Using Spatial Information to Reduce Costs of Controlling Agricultural Nonpoint Source Pollution

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Reducing costs of controlling nonpoint source (NPS) pollution will be a high public priority in the next century. Compliance and transaction costs of reducing nitrogen runoff from dairies in the Lower Susquehanna Watershed by 40% are estimated for perfectly targeted and uniform performance standards. The perfectly targeted standard reduces compliance and transaction costs by almost 75% compared with the uniform standard. Future NPS control policies should use spatial information to target policy resources to priority concerns, areas, and farms. Further research is needed to lower the costs and increase the accuracy of spatial information.

Effective policy design to reduce nonpoint source (NPS) pollution in agriculture will be crucial in the next century because of increasing public desire for water quality protection, limited public funds for reducing water pollution, and ambivalence about command and control approaches. Textbook analyses (Bohm and Russell 1985; Tietenberg 1998) imply that flexible, decentralized instruments for protecting water quality should be favored when polluting firms vary in their resources and pollution control costs. Flexible instruments are policies that can adapt to changing economic conditions (tastes and preferences of consumers, production technologies, resource stocks) in order to achieve the water quality goal (Bohm and Russell 1985). Flexible instruments allow actors to take advantage of their unique information in order to minimize compliance costs of controlling pollution.

But such analyses do not usually consider transaction costs. Transaction costs include all costs incurred by the water quality agency in order to carry out the pollution control policy. Transaction costs include information, contracting, and enforcement costs (Krier and Montgomery 1973). From a water quality agency's viewpoint, information costs include the costs of determining (1) the identification of farms to be targeted; (2) the set of actual farm practices and pollutant loadings in the watershed and linkages between practices and loadings; and (3) which practices need to be adopted to achieve objectives of reduced loadings. Contracting costs are the administrative and staffing costs involved in contacting targeted farms, in reaching agreements with farmers about practices that must be adopted or loadings reductions that must be achieved, and in writing up contracts to create the legal status necessary to implement policies. Enforcement costs are incurred in determining whether pollution-reducing practices have been implemented or pollution reductions achieved and in imposing and extracting penalties from noncomplying farms. Enforcement costs include litigation costs when required practices or pollution reductions are appealed.

Compliance costs are reductions in farmers' net incomes in order to comply with the pollution control goal. Compliance costs are likely to decrease as policies come to be based on more spatial information. The effects of spatial information on transaction costs are unclear and site-specific. Information costs may increase with targeting because of the increased costs of collecting more spatial information used to decide which firms to target. However, contracting and enforcement costs may decline because targeting reduces the number

of farms requiring contracting and enforcement activities.

Advances in knowledge about how to collect and analyze spatial information (e.g., geographic information systems, satellite imagery) suggest that the supply of spatial information is increasing. Because of potential to use spatial information to achieve water quality goals while reducing compliance costs and *perhaps* transaction costs, the demand for spatial information to support water quality protection efforts could also increase. Improved spatial information may yield large benefits to society because NPS pollution is heterogeneous and diffuse. Increases in demand for spatial information depend on whether the net benefits from the use of such information to reduce NPS pollution control costs over uniform policy design can be demonstrated.

In this study, we present a conceptual framework for estimating the value of spatial information in reducing NPS control costs and demonstrate the framework with an empirical application to Lower Susquehanna watershed dairies. This watershed is located mainly in Pennsylvania with a small portion in Maryland. We then suggest implications of the empirical results for future research and policy design to facilitate use of spatial information for NPS control in agriculture.

Conceptual Framework

The value of information in reducing pollution control costs depends on the type of policies used to reduce pollution. Here we focus on the value of information for allocating pollution reductions under a regulatory performance standard. The framework, however, is general and could be applied to other policies, including design standards, practice subsidies, and subsidized pollution reduction.

We assume society has chosen the pollution reduction goal and wishes to minimize costs of achieving the goal. The value of information (V_P) is based on the difference between total control costs of a performance standard uniformly applied to all farms (CC_U) and a perfectly targeted performance standard (CC_P).

$$(1) \qquad V_P = CC_U - CC_P.$$

CC_U is estimated as:

$$(2) \qquad CC_U = \sum_i [C_{iU}(r_i) + TC_{iU}(r_i)].$$

C_{iU}, the ith farm's compliance cost of achieving the uniform standard, is a function of r_i, the level of pollution reduction on the ith farm. Compliance

costs equal the reduction in net farm income resulting from complying with the practices required to reduce pollution. Reductions in net income may be due to the farmer's direct costs of implementing required practices or structures as well as opportunity costs from having to eliminate or reduce profitable enterprises. We assume pollution reduction costs on each farm will increase at an increasing rate with the amount of reduction.

