Journal of Agricultural and Applied Economics, 31,1(April 1999):161–175

Accounting for Spatial Characteristics of Watersheds in Evaluating Water Pollution Abatement Policies

Zeyuan Qiu and Tony Prato

ABSTRACT

This study evaluates three agricultural nonpoint pollution abatement policies: regulating the spatial pattern of agricultural activities, ambient tax, and abatement tax/subsidy. All three policies incorporate spatial characteristics of agricultural emission loading and movement for an agricultural watershed in the Midwest. The effects of spatial variation in natural conditions and landscape features on agricultural emissions and crop yield are evaluated using a newly developed biophysical simulation model and experimental data. While the policies are equally cost effective in reducing agricultural nonpoint source pollution, their implementation feasibility is quite different.

Key Words: atrazine, environmental policy, nonpoint pollution, simulation, watershed management, water quality.

Agriculture is the leading contributor to nonpoint source pollution and the leading source of water pollution in the United States (EPA, 1994b). Agricultural nonpoint source pollution is caused by sediment, nutrients, salts, organics, pesticides, and pathogens (Krivak). It is diffuse, has complicated spatial and temporal dimensions, and is driven by the vagaries of weather. These features make it desirable to evaluate agricultural nonpoint source water pollution in an integrated watershed management framework. A watershed approach is addressed in the Clean Water Act (CWT) and endorsed by many governmental agencies and professional organizations (EPA, 1994a; Water Environment Federation; Adler, Landman, and Cameron).

Two spatial aspects of agricultural nonpoint source pollution are significant for policy analysis. First, agricultural emissions originate on agricultural fields. Second, agricultural emissions are transported by surface runoff or subsurface flow to water bodies where they can pollute surface and ground water. Agricultural nonpoint source water pollution is significantly affected by spatial variation in soil, topographic, hydrologic, geologic, and landscape features in the fields and on the transport path for a pollutant. The close relationship between the pattern of upland agricultural activity and downstream water quality has been recognized (EPA, 1974; Dillon and Kirchner; Omernik; Hopkins and Clausen). Watershed-scale policy analysis should account for the diffuse and spatially diverse nature of agricultural nonpoint source pollution (Jacobs and Casler; Braden et al.; Pan and Hodge).

Zeyuan Qiu is research assistant professor and Tony Prato is professor of resource economics and management, and director, Center for Agricultural, Resource and Environmental System (CARES), Department of Agricultural Economics, University of Missouri-Columbia. Financial support from the U.S. Department of Agriculture is gratefully acknowledged.

This paper explicitly recognizes the importance of spatial variability in agricultural nonpoint source pollution in designing and evaluating pollution control policies. Specifically, the first spatial aspect of agricultural nonpoint source pollution, i.e. the effects of spatial variation in soil, topographic, and hydrologic conditions on agricultural emissions, is evaluated using a newly developed biophysical simulation model called *Soil and Water Assessment Tool* (SWAT) (Arnold et al.). The second spatial aspect, i.e. the impacts of landscape features such as riparian buffers along the streams on agricultural pollution, is considered by integrating experimental data on riparian buffers with SWAT simulation results. Following the least-cost framework originally proposed by Baumol and Oates, a mathematical programming model is developed to evaluate alternative policies for reducing agricultural nonpoint source pollution in an agricultural watershed. Rather than assuming uniform effects on all farms in a watershed, the site-specific impacts of these policies are considered. This approach is consistent with the diffuse and spatially diverse nature of agricultural nonpoint source pollution in a watershed.

Previous Literature on Externality Control

Externality control is typically accomplished with a Pigouvian tax or subsidy. The proper level of a Pigouvian tax (subsidy) on the activities of the generator of an externality is equal to the marginal net damage (benefit) produced by that activity, i.e. the difference between marginal social and private damage. The graphical presentation of the Pigouvian tax can be found in Tietenberg (1988) and Prato. Even though a Pigouvian tax or subsidy results in optimal resource allocation, it has rarely proved feasible because of the inability to measure marginal social damage or benefit. As an alternative, Baumol and Oates suggested an environmental pricing and standards approach for achieving a socially acceptable standard of environmental quality. In general, their approach does not result in a Pareto-efficient allocation of resources. However, the correct set of effluent charges on pollutants can achieve the standard at least cost.

Griffin and Bromley extended Baumol and Oates' framework to the case of nonpoint externality and argued that, under full information and certainty, there were four equivalent policy alternatives to control agricultural runoff: the least-cost effluent charges and standards, and the least-cost management incentives and standards. Shortle and Dunn showed that these four agricultural water pollution control policies have different efficiencies when farmers have better information about economic outcomes of agricultural practices than planners and there are uncertainties about the environmental outcomes. They found that the appropriately specified management incentives should generally outperform the management standard, the effluent charges, and standards. Tietenberg (1974) expanded the traditional general equilibrium model to include the kind of spatial detail that is important for describing pollution and derived theorems which provide the basis for the development of spatially differential, tax-based decision rules. Segerson presented a general incentive scheme for controlling nonpoint source pollution in the presence of uncertainty and difficulties of monitoring the pollution contributions from different polluters.

Along with these theoretical developments, many studies empirically evaluate the policy alternatives to control agricultural nonpoint source pollution. Most of these studies link agricultural practices to their adverse environmental impacts and compare alternative policy instruments for achieving a specific environmental goal (Horner; Jacobs and Casler; Jacobs and Timmons; Braden, Larson, and Herricks; Chowdhury and Lacewell). Recent empirical work evaluates policy alternatives incorporating spatial characteristics and focusing on nitrogen contamination (Wu and Segerson; Helfand and House; Larson, Helfand, and House; and Fleming and Adams).

