

The Performance of Compliance Measures and Instruments for Nitrate Nonpoint Pollution Control Under Uncertainty and Alternative Agricultural Commodity Policy Regimes

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Following Weitzman (1974), there is ample theoretical literature indicating that choice of pollution control instruments under conditions of uncertainty will affect the expected net benefits that can be realized from environmental protection. However, there is little empirical research on the ex ante efficiency of alternative instruments for controlling water, or other types of pollution, under uncertainty about costs and benefits. Using a simulation model that incorporates various sources of uncertainty, the ex ante efficiency of price and quantity controls applied to two alternative policy targets, fertilizer application rates and estimated excess nitrogen applications, are examined under varying assumptions about agricultural income support policies. Results indicate price instruments outperform quantity instruments. A tax on excess nitrogen substantially outperforms a fertilizer tax in the scenario with support programs, while the ranking is reversed in the scenario without support programs.

Key Words: nonpoint pollution policy, policy coordination, uncertainty

Building on the seminal work of Weitzman (1974), a small but important literature has emerged on the choice between environmental policy instruments when there is uncertainty on the part of environmental authorities about the costs and benefits of pollution control. Much of the literature (e.g., Adar and Griffin, 1976; Yohe, 1978; Stavins, 1996) is focused on the choice between emissions-based price and quantity controls. Standard results include the finding that the expected net benefits of optimally designed emissions price and quantity controls will generally differ when policy makers are uncertain about pollution control costs, with the difference

depending on the relative slopes of the marginal benefits and costs, and sign and size of the covariance between marginal benefits and costs.

The emissions-based focus of this research limits its direct relevance to nonpoint source pollution problems since a defining characteristic of nonpoint pollution is that emissions from individual sources cannot be metered at reasonable cost. With unobservable pollutant flows, other constructs must generally be used to monitor performance and as a basis for the application of policy instruments (Griffin and Bromley, 1982; Shortle and Dunn, 1986; Segerson, 1988; Xepapadeas, 1995). Options for nonpoint bases include inputs or techniques that are correlated with pollution flows (e.g., use of polluting inputs such as fertilizers), emissions proxies constructed from observations of inputs or techniques that influence the distribution of pollution flows (e.g., estimates of field losses of fertilizer residuals to surface or ground waters), and ambient environmental conditions (e.g., nutrient concentrations in ground or surface waters) (Braden and Segerson,

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This paper was presented at a workshop sponsored by: USDA-NRI, USDA-ERS, and NERCERD.

1993). With these options, the choice of instruments for nonpoint pollution control involves not only a choice between price or quantity mechanisms (or a mixture), but also a choice between target or bases to which they can be applied.

In this paper, we examine the choice between alternative instruments for reducing nitrate pollution from agricultural nonpoint sources. Reducing nitrates in ground and surface waters from agriculture has emerged as a major water pollution control policy objective (U.S. Environmental Protection Agency, 2000). The instruments considered are differentiated according to compliance measures (nitrate inputs, proxies for nitrate losses to the environment) and the types of regulations applied (prices, quantity controls). Each of these instruments is of practical interest, with instances of their use in the United States and Europe (Ribaud, 2001; Horan and Shortle, 2001; Hanley, 2001).

In addition to our interest in the implications of uncertainty for the choice of instruments, we examine the sensitivity of environmental policy performance to other societal choices that affect welfare effects and producer responses. Specifically, the environmental instruments are modeled with and without agricultural commodity and input market distortions created by agricultural price and income policies. Such distortions are found in many countries and are of concern for their impacts on the environmental performance of agriculture (Shortle and Abler, 1999). Environmental policies leading to changes in the use of inputs and outputs that are subject to policy distortions will affect the deadweight losses associated with the price and income support policies (Lichtenberg and Zilberman, 1986). The changes in deadweight losses will count as benefits (if the change is negative) or costs (if the change is positive) of the environmental policy. The literature on second-best environmental policy design suggests the effects of environmental policy parameters (e.g., emissions tax rates) should be adjusted relative to first-best settings to account for the effects on deadweight losses associated with market distortions (Baumol and Oates, 1988). Further, it is plausible that effects on deadweight losses may alter the comparative ranking of the alternative environmental instruments.

The economic performance of alternative instruments for nitrate nonpoint pollution control has been examined in several articles (e.g., Johnson, Adams, and Perry, 1991; Helfand and House, 1995; Huang and LeBlanc, 1994; Shortle and Laughland, 1994; Stevens, 1988; Swinton and Clark, 1994).

However, prior empirical studies have assumed environmental planners have perfect information about costs and benefits. Expanding on the early work of Weitzman, Wu (2000), and Wu and Babcock (2001) provide theoretical guidance to the choice between price and quantity controls for input-based instruments. As with Weitzman, the generality of the results is limited by the use of linear marginal costs and benefits, with uncertainty affecting the intercept but not the slope of these marginal functions (Malcomson, 1978; Shortle and Abler, 1997).