TC_{iU}, the transaction costs of achieving the performance standard, are also a function of pollution reduction (r) as well as the number of farms (i) to which the standard is applied. A more stringent performance standard (greater pollution reduction) may impose more complex practices and higher costs on farmers, which may increase farmers' incentives to avoid carrying out the practices and the agency's costs of enforcing the standards. Applying the standard to more farms increases the agency's transaction costs, because the agency must cover more area and deal with more farmers in contracting and enforcing the standard. Performance standards have high transaction costs because of the difficulty of measuring or estimating farmers' pollution as well as the large number of pollution-reduction practices available. Measurement is complicated because an individual farm's environmental performance is affected by dynamic and spatial influences, including extreme weather events and topographic features.

Furthermore, setting a performance standard for agroenvironmental pollutants in water presumes that the science exists to trace these contaminants to specific sources and practices. Such knowledge is not available for many agroenvironmental problems, because agricultural NPS pollution is diffuse and often subject to long lags before pollutants are observed in surface water or groundwater. Simulation models may substitute for field monitoring studies, although they too can be handicapped by lack of validated data on the linkages between polluting source and impact (Batie and Ervin 1997; DeCoursey 1985; Negahban et al. 1994; Thomann et al. 1994; Rader 1994). These complications add to transaction costs and an uncertainty of achieving the standard.

Despite these complications, information with respect to on-farm spatial characteristics could be used to estimate farm costs of reducing pollution. Aggregate compliance costs could be lowered by assigning greater reductions to farms with lower compliance costs. Minimum total control costs with perfect information (CC_P) are:

$$(3) \qquad Min\ CC_p = Min\ \sum_i [C_{iP}(r_i) + TC_{iP}(r_i)]$$

subject to:

(4) $0 \le r_i \le e_i$

(5) $\sum_i [e_i - r_i] \le Z.$

Pollution reduction on any farm (r_i) must be less than the farm's unconstrained pollution level (e_i). The standard requires that total pollution from the farms in the watershed not exceed a designated performance standard level Z. The performance standard would need to be set by agency authorities and address a single pollutant such as nitrogen deliveries to surface water and/or groundwater.

Minimization of total costs occurs under the following conditions:

(6) $\dfrac{\partial C_i}{\partial r_i} + \dfrac{\partial TC_i}{\partial r_i} - \lambda \ge 0$

(7) $r_i \left[\dfrac{\partial C_i}{\partial r_i} + \dfrac{\partial TC_i}{\partial r_i} - \lambda \right] = 0$

(8) $\lambda \left[\sum_i [e_i - r_i] - Z \right] = 0$

(9) $\lambda \ge 0.$

Each farm's marginal compliance plus transaction costs must equal or exceed λ, the shadow price for the pollution constraint, which is the incremental cost of a one-unit reduction in Z, the allowable pollution in the watershed (equation [6]). Farms for which the marginal increase in transaction plus compliance costs exceeds λ are not required to reduce pollution $(r_i = 0)$ (equation [7]). Farms required to reduce pollution have marginal compliance plus transaction costs equal to λ. If the sum of pollution is less than Z, the shadow price of the pollution constraint is zero $(\lambda = 0)$ (equation [8]). The pollution shadow price is nonnegative (equation [9]). If farms have unequal compliance costs, an assignment of pollution reductions to farms that satisfies equations (6–9) will reduce total compliance costs compared with a uniform performance standard.

The effects of targeting on transaction costs are uncertain. Information costs for targeted and uniform standards might be similar assuming information on all farms must be collected in either case, although perhaps more detailed information would be needed to decide which farms to target. Contracting and enforcement costs might decline with targeting because fewer farms would have to be selected and monitored for compliance compared with a uniformly applied standard.

Empirical Application

Case Study Area

The study area is the Lower Susquehanna watershed, which drains to the Chesapeake Bay. In 1983, Pennsylvania, Maryland, Virginia, the District of Columbia, the Chesapeake Bay Commission, and the Environmental Protection Agency signed the Chesapeake Bay Agreement, by which all parties agreed to cooperate to protect and restore the Bay. In 1987 these parties agreed to reduce controllable nitrogen and phosphorus entering the bay by 40% by the year 2000. Control efforts have focused on agriculture, which contributes an estimated 39% of nitrogen and 49% of phosphorus entering the bay (Chesapeake Bay Program 1996). An estimated 16% baywide reduction in total phosphorus was achieved from 1984 to 1992, but nitrogen levels did not change significantly (Chesapeake Bay Program 1994). In 1992, the agreement partners reaffirmed their commitment to the 40% reduction goal beyond 2000. They also adopted "tributary strategies," which called on states to target their nutrient reduction strategies according to nutrient problems within each river basin (Virginia Chesapeake Bay Program 1993).