This paper contributes to the empirical literature by evaluating alternative policies for controlling atrazine contamination of stream water in a Midwestern agricultural watershed.

This study makes several significant contributions. First, instead of assuming hypothetical spatial differences as in many previous studies, the study accounts for real spatial differences in a heterogeneous watershed. The watershed is divided into several subwatersheds to account for spatial variability in economic and environmental impacts due to variations in land use, soil, hypsography, and hydrology. Second, instead of evaluating a variable input tax to reduce use of inputs and the resulting environmental impacts, this study evaluates the policy alternatives such as tax/subsidy and regulations that promote the optimal spatial pattern of farming systems for achieving predetermined environmental objectives. Third, the study is motivated by atrazine contamination in surface water, a serious water quality problem in the Corn Belt. This study assumes full information and certainty in the sense that the economic and environmental outcomes of agricultural practices can be accurately predicted using a biophysical simulation model and enterprise budget generator.

Economic Model

Griffin and Bromley's uniform incentive framework is extended to the variable incentive case and then applied in a watershed. Following the notation used by Griffin and Bromley, let y^j be the production bundle of firm j with y^j_n being the n[th] element (positive or negative) of that vector, $j = 1, 2, \ldots, J$. Positive activities represent outputs and negative ones inputs. Pollutants generated by firm j are nonnegative and are denoted by z^j. There are J firms and N goods or activities excluding the pollutants. The production set of firm j is given implicitly by $f^j(y^j, z^j) \leq 0$. We add a notation Z for the cumulative pollutants from all J firms. Instead of assuming additive pollutants, i.e. $Z = \sum_{j=1}^{J} z^j$, cumulative pollutants are assumed to be a function of z^j, i.e. $Z = g(u, z^j)$, where u are exogenous physical parameters affecting pollutant accumulation. Obviously, the former specification is just a special case of the latter. Suppose the cumulative pollutants are limited to Z^*. Society's problem can be formulated as maximizing total profits given productive abilities and the constraint on pollutants. Society's Lagrangian is

$$L = \sum_{j=1}^{J} py^j - \sum_{j=1}^{J} \alpha^j f^j(y^j, z^j) - \mu(g(u, z^j) - Z^*),$$

where p is a price vector, the α^j's and μ are appropriate Lagrange multipliers. Assuming all necessary conditions hold, the following first-order conditions describe the optimal choice of production activities:

$$p_n - \alpha^j \frac{\partial f^j(y^j, z^j)}{\partial y^j_n} = 0 \quad \text{for all } j \text{ and } n,$$

and

$$-\alpha^j \frac{\partial f^j(y^j, z^j)}{\partial z^j} - \mu \frac{\partial g(u, z^j)}{\partial z^j} = 0 \quad \text{for all } j.$$

On the private side, let s^j represent the per-unit incentive for pollutants generated by firm j and Z^*_j the incentive base level for firm j. Pollutant emissions above Z^*_j are taxed at a rate of s^j and emissions below Z^*_j are subsized at the same rate. If firm j maximizes profits in the presence of such an incentive, then the private Lagrangian is $L^j = py^j - \delta^j f^j(y^j, z^j) + s^j(Z^*_j - z^j)$.

The optimality conditions are then given by the following equations:

$$p_n - \delta^j \frac{\partial f^j(y^j, z^j)}{\partial y^j_n} = 0 \quad \text{for all } j \text{ and } n, \text{ and}$$

$$-\delta^j \frac{\partial f^j(y^j, z^j)}{\partial z^j} - s^j = 0 \text{ for all } j.$$

If the private value of productive abilities is equivalent to the social value ($\delta^j = \alpha^j$) and the incentive (s^j) is set equal to $\mu[\partial g(u, z^j)/\partial z^j]$, then the social and private solutions are the same. Griffin and Bromley's other arguments can be extended from here.

It is interesting to note that the incentive depends on the functional form of $g(u, z^j)$. If $g(u, z^j) = \sum_{j=1}^{J} z^j$, as specified by Griffin and Bromley, the incentive should be the same across all firms. Such a decision rule is appropriate for "uniformly mixed fond pollutants" defined by Tietenberg (1988). The damage caused by these pollutants depends on the amount entering soil, water, and atmosphere and is relatively insensitive to where the emissions enter the environmental medium. In contrast to the uniformly mixed fond pollutant, there are "nonuniformly mixed surface pollutants." The damage caused by these pollutants is related to their concentration levels in the medium, which are sensitive to the amount of pollutants, the volume of the medium, and the location of emissions. In this case, $g(u, z^j)$ is not necessarily an additive function. The resulting incentive would vary by firm in order to achieve the desired environmental quality in a cost-effective manner.

Water pollution can be classified as a nonuniformly mixed surface pollutant as discussed above. In this paper, the above framework is used to describe how a fully informed watershed planning authority (WPA) would select farming systems in a watershed so as to achieve a specific water quality objective at minimum loss in total watershed net return (TWNR). A farming system specifies crop rotation, tillage method, and pesticide and fertilizer application rates. An agricultural watershed is divided into several subwatersheds and each subwatershed is treated as a separate source of pollution in accounting for spatial variability in soil, topographic, hydrologic, geologic, and landscape features. Cumulative water pollution at the watershed outlet is a weighted average of water pollution from all subwatersheds. The weights account for the contributions of pollution from subwatersheds to the cumulative pollution at the watershed outlet.

