# Environmental Risk and Agri-Environmental Policy Design[1]

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**Abstract**

Agricultural nonpoint pollution is inherently stochastic (e.g., due to weather). In theory, this randomness has implications for the choice and design of policy instruments. However, very few empirical studies have modeled natural variability. This paper investigates the importance of stochastic processes for the choice and design of alternative nonpoint instruments. The findings suggest that not explicitly considering the stochastic processes in the analysis can produce significantly biased results.

## Introduction

Agricultural nonpoint source pollution, especially nutrient runoff, is a major source of many remaining U.S. water quality problems (USEPA and USDA, 1998). As efforts to control these sources are beginning to take shape (USEPA and USDA, 1998; Ribaudo, Horan, and Smith, 1999), there is a growing need for economic analysis that can guide the selection and design of policy instruments. However, the existing literature on the relative efficiency of alternative nonpoint pollution control instruments (e.g., Helfand and House, 1995; Larson, Helfand, and House, 1996; Flemming and Adams, 1997; Moxey and White, 1992; Shortle, Horan, and Abler, 1998; Claassen and Horan, forthcoming; Horan et al., 1999; Hopkins, Schnitkey, and Tweeten, 1996; Taylor, Adams, and Miller, 1992; Huang, Shank, and Hewitt, 1996; Mapp et al., 1994) is still far from a consensus on what types of instruments represent good economic policies (Shortle, Horan, and Abler, 1998).

One reason for these mixed economic results may be, at least in part, a lack of convention in how nonpoint pollution problems are modeled empirically. Nonpoint problems are characterized by several important features, such as stochastic and unobservable emissions, significant heterogeneity in environmental impacts and large numbers of polluters (Braden and Segerson, 1993; Shortle and Abler, 1997). However, different studies account for these features differently. This lack of convention is troublesome because the importance of specific nonpoint characteristics is unclear. Recent studies have begun to rectify this problem. For instance, several studies indicate that it is important to design instruments to account for heterogeneity (e.g., Carpentier, Bosch, and Batie, 1998; Flemming and Adams, 1997; Claassen and Horan, forthcoming; Horan et al., 1999), although this finding is not applicable to all situations (Helfand and House, 1995). Indeed, since individual studies typically only focus on a particular water quality problem in a particular geographical area, the robustness of any results pertaining to the importance of nonpoint features is a concern.

The highly stochastic nature of nonpoint pollution is of particular interest (Braden and Segerson, 1993; Shortle and Abler, 1997). One implication of stochastic pollution is that the economic benefits of pollution control are also stochastic. If these benefits depend nonlinearly on emissions, then a degree of (environmental) risk is associated with production and pollution control choices.[2] We use the term risk to indicate that there are economic benefits to controlling moments of the distributions of environmental and economic outcomes other than just mean emissions. Thus, we distinguish between risk and the level of stochasticity. Risk depends on both the level of stochasticity and the economic value associated with stochasticity.

Risk is important to the extent that it influences optimal policy design and related outcomes. Indeed, stochastic processes are potentially important in determining the relative efficiency of policy instruments, as some instruments account for risk better than others (Shortle, Horan, and Abler, 1998; Horan et al., 1999). For example, input-based instruments can be designed efficiently to account for the risk-effects created by the use of each input. In contrast, instruments based on mean emissions cannot account for all of these risk-effects, and hence cannot be efficient (Shortle, Horan, and Abler, 1998; Shortle and Dunn, 1986).

Relatively few empirical studies have actually modeled stochastic pollution in a meaningful way (e.g., McSweeny and Shortle, 1990; Mapp et al., 1994; Teague, Bernardo, and Mapp, 1995). Instead, most studies are either based on a deterministic specification or only consider policy goals that limit mean environmental impacts, such as mean emissions or a linear aggregation thereof, and do not account for other distributional moments (of environmental impacts) that may have important economic implications (e.g., Helfand and House, 1995; Larson, Helfand, and House, 1996; Litner and Weersink, 1999; Moxey and White, 1994; Flemming and Adams, 1997; Huang, Shank, and Hewitt, 1996; Hopkins, Schitkey, and Tweeten, 1996; Taylor, Adams, and Miller, 1992; Yiridoe and Weersink, 1998; Carpentier Bosch, and Batie, 1998; Claassen and Horan, forthcoming).[3] In contrast, many actual policies focus on distributional moments other than the mean. Examples include treatment requirements for drinking water or capacity regulations for manure storage, both of which are highly attuned to low probability events (e.g., low levels of pathogens in drinking water or a 100 year flood) that have significant health and/or economic impacts (U.S. EPA, 1993; Ohanian, 1992). Hence, empirical studies that do not consider distrubutional moments other than the mean may provide insufficient or even misleading input to the policymaking process.

The objective of this paper is to investigate the economic consequences of not accounting for stochastic processes when designing and comparing alternative nonpoint pollution control instruments. This important question remains largely unanswered in the literature.[4] We begin with a conceptual model to illustrate the issues involved. Next, a simulation is developed to compare the environmental and economic impacts of various instruments when designed under deterministic and stochastic specifications for nonpoint processes. The simulation is constructed as an experiment to determine potential impacts under a wide variety of situations.

## A Model of Nonpoint Pollution

Following Shortle, Horan, and Abler (1998), assume a particular resource (e.g., a lake) is damaged by a single residual (e.g., nitrogen). Economic damages, $D$, are an increasing function of the ambient concentration of the residual, $a$, i.e., $D=D(a)$, $D^{\prime}>0$. Ambient pollution depends on emissions from nonpoint sources, $r_i$ ($i = 1, 2, \ldots, n$), natural generation of the pollutant, $\zeta$, stochastic environmental variables that influence transport and fate, $\delta$, and watershed characteristics and parameters, $\psi$, i.e., $a = a(r_1, r_2, \ldots, r_n, \zeta, \delta, \psi)$ $(\partial a/\partial r_i \geq 0 \ \forall i)$. Nonpoint emissions cannot be observed directly (at least not at an acceptable cost) and, via stochastic variations in environmental drivers (e.g., weather), are stochastic. Accordingly, nonpoint sources can only influence the distribution of their emissions. Emissions depend on an ($m$x1) vector of variable inputs, $x_i$, site-specific, stochastic environmental variables, $v_i$, and site characteristics (e.g., soil type and topography), $\alpha_i$. The relationship for site $i$ is $r_i = r_i(x_i, v_i, \alpha_i)$.

## Risk and Instrument Design

Throughout this paper, we analyze instruments designed to maximize the expected net social benefits from production.[5] Assuming firms are price-takers operating in undistorted, competitive markets, the expected social net benefits from production are defined as consumers' surplus, plus firm quasi-rents, plus any rents that accrue to factors of production not supplied at constant cost to the industry, minus the expected damages from pollution.

### *First-best input taxes*

To see how risk may be important, consider a set of firm-specific taxes applied to each input that influences emissions. The efficient (first-best) tax rates applied to risk-neutral, profit-maximizing firms are of the form (Shortle, Horan, and Abler, 1998)

$$\tau_{ij} = E\left\{D'(a^*)\right\} E\left\{\frac{\partial a^*}{\partial r_i}\right\} E\left\{\frac{\partial r_i^*}{\partial x_{ij}}\right\} + E\left\{D'(a^*)\right\} cov\left\{\frac{\partial a^*}{\partial r_i}, \frac{\partial r_i^*}{\partial x_{ij}}\right\} + cov\left\{D'(a^*), \frac{\partial a^*}{\partial r_i}\frac{\partial r_i^*}{\partial x_{ij}}\right\} \quad \forall i,j \qquad (1)$$

where $\tau_{ij}$ is the tax applied to the $j$th input of the $i$th firm, and where the superscript * indicates that the RHS expression is evaluated at the ex ante efficient solution. The optimal tax rate for input $j$ for firm $i$ equals expected marginal damages, times the expected marginal increase in ambient pollution levels from firm $i$'s emissions, times the expected increase in emissions from increased use of input $j$ at the margin, plus two covariance terms that act as risk premiums or rewards, depending on the signs. The tax rate may be positive or negative depending on the signs and relative magnitudes of the three RHS terms in equation (1).

The sign of the first RHS term will be positive for pollution-increasing inputs and negative for pollution-decreasing inputs. The signs of the risk terms are ambiguous without further specification. If $a$ is convex in runoff, then the first covariance term is of the same sign as $\partial Var(r_i^*)/\partial x_{ij}$. Thus, when $a$ is convex, risk and hence $\tau_{ij}$ are increased when an increase in the use of the input increases the variance of runoff.[6] Similarly, when $D^{//} > 0$, risk and hence $\tau_{ij}$ are increased when an increase in the use of the input increases the variance of *a*. However, if *a* is concave in runoff and/or if $D^{//} < 0$, then increases in the variance of runoff and/or ambient pollution have the opposite effect on $\tau_{ij}$. Greater variability of environmental outcomes would be socially beneficial in such cases, which are quite plausible. Ecosystem health and associated economic impacts may be realistically modeled by "S" shaped impact functions which have both convex and concave segments (Hershaft et al., 1978). In any case, the first risk term is generally nonzero when ambient pollution is a nonlinear function of emissions, while the second risk term is generally nonzero when damages are a nonlinear function of ambient pollution.