Whereas prior research on the choice of instruments under uncertainty is almost exclusively conceptual, this analysis is based on a simulation model that incorporates various sources of uncertainty and nonlinearities, and allows for estimation of the expected net benefits from alternative instruments. Public uncertainty about economic and environmental outcomes due to imperfect information about parameters of cost and benefit functions (e.g., input substitution elasticities, product demand, and input supply elasticities), and stochastic environmental variables is captured by the model.

Our interest in linkages between price and income policies on environmental policy performance leads us to implement our simulation model at a national level so as to capture the effects of environmental policies on prices and, in turn, the deadweight loss of the price and income support policies. This national-level model is not spatially explicit. The simulation model is based on a set of assumptions about the structure of agricultural production, input and output markets, the relationship between land uses and pollution loadings, and pollution loadings and environmental damages. These assumptions are primarily reflected in the level of aggregation and functional forms. However, within the broad structure, the robustness of our results is increased by considering numerous possible states of nature as defined distributions of key economic and biophysical parameters.

We begin with a conceptual model of the design of input taxes, input standards, a tax on expected emissions, and an expected emissions standard. The design and choice between input taxes and input quantity controls are examined in detail. The discussion serves to clarify the issues addressed in this paper and to facilitate the presentation of the empirical simulation model. Next, the simulation model is developed and results are presented on the relative performance of alternative instruments under alternative commodity policy scenarios.

Nonpoint Policy Choices

For heuristic purposes, we consider the problem of an environmental agency seeking to control nitrate pollution from agriculture. A single competitive agricultural sector is assumed (specifically, corn in our empirical simulation) as the source of nonpoint pollution, and production and emissions are modeled at the industry level. Following prior research on agricultural nonpoint pollution (e.g., Griffin and Bromley, 1982; Shortle and Dunn, 1986; Horan, Shortle, and Abler, 1999), environmental outcomes are a function of inputs used in production (e.g., nitrogen fertilizer) and environmental conditions influencing the formation and fate of nonpoint emissions (e.g., weather). Because of the level of aggregation, other spatially explicit factors are not included. The environmental agency is unable to observe emissions directly, but can formulate expectations using a probabilistic model of the relationship between input use and emissions.

It is assumed the agency has four types of instruments from which it may choose. The instruments are defined by a choice of policy target or base (inputs or expected emissions) and the choice of mechanism for regulating the target (taxes or quantity controls). We envision the agency proceeding as follows when evaluating the options. First, the agency models the change in resource allocation and the resulting social benefits and costs which will result from a specific application of a particular instrument. The agency has imperfect knowledge of producers' cost functions, commodity demands, input supplies, and environmental damage costs, and is therefore uncertain about effects on resource allocation and societal benefits and costs. However, the agency is assumed to have a well-defined information structure consisting of the possible states of the world and their probabilities. The agency can therefore compute state-contingent resource allocation and social net benefit outcomes for specific policy designs, and can compute the expected values over the set of all possible states.

The agency's objective is to maximize the expected societal benefits less costs of pollution control. The optimal design of a particular instrument (e.g. a fertilizer tax) is the design that yields the greatest increase in expected social benefits less costs contingent on the choice of instrument. The preferred instrument is the one that provides the greatest expected net benefit among the four options.

Notation is as follows:

- r / pollution loading;
- Q / industry output;
- x_i / production input i ($i = 1, 2, \dots, m$);
- p_s / price received for output;
- p_d / price paid for output;
- w_i^d / price paid for factor i ;
- w_i^s / price received for factor i ;
- Ω / vector of production parameters;
- α / vector of demand parameters;
- \mathbf{g} / vector of factor supply parameters for factor i ;
- γ / vector of fate and transportation parameters;
- κ / vector of damage cost parameters;
- $r(x_1, x_2, \dots, x_m; \gamma)$ / pollution loading function;
- $Q(x_1, x_2, \dots, x_m; \Omega)$ / output production function;
- $p^d(Q, \alpha)$ / inverse demand for output;
- $w_i^s(x_i, \mathbf{g})$ / inverse supply function for factor i ;
- $D(r, \kappa)$ / pollution damage cost function.

The agency's uncertainty about the response of polluters to price instruments and the economic costs of compliance with both price and quantity instruments results from imperfect information about production technology, output demand, and input supplies. This uncertainty is modeled by treating the vectors Ω , α , and \mathbf{g} ($i = 1, 2, \dots, m$) as random variables with known distributions from the agency's perspective. Regulatory uncertainty about the benefits of the pollution control instruments results from imperfect information about the transport and fate of pollutants, and the economic damage caused by pollution. This uncertainty is modeled similarly, treating γ and κ as random variables with known distributions.

