural income and water quality in the watershed was studied by perturbing the system using taxing and regulatory policies to study resulting changes. We disaggregated the income effects into both the mean and variance to add information on the impact of policies. The impacts were measured as percentage changes from the baseline. Policy instruments evaluated in this study fell into three categories: (1) taxing: increase in per unit price of inputs, (2) restricting input use at the aggregate level, and (3) restricting input use at the per acre level. Thus, a total of six policy scenarios (three types on two inputs, nitrogen and atrazine) were considered in this study. The taxing policy attempts to reach a desirable outcome in water quality through its effect on farm profits, while the aggregate restriction on inputs at the watershed level invokes constraints in allowable rates of pollution for the entire area. The per acre restriction limits the producers in their choice of technologies that do not violate limits on the per acre input use. The implications of policy intensities at various levels were evaluated by parameterizing the DEMP model.

Results and Discussion

Estimates from the EPIC/WQ model (table 1), show that organic nitrogen in runoff was lowest, compared with all other farming systems, under a corn-alfalfa system with low input use. Nitrates in the runoff water were lowest (2.68 lbs. per acre) in continuous corn under no till with low input use, in contrast to corn-soybean-wheat under no till with high input use (13.38 lbs.). Nitrogen pollution in both subsurface and percolate forms was highest in corn-alfalfa rotation with high input use. Exposure to higher levels of nitrates in groundwater could be a cost associated with increased alfalfa acreage in the region (Foltz, Lee, and Martin 1993). Phosphorus contaminants were relatively high under a corn-alfalfa system with high input use. Sediment loading was lowest in a corn-alfalfa rotation with low input use (conventional till) and highest under corn-soybean-wheat with medium input use (minimum till).

The multiyear cropping and pollution loading information was used in the farm decision problem. It was observed that under risk neutral preferences, the optimal solution generated specialization in corn-soybean rotation with medium level of input use and no tillage (CS-MF-NT). This is because the ranking is based on mean income (without considering the risk involved), and CS-MF-NT had the highest mean among cropping systems.

The results of the baseline run (100-acre scale)

[3] GRASS (Geographic Resource Analysis Support System) 4.1 is a GIS software developed by the U.S. Army Corps of Engineers.

Table 1. Per Acre Cropping System Simulation Results

Cropping System[1]	I[2]	ON[3]	NR[4]	NSS[5]	NP[6]	PR[7]	PS[8]	SL[9]
CC-LF-CT	264	20.98	2.68	2.68	4.46	31.22	3.01	0.99
CC-LF-MT	263	17.59	2.68	2.68	4.46	65.12	2.68	0.97
CC-LF-NT	262	17.39	2.68	2.68	5.35	93.66	5.47	0.97
CC-MF-CT	358	21.10	3.57	4.46	7.14	41.03	3.03	1.00
CC-MF-MT	358	17.70	3.57	4.46	8.03	92.77	2.86	0.97
CC-MF-NT	358	17.55	3.57	4.46	8.03	137.37	3.07	0.95
CC-HF-CT	365	21.35	4.46	7.14	16.95	51.74	3.27	1.05
CC-HF-MT	367	17.84	4.46	6.25	17.85	122.21	3.08	0.96
CC-HF-NT	371	17.83	4.46	6.25	17.84	183.76	3.38	0.93
CS-LF-CT	361	20.78	5.35	4.46	16.06	24.09	3.02	1.04
CS-LF-MT	364	17.12	5.35	4.46	18.73	53.52	2.70	0.97
CS-LF-NT	360	16.48	4.46	4.46	18.73	80.29	2.77	0.96
CS-MF-CT	370	20.81	6.24	5.35	24.98	32.11	3.16	1.07
CS-MF-MT	373	17.16	5.35	5.35	27.65	67.80	2.84	0.97
CS-MF-NT	377	16.53	5.35	5.35	28.55	102.59	2.94	0.94
CS-HF-CT	365	20.82	6.24	7.14	33.01	33.01	3.18	1.11
CS-HF-MT	368	17.20	5.35	6.25	36.57	81.18	2.96	0.97
CS-HF-NT	374	16.56	5.35	6.25	38.36	125.78	3.12	0.94
CA-LF-CT	301	8.36	7.14	8.92	62.44	415.69	2.42	0.48
CA-LF-MT	307	7.47	8.03	8.03	62.44	442.45	2.37	0.42
CA-LF-NT	314	7.36	8.92	8.92	62.44	472.78	2.44	0.41
CA-MF-CT	342	8.30	7.14	8.92	65.12	624.43	3.06	0.48
CA-MF-MT	343	7.46	8.02	8.92	64.23	666.36	3.07	0.42
CA-MF-NT	349	7.37	9.81	8.92	65.12	713.64	3.20	0.41
CA-HF-CT	341	8.31	7.14	8.92	67.80	832.28	3.72	0.48
CA-HF-MT	347	7.47	8.03	8.92	66.90	890.26	3.77	0.42
CA-HF-NT	355	7.38	9.81	8.92	67.80	953.60	3.96	0.41
CW-LF-CT	268	22.48	8.92	7.14	40.14	32.11	3.33	1.52
CW-LF-MT	272	17.51	9.81	7.14	40.14	79.39	3.00	1.11
CW-LF-NT	276	16.28	9.81	7.14	40.14	118.64	3.00	1.00
CW-MF-CT	316	22.35	10.70	8.03	41.03	44.60	3.47	1.51
CW-MF-MT	315	17.48	11.60	8.03	41.03	105.26	3.18	1.11
CW-MF-NT	318	16.30	11.60	8.03	41.03	161.46	3.27	1.00
CW-HF-CT	321	22.35	12.49	8.92	41.93	52.63	3.57	1.50
CW-HF-MT	325	17.53	13.38	8.03	41.93	130.24	3.39	1.11
CW-HF-NT	331	16.35	13.38	8.03	41.93	205.17	3.57	1.00