The Lower Susquehanna watershed consists of five million acres, including 1.5 million in agriculture. This agricultural acreage is estimated to be the largest single source of nutrient pollution in the watershed (USEPA 1992). The U.S. Department of Agriculture's Economic Research Service, the National Agricultural Statistics Service, and the Natural Resource Conservation Service conducted a survey of field and farm-level agricultural and conservation practices at Natural Resource Inventory (NRI) sites in this watershed. The Lower Susquehanna Area Studies Survey combines economic and management data with detailed site information. Personal interview surveys determined farming practices on fields encompassing 500 randomly selected NRI sites in the watershed (weighted for soil hydrological groups). The NRI contains numerous physical attributes of randomly selected cropland and pasture land sites (USDA/SCS n.d.). Input quantities, timing of input applications, and management practices are available for each selected field. Sales, total acres in each crop, labor use, livestock numbers, and government program participation are also available for the farm that includes each field.

Our study focused on 246 sites located on dairy farms, of which 232 were located in Pennsylvania and 14 in Maryland. Nine farms were discarded from the analysis because they had missing infor-

mation, leaving 237 farms in the analysis. Of the sample dairy farms 37% had sales between $0 and $99,999, 39% between $100,000 and $249,999, 17% between $250,000 and $499,999, and 7% had sales exceeding $500,000.

Each farmer's constrained optimization problem is to maximize profits (π)

$$(10) \quad Max\ \pi_i = Max \sum_m P_m Y_{im}(X_{ikm}(d_i))$$

subject to:

$$(11) \qquad \sum_m B_{ikm} X_{ikm} \le b_{ik}$$

$$(12) \quad \sum_k \sum_m e_{ikm}(Y_{ikm}, X_{ikm}, d_i) \le z_i(S)$$

Y_{im} is a multioutput production function on the ith farm producing m crop and livestock activities using inputs $x_1, x_2, \ldots x_k$, and P_m is output m's net return. Outputs can be produced with differing technologies, which represent unique combinations of the k inputs. Output also depends on d_i, an index of soil characteristics on the ith farm that affect crop productivity and potential for runoff. Activities are linked by conventional constraints such as limits on total land and labor (b_{ik}). B_{ikm} is use of the kth input in activity m on the ith farm. If a performance standard is imposed, the sum of nitrogen runoff delivery resulting from the k inputs and m outputs (e_{ikm}) must be less than z_i, the farm's allocation of nitrogen runoff delivery. The level of z_i depends on the targeting standard (S) where $S = 1$ is the uniform performance standard and $S = 2$ is the perfectly targeted performance standard.

With the targeted performance standard, a sequential optimization ensures that equations (6) through (9) are satisfied and total compliance plus transaction costs for the watershed are minimized. A watershed level optimization chooses the least costly farms to target as well as the amounts of nitrogen runoff reduction to be achieved by targeted farms when both compliance and transaction costs are considered.

The maximization problem is solved using a linear programming model (SUSFARM) designed for farms in the Lower Susquehanna River watershed (Bosch, Carpenter, and Heimlich 1995). SUSFARM is written in GAMS (General Algebraic Modeling System) (Brooke, Kendrick, and Meeraus 1992) and solved with MINOS (Modular In-core Nonlinear Optimization System). Input files containing the information from the survey specific to each farm, such as management practices and type and amount of land and livestock,

are read by GAMS, and each farm is solved sequentially.

Crop and dairy livestock product sale prices are Pennsylvania weighted average prices for 1991–95, and variable input costs are from Pennsylvania farm enterprise budgets (Pennsylvania Cooperative Extension Service 1992). All costs and prices are expressed in 1996 dollars. Fixed costs are not included. Labor is provided by fixed family labor as well as seasonal labor, which can be hired at $6.50 per hour.

Livestock Enterprises

Livestock enterprises include dairy cow, beef cow-calf, hog farrow-to-finish, and poultry broiler operations. Four rations are available for the dairies: alfalfa-corn silage, corn silage only, alfalfa hay only, and alfalfa haylage. Milk production per cow is a function of herd size (Ford 1992), and feed requirements are in turn a function of milk production. Livestock facilities are assumed to be fixed in the short run, and the herd size cannot exceed the number of livestock each farmer reported in the survey.

Balance equations ensure that all manure produced is spread on the producing farm's cropland. No more than 25% of manure production can be spread in any season unless the farmer reported having manure storage facilities or constructed some storage at a fixed cost per unit of manure capacity. Per-ton manure spreading and storage costs are synthesized based on machinery, labor, and storage facility requirements (Ritter 1990).