We illustrate what is meant here by state-contingent changes for the cases of input taxes and quantity controls. The agency assumes producers maximize profits under perfect information about input and output prices, and technology. Using our industry-level specification of production, the state-contingent input choices when producers are subject to input taxes solve:

$$(1) \quad \max_{x_1, \dots, x_m} \pi' p_s Q(x_1, x_2, \dots, x_m; \Omega) \\ \& \prod_{i=1}^m (w_i^d / \tau_i) x_i,$$

where τ_i is the tax rate for factor i . Individual input tax rates may be positive, negative, or zero depending on whether the agency seeks to discourage or encourage particular input choices. The input demand functions that solve the profit-maximization problem are given as $x_i^\tau(p_s; \tilde{w}_1, \dots, \tilde{w}_m; \Omega)$, where $\tilde{w}_i = w_i^d \circ \tau_i, i = 1, 2, \dots, m$. When subject to input standards, the state-contingent profit-maximizing input choices solve:

$$(2) \quad \max_{x_1, \dots, x_m} \pi^i p_s Q(x_1, x_2, \dots, x_m; \Omega) \\ \& \prod_{i=1}^m w_i^d x_i \\ \text{s.t.: } x_i \leq \bar{z}_i, \quad (i = 1, \dots, m),$$

where the set \bar{z}_i is the regulatory standard defining the set of acceptable input choices. The standards may be lower bounds for pollution-control inputs or upper bounds for pollution-increasing inputs. The input demands that solve the profit-maximization problem with quantity controls are denoted $\bar{x}_i(p_s; w_1^d, \dots, w_m^d; \Omega; \bar{z}_1, \dots, \bar{z}_m), i = 1, 2, \dots, m$. Similar procedures would be followed to obtain state-contingent responses for the other types of instruments.

Given the state-contingent production choices, the optimal design of an instrument is obtained by maximizing the expected social value of the instrument. If it is assumed the output and input markets are undistorted by agricultural price and income policies, the optimal design of a particular instrument maximizes the expected economic surpluses accruing to consumers, producers, and resource suppliers less environmental damage costs (Just, Heuth, and Schmitz, 1992). After some manipulation, the expected sum of the surpluses can be expressed as:

$$(3) \quad ENB^i = E \left[\int_{\Omega} Q(x_1, \dots, x_m; \Omega) p(s, \alpha) ds \right. \\ \& \prod_{i=1}^m \left(w_i(x_i, \mathbf{g}) \& \int_{\Omega} w_i^x(s, \mathbf{g}) ds \right) \\ \& D(r(\alpha, \kappa)) \Big],$$

where the expectation is formed using the agencies' joint probability density function for the random variables $\Omega; \mathbf{g}_1, \dots, \mathbf{g}_m; \alpha; \gamma$; and κ . Ex ante optimal input tax rates, or input restrictions, are determined by plugging the input demands that solve (1) for any τ_i , or (2) for any \bar{z}_i , into (3) and then choosing tax rates or input restrictions to maximize the resulting expression. Specifically, ex ante optimal input taxes maximize (3) subject to $x_i = x_i^\tau(p_s; \tilde{w}_1, \dots, \tilde{w}_m; \Omega), i = 1, 2, \dots, m$. The ex ante optimal tax rate for input

x_i is denoted as τ_i^* , and the resulting optimized expected net benefits are denoted $ENB_{\tau_i}^*$. The ex ante optimal input restrictions maximize (3) subject to $x_i = \bar{x}_i(p_s; w_1^d, \dots, w_m^d; \Omega; \bar{z}_1, \dots, \bar{z}_m), i = 1, 2, \dots, m$. The ex ante optimal restriction for input x_i is written as \bar{z}_i^* , and the optimized expected net benefit function as $ENB_{\bar{z}_i}^*$.

The difference in the expected net benefits of the ex ante optimal input taxes and standards is $\Delta_{\tau\bar{z}} = ENB_{\tau_i}^* - ENB_{\bar{z}_i}^*$. A fundamental result of prior research on the choice of instruments under uncertainty is that it is uncertainty about polluters' control costs which causes differences in the ex ante efficiency of optimized price and quantity controls (e.g., Weitzman, 1974). Accordingly, with perfect information about polluters' responses, $\Delta_{\tau\bar{z}} = 0$. Under imperfect information about control costs, $\Delta_{\tau\bar{z}}$ may be positive or negative. From prior work on the choice between emission taxes and quantity controls, it is clear that under imperfect information little can be said a priori about $\Delta_{\tau\bar{z}}$ without imposing very strong restrictions on the structure of the underlying economic and physical relationships and the joint probability density function of the random variables (Baumol and Oates, 1988; Weitzman, 1974; Stavins, 1996; Malcomson, 1978; Wu and Babcock, 2001; Wu, 2000).

The Simulation Model

For our empirical simulation, we use a model of corn production in the United States and its impacts on nitrate pollution. The model has the same general structure as the conceptual model presented above. This section begins with a description of the specific forms of the production, commodity demand, input supply, pollution loading, and damage cost functions. Uncertainty is modeled by treating parameters of these functions as random variables with known distributions. The functional forms and parameter distributions were selected to provide a reasonable but computationally feasible representation of the problem. The remainder of the section is devoted to the parameter distributions, solution procedures, and other details of the implementation of the model.