[1]CC: continuous corn. CS: corn-soybean. CA: corn-alfalfa. CW: corn-soybean-wheat. LF: low fertilizer level. MF: medium fertilizer level. HF: high fertilizer level. MT: minimum tillage. CT: conventional tillage. NT: no tillage.
[2]I: discounted income stream ($/ac).
[3]ON: organic nitrogen (lb/ac).
[4]NR: nitrogen in run-off (lb/ac).
[5]NSS: nitrogen in subsurface flow (lb/ac).
[6]NP: nitrogen in percolate (lb/ac).
[7]PR: phosphorus in runoff (lb/ac).
[8]PS: phosphorus with sediment (lb/ac).
[9]SL: soil loss (t/ac).

under a relative risk aversion coefficient of one (table 2) showed diversification in cropping activities. Increasing the value of the risk aversion coefficient resulted in lesser allocation to cropping systems with higher risk and further diversification in optimal plans. The optimal plan under relative risk aversion of one was more representative of the study area and hence was used as a baseline run for policy analysis. A comparison with the results under risk neutrality explains the classical financial response to risk through movement along the income-risk frontier. The baseline results generated area-weighted estimates of 55% of corn, 38.82% of soybean, 4.39% of alfalfa, and 0.46% of wheat acreage. The average annual nitrogen use per acre was 124.06 lbs., while phosphorus and potassium use were 56.97 lbs. and 91.76 lbs., respectively. Nearly 38.09, 14.23, and 21.75% of the land were under no till, minimum till, and conventional till, respectively, in the baseline results.

The economic and water quality implications of the baseline plan are also evaluated. An estimated 16.46 lbs. of organic nitrogen pollutants per acre was predicted in the runoff water. Nitrate loss in

Table 2. Percent of Allocation under Baseline Scenario of Optimization

Cropping System[1]	Allocation	Cropping System	Allocation
CC-MF-CT	1.9138	CS-HF-CT	2.2428
CC-MF-MT	1.9195	CS-HF-MT	2.3873
CC-MF-NT	1.8857	CS-HF-NT	2.6401
CC-HF-CT	2.2222	CA-MF-CT	1.1617
CC-HF-MT	2.3332	CA-MF-MT	1.1843
CC-HF-NT	2.4971	CA-MF-NT	1.4627
CS-LF-CT	2.0558	CA-HF-CT	1.0950
CS-LF-MT	2.1958	CA-HF-MT	1.3816
CS-LF-NT	1.9776	CA-HF-NT	1.7663
CS-MF-CT	2.4555	CW-HF-CT	0.1771
CS-MF-MT	2.6270	CW-HF-MT	0.3668
CS-MF-NT	59.4357	CW-HF-NT	0.6153

[1]CC: continuous corn, CS: corn-soybean, CA: corn-alfalfa, CW: corn-soybean-wheat, LF: low fertilizer level, MF: medium fertilizer level, HF: high fertilizer level, MT: minimum tillage, CT: conventional tillage, NT: no tillage.

subsurface flows was 5.55 lbs. per acre, and loss of the mineral nitrogen was 5.73 lbs. per acre. The nitrogen loss through percolation below the crop root zone was 29.55 lbs. per acre. The soluble phosphorus in runoff was estimated at 152.22 lbs. and the phosphorus attached to the sediment was 3.02 lbs. per acre. Sediment generated in the baseline, as estimated by USLE (Universal Soil Loss Equation), was 0.55 tons per acre. Atrazine loading was estimated at 0.192 gms per acre, while the alachlor level was 0.233 gms per acre. The discounted mean of income stream of the farms was $370 per acre, with a variance of $210 per acre.

The implications of policies span dimensions in tradeoffs among various forms of pollutants, targeted and nontargeted contaminants, agricultural income, and income risk (variance income). To enable a clear understanding of these complexities, a graphical representation of the impacts is presented for each policy scenario. By graphing the multidimensional changes into deviational changes from the baseline, the impacts of various policies can be compared. The *X* axis indicates the level of an increment in policy intensity compared with the baseline. For instance, in the case of a tax on nitrogen, at x = 2, the policy intensity is calculated as 100* (2 + 1) = 300% increase in nitrogen price from the baseline. The changes in water quality and economic attributes are represented on the *Y* axis as percentage deviations from the baseline.