Crop Enterprises

SUSFARM includes alfalfa, corn grain, corn silage, grass pasture, wheat, soybeans, oats, grass hay, and rye cover, which account for nearly 93% of crop acreages reported in the survey. Each farm's total land available for crops and pasture is based on the survey. Balance equations require crops produced or feeds purchased to equal or exceed livestock feed requirements plus sales. Alfalfa, corn grain, soybeans, oats, and grass hay can be bought and sold. Wheat can only be sold, while soybean meal can only be purchased. Crop yields are based on the soil type at the sample site (Serotkin 1993). Soil properties at the sample site are assumed to hold for the entire farm.

Crop nutrients can be applied as animal manure and/or commercial fertilizer. Nitrogen is also obtained from precipitation, legume fixation and carryover, and mineralization of soil organic matter. Manure nutrient content equals the amount of

plant-available nitrogen, phosphate, or potash per ton of manure (in dry matter). Manure nitrogen volatilization losses in storage and after spreading and seasonal nitrogen runoff and leaching between the time of spreading and crop uptake are subtracted from nitrogen availability.

SUSFARM distinguishes thirty-six rotations, which refer to sequences of crops and tillage operations. Rotations have some combination of four tillage types (conventional, reduced, no-till, and none), nine crops, and contour strip-cropping or no strip-cropping. Choice of rotation affects potential soluble and sediment-adsorbed nitrogen runoff. The estimated effectiveness of strip-cropping in reducing erosion is taken from the USDA/SCS (1991), while annualized costs of implementing contour strip-cropping are taken from Camacho (1992).

Under the Federal Agricultural Improvement and Reform Act (FAIR, HR 2854) (USDA/ERS 1996) crop production is not restrained by commodity program participation. Government program payments are not estimated because they are not affected by the farm's nitrogen runoff.[1]

Nitrogen Applications and Delivery

Mass-balance equations in SUSFARM require that nitrogen from mineralization of soil organic matter, precipitation, commercial fertilizer, manure, legume fixation, and legume carryover equal or exceed crop uptake after accounting for nitrogen volatilization, leaching, and runoff. Nitrogen contributions to crops and nitrogen runoff from fertilizer and manure spreading are calculated according to how, when, and where spreading occurs. The model contains two methods of nitrogen application (surface applied and incorporated) and four seasons of application. Nitrogen runoff is reduced by applying closer to the season of plant uptake and by incorporation.

Soluble nitrogen runoff in each season depends on precipitation, the partition of precipitation into runoff and infiltration, and the nitrogen available to runoff (Yagow et al. 1990). The precipitation partition into infiltration and runoff depends on the average rainfall per rainfall event as well as crop, tillage, and hydrological soil group (USDA/SCS 1986). All soluble nitrogen runoff is assumed to be delivered to surface water.

Annual nitrogen loss in sediment is a fraction of the annual per-acre sediment erosion for each crop rotation as calculated from the Universal Soil Loss

Equation (USLE) (Wischmeier and Smith 1978; USDA/SCS 1991). Potential to reach surface waters is estimated in SUSFARM with the sediment delivery ratio described by Shanholtz and Zhang (1988). The delivery ratio is a function of the distance to the nearest body of water, the land cover along the flow path from fields to the receiving water body, and the slope of the flow path to the receiving water body.[2]

Transaction Costs

Transaction costs were estimated by identifying and budgeting costs of activities required to target and enforce nitrogen runoff reductions (Carpentier 1996). These activities include the initial activities and costs to collect the information, contract with the farmer, and enforce the agreement for each regulatory standard as well as the activities and costs to update the implementation each year over a ten-year horizon (table 1).[3]

Uniform Performance Standard

For the purposes of this study, it is assumed that a uniform 40% reduction in nitrogen is applied to all sample farms. The agency estimates a baseline pollution loading level e_i and the practices that achieve the 40% reduction on each farm. Because it would not be practical to monitor pollution from each farm, simulation models such as EPIC (Erosion Productivity Impact Calculator) (Williams et al. 1989) that predict the flow and path of pollution under farm conditions are used to estimate the baseline. To run such a model, agency personnel must determine farm physical characteristics and boundaries. An agent must travel to the farms and gather field boundary information, calibrate the simulation model for the area, and estimate a delivery ratio to surface water for each farm's runoff. The simulation model is used to identify practices that achieve a 40% reduction on each farm. An estimated seventy-three hours per farm are required initially to gather the necessary information for the uniform performance standard, and twelve hours are required each year to update the plan to account for changing economic conditions.[4]

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[1] Farms need to comply with conservation requirements on highly erodible land (HEL) in order to receive government payments.