Functional Forms

The industry production function, $F(x, \Omega)$ above, is a two-level nested constant elasticity of substitution (CES) production function (Sato, 1967). The two-level CES production function is parsimonious in

parameters and is a reasonable approximation of aggregate agricultural processes (Kaneda, 1982; Hayami and Ruttan, 1985; Howitt, Ward, and Msangi, 1999). The key parameters of this functional form are the share parameters and elasticities of substitution between inputs. Following Abler and Shortle (1992), output is modeled as a function of a composite mechanical input, M , and a composite biological input, B . The mechanical input is a function of capital and labor. However, prices of capital and labor are held constant in this analysis because they are generally not affected by the agricultural sector (Abler and Shortle, 1992). Consequently, there will be no change in the relative price within the nest, implying no change in the factor proportions. Thus, the inputs can be aggregated into a single input, x_k . The production function is therefore given as:

$$Q = A_1 (s_B B^{\rho_T} s_K x_K^{\rho_T})^{1/\rho_T}$$

where

$$\rho_T = \frac{\sigma_T + 1}{\sigma_T}$$

σ_T is the elasticity of substitution between the mechanical and biological inputs, A_1 is a scaling constant, and $0 < s_B, s_K < 1$ are share parameters. The biological input is a function of land (x_L) and fertilizer (x_F), and is specified as:

$$B = A_2 (s_L x_L^{\rho_B} s_F x_F^{\rho_B})^{1/\rho_B}$$

where

$$\rho_B = \frac{\sigma_B + 1}{\sigma_B}$$

σ_B is the elasticity of substitution between land and fertilizer, A_2 is a scaling constant, and $0 < s_L, s_F < 1$ are the share parameters. Because this is a simulated national production model using national averages for various parameters, climate and other regional variations that would affect yield are not modeled.

The inverse demand function, $p^d(Q, \alpha)$, is assumed to be linear with intercept α_1 and slope α_2 . This specification of the demand function can be viewed as a first-order Taylor series approximation for the true demand function in the neighborhood of the expansion point.

All inputs except land are assumed to be supplied perfectly elastically. In the short run, supply responses are not perfectly elastic, but over time labor and resources used in capital and chemical production can be withdrawn at relatively low cost to nonagricultural use (Gardner, 1987; Abler and

Shortle, 1992). However, even if the long-run supply responses are not perfectly elastic, they can be treated as such because the elasticities are very large (Abler and Shortle, 1992). In the context of the model presented above, this implies $M_i^s/M_i = 0$ for all inputs except land. Land supply is specified as a constant elasticity function of the rental rate of land, $x_L = A_3(w_L)^g$, where g is the elasticity of the land supply and A_3 is a scaling constant.

Emissions are modeled as a proportion of excess nitrogen. Excess nitrogen is defined as applied nitrogen less nitrogen uptake by the plant, $e = x_F - \gamma Q$, where γ is the proportion of nitrogen removed per bushel of corn. Following Roth and Jury (1993), surface runoff is specified as $r = \beta(x_F - \gamma Q)$, where β represents the proportion of excess nitrogen in runoff. Most nitrogen damage is in the form of runoff, but there is also some leaching. However, given the level of aggregation of the model, the details of the fate of nitrogen cannot be fully captured. For the sake of simplicity, damage costs are assumed to be an exponential function of nitrogen runoff, $D = \kappa_2(r)^{\kappa_1}$.

There are no definitive studies on the costs of nitrogen pollution or the form of a nitrogen damage cost function. We use a functional form that has desirable economic properties (increasing marginal damage cost), and is parsimonious in parameters to facilitate the computation of expected values. This form has been used in prior studies in the design of emissions taxes and standards under uncertainty (Kolstad, 1986; Xepapadeas, 1991).

Environmental and Farm Policy Instruments

Our environmental instruments are taxes and standards on purchased nitrogen fertilizer, and taxes and standards on excess nitrogen. These policies are examined with and without corn price supports and land retirement policies. The support policies we consider are intended to illustrate the effect of common types of agricultural interventions rather than to evaluate specific policies. However, the policies examined are analogous to those applied to corn production in the United States prior to farm policy reforms implemented by the 1996 Federal Agriculture Improvement and Reform (FAIR) Act (also known as the "Freedom to Farm Act"). Specifically, the price support program is analogous to the Deficiency Payment Program, and the land retirement policy is analogous to the Acreage Reduction Program (ARP). The 1996 reforms were intended in part to wean agriculture from extensive

government support. The more recent 2002 Farm Bill reverses that direction.

The effect of the land retirement program is a reduction in the supply of land (Gardner, 1992). If L is total land supply, x_L land in use, and ϕ the proportion of land idled in the land retirement program, then total land supply is given as $L = x_L + \phi L$. The land supply equation with the land retirement program is represented by

$$\frac{x_L}{1 + \phi} = A_3(w_L)^g.$$

Following Shortle and Laughland (1994), the price support program is modeled as a per unit subsidy (s) on output. The price received for output (p_s) is therefore equal to the price paid for output (p_d) plus the subsidy.