Nitrogen-targeted policies

The implications of nitrogen-targeted policies on agricultural income and water quality are presented in figures 1 and 2. The impacts (in percentage terms) are developed under taxing, and restrictions are depicted graphically for water quality and economic implications.

Taxing nitrogen. Increasing the price of nitrogen reduced the nitrogen pollutants in surface and subsurface water to a maximum of 2% before substantial income loses were observed (figure 1). To reduce nitrogen pollutants by 1%, a nitrogen tax of 400% from the baseline was necessary. Reduction in sediment is responsive to a twofold increase in nitrogen price. The sediment reduction is substantial (a maximum of 16%) with a ninefold increase in nitrogen tax level. The pesticide contaminants declined at a 400% tax level, with a maximum reduction of only 5 to 6% of contamination. One notable observation is that nitrogen taxing increased the nitrogen contamination in groundwater, due to shifts in cropping systems toward alfalfa-based cropping, which uses less nitrogen but has higher estimates of percolate nitrogen. The nitrogen tax also resulted in a slight increase in total phosphorus (runoff and sediment) pollution, due to an increase in the area under a cropping system with lesser nitrogen use and phosphorus pollution than the baseline levels.

The economic impacts of this policy are presented in the lower panel of figure 2. As expected, the use of nitrogen declined at a constant rate for each increase in nitrogen taxation up to a 600% level. A maximum achievable level of reduction was roughly 20% from the baseline, with an elevenfold increase in nitrogen price. While atrazine and alachlor use declined similarly to nitrogen use, the rate of decline in atrazine use was higher than that of alachlor use in percentage terms. The use of phosphorus declined very gradually until a maximum reduction of 1.2% from the baseline was reached. The loss in income was almost linear in nitrogen price rise. Risk (measured as variance in income) faced by the farmers decreased because of a shift in optimal cropping mix with a significant drop after the 600% level of tax, until a maximum decrease of 10% was achieved.

Increasing the tax on nitrogen reduced surface sources of the pollutant but resulted in an increase of nitrogen contamination of groundwater. Nontargeted pollutants like pesticides and sediments were responsive to nitrogen taxing. Because farm income declined and groundwater nitrogen contamination increased under nitrogen taxing, an optimal level of nitrogen taxes can be fixed after accounting for the tradeoffs in water quality benefits.

Aggregate nitrogen use restriction. Aggregate restrictions on nitrogen use showed varying effects on water quality and are presented in figure 2. With