The risk terms are clearly important to the extent that they affect instrument levels and have economic consequences. Accordingly, policy prescriptions could be subject to significant error if instrument levels are derived from a deterministic model (a mis-specified model in which the distributions of all random variables are ignored) when, in reality, pollution is stochastic and risk is important. This can be seen by comparison of (1) to tax rates derived from a mis-specified model, which are of the form

$$\tau_{ij}^{/} = D^{/}(a^{**}) \frac{\partial a^{**}}{\partial r_i} \frac{\partial r_i^{**}}{\partial x_{ij}} \qquad \forall i, j \tag{2}$$

where the supercript ** denotes that the RHS expression is evaluated at the optimal solution from the mis-specified model.

The tax rates defined in (2) differ from those in (1) in three respects. First, there are no expectations operators in (2) since the mis-specified model is deterministic. Thus, other things being equal, the first RHS term in (1) will differ from the RHS of (2). Analytically, the sign of this difference is ambiguous and depends on the distributions of the random variables and how they enter into environmental relations and marginal damages. Consider the marginal damage term as an example. Only the means of random variables will matter if marginal damages are a linear function of the random variables, whereas the means, variances, and covariances of the random variables will matter if marginal damages are a quadratic function of the random variables. Failure to account for these moments can therefore affect the level of the tax and, accordingly, input use by firms -- even those inputs that have no risk-effects. Other things being equal, a positive (negative) difference between the first RHS term in (1) and the RHS of (2) indicates that taxes derived from the mis-specified model will be too low (high) and will not fully transmit the costs (benefits) that this term represents.

A second difference is that there are no risk terms in (2). The tax rate $\tau_{ij}^{/}$ therefore does not account for social costs stemming from environmental risk. The implication is that more risk will result in larger taxes and smaller subsidies in (1) than in (2), other things being equal, and thus the allocation of pollution control efforts across input choices and individual firms will differ between the two models. There will be incentives under the mis-specified model for firms to use risk-increasing inputs at inefficiently high levels and to under-employ risk-reducing inputs. Additionally, firms with greater contributions to environmental risk at the margin will face incentives to adopt inefficiently lax pollution controls while firms with smaller risk contributions will face incentives to adopt inefficiently stringent controls when the mis-specified model is applied. The differences in the pollution control allocations that arise from the two models may have important implications for the allocation of economic gains and losses to those with an economic interest in pollution control.

A final point of comparison is that the allocations at which (1) and (2) are evaluated will differ. The quantitative implications of this difference are generally ambiguous without further specification. As an example, suppose pollution-increasing and risk-increasing inputs are positively correlated and that damages and all environmental relations are convex. With risk-increasing inputs being used in larger quantities under (2) (due to the lack of risk-terms), marginal damages and marginal environmental impacts will be larger. The effect is a larger tax in (2), somewhat offsetting the lower tax rates that result when there are no risk-terms. Other specifications may exacerbate differences in tax rates. Generally, differences between (1) and (2) depend on the specification of the model, with the shape of the damage, ambient, and emissions functions, as well as substitution, output, and price effects being particularly important.

*Second-best instruments*

Similar comparisons can be made for second-best instruments. The form of second-best input taxes that are applied uniformly across producers (when a differentiated structure is preferred) to only a subset of inputs that affect emissions, and the form of second-best, uniformly applied taxes based on expected emissions are derived in Shortle, Horan, and Abler (1998) and Horan et al. (1999) and are presented in Table 1 for both the correctly-specified and mis-specified models.[7] In each case, the single correct tax rate depends on covariance terms involving all producers and all inputs, whereas a first-best tax rate depends only on convariances involving a single input used by a single producer. These additional covariances, which occur because the instruments are not differentiated across producers and their input use, represent additional sources of divergence between the correctly-specified and mis-specified models, relative to the first-best case.

Deterministic modeling efforts can affect the (perceived and actual) relative economic performance of alternative instruments in addition to their absolute performance. For example, ambient taxes and taxes based on mean emissions may appear to be first-best under a deterministic specification, but they can only be second-best in a stochastic setting (Horan, Shortle, and Abler, 1998; Shortle and Dunn, 1986). This is because instruments based on mean environmental performance do not adequately provide incentives for firms to consider how their choices impact the variance and higher moments of the distribution of environmental outcomes.

**A Simulation Model**

Little can be said analytically about how the design of different instruments and their associated economic consequences might be affected by deterministic modeling when pollution is stochastic. However, we should be able to say something of policy relevance by specifying the model in a realistic fashion. Therefore, to gain further insight, we have developed a simulation experiment involving one thousand independent, hypothetical watersheds. The use of hypothetical watersheds permits complete control over the design of the experiment and, by comparison to one or a small number of case studies of actual watersheds, increases our ability to investigate these issues for a variety of conditions. Although the watersheds are hypothetical, significant effort was taken to ensure the relationships are representative of more realistic settings, particularly those involving agricultural sources -- the most important source of remaining water quality problems in the U.S. (USDA and USEPA, 1998).

The simulation model has the same general structure as standard conceptual models of agricultural nonpoint pollution (e.g., Shortle and Dunn, 1986; Shortle, Horan, and Abler, 1998). Specifically, each watershed contains four nonpoint sources, where each source essentially represents classes of producers that vary according to cost structure and environmental impacts. These variations are taken to occur at the sub-watershed level, so that each source represents aggregate production within a region. Producers in each region operate in competitive markets, taking prices as given, although the commodity (corn) price is endogenous to the watershed and land prices are endogenous to each region.[8] Production is a two-level CES function of a composite 'biological' input (land and fertilizer) and a composite 'mechanical' input (capital and labor). Details of production and input and output markets are provided in the appendix.

Nonpoint emissions (runoff) per acre are influenced by excess fertilizer use (i.e., fertilizer not taken up by the crop) per acre as well as a stochastic, weather related term. Specifically, farm *i*'s runoff per acre, $r_i/x_{i1}$ (where $x_{i1}$ is land), is a second-order approximation of actual per acre runoff, which is taken to be an increasing, convex function of excess fertilizer use per acre, $g_i$, i.e, $r_i/x_{i1} = b_{1i}g_i + b_{2i}g_i^2 + v_i g_i$, where $g_i = x_{i2}/x_{i1}$, $x_{i2}$ is excess fertilizer, and $v_i$ is a random variable with zero mean. The specification for the random term is consistent with that of Just and Pope (1978). In particular, a larger value of $g_i$ (due to either more fertilizer or less land) results in a larger mean and variance of $r_i/x_{i1}$.

Runoff from each source is transported to a water body according to a stochastic (due to weather) process, although only a fraction of the runoff generated at each site becomes part of the ambient pollution concentration in the water body. The proportion of the runoff that is transported is modeled as a constant transport coefficient, $\varpi_i$. In aggregate, pollution transport and the resulting ambient pollution levels are reasonably represented by a first-order approximation (Roth and Jury, 1993) based on the sum of the transported runoff (loadings) from all sources, $a = (\psi + \delta)L$, where $L = \sum_{i=1}^{n} \varpi_i r_i$ is loadings, $\psi$ is a deterministic parameter, and $\delta$ is a random variable with zero mean. Thus, more loadings result in a greater mean and variance of *a*. Finally, the resulting ambient pollution concentration creates economic damages, denoted *D*. Economic damages are a second-order approximation of actual damages, which is taken to be an increasing, convex function of *a*, i.e., $D = d_1 a + d_2 a^2$.

More details of the model and data used for calibration are provided in the appendix. In particular, the elasticities and other parameters used to calibrate the model are drawn from a literature that reports a range of values. This parameter uncertainty is dealt with through a Monte Carlo (sensitivity) analysis in which the uncertain values are randomly distributed. Each of the one thousand watersheds in the model is developed from a single draw of all uncertain parameter values, and the results from each watershed are used to form a distribution of results. More details of this procedure are also provided in the appendix. However, note the distinction between parameter uncertainty and stochastic variables. In each draw, parameters are treated as deterministic while stochastic variables remain stochastic.