The following mathematical programming model applies the general framework discussed above and determines the cost-effective spatial pattern of farming systems in the watershed, $\{x_{ij}^*\}$, and the efficient abatement of pollutant k in subwatershed j, $\{y_{kj}^*\}$:

$$(1) \qquad \text{maximize for } x_{ij} \qquad \sum_{i=1}^{I} \sum_{j=1}^{J} \pi_{ij} x_{ij} C_j$$

subject to:

$$(2) \qquad b_{kj} - y_{kj} - \sum_{i=1}^{I} e_{ijk} x_{ij} \geq 0 \qquad \text{for all } k \text{ and } j,$$

$$(3) \qquad B_k - \sum_{j=1}^{J} y_{kj} w_{kj} \leq R_k \qquad \text{for all } k,$$

$$(4) \qquad \sum_{i=1}^{I} x_{ij} \leq 1 \qquad \text{for all } j,$$

$$(5) \qquad x_{ij} \geq 0,$$

$$(6) \qquad y_{kj} \geq 0,$$

where i is a farming system index, j is a subwatershed index, k is a water quality index, C_j is the total cropland acreage in subwatershed j, π_{ij} is the per-acre annual net return for farming system i in subwatershed j as discussed below, e_{ijk} is the amount of pollutant k generated by farming system i in subwatershed j as estimated by the SWAT model, b_{kj} is the baseline level of water pollutant k at the outlet of subwatershed j, B_k is the baseline level of water pollutant k at the watershed outlet, w_{kj} is the contribution rate of subwatershed j to pollutant k as estimated by the SWAT model, x_{ij} is the proportion of cropland in farming system i in subwatershed j, y_{kj} is the efficient abatement of water pollutant k in subwatershed j, and R_k is the desired level of water pollutant k at the watershed outlet as selected by the WPA.

The baseline solution maximizes total watershed net return (TWNR), given in equation (1), only subject to resource constraints given by equations (4) and (5). The water quality indicators for the baseline (b_{kj} and B_k) are estimated from the baseline solution $\{x_{ij}^*\}$, namely $b_{kj} = \Sigma_{i=1}^{I} e_{ijk} x_{ij}^*$ for all k and j, and $B_k = \Sigma_{j=1}^{J} w_{kj} b_{kj}$ for all k. The biophysical simulation model accounts for the delivery of agricultural pollutants within subwatersheds. Contribution rates used in the second water quality constraints (w_{kj}) capture the delivery of pollutants from subwatershed outlets to the watershed outlet.

The two water quality constraints require water quality compliance at each subwatershed outlet (equation (2)) and water quality compliance at the watershed outlet (equation (3)). The simultaneous relationship between water quality constraints at the watershed and subwatershed outlets is controlled by y_{kj} which is the efficient abatement of water pollutant k in subwatershed j and varies substantially across subwatersheds and depends on b_{kj}, e_{ijk} and w_{kj}. The latter are determined by the spatial characteristics of the watershed. The above mathematical programming model was solved using the General Algebraic Modeling System (GAMS) (Brooke, Kendrick, and Meeraus). The shadow prices for equations (2) and (3) are the per-unit marginal costs of reducing water pollution at the subwatershed and watershed outlets, respectively, and can be used to set the incentive levels at subwatershed and watershed levels: $\{x_{ij}\}$ define the cost-effective spatial pattern of farming systems, and $\{y_{kj}\}$ define the efficient abatement levels across all subwatersheds.

Data Development

Study Area

The study area is Goodwater Creek watershed in north central Missouri. It is part of the Central Claypan Soils Major Land Resource Area (MIRA 113) that includes about 10 million acres in the Midwest. The watershed is also the study site of the Missouri Management System Evaluation Area (MSEA) project. Broad nearly flat divides and gentle sideslopes characterize topography in the watershed. Broad alluvial valleys are often dissected by small streams. The principal agricultural activity in the watershed is crop production. Cropland accounts for 72% of the watershed. There are well-developed treelines and forest along the streams and waterways in the watershed. The area occupied by treelines and forest is equivalent to a 105-foot-wide riparian buffer strip on each side of the stream.

Current and potential crop production methods in the watershed are represented by 37 farming systems which are patterned after the six farming systems evaluated in the Missouri MSEA project. Each farming system consists of a specific crop rotation (corn-soybean, sorghum-soybean or corn-soybean-wheat), tillage system (minimum tillage or no tillage), fertilizer application rate (low, medium, or high) and pesticide application rate (low or high). The 37th farming system consists of a cool-season grass and legume with no pesticide and fertilizer use. It approximates conditions for cropland enrolled in the CRP. A summary of the 37 alternative farming systems is presented in Table 1. Detailed information about the specific amounts and methods of fertilizer and pesticide application and field operations for each crop in a farming system can be found in Qiu.

Estimation of Water Quality Impacts

SWAT is used to estimate water quality effects of alternative farming systems. It is a continuous-time, basin-scale simulation model that operates on a daily time step. It allows a basin to be divided into hundreds or thousands of grid cells or subbasins. SWAT predicts the impacts of agricultural management systems on soil erosion, surface water quality and ground water quality in large ungaged rural basins. SWAT embeds a flexible configuration structure that distinguishes the different activities within a watershed and provides a flexible routing structure to control the movement of water, sediment, nutrients, and pesticides. It is suited for evaluating alternative watershed management schemes. Studies show that SWAT provides reasonable estimates of runoff and hydrologic budget at the watershed level (Bingner; Arnold and Allen). As an effort to better understand the physical, biological, chemical, and hydrological processes in Goodwater Creek watershed, SWAT was calibrated and validated using the measured data in the Missouri MSEA project (Heidenreich, Zhou, and Prato).