[2] More details on nitrogen runoff calculations are described in Bosch, Carpentier, and Heimlich 1995.

[3] Estimated enforcement costs do not include litigation costs.

[4] Information costs could also be incurred for monitoring of ambient water quality, identification of the pollution contributions from point and nonpoint sources, consideration of stream flow and temperatures, and possible resetting of the performance standard, Z (equation [5]), or respecification of the linkages in the simulation model between farm practices and environmental impacts. These complications of monitoring and adaptive feedback strategies are not addressed in this study.

Table 1. Activity Hours and Transaction Costs Per Farm to Implement Performance Standards[a]

	Activities			
	Information	Contracting	Enforcement	Total
Uniform performance				
Initial (hrs)	73.00	0.00	0.00	73.00
Update (hrs/yr)	12.00	4.00	3.40	19.40
Annualized cost ($)	496.00	109.00	86.00	691.00
Targeted performance				
Nontargeted farms				
Initial (hrs)	77.00	0.00	0.00	77.00
Update (hrs/yr)	0.00	0.00	0.00	0.00
Annualized cost ($)	233.00	0.00	0.00	233.00
Targeted performance				
Targeted farms				
Initial (hrs)	80.00	0.00	0.00	80.00
Update (hrs/yr)	16.00	7.00	3.40	26.40
Annualized cost ($)	610.00	158.00	86.00	854.00

Initial hours represent hours required in the first year to implement the standard. Update hours are the hours required each of the following years to maintain the standard.

[a]Costs are based on a ten-year horizon and a real 5% interest rate. Annualization factor is equal to 7.7217 (Lee et al. 1980, appendix table 4, $n = 10$, $i = 5$). Travel costs include an average of 30 miles at $0.25/mile. Total costs depend on number of hours required for tasks and types of professionals performing tasks. Hourly wages are $23, $25, and $32 for a technician, an agronomic expert, and an attorney, respectively.

Contracting the uniform performance standard involves returning to each farm to discuss alternative pollution prevention and control plans with the farmer and to reach an agreement. The farmer must be presented with the set of practices for his/her farm and allowed to choose the preferred best management practices (BMPs), a procedure that is assumed to take two hours. An attorney can then write the contract including the practices agreed upon, and the farmer has to visit the office to sign. Four hours are allowed each year for contracting, at an annualized cost of $109.

Enforcement in this study consists of verifying that the practices in place are those agreed to by the farmer and formalized in the contract. Farm adjustments are likely to be complex and involve more than one BMP as well as a nutrient management plan; thus 3.4 hours are required annually, at an annualized cost of $86. Total annualized information, contracting, and enforcement costs are $691 per farm.

A pure performance standard would have the agency set the level of pollution allowed and the farmer select the preferred practices. Enforcement would be based on whether the stipulated performance level were met. The policy instrument described here has characteristics of a design standard because enforcement is based on whether the farmer follows agreed-upon pollution control practices. However, we refer to the instrument as a performance standard because the farmer has the choice of practices to follow as long as the simulation model predicts that the selected practices will achieve the goal of reduced nitrogen runoff.

Targeted Performance Standard

The allocation of farm compliance requirements that achieves the 40% reduction in nitrogen delivery at least cost is found using sequential optimization. Costs are minimized—first at the farm and then at the watershed level. Costs of reducing nitrogen runoff delivery by a specified amount are minimized for each farm, and shadow prices for the nitrogen delivery constraint are estimated. In this study, shadow prices for 20%, 40%, 60%, and 80% reductions in nitrogen delivery are used to derive each farm's marginal compliance cost curve. A mixed integer programming model, ALLOCATI, is then used to minimize watershed costs of achieving the desired aggregate reduction in nitrogen delivery (Carpentier 1996). Watershed costs equal the sum of compliance costs (approximated by shadow prices of nitrogen reduction on each farm) plus transaction costs.

The initial costs of visiting the farm and calibrating the crop simulation model are the same as for the uniform standard, but an additional bioeconomic model is required to simulate shadow prices. These shadow prices are subsequently used in ALLOCATI to find the allocation of requirements among farms that minimizes compliance plus

transaction costs (as identified by equations [6–9]). Farms for which the marginal increase in transaction costs plus compliance costs always exceed the incremental cost of pollution reduction in the watershed (λ) are not targeted.

Transaction costs are estimated separately for targeted and nontargeted farms. Nontargeted farms are assigned an annualized information cost of $233 per farm for seventy-seven hours required to estimate their marginal compliance costs. Nontargeted farms are assigned zero reduction because of their high estimated costs of pollution reduction and have no contracting and enforcement costs.