We obtain results for four alternative schemes to reduce nutrient runoff: efficient input taxes (defined by firm-specific taxes applied to fertilizer and land), uniform fertilizer taxes, firm-specific taxes based on mean runoff, and uniform mean runoff taxes. These schemes have real world analogues. Measures to regulate fertilizer use, primarily in the form of fertilizer quotas or taxes, are a common feature of policy proposals to reduce nutrient pollution, and have been implemented in some states in the U.S. and Europe (Leuk, 1994; Ribaudo, 1998). Crop land retirement, in the form of the Conservation Reserve Program, is a major approach to agricultural nonpoint pollution control in the U.S. (USDA-ERS, 1997).

For each scheme, first- or second-best instruments and associated welfare measures are determined taking all stochastic components into consideration (the correct model). Values of all instruments and associated welfare measures are also determined optimally under an incorrect, deterministic specification in which all random variables are evaluated at their means (the perceived results of the mis-specified model). In addition, the actual distributions of environmental and economic results (i.e., taking the stochastic nature of pollution into consideration) are determined given the values of the instruments derived using the mis-specified model (the actual results of the mis-specified model). (The model is constructed so that the perceived and actual results of the mis-specified model will only differ with respect to expected damages and hence expected social net benefits). Large differences in results from the correctly specified and mis-specified models indicate the need for a stochastic specification, while small differences suggest a deterministic specification may be adequate.

Obviously, the specification of stochastic processes and risk may have an important impact on results.[9] To ameliorate any potential biases due to model construction, the stochastic processes and risk components have been specified quite simplistically (although realistically). Ambient pollution is a linear function of the random components, each of which is independently distributed. Consequently, the quadratic damage term is the only source of risk in the model (e.g., the first covariance term on the RHS of (1) vanishes) given the linear and independent specification of stochastic terms. The Monte Carlo analysis also helps to alleviate any bias due to model construction by providing results for a range of risk components and distributions of stochastic variables. This allows us to compare results from watersheds with little variability in stochastic components and little risk with results from watersheds in which these stochastic and risk elements are more important.

## Results

### *Tax rates*

We begin by comparing, for each policy scenario, the optimal tax rates derived in the correctly specified model (the correct tax rates) with those derived in the mis-specified model (the incorrect tax rates). The results are expressed in Table 2 as the percentage difference between the correct and incorrect tax rates (i.e., 100(correct tax - incorrect tax)/correct tax).

First, consider the input tax scenarios. The efficient and uniform fertilizer tax results are similar, with the correct fertilizer tax rates being larger than the incorrect tax rates in all samples. Specifically, the difference between correct and incorrect tax rates range from 0.12% (in the case of very little risk) to 84% (in the case of substantial risk), with a sample average difference of 30% to 37%. The incorrect tax rates are smaller in each case because fertilizer is a risk-increasing input and, as described above, there are incentives under the mis-specified model for farms to use risk-increasing inputs at inefficiently high levels. Note that the difference between correct and incorrect tax rates is proportionately larger in Regions 1 and 3, which are greater contributors to risk due to the greater proportion of runoff that is transported from these regions. Also, on average, the difference in correct and incorrect tax rates are proportionately smaller in Region 4 than in Region 2, which contributes more to risk since fertilizer is used more intensively in this region.

For the case of efficient land subsidies, the correct subsidies are 34% larger than incorrect subsidies in Region 1 and 10.5% larger than incorrect subsidies in Region 3, on average. The range of differences in these regions is large, however, extending from -50% to 248%. Positive differences generally occur in Regions 1 and 3 because land is a risk-reducing input (in terms of its impact on the variance of ambient pollution), and so there will be marginal incentives under the mis-specified model for farms in these regions to under-employ land (i.e., smaller subsidies). However, the larger fertilizer taxes in these regions under the correct model result in proportionately larger output effects and hence proportionately smaller derived demands for land. The net effect is that less land is employed in Regions 1 and 3 under the correct model than in the mis-specified model. In contrast, the correct subsidies are 26% smaller than incorrect subsidies in Region 2 and 27% smaller than incorrect subsidies in Region 4, on average. The range of differences in these regions is also large (but smaller than that of Regions 1 and 3), extending from –80% to 25%. Negative differences generally occur in Regions 2 and 4. This is because the larger output effects in Regions 1 and 3 under the correct model drive up the output price, resulting in proportionately larger increases in the dervied demand for land in Regions 2 and 4 under the correct model. Thus, less of a subsidy is required than in the mis-specified model.

Now consider the expected runoff tax scenarios. The expected runoff tax encourages farms to substitute land for fertilizer, and also to reduce output and hence the use of both of these inputs. Reducing fertilizer use and/or increasing land use reduce risk; however, the incorrect model does not take these additional benefits into account (even in the correct model, these risk-impacts cannot both be efficiently managed using an expected runoff tax (Shortle and Dunn, 1986)). Accordingly, for almost every sample, fertilizer use and land use are too high in Regions 1 and 3 (which have the greatest risk-impacts due to larger transport coefficients) in the mis-specified model, with a net effect of too much risk due to greater fertilizer use. These input use decisions correspond to smaller incorrect taxes relative to the correct taxes. Specifically, the difference between correct and incorrect tax rates in these regions range from 0.16% to 89%, with sample average differences of 39.5% and 38%. Greater fertilizer use and land use in the mis-specified model are accompanied by more output and a smaller output price, reducing the derived demand for land and fertilizer in Regions 2 and 4. This also results in smaller incorrect taxes relative to correct taxes for the vast majority of the samples, although with less fertilizer and land (and hence output) due to the smaller output price. However, these smaller taxes are not reflected in Table 2 as the incorrect taxes are substantially larger than the correct taxes on average. These large mean differences occur in a small percentage of samples in which the correct taxes for Regions 2 and 4 are essentially zero due to the small contribution these regions make towards ambient pollution and also risk, while the taxes for Regions 1 and 3 are large due to their significant risk contributions in these samples. Thus, any percentage change from the correct taxes in Regions 2 and 4 in these cases will necessarily be large enough to bias the entire distribution of results. As it happens, the incorrect tax rates are actually slightly positive in these limited samples, primarily because pollution control is reallocated from Regions 1 and 3 since the mis-specified model does not recognize their contribution to risk (and thus the mean contributions of Regions 2 and 4 towards ambient pollution are seen as more important). Thus, farmers who would not optimally bear pollution control costs are subjected to taxes when risk is not considered.

*Welfare levels*

We now compare welfare levels resulting from the correctly specified model (correct welfare) with those resulting from the mis-specified model (incorrect welfare, i.e., actual welfare, as measured by the correct model, based on producer responses to the taxes derived in the incorrect model). We do this in two ways. First, for each policy scenario, we compare correct and incorrect welfare directly (Table 3), as well as incorrect welfare and the perceived welfare that results under the mis-specified model (perceived welfare, i.e., welfare predicted by the mis-specified model) (Table 4). Second, we compare the relative (perceived) performance of the various policy approaches under the two models, where performance is measured in terms of the various welfare measures (Table 5).

**Differences in absolute performance (actual and perceived).** First, consider a direct comparison of correct and incorrect welfare. The results are expressed in Table 3 as percentage differences from the correct welfare measures. As is required, the correct model results in larger expected net social benefits, although these benefits do not differ much between the correct and mis-specified models. Even in samples in which risk is significant, the differences are only moderate at around 8% for all instruments. However, differences in the welfare accruing to different groups with an interest in production are significant in most cases. Differences in incorrect and perceived welfare (Table 4, where the results are expressed as percentage differences from the incorrect welfare measures) follow an opposite pattern. The incorrect model consistently overestimates expected social net benefits, by as much as 22% in some samples. These differences are due to inaccurate estimates for expected damages, which are discussed below.

Consumers' surplus is smaller under the correct model for all samples, indicating that optimally managing risk results in an output reduction (Table 3). The reductions in consumers' surplus range from minuscule in samples with minimal risk to almost 40% in samples with significant risk, with an average reduction of about 6% for differentiated policy instruments to more than 11% for uniform instruments. Since consumers' surplus is unaffected by risk, there are no perceived differences from actual results.

The biggest differences in welfare occur with respect to expected damages (Table 3). Expected damages are smaller under the correct model for all samples, indicating a significant welfare improvement from accounting for risk. The reductions in expected damages range from almost nothing in samples with minimal risk to almost 386% in samples with significant risk, with an average reduction of about 71% for differentiated policy instruments to about 55% for uniform instruments. Differences in incorrect and perceived damages (Table 4) follow an opposite, although less pronounced, pattern. The mis-specified model consistently underestimates expected damages by 41% to 44% on average across samples, and by as much as 91% in some samples, due to the fact that the mis-specified model does not value risk.