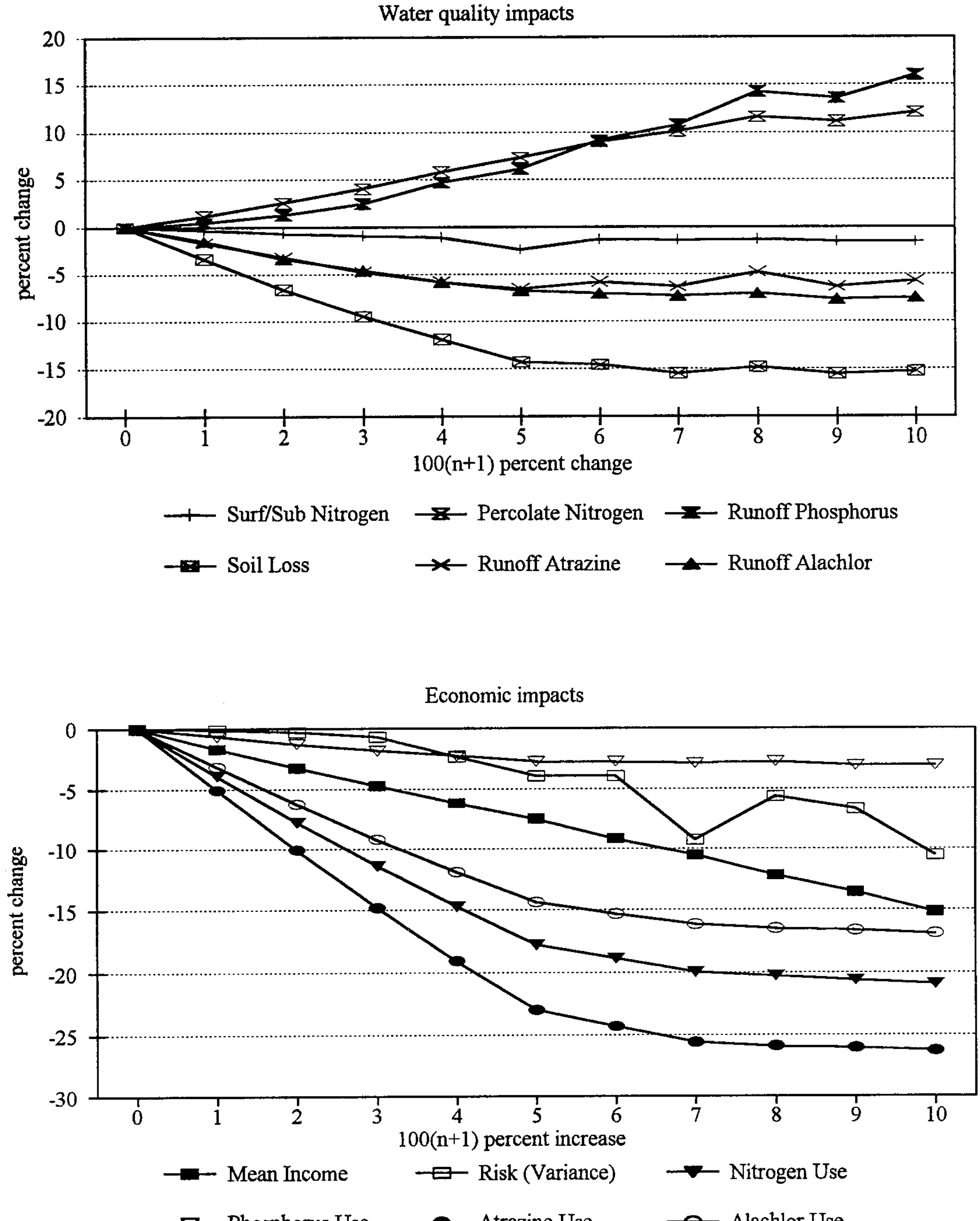

Figure 1. Economic and Water Quality Implications of Nitrogen Tax.

a 40% restriction in nitrogen use, the groundwater nitrogen contamination reduced to a maximum level of 30%. The levels of these contaminants increased for restriction above 30% from the baseline. Restricting above 20% of nitrogen use from the baseline level reduced pesticide pollution in runoff water. Response of total phosphorus was similar to that of nitrogen in groundwater but differed in magnitude. The sediment in runoff water was reduced considerably for restriction levels above 20%. The surface and subsurface nitrogen contaminants remain unchanged until a 40% re-