Eighty hours are required to gather initial information for targeted farms. More time is needed on targeted farms compared with nontargeted farms in order to specify practices that minimize compliance plus transaction costs over all targeted farms. Targeted farms require sixteen hours to update the information annually and total annualized information costs are $610.

Contracting costs may be higher for the targeted performance standard than for the uniform performance standard because (1) farmers may have preferences for farming practices other than those estimated to be cost-minimizing; (2) farmers may disagree about which practices are cost-minimizing on their farms; or (3) farmers may disagree with the level of reduction imposed on their farms. Assuming that the allocation is fixed but that farmers can offer alternative practices, the technician is assumed to have to go back to the office and repeat the analysis with the proposed alternatives. Contracting therefore requires two visits to each targeted farm to evaluate farmers' proposed alternative practices to achieve their performance standard, adding $49 to the annualized cost of the uniform allocation. Enforcement activities and costs per targeted farm are assumed to be the same as for the uniform standard. Total annualized transaction costs are $854 per farm.

Results

Baseline

In the baseline (unrestricted) case, average total gross margins are $131,934 and nitrogen delivery averages 893 pounds per farm (table 2). Farms milk an average of 136 cows and have 278 acres of harvested crops, over a third of which are alfalfa. No-till is the leading form of tillage—indicating its higher profitability in this watershed compared with reduced and conventional till.

Table 2.   Effects of Targeted and Uniform Performance Standards on Dairy Farms

Outcomes[b]	Baseline	Performance Standard[a]	
		Uniform	Targeted
Total gross margins ($)	131,934	119,689	129,158
Nitrogen delivery (lbs)	893	536	536
Leaching (lbs)	3,352	3,015	3,204
Livestock Enterprises (breeding units)			
Cows	136	135	136
Hogs	37	34	35
Poultry[c]	2,014	2,014	2,014
Other cattle	20	8	18
Manure (tons dm)			
Winter spreading	31	4	20
Spring spreading	116	100	116
Summer spreading	105	105	109
Fall spreading	42	73	47
Total manure production	293	281	291
Manure incorporated	40	120	57
New manure storage	0	14	3
Crop Enterprises (ac)			
Alfalfa	104	85	104
Corn grain	66	36	54
Corn silage	58	52	57
Wheat	50	106	64
Total harvested crops[d]	278	279	279
Grazed pasture	44	34	41
Idle pasture	64	74	66
Conventional till	21	17	21
Reduced till	58	132	75
No-till	124	66	107
Strip-cropped acres	93	156	140
Commercial nitrogen (tons)	19	12	16

[a]Standards are for a 40% reduction in nitrogen delivery.
[b]All values are averaged over 237 farms.
[c]Poultry reported as number of birds sold.
[d]Total values may differ because of rounding.

Uniform Performance Standard

Average total gross margins for 237 farms decrease by 9% from the baseline to $119,689 (table 2). Total nitrogen delivery declines to an average of 536 pounds per farm. Total leaching toward groundwater, which was not allowed to exceed the baseline on any farm, decreases slightly to 3,015 pounds per farm. The numbers of dairy cows, other cattle, and hogs decrease slightly, while poultry numbers remain unchanged. Alfalfa, corn silage, and corn grain decline, while wheat, which provides winter cover as well as revenue, more than doubles.

Strip-cropping, incorporation of manure, and shifting the timing of manure application (from winter to fall when manure nutrients can be used by wheat) are major intensive margin adjustments to achieve the 40% performance standard. The

construction of an average of 14 tons of manure storage (dry matter basis) allows winter spreading to decrease to 4 tons (from 31 tons). Manure incorporation into soil triples to an average of 120 tons. On average, acres of reduced tillage increase from 58 to 132 acres, while no-till acreage is reduced to allow the increase in manure incorporation. Over 50% of planted acres on the average farm are strip-cropped. Idle pasture land increases by ten acres. No cropland is idled.

Targeted Performance Standard

The targeted performance standard is achieved by the same management practices as the uniform standard, but the practices are limited to fewer farms, resulting in a mean total gross margin of $129,158 for the total sample of 237 farms, $9,469 more per farm (averaged over 237 farms) than under a uniform performance standard (table 2). Strip-cropped acreage, new manure storage construction, fall manure spreading, manure incorporation, and wheat acres increase on targeted farms relative to the baseline, while winter manure spreading declines.

Control Cost Comparison

While both the uniform performance and targeted performance standards reduce nitrogen delivery by 84,628 pounds, the total control costs of the targeted standard are one-fourth of control costs for the uniform standard ($770,886 versus $3,065,832). Compliance costs and transaction costs are each less for the targeted performance standard compared to the uniform performance standard (table 3). Compliance costs are $657,912 ($2,776 per farm averaged over 237 farms) for the targeted performance standard compared with $2,902,065 ($12,245 per farm) for the uniform standard). Total compliance costs are lower for targeting, because runoff control practices are required on fewer acres in the watershed when farms with large nitrogen deliveries and low marginal costs of reducing deliveries are selectively targeted for larger reductions. Only 93 of the 237 farms are targeted.