To incorporate the spatial pattern of soil, topography, and stream drainage pattern, Goodwater Creek watershed is divided into 32 subwatersheds based on hydrologic conditions. Complete field boundaries are incorporated into the watershed delineation. Each subwatershed is treated as a subbasin in the model. Agricultural activities within a subbasin are distinguished using "virtual subbasins." A virtual subbasin is a portion of land in a subbasin devoted to specific land uses and

Table 1. Summary of Water Quality and Economic Impacts for 37 Alternative Farming Systems in Goodwater Creek Watershed

Farming System Name[a]	Crop Rotation[b]	Tillage System	Fertilizer Application[c]	Pesticide Application[c]	Net Return[d] ($/acre)	Sediment Yield[d] (TAY)	NCSW[d] (ppm)	ACSW[c] (ppb)
CBMHH	C-B	Min-Till	High	High	96.17	0.49	3.82	45.09
CBMMH	C-B	Min-Till	Medium	High	73.86	0.49	3.23	45.09
CBMLH	C-B	Min-Till	Low	High	51.42	0.49	2.65	45.09
CBMHL	C-B	Min-Till	High	Low	93.17	0.49	3.82	28.18
CBMML	C-B	Min-Till	Medium	Low	72.29	0.49	3.23	28.18
CBMLL	C-B	Min-Till	Low	Low	51.34	0.49	2.65	28.18
SBMHH	S-B	Min-Till	High	High	91.69	0.50	3.39	33.23
SBMMH	S-B	Min-Till	Medium	High	68.30	0.50	2.81	33.23
SBMLH	S-B	Min-Till	Low	High	45.38	0.50	2.24	33.23
SBMHL	S-B	Min-Till	High	Low	88.02	0.50	3.39	22.16
SBMML	S-B	Min-Till	Medium	Low	66.17	0.50	2.81	22.16
SBMLL	S-B	Min-Till	Low	Low	44.79	0.50	2.24	22.16
CBWMHH	C-B-W	Min-Till	High	High	77.69	0.84	1.88	26.53
CBWMMH	C-B-W	Min-Till	Medium	High	55.26	0.84	1.65	26.53
CBWMLH	C-B-W	Min-Till	Low	High	32.75	0.84	1.43	26.53
CBWMHL	C-B-W	Min-Till	High	Low	72.92	0.84	1.88	16.58
CBWMML	C-B-W	Min-Till	Medium	Low	51.91	0.84	1.65	16.58
CBWMLL	C-B-W	Min-Till	Low	Low	30.81	0.84	1.43	16.58
CBNHH	C-B	No-Till	High	High	81.42	0.48	3.82	45.21
CBNMH	C-B	No-Till	Medium	High	59.21	0.48	3.24	45.21
CBNLH	C-B	No-Till	Low	High	36.86	0.48	2.65	45.21
CBNHL	C-B	No-Till	High	Low	73.75	0.48	3.82	27.13
CBNML	C-B	No-Till	Medium	Low	52.96	0.48	3.24	27.13
CBNLL	C-B	No-Till	Low	Low	32.03	0.48	2.65	27.13
SBNHH	S-B	No-Till	High	High	77.16	0.49	3.38	27.69
SBNMH	S-B	No-Till	Medium	High	53.69	0.49	2.81	27.69
SBNLH	S-B	No-Till	Low	High	30.67	0.49	2.24	27.69
SBNHL	S-B	No-Till	High	Low	69.10	0.49	3.38	22.15
SBNML	S-B	No-Till	Medium	Low	47.21	0.49	2.81	22.15
SBNLL	S-B	No-Till	Low	Low	25.73	0.49	2.24	22.15
CBWNHH	C-B-W	No-Till	High	High	69.73	0.84	1.89	26.36
CBWNMH	C-B-W	No-Till	Medium	High	47.38	0.84	1.66	26.36
CBWNLH	C-B-W	No-Till	Low	High	24.93	0.84	1.44	26.36
CBWNHL	C-B-W	No-Till	High	Low	61.86	0.84	1.89	15.81
CBWNML	C-B-W	No-Till	Medium	Low	40.92	0.84	1.66	15.81
CBWNLL	C-B-W	No-Till	Low	Low	19.88	0.84	1.44	15.81
GLCNN	G-L	Conv-Till	None	None	27.34	0.11	0.44	0.00

[a] The first two or three letters indicate crop rotation. The last three letters indicate tillage system, fertilizer and pesticide application levels, respectively.
[b] C = Corn, B = Soybean, S = Sorghum, W = Wheat, G = Grass, L = Legume.
[c] Detailed information about application rates can be found in Qiu (pp. 79 and 81).
[d] Watershed average levels. Values at the subwatershed level can be found in Qiu (pp. 101, 105, 109 and 113).

soil types. Each non-cropland use in a subbasin is treated as a virtual subbasin. Several soil types may be associated with a non-cropland use, but only the soil type with largest percentage is assigned to the virtual subbasin. Cropland in a subbasin is divided into several virtual subbasins depending on the soil types. Thirty-two subbasins are delineated. Each consists of one to seven virtual subbasins. A total of 207 virtual subbasins are delineated for Goodwater Creek watershed. ArcInfo GIS is used to delineate the watershed, manage the landuse, soil, hydrologic, and topographic information, and to generate land slope, stream slope and length, and other input information for SWAT. SWAT is run for each farming system using daily precipitation from 1971 to 1994 in Kingdom City near the watershed.

SWAT provides three options for its output files: daily, monthly, and annual. Monthly outputs are used in this application. While SWAT simulates several water quality indicators, only three are considered in this study: sediment yield (SY), nitrogen concentration in stream water (NCSW), and atrazine concentration in stream water (ACSW). Sediment yield is measured using annual average values, NCSW is measured using monthly average values for the May–October period, and ACSW is measured using monthly average values for the May–July period based on the 24-year period used in the SWAT simulations. These indicators and time periods are chosen because they are of interest to the public and researchers (Missouri MSEA Management Team).