Finally, consider the differences in landowners' surplus under the correct and mis-specified models (Table 3). In aggregate, the differences are generally small on average, with moderate positive or negative differences occurring in some samples. However, differences in the returns to landowners in particular regions may be quite large, even on average. As with consumers' surplus, there are no perceived differences from actual results since landowners' surplus is unaffected by risk.

**Differences in perceived relative performance.** Now consider the relative (perceived) performance of the various policy approaches under the two models, where performance is measured in terms of the various welfare measures (Table 5). In the correct model, for example, efficient taxes always result in greater expected social net benefits, greater consumers' surplus, and smaller expected damages than any of the other policy scenarios, and generally result in greater landowners' surplus than any of the other scenarios (except for the uniform expected runoff tax). In contrast, the mis-specified model predicts that the welfare impacts of efficient taxes are equivalent to those of non-uniform expected runoff taxes, and overpredicts the number of samples for which the efficient tax produces greater landowners' surplus than the uniform fertilizer tax and the uniform expected runoff tax. However, incorrect predictions are not necessarily a problem unless the welfare measures being compared are significantly different from each other. For example, mean social net benefits under efficient taxes are only 0.01% larger than those under a non-uniform expected runoff tax, and similar mean differences with respect to the other welfare measures are also less than 1% in this case. Thus, it matters little whether efficient taxes or non-uniform runoff taxes are applied, which is interesting because much has been made of the fact that non-uniform expected runoff taxes are inefficient due to their inability to account for risk (Shortle and Dunn, 1986).[10] However, differences in landowners' surplus are almost 3% larger in Region 1 and 1.5% larger in Region 3 on average (and larger in many other cases) under non-uniform taxes (not reported in Table 5), which may be significant in monetary terms.

For comparisons involving other combinations of instruments, the mis-specified model generally makes accurate comparisons on the basis of expected social net benefits, consumers' surplus, and expected damages. The only significant exceptions are for consumer's surplus and expected damages for comparisons involving uniform fertilizer taxes and uniform expected runoff taxes. The mis-specified model has more difficulty when comparing landowners' surplus, particularly when broken down by region.

*Extreme events*

Finally, accounting for environmental risk is important in terms of how policy instruments influence the probability of extreme events. Our particular concern lies with the probability of unwanted extreme events, such as excessive runoff and associated levels of ambient pollution and economic damages. A point of reference is needed to define an extreme event. Therefore, for each policy scenario and for each environmental performance measure, we define an extreme event as one in which the performance measure takes on a value in excess of two standard deviations above the mean, where the relevant mean and standard deviation are those resulting in an optimal solution using the correct model. For example, consider a uniform fertilizer tax. First, the optimal tax is derived in both the correct and mis-specified models. Next, the mean and standard deviation of each performance measure (runoff, ambient pollution, and damages) is calculated given the production choices resulting from the correct tax. These statistics are used to determine extreme values as described above. Finally, the probability of exceeding these values is calculated and compared for the uniform fertilizer taxes in the correct and incorrect models.

The results of this exercise are presented in Table 6. For simplicity, a Monte Carlo approach was not used in deriving these results. Instead, mean values were used for all uncertain parameter values. The associated extreme critical values for each policy are as follows: runoff critical values are on average 116% larger than mean runoff in each region, ambient pollution critical values are on average 139% larger than mean ambient pollution, and damage critical values are on average 196% larger than mean damages.

Table 6 clearly shows that the probability of extreme events is larger when the mis-specified model is applied. This is expected. However, the differences in the probabilities associated with extreme events is substantial, differing by a factor of three or four for the case of ambient pollution, and a factor of three for damages. Thus, policy measures that do rely on risk information are much less likely to result in severe environmental outcomes, such as a massive fish kill due to nutrient over-enrichment, than policy measures that take this information into account.

## Conclusion

Frequently, policy regarding the control of pollution or of natural events, such as floods, focuses on reducing the risk of the extreme events. This focus is natural: mean levels of these events may affect more people more of the time, but of greater political concern are relatively rare extreme events, which may affect only a few people, but in catastrophic ways. An example is a pathogenic outbreak in a local water supply. Large sums of money are expended to protect drinking water from such occurrences. The political costs of an outbreak can be high, particularly when children or the elderly become ill and/or die as a result. An economic model that acknowledges only mean water quality events will not capture the range of possible welfare impacts. Hence, a model that does not consider the stochastic aspects of physical processes, as is the case for most existing, economic models for the control of pollution or other natural events, will be less than satisfactory for policymaking purposes. Furthermore, even the mean welfare impacts estimated by these models may be incorrect.

This paper examines the implications of explicitly considering the stochastic nature of environmental processes when developing nonpoint pollution control policies. We find that even with a quite simplistic specification for risk, several important results arise from this analysis. First, risk has important impacts on the magnitudes of policy instruments. Second, the impacts of risk on expected net benefits are relatively small, while the impacts of risk on the allocation of welfare are relatively large in many cases. This result is significant for policymaking purposes, given that the allocation of welfare may be of more importance than the aggregate welfare level. Third, the perceived welfare calculated from a mis-specified model differs significantly from actual welfare levels that result when using policy choices derived from the mis-specified model. Thus, the actual impacts of a policy are likely to differ significantly from the predictions of naive, deterministic models. Somewhat surprisingly, these incorrect perceptions actually result in the mis-specified model and the correct model yielding almost identical comparisons of the relative performance of various first-best and second-best instruments, although the mis-specified model does not always make accurate comparisons regarding the returns to consumers and landowners under the various policy approaches. Finally, we find that deriving optimal policy instrument levels using risk-based models significantly reduces the probability of unwanted, extreme events such as excessive pollution and damage levels.

**Appendix**

The simulation model closely follows that of Claassen and Horan (Forthcoming). In each watershed, four nonpoint sources produce a single, identical agricultural commodity (corn) according to a constant returns to scale, two-level CES technology (Sato, 1967). Corn production depends on a composite biological input and a composite mechanical input. The biological input is produced using land and fertilizer according to a constant returns to scale CES technology. The mechanical input depends on labor and capital, but is not decomposed into these inputs because labor and capital prices are held fixed and hence labor and capital are used in constant proportions. Production heterogeneity is created through input cost shares (Table A1), with farms 1 and 2 using fertilizer more intensively on a per acre basis than farms 3 and 4. Initial outputs and costs are identical across farms to reduce the impacts of scale effects among sources since heterogeneity does not occur along these lines. Aggregate revenue and costs for this sector equal one. With all input and output prices set equal to one initially, output equals revenue and inputs equal factor costs.

Output and land prices are endogenous. The output market is at the watershed level and output demand is modeled as a first-order approximation of actual demand. In contrast, land supply takes a constant elasticity form and is defined for each source (i.e., if each source represents aggregate production in a region of the watershed, then land supply is defined at the regional level).

On the environmental side, runoff, ambient pollution, and damage functions are desribed in the text. Environmental heterogeneity is created by farms 1 and 3 having larger initial average runoff per acre (i.e., $r_i/(x_{i1}g_i)$) on average (Table A1; see also discussion of Monte Carlo analysis below). Transport coefficients represent another important source of environmental heterogeneity as farms 1 and 3 on average have higher transport coefficients than other farms (Table A1; see also discussion of Monte Carlo analysis below). Mean ambient pollution equals one initially. Finally, economic damages from pollution is calibrated by setting initial expected damages equal to 20% of initial net benefits (similar to an upper bound reported by Smith (1992) for groundwater damages) and by choosing an elasticity of expected damages (Table A1).

The impacts of stochastic environmental terms are modeled using a Gaussian Quadrature to provide an exact measure of expected damages and related terms (Miller and Rice, 1983; Preckel and DeVuyst, 1992). Since $r_i$ and $a$ are linear in the random variables and damages are quadratic, each random variable only needs to be evaluated at two points to provide an exact measure of all relevant expected values (Miller and Rice, 1983). The joint distribution for the five random variables therefore consists of 32 points.

A number of elasticities and other parameters are needed to calibrate the model. However, the literature reports a range of values. To deal with this parameter uncertainty, we follow Abler and Shortle (1995) and Davis and Espinoza (1998) and perform a Monte Carlo (sensitivity) analysis to obtain a distribution of ex post results.[11] Specifically, the model is solved one thousand times, taking many parameter values as randomly and independently distributed. Each iteration represents a single draw of all uncertain parameter values and, at each iteration, parameter values are assumed known with certainty. In effect, each iteration represents an individual watershed. Uncertain parameter values are all assumed to be uniformly distributed according to reasonable bounds suggested by the literature. The parameters and their distributions are also reported in Table A1. Source-specific values are allowed to differ at each iteration, although source-specific values of a particular parameter are all taken from the same distributions (unless specified otherwise). The sample size of one thousand is large enough to obtain fairly tight confidence intervals around the sample expected net benefits for each scheme.