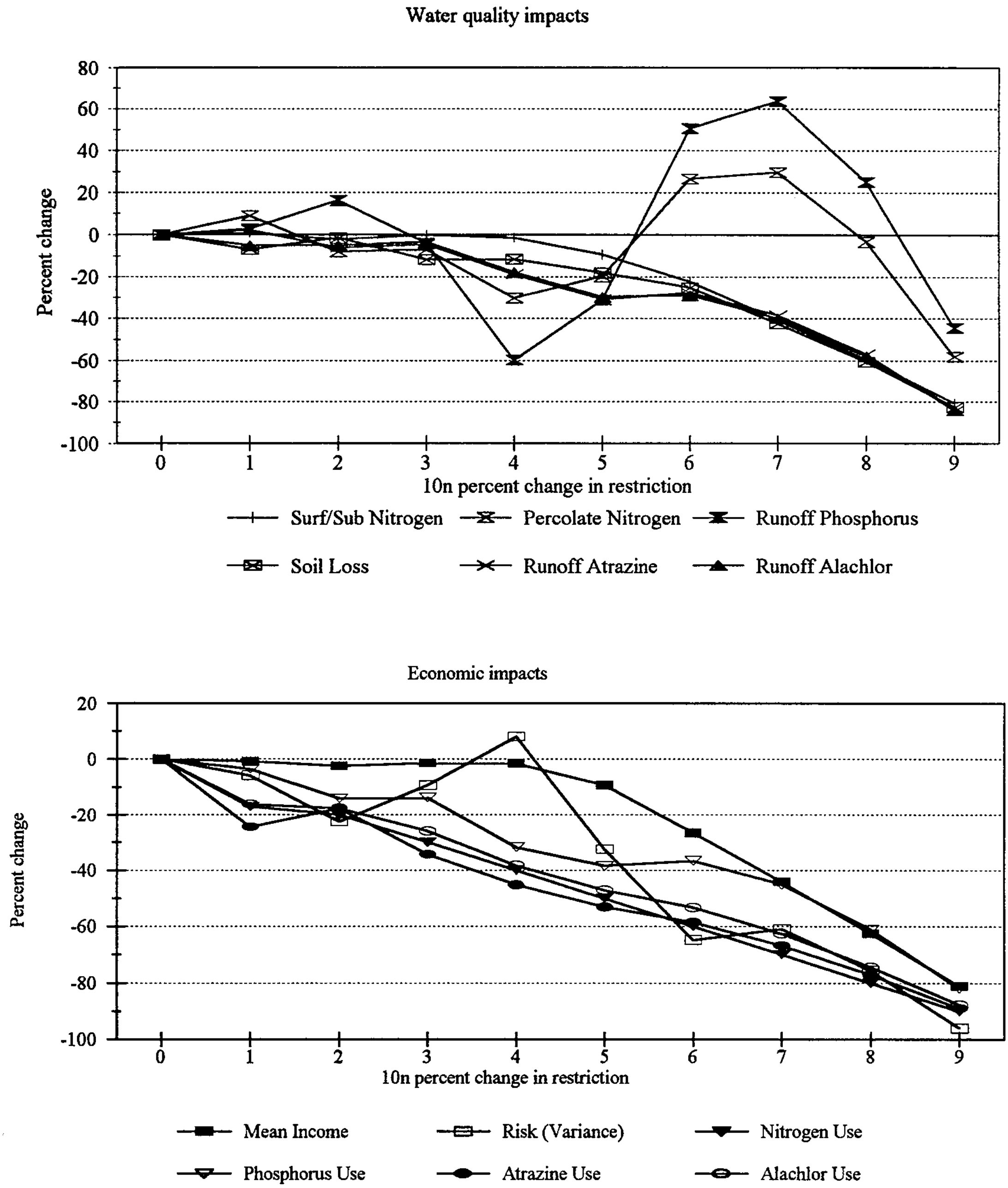

Figure 2. Economic and Water Quality Implications of Restricting Aggregate Nitrogen Use.

striction in nitrogen use occurred. Restriction of nitrogen use in aggregate had a significant impact on use of all inputs. The use of phosphorus and pesticides decreased under this policy. The effect on farm income was not significant for restrictions less than 40%, after which the income loss was substantial. Agricultural income risk increased by 10% for a 40% increased restriction and declined for increases above 50% in nitrogen use.

The aggregate restriction in nitrogen use above 40% was detrimental to agricultural income. Reduction of agricultural NPS pollution can be achieved with restriction in nitrogen use up to 40%. For restrictions higher than 40%, a rise in nitrogen contamination of groundwater and in phosphorus levels of surface water was predicted.

Per acre nitrogen use restriction. Constraining per acre use of nitrogen restricts entry into the optimal cropping plans of certain technologies using high amounts of nitrogen fertilizer. The per acre restriction levels had varying effects on water quality. While not shown graphically, large shifts in cropping systems occurred for restrictions of 6, 30, and 54% from the baseline nitrogen use per acre. At these levels, changes in technologies and shifts in cropping systems were observed, while water quality improved substantially from the baseline. Under all levels of restriction, a reduction of 32% of phosphorus loading was observed. Estimated soil loss declined by 13% from the baseline for a 6% restriction level and by 16% for both 30 and 54% restriction levels. Pesticide contaminants declined by more than 10% for a 6% restriction in nitrogen use. The reduction in nitrogen contamination of surface and subsurface water (targeted pollution) was less effective under this policy type than under the aggregate restriction scenario. Significant drops in total nitrogen use by 5%, in phosphorus by 15%, and in pesticides by 20% were observed for a 6% restriction from the baseline. This 6% restriction in nitrogen use per acre reduced the choice set of available cropping systems and resulted in a 5% drop in total nitrogen used in the entire farm. Though the mean income of producers was not severely affected, the risk in farm income increased substantially for all per acre restrictions.

To summarize, only certain levels of per acre restrictions were effective in protecting water quality. The most effective restriction level was 30% from the baseline application rates. The per acre policy was effective in reducing general deterioration of water quality only for a certain percentage reduction. Similar results were obtained by Mapp et al. (1994) in the Central High Plains. Under the per acre restriction scenario, risk in income was greatest because of limited adjustments in optimal cropping systems.

Pesticide-targeted policies

The pesticide-targeted policies were applied to atrazine in this study. The three policies on pesticides were similar in magnitude to those of nitrogen-targeted policies.

Taxing atrazine. Taxing atrazine pesticide showed little response for levels below 500% from the baseline. Taxing beyond this level resulted in reductions in levels of phosphorus, pesticides, and sediment loading (figure 3). The reduction in sediments in runoff was constant for each increase in tax above 600% in price level. The alachlor level in water dropped by 4% from the baseline level for an elevenfold increase in tax, while atrazine pollutants declined at an eightfold tax. Nitrogen pollution in groundwater increased from the baseline for each increase in atrazine tax. Surface and subsurface nitrogen was unaffected by atrazine taxes for 500% increases, after which these contaminants increased.

The effect of atrazine taxes on input use, agricultural income, and risk are presented in the lower panel of figure 3. The use of atrazine and alachlor declined steadily for all levels of taxing. Nitrogen use declined by 2% for a 100% increase in taxing. The use of phosphorus was less affected (2%) by this tax. Farm income declined by approximately a 5% level at all tax levels. The risk increased from the baseline, excepting for a 600% level.

In summary, most of the water pollutants can be effectively reduced with taxing up to a 500% increase from the baseline. An atrazine tax policy increased nitrogen contaminants in groundwater because of shifts in cropping systems involving higher percolate nitrogen. Risk level was at the minimum at a 500% atrazine tax. Pollutant reduction was maximum at a 500% tax level. The reduction in financial risk is due to increasing allocation of a cropping system that uses less atrazine but has lower financial risk and income.