Since water quality impacts of riparian buffers cannot be simulated by SWAT, experimental data on riparian buffers are combined with SWAT simulation results to model the water quality impacts of riparian buffers. A two-step procedure is used to estimate the impacts of riparian buffers. First, sediment, NCSW, and ACSW are simulated ignoring the area of riparian buffers in a subbasin. Second, experimental data on riparian buffers are used to adjust the three water quality indicators downward to account for the reduction in pollutant loading to streams from riparian buffers.

Schultz et al. reported that restored multispecies riparian buffer strip systems reduced sediment and chemicals moving with surface runoff by trapping over 90% of the material in the buffer zone where the plants and soil microbes can immobilize and metabolize them. Experiments on riparian buffer strips in Nebraska conducted by Hoagland showed that for newly planted, 50-foot-wide buffer strips, pollution reductions are 72% for sediment yield, 24% for total nitrogen concentration, 35% for atrazine concentration, and 61% for runoff volume. For well-established riparian buffer strips, these percentages are 89%, 73%, 88%, and 99%, respectively.[1] Robinson, Ghaffarzadeh, and Cruse reported that an initial 9.8-foot-wide vegetative filter strip removed more than 70% of the sediment from runoff, while the 30-foot-wide vegetative filter strip removed 85% in their study area. The following percentage reductions in water pollutant loading are used: sediment yield, 85%; NCSW, 75%; and ACSW, 85%. Average values of the estimated water quality indicators for watershed are presented in Table 1.

Contribution rate determines how the pollutants generated in a subwatershed affect water quality at the watershed outlet. It simplifies the modeling of emission transport by assuming that all agricultural emissions generated in a subwatershed are transported to the watershed outlet in the stream flow (no deposition or degradation). This assumption is reasonable for a small watershed because the residence time of pollutants in stream water is typically quite short. It is more appropriate for dissolved pollutants such as nitrate and atrazine than for suspended pollutants such as sediment. Contribution rate is measured by the ratio of the area in a subwatershed to the total area of the watershed for SY and by the ratio of runoff water volume in each subwatershed to total water runoff volume in the watershed for NCSW and ACSW. The numerator and denominator of these ratios are calculated using monthly output from SWAT.

Estimation of Net Return

Net return for each farming system varies across subwatersheds due to spatial variability in crop yields caused by soil, hydrologic, and topographic conditions. Net return equals gross revenue minus average production cost.

[1] The experimental report was obtained from Douglas Wallace, NRCS, USDA, Columbia, Missouri.

Gross revenue depends on crop yields and prices, and production cost depends on cultural practices, input use, and input prices. This study uses a composite crop yield that is the product of a crop yield goal and a crop yield index. The crop yield goal reflects farmers' expectation of crop yield and captures management factors, such as fertilizer and pesticide use and field operations. It is determined based on field experimental data from the Missouri MSEA project, Missouri average crop yields for the two counties in which the watershed is located, and fertilizer/pesticide use for each farming system. Even though the yield goal is the same, actual crop yield may be higher or lower due to natural conditions. The crop yield index captures the impacts of natural (soil, topographic, and hydrologic) conditions in each subbasin on crop yield. It is calculated using crop yields simulated with SWAT and is the ratio of the simulated crop yield in a subwatershed to the average simulated crop yield in the watershed. The simulated crop yields are not used to calculate net return because they are not sensitive to management factors such as changes in fertilizer use as noted by Heidenreich, Zhou, and Prato. However, they vary across subbasins due to variation in natural conditions.

Production costs are estimated using the CARE (Cost and Return Estimator) enterprise budget generator (USDA SCS). Production costs include the variable costs of machinery ownership and operation, materials, fuel and labor, capital, drying, and management. Gross revenue for a farming system is the sum of the products of composite crop yields and their respective market prices. Agricultural input and output prices used to estimate net return are average 1990–94 prices for the Midwest. Annual average net return for a farming system is obtained by dividing total net return for all crops in the rotation by the number of years in the rotation. Annual gross revenue for the 37th farming system is assumed to be \$60 per acre. This is the rental rate in Missouri for cropland enrolled in the CRP. The average annual net return for each farming system is presented in Table 1.

Results and Discussion

The baseline solution maximizes total watershed net return (TWNR), given in equation (1), subject to resource constraints given by equations (4) and (5). Since CBMHH (corn-soybean rotation, minimum tillage with high pesticide and high fertilizer application rates) is the most profitable farming system for all cropland in the watershed, it is designated as the baseline farming system. CBMHH is the most used farming system in this watershed, so the baseline actually reflects production in this watershed. In the baseline, average annual net return (AANR) is \$96.51 per acre, TWNR is \$1,328,535, SY is 0.505 tons per acre per year (TAY), NCSW is 3.86 parts per million (ppm) and ACSW is 44.47 parts per billion (ppb) at the watershed outlet. These results indicate that ACSW is the most serious water quality problem in the watershed. An ACSW of 44.47 ppb is almost 15 times greater than the drinking water standard of 3 ppb established by EPA. This result is consistent with the findings of the Missouri MSEA project (Missouri MSEA Management Team). Four atrazine reduction objectives are evaluated: reducing concentrations at the watershed outlet from 44.47 ppb (baseline value) to 24, 12, 6, or 3 ppb.