## References


Abler, D.G., and J.S. Shortle. "Technology as an Agricultural Pollution Control Policy." *American Journal of Agricultural Economics* 77 (Feb. 1995): 20-32.

Binswanger, H.P. "The Measurement of Technical Change Biases with Many Factors of Production." *American Economic Review* 64(Dec. 1974): 964-76.

Braden, J.B. and K. Segerson. "Information Problems in the Design of Nonpoint-Source Pollution Policy." *Theory, Modeling, and Experience in the Management of Nonpoint-Source Pollution*. C.S. Russell and J.F. Shogren, eds. Boston: Kluwer Academic Publishers, 1993.

Carpentier, C.L., D.J. Bosch, and S.S. Batie. "Using Spatial Information to Reduce Costs of Controlling Nonpoint Source Pollution." *Agricultural and Resource Economics Review* 27 (1998): 72-84.

Chambers, R.G. and U. Vasavada. "Testing Asset Fixity for U.S. Agriculture." *American Journal of Agricultural Economics* 65(Nov. 1983): 761-69.

Chavas, J.P. and M.T. Holt. "Acreage Decisions Under Risk: The Case of Corn and Soybeans." *American Journal of Agricultural Economics* 72(Aug. 1990): 529-38.

Claassen, R., and R.D. Horan. "Uniform and Non-Uniform Second-Best Input Taxes: The Significance of Market Price Effects on Efficiency and Equity." *Environmental and Resource Economics*, Forthcoming.

Davis, G.C. and M.C. Espinoza. "A Unified Approach to Sensitivity Analysis in Equilibrium Displacement Models." *American Journal of Agricultural Economics* 80(Nov. 1998): 868-879.

Fernandez-Cornejo, J. "Short- and Long-Run Demand and Substitution of Agricultural Inputs." *Northeast Journal of Agricultural and Resource Economics* 21(Apr. 1992): 36-49.

Fisher, D., J. Ceraso, T. Mathew, and M. Oppenheimer. *Polluted Coastal Waters: The Role of Acid Rain*. New York: Environmental Defense Fund, 1988.

Flemming, R.A., and R.M. Adams. "The Importance of Site-Specific Information in the Design of Policies to Control Pollution." *Journal of Environmental Economics and Management* 33 (1997): 347-358.

Hallberg, G.R. "Nitrates in Groundwater in Iowa," in *Rural Groundwater Contamination*, F.M. D'Itri and L.G. Wolfson, eds. Chelsea, MI: Lewis Publishing, 1987.

Helfand, G. and B. House. 1995. "Regulating Nonpoint Source Pollution Under Heterogeneous Conditions." *American Journal of Agricultural Economics* 77 (Nov. 1995): 1024-1032.

Hershaft, A., A.M. Freeman III, T.D. Crocker, and J.B. Stevens. *Critical Reviews of Estimating Benefits of Air and Water Pollution Control*. Washington, DC: U.S. Environmental Protection Agency, EPA-600/5-78-014, 1978.

Hertel, T.W. "Negotiating Reductions in Agricultural Support: Implications of Technology and Factor Mobility." *American Journal of Agricultural Economics* 71(Aug. 1989): 559-73.

Holt, M. T. "A Multimarket Bounded Price Variation Model Under Rational Expectations: Corn and Soybeans in the United States." *American Journal of Agricultural Economics* 74(Feb. 1992): 10-20.

Hopkins, J., G. Schnitkey, and L. Tweeten. "Impacts of Nitrogen Control Policies on Crop and Livestock Farms at Two Ohio Farm Sites." *Review of Agricultural Economics* 18 (1996):311-324.

Horan, R.D., J.S. Shortle, and D.G. Abler. "Ambient Taxes when Polluters Have Multiple Choices." *Journal of Environmental Economics and Management*, 36 (1998): 186-199.

Horan, R.D., J.S. Shortle, M.O. Ribaudo, and D.G. Abler. "Second-Best Markets for Point/Nonpoint Pollution Trading." Contributed Paper, the Ninth Annual Conference of the European Association of Environmental and Resource Economists, Oslo, Norway, June 25-27, 1999.

Huang, W-Y, D. Shank, and T.I. Hewitt. "On Farm Costs of Reducing Residual Nitrogen on Cropland Vulnerable to Nitrate leaching." *Review of Agricultural Economics* 18 (1996): 325-339.

Just, R.E., and R.D. Pope. "Stochastic Specifications of Production Functions and Economic Implications." *Journal of Econometrics* 7 (1978): 67-86.

Kawagoe T., K. Otsuka, and Y. Hayami. 1985. "Induced Bias of Technical Change in Agriculture:The United States and Japan, 1880-1980." *Journal of Political Economy*. 94(3):523-544.

Keeney, D.R. "Nitrogen Management for Maximum Efficiency and Minimum Pollution: Farmed Soils, Fertilizer, Agro-systems." *Nitrogen in Agricultural Soils*, F.J. Stevenson, ed. Madison, WI: American Society of Agronomy, 1982.

Koutsoyiannia, D. "Optimal Decomposition of covariance matrices for multivariate stochastic models in hydrology." *Journal of the American Water Resources Association* **35** (1999): 1219-1229.

Larson, D., G. Helfand, and B. House. "Second-Best Tax Policies to Reduce Nonpoint Source Pollution." *American Journal of Agricultural Economics* 78 (Nov. 1996): 1108-1117.

Lee, D.R. and P.G. Helmberger. "Estimating Supply Response in the Presence of Farm Programs." *American Journal of Agricultural Economics* 67(May 1985): 193-203.

Leuck, D. "The EC Nitrate Directive and its Potential Effects on EC Livestock Production and Exports of Livestock Products", in *Environmental Policies: Implications for Agricultural Trade* (J. Sullivan, Ed.), U.S. Department of Agriculture, Economic Research Service FAER-252, Washington, DC, 1994.

Litner, A.M., and A. Weersink. "Endogenous Transport Coefficients: Implications for Improving Water Quality from Multi-Contaminants in an Agricultural Watershed." *Environmental and Resource Economics* 14 (1999): 269-296.

Manguerra, H.B., and B.A. Engel. "Hydrologic Parameterization of Watersheds for Runoff Prediction Using SWAT." *Journal of the American Water Resources Association* **34** (1998): 1149-1162.

Mapp, H.P., D.J. Bernardo, G.J. Sabbagh, S.Geleta, and K.B. Watkins. "Economic and Environmental Impacts of Limiting Nitrogen Use to Protect Water Quality: A Stochastic Regional Analysis" *American Journal of Agricultural Economics* 76 (Nov. 1994): 889-903.

McSweeny, W.T., and J.S. Shortle. "Probabilistic Cost-Effectiveness in Agricultural Nonpoint Pollution Control." *Southern Journal of Agricultural Economics* 22 (1990): 95-104.

Miller, A.C. and T.R. Rice. "Discrete Approximations of Probability Distributions." *Management Science* **29** (1983): 352-362.

Moxey, A., and B. White. "Efficient Compliance with Agricultural Nitrate Pollution Standards." *Journal of Agricultural Economics* 45 (1994): 27-37.

National Research Council. *Soil and Water Quality: An Agenda for Agriculture*. Washington, DC: National Academy Press, 1993.

Ohanian, E.V. "New Approaches in Setting Drinking Water Standards." *Journal of the American College of Toxicology* **11** (1992): 321-324.

Peterson, G.A. and W.W. Frye. "Fertilizer Nitrogen Management." *Nitrogen Management and Ground Water Protection*. R.F. Follet, ed. Amsterdam: Elsevier, 1989.

Preckel, P.V., and E. DeVuyst. "Efficient Handling of Probability Information for Decision Analysis under Risk." *American Journal of Agricultural Economics* **74** (1992): 655-662.

Ray, S.C. "A Translog Cost Function Analysis of U.S. Agriculture, 1939-77." *American Journal of Agricultural Economics* 64(Aug. 1982): 490-98.

Ribaudo, M. "Managing Agricultural Non-Point Source Pollution: Are States Doing the Job?" *Devolution in Environmental Policy.* D. Southgate, ed., Proceedings of the Resource Policy Consortium Symposium, Washington, DC, 1998.

Ribaudo, M.O., R.D. Horan, and M.E. Smith. *Economics of Water Quality Protection from Nonpoint Sources - Theory and Practice*, ERS Agricultural Economics Report, AER-782, November, 1999.

Roth K., and W.A. Jury. "Modeling the Transport of Solutes to Groundwater Using Transfer Functions." *Journal of Environmental Quality* **22** (1993): 487-493.

Sato, K. "A Two-Level Constant-Elasticity of Substitution Production Function." *Review of Economic Studies*, 34 (1967): 210-18.