Seven farms abate 60,235 pounds (75% of their baseline on average) or 71% of the required reduction in nitrogen delivery for the whole watershed. A few farms contribute large reductions because (1) they tend to be on sites with the highest nitrogen delivery potential (steep slopes and close to water), and (2) in the baseline they tend not to use any management practices to reduce nitrogen runoff, such as manure storage and strip-cropping.

Table 3. Costs of Reducing Nitrogen Delivery

	Performance Standards	
	Uniform	Targeted
Reduction in nitrogen delivery		
Total pounds reduction (237 farms)	84,628	84,628
Pounds reduced per farm[a]	357	357
Number of farms targeted	237	93
Pounds reduced per farm targeted	357	910
Compliance costs ($)		
Total compliance cost (237 farms)	2,902,065	657,912
Per farm compliance cost[a]	12,245	2,776
Compliance cost per pound	34	8
Transaction costs ($)		
Total information cost[b] (237 farms)	117,552	90,282
Information cost targeted farms	117,552	56,730
Information cost nontargeted farms	0	33,552
Contracting and enforcement costs (237 farms)	46,215	22,692
Total transaction costs (237 farms)	163,767	112,974
Per farm transaction cost[a]	691	477
Transaction costs per pound	2	1
Control costs ($)		
Total control costs (237 farms)	3,065,832	770,886
Per farm control costs[a]	12,936	3,253
Control cost per pound	36	9

[a]Averaged over 237 farms.
[b]Information for nontargeted farms is needed to decide which farms should not be targeted. Information for targeted farms is needed to decide how much reduction to impose.

This combination of large initial delivery and few management practices in use (and thus many available alternatives to control delivery) results in very low marginal compliance costs for these farms. For example, the seven farms that reduce 75% of their baseline have mean baseline delivery of 7,587 pounds per farm and mean shadow prices of $2, $3, $13, and $27 per pound for the first 20%, 40%, 60%, and 80% reductions, respectively. Farms not targeted have mean baseline delivery of only 391 pounds per farm, and the associated shadow prices are $109, $271, $668, and $1,074 per pound for the 20%, 40%, 60%, and 80% reduction levels, respectively.

Transaction costs are $112,974 ($477 per farm averaged over 237 farms) for the targeted performance standard, compared with $163,767 ($691 per farm) for the uniform performance standard. Total information costs are lower for the targeted standard than for the uniform standard because the information on 237 farms has to be updated every year for the uniform standard, while the information on loadings and required reductions has to be updated for only 93 farms for the targeted performance standard (table 3). This result assumes that nothing occurs in subsequent years that would

cause nontargeted farms to have lower compliance costs than targeted farms. Contracting and enforcement costs are lower under targeting than under the uniform standard because there are fewer farms on which standards must be contracted and enforced (93 farms versus 237 farms).

Compliance costs are more unequally distributed under targeting. The 20 targeted farms bearing the highest compliance costs have a total of $511,700 compliance costs or 78% of the total. Under the uniform standard, the 20 farms bearing the highest compliance costs have a total cost of $1,111,313 or 38% of the total. Equity concerns of targeting could be addressed by encouraging pollution reduction trading between farms with high and low costs of reducing nitrogen runoff (Malik, Larson, and Ribaudo 1994). For example, all farms could be required to achieve an equal percentage reduction in runoff. However, farms with high costs per pound of reduced nitrogen runoff could purchase pollution permits from low-cost farms that would allow them to exceed their alloted runoff standard by a certain amount. Low-cost farms that sold permits would reduce their runoff below their alloted standard.

Implications for Research

The preceding analysis suggests that the use of spatial information with adequate technologies and institutions can reduce costs of controlling NPS pollution. Further research is needed in the following areas to better assess potential cost savings from using spatial information to target NPS pollution control.

Intrafarm Spatial Variability

The procedures described here assume that each farm has homogenous soil and topographical characteristics based on a single NRI sample site drawn from that farm. Relaxing this assumption to allow intrafarm heterogeneity in soil characteristics and field topography might change the estimated value of spatial information. If more sample sites were drawn from within each farm, it is likely that the variability among farms in terms of their average soil and topographical characteristics would decline. In the case study, farms with the lowest costs of reducing nitrogen runoff have fields with steep slopes that are located close to water. If field characteristics on these farms vary, then low-cost reductions may be achieved on only part of the farm.