With these specifications, a market equilibrium for any particular parameter set and instrument solves equations (4)–(10) in table 1 for consumer and producer output prices, total output, profit-maximizing input use, land supply, and the shadow prices μ , of a fertilizer standard, or λ , of an excess nitrogen standard, if those instruments are operative. Equation (4) defines the producer price. The policy parameter s in this equation is zero in simulations with no commodity policies and a positive constant in simulations with commodity policies. Equation (5) is the inverse demand function and defines the consumer price that would clear the market given total production. Equation (6) defines the level of output.

Equations (7)–(9) are the first-order conditions for profit maximization. The shadow price μ in equation (7) is zero in simulations without the fertilizer standard. It will be positive and an equilibrium result of simulations with the standard. Similarly, the shadow price λ in equations (7)–(9) is zero in simulations without the excess nitrogen standard, and positive and an equilibrium result in simulations with the standard. The fertilizer tax rate, τ_F , is positive in simulations with fertilizer tax and zero otherwise. The excess nitrogen tax rate, τ_e , is positive in simulations with the excess nitrogen tax and zero otherwise. Equation (10) is the land supply function. The agricultural policy parameter ϕ is zero in simulations without commodity policies and a positive constant in simulations with commodity policies. Equations (11)–(13) define relationships between production choices and environmental damages.

Table 1. Model Equations

Price Definition/Consumer Behavior:

$$(4) \quad p_s = p_d + s$$

$$(5) \quad p_s = \alpha_1 + \alpha_2 Q$$

Production:

$$(6) \quad Q = A_1 (s_B B^{p_r} s_K x_K^{p_r})^{1/\rho_r},$$

where $B = A_2 (s_L x_L^{p_B} s_F x_F^{p_B})^{1/\rho_B}$

Profit Maximization:

$$(7) \quad p_s \frac{M_D}{M_F} + (w_F + \tau_F \mu) + (\tau_e \lambda) \frac{M_F}{M_F} = 0,$$

$$\mu(\bar{z}_F + x_F) = 0, \quad \lambda(\bar{e} + e) = 0$$

$$(8) \quad p_s \frac{M_D}{M_L} + w_L + (\tau_e \lambda) \frac{M_F}{M_L} = 0$$

$$(9) \quad p_s \frac{M_D}{M_K} + w_K + (\tau_e \lambda) \frac{M_F}{M_K} = 0$$

Land Supply:

$$(10) \quad \frac{x_L}{1 + \phi} = A_3(w_L)^g$$

Excess Nitrogen:

$$(11) \quad e = x_F + \gamma Q$$

Runoff:

$$(12) \quad r = \beta(x_F + \gamma Q)$$

Environmental Damages:

$$(13) \quad D = \kappa_2(r)^{\kappa_1}$$

Net Social Benefit:

$$(14) \quad NB = \int_{\Omega} Q(x_1, \dots, x_m; \Omega) p(s, \alpha) ds$$

$$+ \sum_{i=1}^m \left(w_i(x_i, g_i) + \frac{x_i}{\eta_D} w_i(s, g_i) \right) + D(r; \kappa)$$

Notes: For an input tax alone, $\mu_i, t_e, \lambda = 0$; for an input standard alone, $t_i, t_e, \lambda = 0$; for an excess nitrogen tax alone, $\mu_i, t_i, \lambda = 0$; and for an excess nitrogen standard alone, $\mu_i, t_i, t_e = 0$.

Model Calibration and Parameter Distributions

Public uncertainty is modeled as uncertainty by environmental regulators about economic and environmental parameters—specifically, the elasticity of substitution between land and fertilizer (σ_{LF}), the output demand elasticity (η_D), the land supply elasticity (g), the damage cost function coefficient (κ_1), and the runoff coefficient (β). The uncertain parameters are assumed to be independent continuous random variables with known distributions. Uniform and normal are used in the simulations.

Moments of the production and market parameters [the elasticity of substitution between land and

fertilizer (σ_{LF}), land supply elasticity (g), and output demand elasticity (η_D), are derived from Abler and Shortle (1995). Specifically, the parameters are distributed with means 1.0, 0.2, and 0.7 and standard deviations 1.0, 0.057, and 0.3, respectively. Using the expression $\sigma_B = s_B \sigma_{LF} + (1 - s_B) \sigma_T$, the elasticity of substitution between land and fertilizer (σ_B) is calculated. Following Abler and Shortle (1995), $\sigma_T = 0.50$. The slope and intercept of the inverse demand function for output are derived as

$$\alpha_2 = \frac{P}{Q \times \eta_D} \quad \text{and} \quad \alpha_1 = P - \alpha_2 Q,$$

where η_D is the elasticity of demand at the expansion point (P, Q).