Shortle, J.S. and D.G. Abler. "Nonpoint Pollution."*International Yearbook of Environmental and Natural Resource Economics 1997/98*. H. Folmer and T. Tietenberg, eds. Cheltenham, UK: Edward Elgar, 1997.

Shortle, J.S., and J.W. Dunn. "The Relative Efficiency of Agricultural Source Water Pollution Control Policies." *American Journal of Agricultural Economics* **68** (1986): 688-697.

Shortle, J.S., R.D. Horan, and D.G. Abler. "Research Issues in Nonpoint Pollution Control." *Environmental and Resource Economics* 11 (1998): 571-585.

Smith, R.A., G.E. Schwarz, and R.B. Alexander, "Regional Interpretation of Water-Quality Monitoring Data." *Water Resources Research* 33 (1997): 2781-2798.

Smith, V.K. "Environmental Costing for Agriculture: Will it be Standard Fare in the Farm Bill of 2000?" *American Journal of Agricultural Economics* 74 (Feb. 1992): 1076-1088.

Taylor, M., R.M. Adams, and S.F. Miller. "Farm-Level Response to Agricultural Effluent Control Strategies: The Case of Willamette Valley." *Journal of Agricultural and Resource Economics* 17 (1992): 173-185.

Teague, M.L., D.J. Bernardo, and H.P. Mapp. "Farm Level Economic Analysis Incorporating Stochastic Environmental Risk Assessment." *American Journal of Agricultural Economics* 77 (1995): 8-19.

Tegene, A., W.E. Huffman, and J.A. Miranowski. "Dynamic Corn Supply Functions: A Model with Explicit Optimization." *American Journal of Agricultural Economics* 70(Feb. 1988): 103-11.

Thirtle, C.G. "Accounting for Increasing Land-Labour Ratios in Developed Country Agriculture." *Journal of Agricultural Economics* 36(May 1985): 161-69.

U.S.D.A., Economic Research Service, *Agricultural Resources and Environmental Indicators, 1996-97*, ERS Agricultural Handbook Number 712, U.S. Department of Agriculture, Washington, DC, 1997.

U.S.D.A., Economic Research Service. *State-Level Cost of Production - Major Field Crops*. ERS world-wide web site, http://www.econ.ag.gov.

U.S.D.A., National Agricultural Statistical Service. *Agricultural Statistics 1990.* Washington D.C: U.S. Government Printing Office, 1990.

U.S. EPA, *Water Quality Standards Handbook, Second Edition*. Washington, DC: U.S. Environmental Protection Agency, Office of Water, 1993

U.S. E.P.A. and U.S.D.A. *Clean Water Action Plan: Restoring and Protecting America's Waters*. EPA-840-R-98-001. Washington, DC: U.S. Environmental Protection Agency, 1998.

Weinberg, M. and C. L. Kling. "Uncoordinated Agricultural and Environmental Policy Making: An Application to Irrigated Agriculture in the West." *American Journal of Agricultural Economics* 78(Feb. 1996): 65-78.

Yiridoe, E.K., and A. Weersink. "Marginal Abatement Costs of Reducing Groundwater-N Pollution with Intensive and Extensive Farm Management Choices." *Agricultural and Resource Economics Review* 27 (1998): 167-185.

Table 1. Second-Best Taxes Derived Under Correct and Mis-Specified Models

| Tax base | Correctly Specified Model | Mis-Specified Model |
|---|---|---|
| Input $u$ | $b_u^{\#} = \sum_{i=1}^{n} E\{D(a^{\#})\} E\left\{\frac{\partial a^{\#}}{\partial r_i}\right\} E\left\{\frac{\partial r_i^{\#}}{\partial z_{iu}}\right\} \rho_{iu}^{\#} + \sum_{i=1}^{n} \sum_{j \neq u}^{m'} \left[ E\{D'(a^{\#})\} E\left\{\frac{\partial a^{\#}}{\partial r_i}\right\} E\left\{\frac{\partial r_i^{\#}}{\partial z_{ij}}\right\} - \frac{\partial \pi_{ri}^{\#}}{\partial z_{ij}} \right] \omega_{iju}^{\#} \rho_{iu}^{\#} + \sum_{i=1}^{n} \sum_{j=1}^{m'-m} E\{D'(a^{\#})\} E\left\{\frac{\partial a^{\#}}{\partial r_i}\right\} E\left\{\frac{\partial r_i^{\#}}{\partial y_{ij}}\right\} \gamma_{iju}^{\#} \rho_{iu}^{\#} + \sum_{i=1}^{n} \left[ E\{D'(a^{\#})\} cov\left\{\frac{\partial a^{\#}}{\partial r_i}, \frac{\partial r_i^{\#}}{\partial z_{iu}}\right\} + \sum_{i=1}^{n} cov\left\{D'(a^{\#}), \frac{\partial a^{\#}}{\partial r_i} \frac{\partial r_i^{\#}}{\partial z_{iu}}\right\} \right] \rho_{iu}^{\#} + \sum_{i=1}^{n} \sum_{j!=u}^{m'} \left[ E\{D'(a^{\#})\} cov\left\{\frac{\partial a^{\#}}{\partial r_i}, \frac{\partial r_i^{\#}}{\partial z_{iu}}\right\} + cov\left\{D'(a^{\#}), \frac{\partial a^{\#}}{\partial r_i} \frac{\partial r_i^{\#}}{\partial z_{iu}}\right\} \right] \omega_{iju}^{\#} \rho_{iu}^{\#} + \sum_{i=1}^{n} \sum_{j=1}^{m-m'} \left[ E\{D'(a^{\#})\} cov\left\{\frac{\partial a^{\#}}{\partial r_i}, \frac{\partial r_i^{\#}}{\partial z_{iu}}\right\} + cov\left\{D'(a^{\#}), \frac{\partial a^{\#}}{\partial r_i} \frac{\partial r_i^{\#}}{\partial z_{iu}}\right\} \right] \gamma_{iju}^{\#} \rho_{iu}^{\#} \quad \forall u$ | $b_u^{\#\#} = \sum_{i=1}^{n} D(a^{\#\#}) \frac{\partial a^{\#\#}}{\partial r_i} \frac{\partial r_i^{\#\#}}{\partial z_{iu}} \rho_{iu}^{\#\#} + \sum_{i=1}^{n} \sum_{j \neq u}^{m'} \left[ D'(a^{\#\#}) \frac{\partial a^{\#\#}}{\partial r_i} \frac{\partial r_i^{\#\#}}{\partial z_{ij}} - \frac{\partial \pi_{ri}^{\#\#}}{\partial z_{ij}} \right] \omega_{iju}^{\#\#} \rho_{iu}^{\#\#} + \sum_{i=1}^{n} \sum_{j=1}^{m'-m} D'(a^{\#\#}) \frac{\partial a^{\#\#}}{\partial r_i} \frac{\partial r_i^{\#\#}}{\partial y_{ij}} \gamma_{iju}^{\#\#} \rho_{iu}^{\#\#} \quad \forall u$ |
| Expected Emissions | $p^{\dagger} = \sum_{i=1}^{n} E\{D'(a^{\dagger})\} E\left\{\frac{\partial a^{\dagger}}{\partial r_i}\right\} \beta_{ij}^{\dagger} + \frac{\sum_{i=1}^{n} \sum_{j=1}^{m} \left[ E\{D'(a^{\dagger})\} cov\left\{\frac{\partial a^{\dagger}}{\partial r_i}, \frac{\partial r_i^{\dagger}}{\partial x_{ij}}\right\} + cov\left\{D'(a^{\dagger}), \frac{\partial a^{\dagger}}{\partial r_i} \frac{\partial r_i^{\dagger}}{\partial x_{ij}}\right\} \right] \kappa_{ij}^{\dagger}}{\sum_{i=1}^{n} \sum_{j=1}^{m} E\left\{\frac{\partial r_i^{\dagger}}{\partial x_{ij}}\right\} \kappa_{ij}^{\dagger}}$ | $p^{\dagger\dagger} = \sum_{i=1}^{n} D'(a^{\dagger\dagger}) \frac{\partial a^{\dagger\dagger}}{\partial r_i} \beta_i^{\dagger\dagger}$ |

where $\kappa_{ij}^{**} = \frac{\partial x_{ij}^{**}/\partial p}{\sum_{i=1}^{n} \sum_{j=1}^{m} \partial x_{ij}^{**}/\partial p}$ , $\beta_{ij} = \frac{E\{\partial r_i^{\dagger}/\partial x_{ij}\} \kappa_{ij}^{\dagger}}{\sum_{i=1}^{n} \sum_{j=1}^{m} E\{\partial r_i^{\dagger}/\partial x_{ij}\} \kappa_{ij}^{\dagger}}$, $\rho_{iu}^{\#} = \frac{\partial z_{iu}^{\#}/\partial b_u}{\sum_{i=1}^{n} (\partial z_{iu}^{\#}/\partial b_u)}$, $\omega_{iju}^{\#} = \frac{\partial z_{ij}^{\#}/\partial b_u}{\partial z_{iu}^{\#}/\partial b_u}$, $\gamma_{iju}^{\#} = \frac{\partial y_{ij}^{\#}/\partial b_u}{\partial z_{iu}^{\#}/\partial b_u}$, $z_i$ is the (1x$m'$) vector of taxed inputs, and $y_i$ is the (1x($m$-$m'$)) vector of untaxed inputs.