Aggregate atrazine use restriction. The impact on water quality was within 20% of the baseline for restrictions up to 40% from the baseline (figure 4). The phosphorus pollutants declined initially to 13% and increased to a high of 75% at a 70% level of restriction. It was possible to reduce pesticide contamination with higher than 15% restriction in use. Estimated soil sediments declined at most restriction levels, except for 60 and 80%. The increase at these levels is due to adjustment in crop plans with the entry of a new cropping system. Surface and subsurface nitrogen pollutants declined for restrictions above 40%. As expected, pesticide use declined at all levels of restriction. Most of the pollutants declined from their baseline for restrictions above a 20% level. A significant drop in income was observed with restrictions higher than 20% from the baseline. The risk in income dropped, as did mean farm income, with restrictions higher than 20% from the baseline. This was due to entry of low-income cropping systems with lower risk.

Restricting atrazine use on the aggregate reduced the use of most farm chemicals. Pesticide contamination in both surface and groundwater dropped for restrictions greater than 10%. However, phosphorus and nitrogen loadings to groundwater increased to significant levels under this

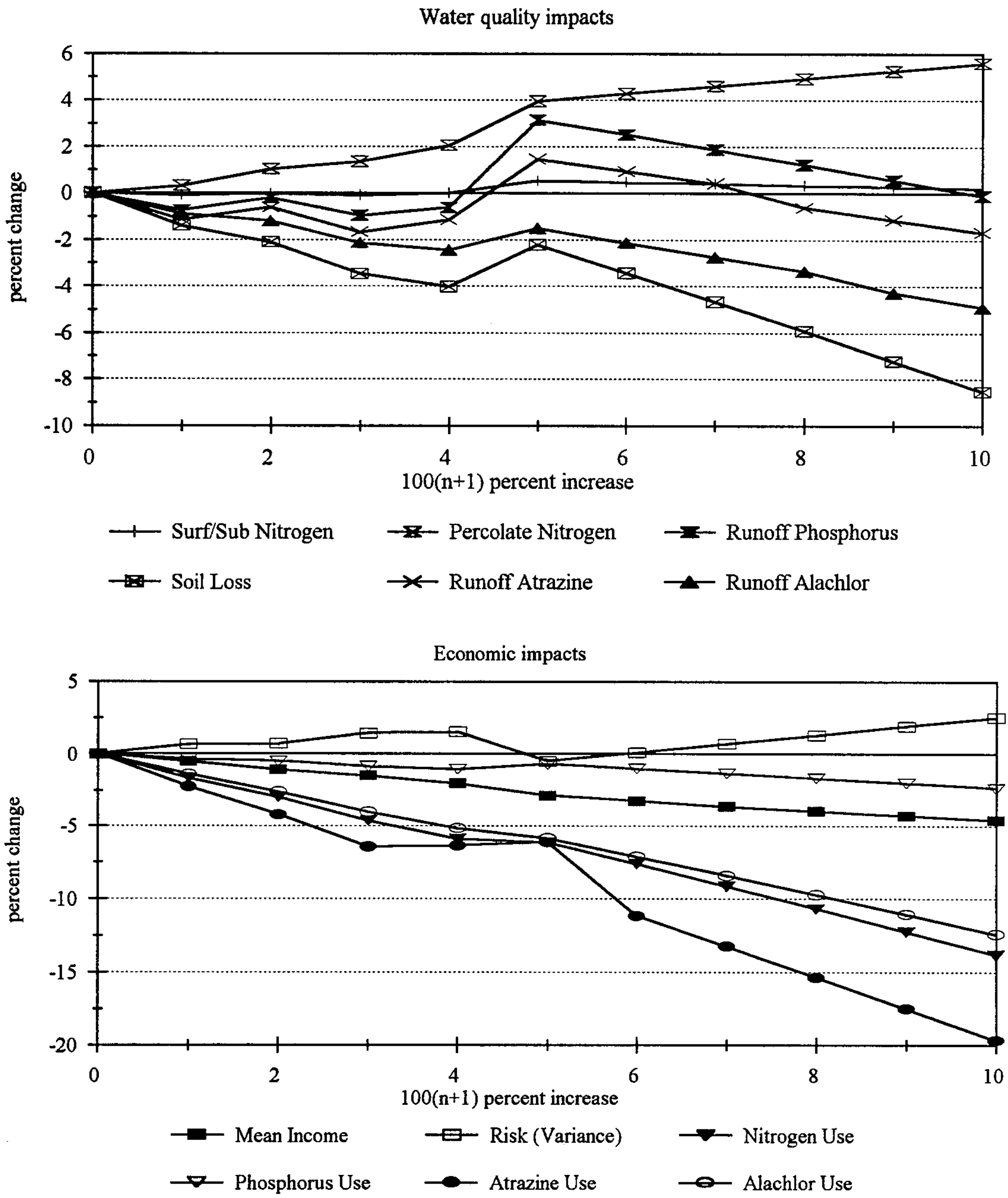

Figure 3. Economic and Water Quality Implications of Pesticide Tax.

policy. Restricting aggregate atrazine use more than 20% from the baseline was found to drastically affect farm income and risk.