The uptake parameter (γ) is treated as a constant and equal to 0.66 (National Research Council, 1993; Abrahams, 1996). The runoff parameter (β) is assumed to be distributed with mean of 0.60 and variance of 0.115. This keeps the parameter within the plausible range of zero and unity. There is little information on the curvature of the damage function in the literature. Peck and Teisberg (1993) found optimal emissions control policies are more sensitive to the degree of nonlinearity in the damage function than to the scale of the damage function. We assume κ_1 to have a mean of 2 and a variance of 0.57. The marginal damage cost function therefore varies from a constant to a cubic function.

Very little is known about the economic cost of nitrate pollution, despite some attempts at quantifying it. Smith (1992), in a survey of environmental damage estimates, suggests it probably lies within a range of 0.27% to 18.24% of total crop value. For this study, damages are assumed to be 10% of total crop value, which is about the mid-point of the range reported by Smith. The scale parameter (κ_2) is calculated as D/κ_1 .

Although we assume continuous distributions of the unknown parameters, we use discrete approximations in the simulations. This is because there is no closed-form solution to equations (4)–(10). Consequently, the net benefit function [equation (14)] cannot be integrated directly to obtain the expected net benefit as a function of the instrument specifications for these distributions. The approximation is obtained using values and probabilities from discrete approximations of the random variables. The discrete approximations were derived using an iterative procedure outlined in figure 1.

First, n points are selected for each random variable. The points and their probabilities are selected to preserve moments of the original distribution.

Points and probabilities for standardized distributions for this purpose can be found in Stroud and Secrest (1966). Values for similar distributions with different means and standard deviations can be obtained by performing the same affine transformations needed to transform the standard distribution to the desired distribution. With five random variables, the joint probability space is 5^n . Each element in this set implies a different realization of the model. Because each realization is used in the simulations (see below), the number of models we must solve to obtain results increases exponentially with n . Therefore, while increasing n improves the approximation, it also increases the computational requirements exponentially.

Next, the scale parameters (A_1, A_2 , and A_3) and the demand parameters (α_1 and α_2) are calibrated so that the solution to equations (4)–(10) with no environmental policies operative corresponds to the average values of the market price, production, and input variables for the years 1989–1993, which is the baseline equilibrium used for this analysis. This period was selected as the baseline because of our interest in exploring how distortionary commodity policies can influence the comparative performance of environmental policies. This baseline period captures the last few years of the major distortionary corn commodity policies in the United States. The baseline data were drawn from the U.S. Agricultural Resources Model (Quiroga, Konyar, and McCormick, 1992). The market price for corn in this model is \$2.28 per bushel, while the target price is set at \$2.75 per bushel. This implies the subsidy per bushel is \$0.47. In the time period used for this analysis (1989–1993), farmers were required to idle 10% of their acreage when they participated in the Acreage Reduction Program (i.e., $\phi = 0.1$).

Next, the implied expected net benefit for the benchmark equilibrium is computed as the probability weighted sum of the net benefit over the joint probability space. We initiated this procedure with $n = 2$ and tried larger values up to $n = 7$. There was little change in the expected net benefit after $n = 5$ for both the uniform and normal distributions. This finding suggests that the discrete approximations with $n = 5$ does a reasonable job of preserving the mean of the underlying continuous distribution. For example, there was only about a 1% change in expected net benefit between $n = 5$ and $n = 7$, while the joint probability space went from 3,125 points to 78,125. The final set of discrete points and associated probabilities for the random parameters in the model are presented in table 2.

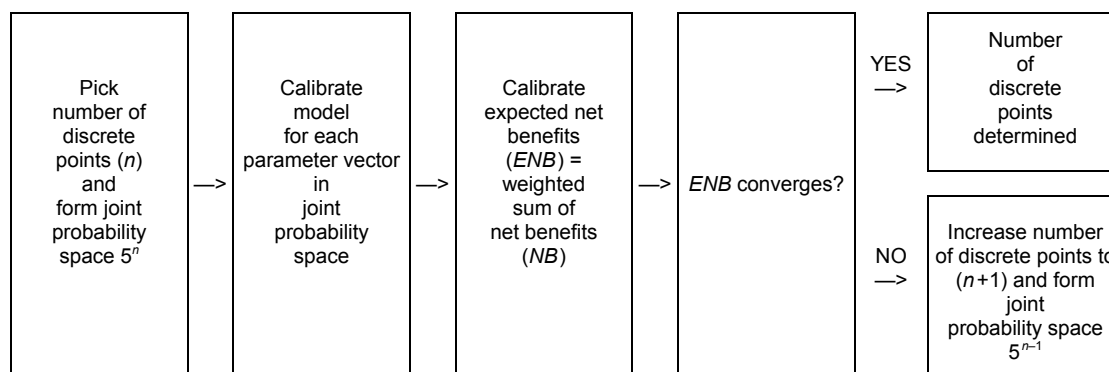


Figure 1. Determining number of discrete points in discrete approximation when number of uncertain parameters = 5

Table 2a. Discrete Points and Probabilities for Uniform Approximation of Parameters

Elasticity of Substitution	Output Demand Elasticity	Land Supply Elasticity	Runoff Coefficient	Damage Function Exponent	Probability
0.09	! 0.31	0.11	0.42	1.09	0.118
0.34	! 0.38	0.15	0.49	1.46	0.239
0.69*	! 0.50*	0.20*	0.60*	2.00*	0.284
1.04	! 0.66	0.25	0.71	2.54	0.239
1.29	! 0.80	0.29	0.78	2.91	0.118

An asterisk (*) denotes variable mean.