Note: Superscripts #, ##, †, and †† denote the variables are evaluated at their optimal value given the specification of the model and instruments being used.

Table 2. Percent Differences Between Optimal Tax Rates
from Correctly Specified Model and Optimal Tax Rates from Mis-Specified Model

| Region on Which Tax is Imposed | Tax Base | Policy Instrument | | | |
|---|---|---|---|---|---|
| | | Efficient Fertilizer Taxes and Land Subsidies | Uniform Fertilizer Taxes | Non-Uniform Expected Runoff Taxes | Uniform Expected Runoff Taxes |
| Region 1 (or uniform rate) | Fertilizer | 32.98[a] (21.79)[b] [0.16, 81.08][c] | 37.24 (23.56) [0.19, 81.88] | | |
| | Land | 34.0 (62.16) [-41.43, 421.49] | | | |
| | Expected Runoff | | | 39.45 (24.58) [0.2, 84.59] | 39.23 (24.28) [0.22, 82.17] |
| Region 2 | Fertilizer | 31.55 (21.79) [0.16, 81.08] | | | |
| | Land | -26.38 (20.08) [-76.37, 25.17] | | | |
| | Expected Runoff | | | -20,957.16 (340287.62) [-6460060, 85.5] | |
| Region 3 | Fertilizer | 33.5 (23.17) [0.12, 84.15] | | | |
| | Land | 10.56 (36.19) [-49.67, 248.02] | | | |
| | Expected Runoff | | | 38.1 (25.5) [0.16, 89.02] | |
| Region 4 | Fertilizer | 30.21 (21.12) [0.13, 84.15] | | | |
| | Land | -26.89 (19.97) [-80.62, 0.45] | | | |
| | Expected Runoff | | | -16343.05 (366085.5) [-8185890, 86.37] | |

Notes: [a]Sample mean of percent differences: 100(correct tax - incorrect tax)/correct tax.
[b]Sample standard deviation of percent differences.
[c]Sample range of percent differences.

Table 3. Percent Differences between Actual Welfare
from Correctly Specified Model and Actual Welfare from Mis-Specified Model

| Welfare Measure | Policy Instrument | | | |
|---|---|---|---|---|
| | Efficient Fertilizer Taxes and Land Subsidies | Uniform Fertilizer Taxes | Non-Uniform Expected Runoff Taxes | Uniform Expected Runoff Taxes |
| Expected Social Net Benefits | 1.32[a] | 1.69 | 1.31 | 1.42 |
| | (1.54)[b] | (1.81) | (1.54) | (1.61) |
| | [0.00, 7.74][c] | [0.00, 8.09] | [0.00, 7.74] | [0.00, 7.98] |
| Consumers' Surplus | -6.26 | -13.01 | -6.08 | -10.23 |
| | (4.74) | (9.19) | (4.62) | (7.32) |
| | [-20.34, -0.02] | [-39.86, -0.06] | [-20.3, -0.02] | [-34.2, -0.05] |
| Expected Damages | -70.9 | -54.48 | -70.84 | -56.29 |
| | ( 71.45) | (45.33) | (71.38) | (50.56) |
| | [-385.72, -0.1] | [-203.59, -0.15] | [-385.62, -0.1] | [-299.73, -0.12] |
| Landowners' Surplus (by region) | | | | |
| Region 1 | -22.28 | -6.13 | -18.33 | -4.98 |
| | (27.64) | (6.82) | (23.11) | (11.01) |
| | [-238.22, 10.66] | [-41.66, 4.77] | [-183.01, 11.03] | [-83.02, 18.42] |
| Region 2 | 7.34 | -6.07 | 7.75 | 1.64 |
| | (7.18) | (6.12) | (6.96) | (6.58) |
| | [-33.46, 30.05] | [-38.02, 4.67] | [-27.52, 29.64] | [-29.73, 25.12] |
| Region 3 | -3.72 | 5.52 | -2.38 | 4.07 |
| | (7.53) | (4.49) | (6.42) | (4.47) |
| | [-43.32, 16.3] | [0.01, 22.38] | [-33.94, 16.19] | [-9.59, 22.39] |
| Region 4 | 6.82 | 5.4 | 6.82 | 6.16 |
| | (5.39) | (4.31) | (5.33) | (5.14) |
| | [-4.69, 27.38] | [0.01, 18.51] | [-4.18, 27.23] | [-2.99, 25.83] |
| Landowners' Surplus (aggregate) | 1.48 | 1.95 | 2.2 | 3.33 |
| | (2.05) | (2.82) | (2.27) | (3.18) |
| | [-3.01, 12.38] | [-3.93, 13.88] | [-1.96, 12.84] | [-0.86, 16.8] |

Notes: [a]Sample mean of percent differences: 100(correct welfare - incorrect welfare)/correct welfare.
[b]Sample standard deviation of percent differences.
[c]Sample range of percent differences.

Table 4. Percent Differences between Actual Welfare

from Mis-Specified Model and Perceived Welfare from Mis-Specified Model

| Welfare Measure | Policy Instrument | | | |
|---|---|---|---|---|
| | Efficient Taxes/Subsidies on Fertilizer and Land | Uniform Fertilizer Taxes | Non-Uniform Expected Runoff Taxes | Uniform Expected Runoff Taxes |
| Expected Social Net Benefits | -3.64[a] | -6.05 | -3.64 | -4.83 |
| | (3.23)[b] | (4.91) | (3.23) | (4.04) |
| | [-14.32, -0.01][c] | [-21.71, -0.01] | [-14.32, -0.01] | [-17.88, -0.01] |
| Expected Damages | 41.4 | 44.25 | 41.4 | 43.23 |
| | (28.08) | (28.03) | (28.08) | (28.05) |
| | [0.11, 91.6] | [0.14, 90.43] | [0.11, 91.6] | [0.14, 91.41] |

Notes: [a]Sample mean of percent differences: 100(actual welfare - perceived welfare)/actual welfare.
[b]Sample standard deviation of percent differences.
[c]Sample range of percent differences.

Table 5. Comparison of Instruments in Correctly Specified Models and in Mis-Specified Models, and Number of Mistakes from Using Mis-Specified Models

| Policy Instruments | Welfare Measure | Percent of Samples in which Row Instruments Outperform Column Instruments (by welfare measure) | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Uniform Fertilizer Tax | | | | Non-Uniform Expected Runoff Taxes | | | | Uniform Expected Runoff Tax | | | |
| | | Correct Model | Mis-Specified Model | | | Correct Model | Mis-Specified Model | | | Correct Model | Mis-Specified Model | | |
| Efficient taxes/ subsidies | Expected Net Social Benefits | 100 | 100 | (0)[a] | ----[b] | 100 | Equivalent | (100) | [0.01] | 100 | 100 | (0) | ---- |
| | Consumers' Surplus | 100 | 100 | (0) | ---- | 100 | Equivalent | (100) | [-0.04] | 100 | 99.6 | (0.4) | [8.84] |
| | Expected Damages | 100 | 100 | (0) | ---- | 80.4 | Equivalent | (100) | [-0.01] | 100 | 100 | (0) | ---- |
| | Landowners' Surplus | 68 | 82.2 | (15) | [-1.64] | 100 | Equivalent | (100) | [-0.17] | 5.8 | 6.8 | (3.2) | [-1.06] |
| Uniform Fertilizer Tax | Expected Net Social Benefits | ---- | ---- | ---- | ---- | 0 | 0 | (0) | ---- | 6.4 | 6.4 | (1.6) | [-0.25] |
| | Consumers' Surplus | ---- | ---- | ---- | ---- | 0 | 0 | (0) | ---- | 98 | 3.6 | (3.2) | [-2.77] |
| | Expected Damages | ---- | ---- | ---- | ---- | 0 | 0 | (0) | ---- | 8.2 | 7 | (2.8) | [-1.56] |
| | Landowners' Surplus | ---- | ---- | ---- | ---- | 25.5 | 17.8 | (9.6) | [1.31] | 1.6 | 1 | (0.6) | [0.7] |
| Non-Uniform Expected Runoff Taxes | Expected Net Social Benefits | ---- | ---- | ---- | ---- | ---- | ---- | ---- | ---- | 100 | 100 | (0) | ---- |
| | Consumers' Surplus | ---- | ---- | ---- | ---- | ---- | ---- | ---- | ---- | 100 | 99.6 | (0.4) | [9.0] |
| | Expected Damages | ---- | ---- | ---- | ---- | ---- | ---- | ---- | ---- | 100 | 100 | (0) | ---- |
| | Landowners' Surplus | ---- | ---- | ---- | ---- | ---- | ---- | ---- | ---- | 8.8 | 8.6 | (2.2) | [-0.12] |

Notes: [a]Percent of samples in which mis-specified model predicts relative performance incorrectly.
[b]Sample mean of actual percentage differences in welfare from the use of row versus column instruments, computed using correct model and for only those samples in which the mis-specified model is incorrect.