First the uniform atrazine reduction plan is compared with the least-cost plan for atrazine reduction. The uniform reduction plan reduces the atrazine concentration in each subwatershed to the desired level defined above without regard to variations in natural conditions among different subwatersheds and achieves the objective at the watershed outlet. The least-cost reduction plan reduces the atrazine concentration at the watershed outlet to the desired level, but atrazine reduction in subwatersheds depends on the natural conditions in subwatersheds. The resulting atrazine concentrations in subwatersheds are not necessarily the same and are arranged in a least-cost fashion. Table 2 gives the watershed-scale abatement costs for each objective under the two reduction plans. Total abatement cost is measured by the reduction in TWNR resulting from the achievement of a particular objective.

Table 2. Total and Marginal Abatement Costs, and Abatement Taxes for Atrazine Abatement in Goodwater Creek Watershed

	Abatement Cost					
	Uniform Reduction[a]		Least-Cost Reduction[b]			
Target ACSW (ppb)	Total ($)	Average ($/acre)	Total ($)	Average ($/acre)	Marginal Cost[c] ($/ppb)	Total Ambient Tax[d] ($)
44.47	7,775	0.66	0	0.00	1,558	0
24	106,449	7.73	78,461	5.70	12,846	308,304
12	490,038	35.60	432,762	31.44	36,512	438,144
6	720,604	52.35	676,130	49.12	43,461	260,766
3	836,388	60.76	808,470	58.73	45,418	136,254

[a] Achieves the water quality objective shown in Column 1 at each subwatershed outlet.
[b] Achieves the water quality objective shown in Column 1 at the watershed outlet and atrazine abatement varies by subwatershed and is determined in a least-cost fashion.
[c] For atrazine abatement at the watershed outlet. Determined by the least-cost framework and equals the shadow price.
[d] Marginal cost times the unrealized atrazine abatement. It is the products of Column 1 and Column 6.

The average abatement cost is the total abatement cost divided by the total cropland acreage in the watershed. The abatement cost under the uniform reduction plan is consistently higher than under the least-cost reduction plan. This result implies that accounting for spatial characteristics of a watershed is important for implementing alternative water pollution abatement policies.

Table 2 also presents the marginal cost and total effluent tax under the least-cost framework. Marginal cost of abating ACSW is the shadow price for water quality improvement at the watershed outlet and is in the solution set of the mathematical programming model presented above. As an ambient tax rate is set equal to the marginal cost of abatement proposed by Baumol and Oates, total ambient tax is approximated by the ambient tax rate times the unrealized abatement in water pollution. For example, the ambient tax rate is $12,846 per ppb for achieving 24 ppb of ACSW and the total ambient tax is $308,304, which is the product of the ambient tax rate and the unrealized abatement of 24 ppb.

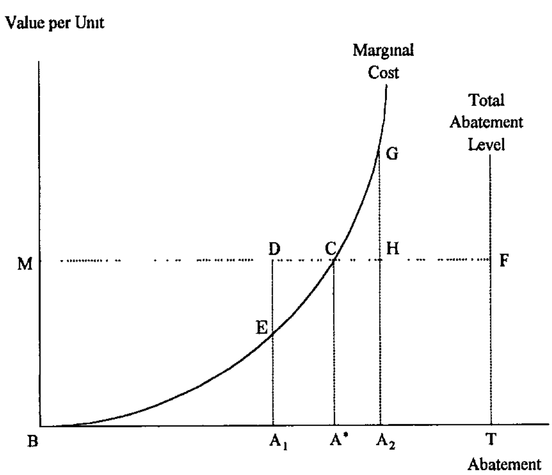

Figure 1. Marginal Cost Curve for Water Pollution Abatement

The relationships among abatement cost, marginal cost, and ambient tax can be explained using the marginal cost of abatement curve illustrated in Figure 1. Let BECG be the marginal cost curve for water pollution abatement, B the pollution level in the baseline, BT the level of complete abatement and BA* the desired level of abatement. As shown in Figure 1, the marginal cost is BM and the total abatement cost is the area A*BC for achieving the abatement level of BA*. By setting the ambient tax rate for achieving BA* equal to the marginal cost of BM, the total ambient tax is the area A*TFC, the product of ambient tax rate (BM) and the unrealized abatement (A*T).

Figure 1 also shows how the desired abatement level of BA* can be achieved by imposing the ambient tax rate of BM. Consider a reduction in abatement from A* to A_1 (the farmer is choosing to abate less than desired).

Here the added tax A^*A_1DC is greater than the abatement cost A^*A_1EC by EDC. To minimize cost, the farmer would choose to pay the cost of abatement and avoid the tax by returning to A*. On the other hand, if farmers reduce pollution by more than A*, such as A_2, the ambient tax is the area A_2HFT and the abatement cost is A_2BG. Farmers bear higher abatement costs (area CHG). When farmers choose an abatement level of BA*, the loss in net returns is minimized (area A_1DFT + area A_1BE). Therefore, an ambient tax is an ideal mechanism for achieving the desired abatement level if farmers act rationally. However, an ambient tax overburdens the farmers as Jacobs and Casler argue. With an ambient tax, farmers not only pay the ambient tax but also bear the abatement cost. We evaluate three policies that achieve the desired water quality in a watershed with least cost and overcome the drawback of the original ambient tax.