Table 2b. Discrete Points and Probabilities for Normal Approximation of Parameters

Elasticity of Substitution	Output Demand Elasticity	Land Supply Elasticity	Runoff Coefficient	Damage Function Exponent	Probability
0.04	! 0.21	0.04	0.27	0.35	0.011
0.17	! 0.33	0.12	0.44	1.22	0.222
0.69*	! 0.50*	0.20*	0.60*	2.00*	0.533
1.21	! 0.75	0.28	0.76	2.78	0.222
1.78	! 1.17	0.36	0.93	3.65	0.011

An asterisk (*) denotes variable mean.

Policy Simulation Procedures

A numerical procedure is used to solve for ex ante optimal instrument values (i.e., the designs that maximize expected net benefits). The steps are:

1. Solve the market model for each parameter set of the joint distribution for given tax rates or standards and commodity program structure.
2. Calculate the net benefit for each parameter set.
3. Construct a probability weighted average of outcomes for each tax rate or standard. This generates a set of expected net benefits for each policy level.
4. Search for the tax rate (or standard) that gives the highest expected net benefit.

Results

Results for the expected net benefits from the optimized environmental instruments are presented in table 3 for two scenarios concerning agricultural income support programs and two alternative assumptions about the form of the distribution of the unknown parameters. In the scenario with the agricultural income support policies, the expected net benefit of an optimized instrument is the increase in the expected social surplus which would result from the implementation of the instrument given that the land and commodity markets are distorted by the income support policies. In scenarios without the agricultural income policies, the expected net benefit of an optimized instrument is the increase in the expected social surplus which would result from implementation of the instrument when land and commodity markets are not distorted by the income support policies. The expected net benefits for each instrument in each scenario are derived using uniform or normal distributions for the unknown parameters.

Expected Net Benefits

The expected net benefit of each of the instruments is substantially greater in the “with” income support programs scenario for both distributions (table 3). There are two reasons for this difference. One is that the agricultural income support programs encourage fertilizer use, and therefore increase environmental damage costs relative to the case without the supports. In other words, there are larger gains to be had from pollution reductions. The greater fertilizer use with agricultural income support programs is a consequence of the increased production resulting from supporting the producer price above the perfectly competitive equilibrium level, and substitution of fertilizer (and other inputs) for land resulting from the implicit tax on land created by the Acreage Reduction Program.

The second reason for this difference has to do with the deadweight costs of the income support programs. The income support programs distort the output and land markets, resulting in less land use but more output than would occur in the perfectly competitive equilibrium. By reducing output and penalizing fertilizer use relative to other inputs, the environmental policies reduce the economic costs of the market distortions in the scenario with the income support programs. This “extra dividend” from the environmental policies is not present in the scenario without income support programs.

In the scenario without income support policies, the tax instruments produce substantially greater benefits than the standards, with the tax on excess nitrogen substantially outperforming the fertilizer tax. As noted previously, optimized taxes and standards for any particular base would be equally efficient in the absence of uncertainty about polluters’ control costs. Thus, our results suggest uncertainty about producer responses can be a very important factor in the choice between price and quantity controls for nitrate pollution control from agriculture. Because our model is highly nonlinear in both endogenous and random variables, the explanation of the relative performance of the taxes and standards for each base cannot be reduced to the differences in the slopes of linear marginal benefit and cost curves offered by Weitzman (1974), or more recently by Wu (2000), and Wu and Babcock (2001).

While the relative performance of taxes and standards for each base is inherently an empirical question when there is uncertainty about polluters’ control costs, the superior performance of the tax on excess nitrogen relative to the fertilizer tax is to be expected a priori in the scenario without income support policies. In our model, nitrogen pollution is proportional to excess nitrogen. Excess nitrogen is in turn equal to nitrogen applications less uptake. Because uptake is proportional to production, which is a function of all inputs, fertilizer is not the only input that affects excess nitrogen. A tax on excess nitrogen will induce producers to choose the combination of fertilizer and other inputs that minimize the cost of reducing nitrogen pollution. A tax on fertilizer will induce responses that minimize the costs of reducing fertilizer use, but these responses would be equivalent to least-cost reductions in nitrogen pollution only if pollution were proportional to fertilizer use. Thus, the costs of a pollution reduction will be greater with the fertilizer tax than with the tax on excess nitrogen. Our findings show the difference in the results is substantial.