Table 6. Probability of Extreme Events Resulting From Policy Choices in Correctly and Mis-Specified Models[a]

| Probability | Policy Instruments | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Efficient Taxes/Subsidies on Fertilizer and Land | | Uniform Fertilizer Taxes | | Non-Uniform Expected Runoff Taxes | | Uniform Expected Runoff Taxes | |
| | Correct Model | Mis-Specified Model | Correct Model | Mis-Specified Model | Correct Model | Mis-Specified Model | Correct Model | Mis-Specified Model |
| Prob(Region 1's runoff > two standard deviations above the mean)[b] | 0.00 | 0.31 | 0.00 | 0.18 | 0.00 | 0.3 | 0.00 | 0.17 |
| Prob(Region 2's runoff > two standard deviations above the mean) | 0.00 | 0.00 | 0.00 | 0.18 | 0.00 | 0.00 | 0.00 | 0.12 |
| Prob(Region 3's runoff > two standard deviations above the mean) | 0.00 | 0.17 | 0.00 | 0.07 | 0.00 | 0.17 | 0.00 | 0.1 |
| Prob(Region 4's runoff > two standard deviations above the mean) | 0.00 | 0.00 | 0.00 | 0.081 | 0.00 | 0.00 | 0.00 | 0.05 |
| Prob(ambient pollution > two standard deviations above the mean) | 0.04 | 0.14 | 0.04 | 0.12 | 0.04 | 0.15 | 0.04 | 0.12 |
| Prob(damages > two standard deviations above the mean) | 0.05 | 0.16 | 0.05 | 0.15 | 0.06 | 0.17 | 0.05 | 0.15 |

Notes: [a]Results are calculated with uncertain parameter values evaluated at their mean values.
[b]Means and standard deviations are calculated from the correctly specified model, given the policy instruments under consideration

Table A1. Factor Cost Shares and Distribution of Uncertain Parameters

| Region | Cost Shares | | |
|---|---|---|---|
| | Biological | | Mechanical |
| | Land | Fertilizer | |
| Region 1 | 0.25 | 0.35 | 0.4 |
| Region 2 | .025 | 0.35 | 0.4 |
| Region 3 | 0.4 | 0.2 | 0.4 |
| Region 4 | 0.4 | 0.2 | 0.4 |

| Uncertain Parameters | Distribution | Mean | Variance | Sources and/or Justification for Parameter Ranges |
|---|---|---|---|---|
| Elasticity of demand | U(-1.2, -0.45) | -0.825 | 0.0469 | Consistent with the domestic elasticity of demand for corn in the Corn Belt and Lake States. See Claassen and Horan (forthcoming) for derivation. |
| Elasticity of land supply | U(0.15, 0.45) | 0.3 | 0.0075 | Chavas and Holt (1990); Holt (1990); Lee and Helmberger (1985); Tegene, Huffman, and Miranowski, (1988) |
| Elasticity of substitution between composite inputs | U(0.1, 0.9) | 0.5 | 0.0533 | Binswanger, (1974); Chambers and Vasavada, (1983); Fernandez-Cornejo, (1992); Hertel, (1989); Kawagoe, Otsuka, and Hayami, (1985); Ray, (1982); Thirtle, (1985) |
| Elasticity of substitution between land and fertilizer | U(1.1, 1.4) | 1.25 | 0.025 | Binswanger, (1974); Chambers and Vasavada, (1983); Fernandez-Cornejo, (1992); Hertel, (1989); Kawagoe, Otsuka, and Hayami, (1985); Ray, (1982); Thirtle, (1985) |
| Average per acre runoff: | | | | |
| Farms 1 and 3 | U(0.2, 0.4) | 0.3 | 0.0033 | NRC, (1993); Peterson and Frye, (1989); Smith, Schwarz, and Alexander, (1997) |
| Farms 2 and 4 | U(0.1, 0.3) | 0.2 | 0.0033 | |
| Uptake | U(0.6, 0.8) | 0.7 | 0.0033 | Keeney, (1982); Peterson and Frye (1989), NRC (1993) |
| Elasticity of per acre runoff | U(1, 2) | 1.5 | 0.0833 | The chosen bounds ensure an increasing, convex function, e.g., Hallberg, (1987); NRC, (1993); Weinberg and Kling, (1996). |
| Coefficient of variation: ambient pollution (CVA) | U(0.1, 3) | 1.55 | 0.7008 | Koutsoyiannia, (1999); Manguerra and Engel, (1998) |
| Coefficient of variation: runoff | U(0.1, CVA) | 0.825* | 0.1752* | Koutsoyiannia, (1999); Manguerra and Engel, (1998) |
| Runoff Transport | | | | |
| Farms 1 and 3 | U(0.6, 0.9) | 0.75 | 0.0075 | Fisher et al., (1988); Smith, Schwarz, and Alexander, (1997) |
| Farms 2 and 4 | U(0.01, 0.3) | 0.155 | 0.0070 | |
| Elasticity of damages | U(1.2, 2) | 1.6 | 0.5333 | The chosen bounds ensure an increasing, convex function |

Note: Cost shares are consistent with the range of estimates for corn production in the Corn Belt and Lake States (Claassen and Horan, forthcoming; USDA-ERS; USDA-ERS, 1990). *Expected mean and variance based on CVA

## Endnotes

1. The views expressed here are those of the authors and do not necessarily reflect those of USDA-ERS.

2. There are two ways that benefits could be a nonlinear function of emissions: (1) benefits are a nonlinear function of environmental quality, or (2) environmental quality is a nonlinear function of emissions.

3. The degree to which risk is accounted for in these and other studies is not always made clear.

4. A few studies in which instruments are designed to achieve an exogenous environmental constraint with a given probability do evaluate how instruments and control costs respond to increases in this probability (McSweeny and Shortle, 1990; Teague, Bernardo, and Mapp, 1995). This provides some indication as to the importance of stochastic processes.

5. The alternative to maximizing the net social benefits from production would be to maximize the net private benefits from production subject to an exogenously defined, probabilistic environmental constraint (i.e., a cost-effectiveness approach). We maximize net social benefits because this approach provides greater insight into the economic merits of modeling risk and because it eliminates the need to specify the type of constraint (there are many possibilities) and the level of the constraint.

6. Let $f$=$f$($q$) ($f^{\prime}, f^{\prime\prime} > 0$), where $q$=$q$($h$). Then $cov\{f^{\prime}(q), \partial q/\partial h\}$ is of the same sign as $cov\{q, \partial q/\partial h\} = .5(\partial var\{q\}/\partial h)$, where this equality follows from: $\partial var\{q\}/\partial h =$ $\partial(E\{q^2\} - E\{q\}^2)/\partial h = 2(E\{q \partial q/\partial h\} - E\{q\}E\{\partial q/\partial h\}) = 2cov\{q, \partial q/\partial h\}$. This result is used throughout the paper, although with different definitions for $f$, $q$, and $h$.

7. These taxes are not described here, but are described in detail in Shortle, Horan, and Abler (1998) and Horan et al. (1999) .

8. The geography of watersheds are such that they may vary greatly in size and in terms of economic importance. We assume a watershed of sufficient size/importance that changes in aggregate production have market price impacts. The elasticity of demand is varied across watersheds to permit a range of price effects.

9. The same stochastic processes affecting environmental outcomes are also likely to influence production. In the present model, we model production deterministically to focus on stochastic environmental processes. However, it is generally important that these processes be taken into consideration as well.

10. The non-uniform expected runoff taxes perform very well because the mean and variance of runoff are positively correlated in the model. Thus, reducing mean runoff indirectly reduces the variance of runoff, an important source of risk in the model.

11. Ex post results describe the expected outcome of a situation in which all parameter values will be known when policies are designed and implemented, even if many parameter values are uncertain at present. In contrast, ex ante results describe the expected outcome of a situation in

which at least some parameter values remain uncertain even when policies are designed and implemented.