The first policy is to regulate the use of farming systems in the watershed so as to achieve the cost-effective spatial pattern of farming systems given by the mathematical programming model, namely $\{x_{ij}^*\}$. The cost-effective spatial pattern of farming systems in the watershed for achieving 12 ppb of ACSW is given in Table 3. In order to achieve 12 ppb of ACSW at the watershed outlet, all cropland in subwatersheds 7, 8, 11, 12, 14, 16, 17, 20, 22, 27, 28, 30, and 32 has to be in farming system SBMHL; cropland in subwatersheds 10, 15, 18, 21, 24, and 25 has to be in farming system CBWMHL; cropland in subwatersheds 1, 2, 3, 4, 6, 9, 13, 19, 23, 26, and 29 has to be taken out of crop production and enrolled in the CRP; and cropland in subwatershed 5 has to be split between CRP and farming system CBWMHL. Subwatershed 31 does not show up in Table 3 because there is no cropland in this subwatershed. This policy achieves a given water quality objective at minimum cost in terms of the reduction in TWNR, i.e. the abatement cost is at a minimum and there are no other charges to farmers. Implementation cost is low because only the spatial pattern of farming systems needs to be monitored which is relatively easy. It is a simple policy because it does not involve economic incentives. As presented in Table 3, a regulatory policy has differential impacts on farms in the watershed. Regulation of farming systems is likely to be opposed by property rights advocates. For comparison, Table 3 also presents the spatial pattern of farming systems under a uniform pollution reduction plan, which is more complicated than under the least-cost reduction plan. Under the uniform reduction plan, cropland in each subwatershed has to be split into two farming systems.

The second policy is the alternative ambient tax. The alternative ambient tax approach sets the tax rate equal to the marginal cost of water pollution abatement, which is the same as the ambient tax discussed above. However, the total alternative ambient tax is approximated by the tax rate times the difference between the desired and measured water quality instead of the unrealized pollution abatement. As shown in Figure 1, with an alternative ambient tax, if farmers decide not to reduce water pollution, they have to pay a total tax given by the area A*BMC. If they reduce water pollution to the level indicated by A_1, they bear an abatement cost given by the area A_1BE and pay a tax given by the area A^*A_1DC. With an alternative ambient tax, there is no extra charge for farmers if water pollution is reduced to the desired level A* or lower. Hence, not reducing pollution and paying an ambient tax is always more expensive than reducing pollution and bearing the abatement cost. When water pollution abatement is more than the desired level of A*B, farmers do not pay the tax but bear more abatement cost. If farmers are rational, they would just reduce water pollution to the desired level (A*). Therefore, the desired water quality objective can be achieved by imposing the alternative ambient tax. The alternative ambient tax functions like the ambient tax but does not overburden farmers.

Two options exist for implementing an alternative ambient tax using the marginal costs of water pollution abatement at both watershed and subwatershed levels. First, the ambient tax can be implemented at the watershed level. For example, if the objective is to achieve 24 ppb for ACSW, the tax rate is

Table 3. Proportion of Cropland in Farming Systems for Subwatersheds under Different Plans for Achieving 12 ppb of ACSW in Goodwater Creek Watershed

Sub-watershed	Uniform Reduction Plan[a]				Least Cost Reduction Plan[b]		
ID	CBMHL	SBMHL	CBWMHL	GLCNN	SBMHL	CBWMHL	GLCNN
1		0.463		0.537			1
2		0.508		0.492			1
3		0.518		0.482			1
4		0.924		0.076			1
5			0.759	0.241		0.433	0.567
6		0.530		0.470			1
7			0.955	0.045	1		
8			0.669	0.331	1		
9		0.469		0.531			1
10			0.977	0.023		1	
11		0.483		0.517	1		
12		0.487		0.513	1		1
13		0.554		0.446			
14		0.505		0.495	1		
15			0.647	0.353		1	
16			0.637	0.363	1		
17			0.634	0.366	1		
18			0.664	0.336		1	
19		0.539		0.461			1
20			0.997	0.003	1		
21			0.671	0.329		1	
22			0.743	0.257	1		
23		0.509		0.491			1
24			0.654	0.346		1	
25			0.670	0.330		1	
26			0.690	0.310			1
27			0.745	0.255	1		
28		0.468		0.532	1		
29		0.559		0.441			1
30	0.493	0.507			1		
32			0.730	0.270	1		

[a] Achieves 12 ppb of ACSW at each subwatershed outlet.

[b] Achieves 12 ppb of ACSW at the watershed outlet and atrazine abatement varies by subwatershed and is determined in a least-cost fashion.

$12,846 per ppb. Second, the ambient tax can be implemented at the subwatershed level. To achieve the same objective, the tax rate is $413.65 per ppb in subwatershed 1 and $503.57 per ppb in subwatershed 2. Tax rates vary across subwatersheds. However, implementation is difficult because it requires monitoring water quality at the subwatershed and watershed outlets. In addition, the alternative ambient tax is based on an ambient concentration of atrazine in stream water so it is very difficult to link the tax to specific farms in the watershed.

The third policy is an abatement tax/subsidy. The policy is a variant of the alternative ambient tax and has properties similar to the alternative ambient tax. It achieves water quality in a cost-effective fashion but does not overburden farmers. With an abatement tax/subsidy, farmers bear only the total abatement cost of achieving the desired water quality. If farmers do not intend to change their farming practices, they have to pay the total abatement cost as a tax. The tax is then paid back as an incentive to farmers who adopt the effective spatial pattern of farming systems in the watershed.

Implementation of this policy assumes that all farmers in the watershed are responsible for water pollution abatement. Therefore, the abatement tax should be levied based on total cropland acreage. The tax rate equals the total abatement cost divided by the total cropland acreage in the watershed, that is, the average abatement cost shown in Column 5 of Table 2 and the last row in Table 4. Suppose the water quality objective is 12 ppb for ACSW at the watershed outlet. The total abatement cost is $432,762 and the abatement tax rate is $31.44 per acre of cropland.