An interesting feature of our results is that, in the absence of income supports, a tax on fertilizer substantially outperforms an excess nitrogen standard. This again underscores the importance of uncertainty about producers’ control costs for the choice between instruments. In the absence of uncertainty about producers’ control costs, the reasoning presented above would imply an excess nitrogen standard would outperform a fertilizer tax. However, in our results, the gains from the use of a tax on the suboptimal base (fertilizer) greatly outweigh

Table 3. Expected Net Benefits (million dollars)

Instrument	With Support Programs		Without Support Programs	
	Uniform	Normal	Uniform	Normal
Fertilizer Tax	492.2	397.0	52.5	53.2
Fertilizer Standard	422.4	334.3	0.01	0.01
Excess Nitrogen Tax	470.0	374.7	71.5	71.0
Excess Nitrogen Standard	452.0	368.5	0.01	0.01

the benefits of using an optimal base (excess nitrogen) with a standard. Thus, recommendations about the appropriate base for nitrate and other types of nonpoint pollution control should not be made without reference to the type of mechanism used to induce changes in polluters' behavior. While taxes dominate for both bases, the same caution should be applied to recommendations about the types of mechanisms used to induce behavioral change.

The relative performance of the instruments in the scenario with income support policies differs from the scenario without these programs. The tax option again substantially dominates the standard for each base. Thus, uncertainty about polluters' control costs remains an important factor, with taxes outperforming standards within a category. However, in this case, the fertilizer tax dominates the tax on excess nitrogen. The reason for the change in ranking has to do with the relative effects of the two instruments on the deadweight loss associated with the income support programs. Both instruments, by reducing total output and the incentives for substituting fertilizer and other inputs for land, yield an extra dividend in the form of reduced deadweight costs from commodity and factor market distortions. While less efficient as an environmental instrument, the fertilizer tax has a better overall economic outcome because it produces a larger reduction in the deadweight costs of the market distortions. Given the definition of excess nitrogen, the excess nitrogen tax is, in effect, a tax on fertilizer less an output subsidy. The fertilizer tax component will encourage substitution of other inputs for fertilizer, and will have a negative output effect as a consequence of increased marginal costs. The output subsidy component will encourage the use of all inputs. Because the fertilizer tax has the stronger effect, the net result is to reduce output and fertilizer use, but to a lesser degree than the fertilizer tax alone.

Although still dominated by the tax instruments, the performance of the fertilizer and excess nitrogen standards is much better in the scenario with income supports than without. The difference is the

larger external costs and the extra dividend from reducing deadweight costs in the scenario with income support policies.

Conclusion

This analysis has examined the choice between alternative instruments for reducing nitrate pollution from agricultural nonpoint sources. Whereas prior research on the choice of instruments under uncertainty is almost exclusively conceptual, this analysis is based on a simulation model that incorporates various sources of uncertainty and nonlinearities, and allows us to estimate the expected net benefits from alternative instruments. The instruments considered are differentiated according to compliance measures (nitrate inputs, proxies for nitrate losses to the environment) and the types of regulations applied (prices, quantity controls).

Public uncertainty about economic and environmental outcomes due to imperfect information about parameters of cost and benefit functions (e.g., input substitution elasticities, product demand, and input supply elasticities), and stochastic environmental variables, is captured by the model by treating the parameters of interest as random variables with known distributions.

Because of our interest in linkages between price and income policies on environmental policy performance, a simulation model was implemented at a national level so as to capture the effects of environmental policies on prices and, in turn, the deadweight loss of the price and income support policies. The expected net benefits from the optimized environmental instruments are presented for two scenarios concerning agricultural income support programs and two alternative assumptions about the form of the distribution of the unknown parameters. The expected net benefit of an optimized instrument is the increase in the expected social surplus which would result from the implementation of the instrument under alternative agricultural income support programs.

The results show that economically efficient nitrate policy choices are sensitive to agricultural income support programs and uncertainty. Economic theory indicates that with perfect information, quantity instruments are as efficient as the tax instruments. However, under conditions of uncertainty or asymmetric information, the performance of quantity and price instruments is an empirical issue. With uncertainty, our findings indicate price instruments give higher expected net benefits than quantity controls, and are therefore more efficient in the control of nonpoint source pollution with or without agricultural income support programs. In the presence of agricultural income support programs, the preferred instrument is a fertilizer tax, whereas without the agricultural income support programs, the preferred policy is an excess nitrogen tax.

Our specific results are clearly contingent on the structure of our simulation model. The major limitations are the modeling of a single crop, the lack of spatial detail, and the assumptions regarding functional forms. Advances in our understanding of national agri-environmental policies under uncertainty would be achieved by increasing the spatial resolution, the comprehensiveness of the modeling of production, and consideration of multiple functional forms. An enormous and obvious challenge in expanding the analysis is, however, the "curse of dimensionality."

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