Per acre atrazine use restriction. The results of this scenario are not presented graphically but are discussed. By restricting pesticide use at the per acre level, nitrogen contamination in groundwater declined at restriction levels of 6 to 25% from the baseline. For higher levels of restriction, percolate nitrogen increased from the base levels. A large drop in sediments was observed at restriction higher than 24% from the baseline and a maximum reduction of 16% was observed. The pesticide loading dropped by 16% for restrictions that had higher than a 36% level from the baseline.

There was a major decline in pesticide use for each increase in per acre use restriction. Nitrogen use increased for restrictions above 6% and re-

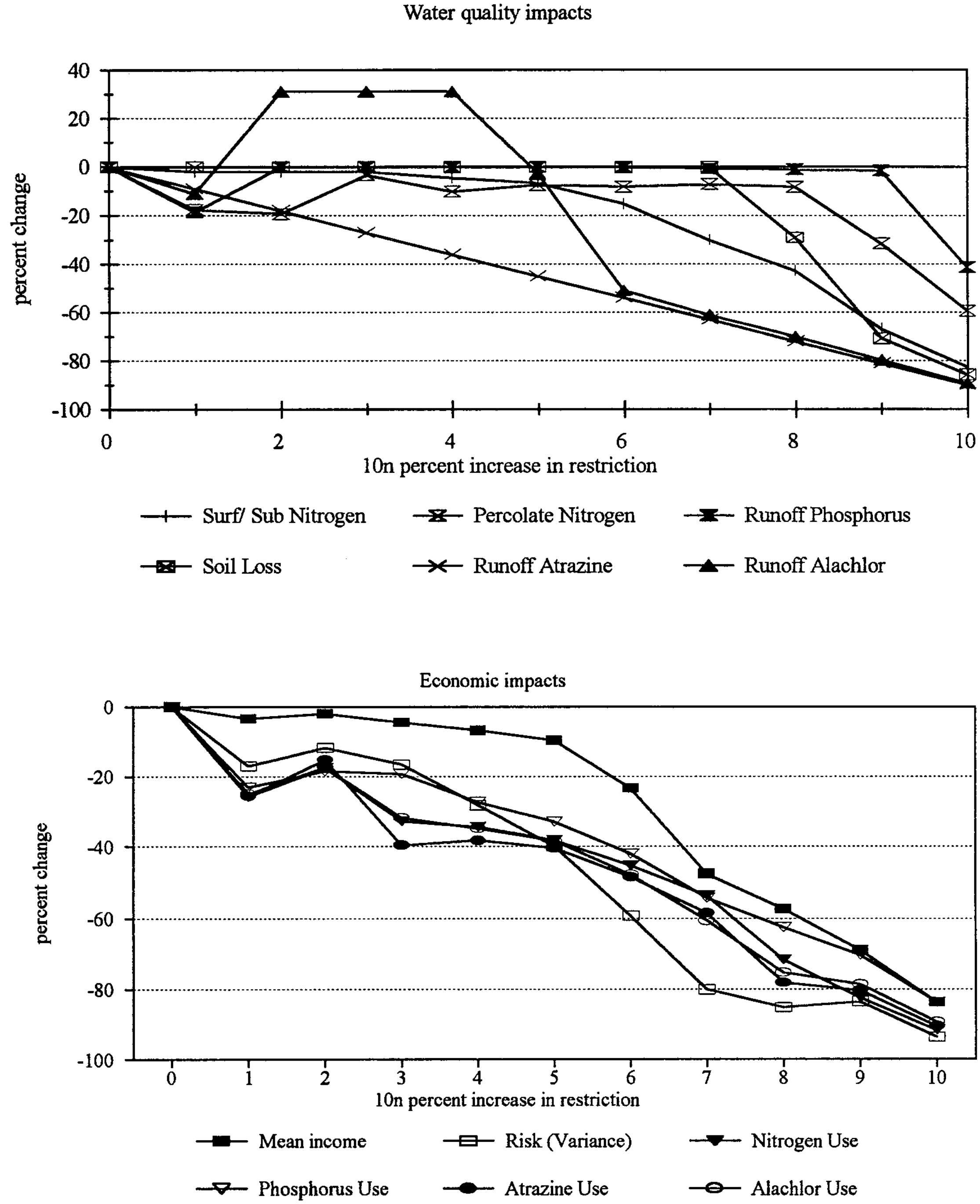

Figure 4. Economic and Water Quality Implications of Restricting Aggregate Pesticide Use.

mained at baseline levels for a 30% restriction. Farm income stabilized after an initial drop, while risk in farm income was the highest at the 6 and 30% levels. There were significant adjustments in technologies at these levels.

In summary, a per acre restriction policy was ineffective in reducing pesticide contamination of water because of a shift to cropping systems that used less pesticides but with a higher loading rate into runoff water. Most of the pollutants affecting water quality increased for restrictions less than 30% from the baseline.