The tax revenue is then used to subsidize farmers who use the farming systems given by Table 3. The incentive mechanism can be designed based on average abatement costs in subwatersheds. Average abatement cost in a subwatershed is the reduction in subwatershed net return per acre from achieving a particular atrazine reduction objective at the watershed outlet. It is calculated by dividing total abatement cost by the cropland acreage in the subwatershed. Average abatement cost varies over subwatersheds as shown in Table 4. Achieving 12 ppb of ACSW entails an average abatement cost that ranges from zero in subwatershed 31 (which has no cropland) to $68.45 per acre in subwatershed 29. By changing their farming system from CBMHH (the baseline farming system) to GLCNN in subwatershed 29 as shown in Table 3, farmers in this subwatershed can be compensated up to $68.45 per acre of cropland. The same rule can be applied to other subwatersheds. Through a well-designed incentive mechanism, the abatement tax/subsidy policy should induce the effective spatial pattern of farming systems and achieve the desired improvements in water quality.

An abatement tax/subsidy policy entails the minimum reduction in TWNR for achieving a desired water quality objective. Implementing the abatement tax/subsidy requires the monitoring of farming systems, which is much easier than monitoring pollution. Another advantage of the abatement tax/subsidy is that it can be applied to specific farms based on their location in the watershed.

Table 4. Average Atrazine Abatement Costs in Subwatersheds of Goodwater Creek Watershed under the Least Cost Plan, Dollars per Acre

Sub-watershed ID	Target ACSW (ppb)				
	44.47	24	12	6	3
1	0.00	4.89	54.61	54.61	54.61
2	0.00	7.20	66.97	66.97	66.97
3	0.00	7.20	66.97	66.97	66.97
4	0.00	6.15	54.61	54.61	54.61
5	0.00	2.91	47.01	66.97	66.97
6	0.00	5.93	66.97	66.97	66.97
7	0.00	6.79	6.79	6.79	6.79
8	0.00	3.06	8.90	69.94	69.94
9	0.00	8.95	68.03	68.03	68.03
10	0.00	2.89	18.77	66.55	66.55
11	0.00	6.36	6.36	69.94	69.94
12	0.00	6.79	6.79	44.34	72.91
13	0.00	6.59	57.58	57.58	57.58
14	0.00	8.28	8.28	74.39	74.39
15	0.00	7.24	19.65	65.07	65.07
16	0.00	8.28	8.28	23.61	23.61
17	0.00	3.43	9.98	24.70	56.98
18	0.00	2.96	22.13	68.03	68.03
19	0.00	8.30	63.58	63.58	63.58
20	0.00	5.52	7.85	21.13	21.13
21	0.00	3.21	22.13	22.13	22.13
22	0.00	3.21	10.60	25.12	72.91
23	0.00	7.24	65.07	65.07	65.07
24	0.00	3.28	22.99	22.99	74.39
25	0.00	2.81	20.73	65.07	65.07
26	0.00	7.03	63.58	63.58	63.58
27	0.00	3.13	9.12	9.12	71.42
28	0.00	8.72	8.72	8.72	8.72
29	0.00	8.68	68.45	68.45	68.45
30	0.00	3.43	9.56	9.56	9.56
32	0.00	3.21	9.33	25.12	72.91
Average[a]	0.00	5.70	31.44	49.12	58.73

[a] At the watershed level same as Column 5 of Table 2.

Summary and Conclusions

Spatial characteristics of a watershed appear to significantly affect pollution abatement cost. Comparison between the uniform and least-cost reduction plans shows that ignoring spatial differences in a watershed is likely to increase abatement cost. This paper uses a spatial mathematical programming model to evaluate three pollution control policies that account for the spatial characteristics of the watershed and achieve the desired improvements in water quality at minimum cost in Goodwater Creek watershed, Missouri. These three policies are regulating the spatial pattern of farming systems, an alternative ambient tax, and an abatement tax/subsidy. The three policies are equally cost-effective because they minimize the abatement cost (loss in TWNR) of reducing atrazine concentration in surface water. It should be noted that the solution set consists of packaged farming systems. If the solution set were more flexible with respect to crop rotation, input use, and tillage practice, it is unlikely that the three policy alternatives would yield identical minimum costs.

The policy alternatives' implementation feasibility is quite different. Regulation is likely to be viewed as an infringement on the property rights of farmers. An ambient tax would require water quality monitoring which is expensive and time consuming. Implementation of the abatement tax/subsidy requires monitoring the spatial pattern of farming systems. Even though it is relatively easy to monitor crop rotation and tillage practices, it is quite difficult to monitor input use. Implementation of the abatement tax/subsidy would have similar difficulties as regulation of input use.

One problem with the integrated watershed management approach to control agricultural nonpoint source pollution is the inconsistency between a hydrological unit such as a watershed and subwatershed and an agricultural management unit such as a farm. Water pollutants in a stream can come from different farms and a farm can be a pollution source for several streams. This paper shows that it is feasible to apply an abatement tax/subsidy policy to farms within a subwatershed. Specifically, an abatement tax/subsidy can be tailored to specific farms based on their location in the subwatershed.

Selection of a policy is much more complicated than indicated in this paper. Abatement cost is just one aspect of feasibility. Policy implementation costs, political support, and stakeholders' preferences are other factors that need to be considered in selecting a policy.

Another major finding of this paper is that it would be very costly to reduce atrazine concentration at the outlet of Goodwater Creek watershed to the drinking water standard of 3 ppb using the farming systems examined here. Total abatement cost would be $808,470 and average abatement cost would be $58.51 per acre, which amounts to 61% of TWNR and average annual net returns in the baseline. In addition, CRP rental payments to farmers would amount to $683,347. This finding suggests that other technical and policy options are needed to achieve the atrazine standard.

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