Practical Caveat

The existence of multiple and cross effects among pollutants is very important in effluent reduction policies. Targeting policies to reduce a particular pollutant can increase other forms of the pollutant or other types of pollutants. Taxation of inputs can affect effluent discharges provided a careful choice is made on the tax level based on relative tradeoffs in the economic and water quality variables. Regulatory policy can also be effective in reducing all pollutants but often entails high monitoring and enforcement costs. Monitoring the pollutants through regular sampling and incentives for strict compliance with the standards can improve enforcement. An incentive (penalty) structure for compliance (noncompliance) needs to be developed on the existing institutional structure of farming.

Another approach is to proceed indirectly through the plan developed by the mathematical modeling. Decisions on the levels of pollutants are made first and entered into the model. The optimal plans generated are taken as a target in the crop management and extension programs. Other regulatory policies and taxes can be directed to achieve this target. Compliance to aggregate restrictions can be achieved through encouraging regional cooperation in maintaining water quality. Incentives to cooperate arise from a higher disparity in penalties between a noncooperative pollution level and a cooperative level. The cross effects of the choice of cropping systems on pollutant loads constitute important information that needs to be made available to producers in the watershed. Carrying out per acre restrictions involves identification of cropping systems that have higher pollution potential and regulating allocation to those activities. Again, a penalty system that charges for each acre of restricted cropping system can be used to achieve this approach. For example, nitrogen in runoff water is high under a corn-soybean-wheat cropping system with high fertilizer use and no till. By reducing per acre allowable nitrogen loading to 13 lbs., those acres under this cropping system can be regulated by a higher penalty for each pound above the allowable limit.

Conclusion

Water quality deterioration through agricultural NPS pollution is a serious problem facing policy makers. Indirect policy instruments that change input use, cropping systems, and technologies have been successfully used as water quality policies. However, policies targeted toward a single pollutant can have cross effects on other forms of pollutants, agricultural incomes, and risk. These implications for nontargeted pollutants and farm risk have not always been considered in previous studies. There is also a need to study the nature of simultaneous adjustments in cropping sequences, input use levels, substitution in cropping systems, profits, and risks facing the producers in a holistic framework. For this a direct expected utility risk programming framework that optimizes cropping systems over time can capture producer behavior and lead to a better estimation of policy implications. This is because of the model's lower reliance on assumptions underlying the distribution of stochastic variables. This study attempts to fill some of these voids in the water quality literature by studying the economic and environmental cross-implications of water quality policies using a direct expected utility programming framework and biophysical simulation.

The results of this study show that policies targeted toward a particular pollutant involved tradeoffs with nontargeted pollutants, economic returns, and risks facing farmers. Overlooking these implications can have undesirable spillover effects (new pollutant problems, increased risk, and so forth). Careful analysis of these tradeoffs can lead to cost-effective policies toward protecting water quality. Input price policies, besides directly reducing targeted pollutants, can also reduce certain nontargeted pollutants but need higher price increases to achieve a relatively small impact on water quality. It was also observed that nitrogen tax actually increased groundwater pollution of nitrogen because economic and environmental implications of this policy involved movement along a multidimensional surface involving tradeoffs among attributes. Cropping systems that use lower levels of nitrogen but have higher groundwater nitrogen pollution entered the optimal plan under a nitrogen tax policy.

Particular levels of taxing and restriction of input use allowed general improvement in water quality, with less loss in farm income and financial risk than did others. Nonetheless, most policies were ineffective in reducing groundwater contamination of nitrogen and in certain cases increased the level. Targeting for per acre reduction in nitrogen and pesticides decreased certain forms of targeted pollutants but increased the levels of other nontargeted pollutants. Per acre regulation was highly effective in achieving substantial reduction of nonpoint pollution in the watershed. The methods of regulation (per acre or aggregate) had varying effectiveness on water quality. Regulating aggregate input use levels was also effective and allowed flexibility in choice of technologies (according to site considerations of the watershed). Input use and pollutant loadings responded well to per acre regulations but increased financial risk. The choice of a particular policy or combination of policies to protect water quality depends on the dimension of economic and environmental implications. Superiority of the restriction policy indicates the need for cooperative solutions at the watershed level to reduce enforcement costs.

The impact on risk and nontargeted pollutants was significant under all policies that regulate input use. The aggregate and per acre restrictions need cooperative actions in the watershed to comply with the regulatory standards. The existing indirect markets can be effectively used in protecting water quality, compared with alternative policies that require creation of permit markets or bid markets, as in land retirement.

This type of study can be extended to other watersheds to evaluate the economic and water quality impacts of various policy options. Other policy instruments, such as trading between point and nonpoint pollutant permits, land retirement, direct restriction on pollutant loads, micro-targeting on spatial locations, and so forth, can also be assessed under this framework. A further area of development would be to integrate this modeling framework with a geographic information system to incorporate micro-level information and NPS spatial dynamics, which can be used to assess spatially targeted policies.

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