To assess the effects of varying spatial attributes

within farms on the value of information, we changed eight locational and biophysical characteristics most correlated with nitrogen delivery in the watershed (Carpentier 1996). These parameters were changed so as to lessen nitrogen delivery potential on the seven farms controlling 71% of the required reduction in nitrogen delivery for the targeted standard. Our assumption was that if more sample sites had been drawn from each of these farms, the average nitrogen delivery potential of each farm would have been reduced because of the variability of soils and topography within the farm. The slope steepness (slope percent) and slope length were decreased by 50%. The clay content of the upper soil layer was decreased to its suggested lower bound (USDA/SCS n.d.), an average reduction of 43%.[5]

Parameters increased by 50% were distance along the flow path from the field to the nearest water body, the flow path slope function, and weighted cover[6] of the flow path from the farm to the nearest body of water. Finally, the soil hydrological group assigned to each of the seven farms was raised one level to increase runoff potential. Soil hydrological groups (A to D) are a classification given by the Natural Resources Conservation Service to U.S. soils to describe their infiltration[7] and runoff potential (Novotny and Chesters 1981). Soil group A indicates the lowest potential for soluble runoff and D has the highest potential. Six of the seven farms were located on soil hydrological group C and were moved to B; one farm was on soil hydrological group B and was moved to A.

As shown in table 4, the value of information was sensitive to changes in farm characteristics. In all cases, the value of information decreases from the baseline, because compliance costs increase as farms with smaller nitrogen delivery potential and higher costs of reducing nitrogen are targeted, and because transaction costs increase as more farms

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[5] In the baseline, the midpoint of the suggested range for clay content of the NRI sample point was used on each farm.

[6] The slope function measures the effect of intervening flow path slope (between field and surface water) on the tendency for sediment particles to be trapped or slowed before they reach surface water. The slope function is inversely related to flow path slope (Heatwole et al. 1987). Weighted cover is a measure of the potential for cover along the flow path to capture eroding sediments (Shanholtz and Zhang 1988). Flow path slope and cover between a farm and surface water are constructed as weighted averages of slope and cover reported by all NRI points in the NRI polygon where the farm is located. Cover types are woodland, pasture, and cropland. An NRI polygon refers to the polygon formed by the intersection of the nearest county boundaries, major land resource area boundaries, and eight-digit hydrologic unit area boundaries surrounding the NRI point.

[7] Infiltration depends on permeability of soils and subsoils, soil moisture, vegetation cover, and other parameters (Novotny and Chesters 1981).

Table 4. Effects on Value of Information of Changes in Farm Characteristics for Selected Farms[a]

Parameter Changed	Change in Parameter Value	Value of Information[b] ($)	Percentage Change in Value of Information	Sensitivity Index[c]	Number of Farms Targeted[d]
Before changes		2,294,946		———	93
Field slope	↓ by 50%	2,144,781	−6.5	0.13	133
Field slope length	↓ by 50%	2,208,040	−3.8	0.08	116
Soil upper layer clay content	↓ by 43%	2,249,846	−2.0	0.05	109
Distance to nearest water body	↑ by 50%	2,235,597	−2.6	−0.05	111
Slope function of the flow path to surface water	↑ by 50%	2,235,336	−2.6	−0.05	111
Cover of flow path to surface water	↑ by 50%	2,235,576	−2.6	−0.05	111
Hydrological group	If C→B, B→A	2,274,294	−0.9	———	93

[a]Farms selected are the seven farms that have both the largest amount of nitrogen loss reductions and the lowest cost nitrogen loss reductions.
[b]Value of information = uniform standard control costs minus targeted standard control costs.
[c]Sensitivity index = percentage change in value of information divided by percentage change in parameter.
[d]Farms targeted from a total of 237 sample farms.

are targeted. For instance, a 50% decrease in the slope steepness on these seven farms increases the number of farms targeted from 93 to 133 (43% increase) and increases all farms' average compliance costs by 19%. As a result, the value of information declines by about 6.5% for a 50% decrease in field slope on the farm. The resulting sensitivity index, computed as the percentage change in the value of information for a 1% change in the farms' characteristics, is 0.13 (6.5/50). The field slope length comes next with an 0.08% decrease in the value of information for every 1% change in the field slope length on these seven farms.

Results are also sensitive to flow path slope and cover, soil upper layer clay content, and distance to stream, each having a 0.05 sensitivity index. The soil hydrological group had the least effect on the value of information (though the sensitivity index cannot be computed). In the model, the soil hydrological group is assumed to affect soluble nitrogen runoff but not erosion. Estimated soluble nitrogen runoff is smaller than estimated sediment-adsorbed nitrogen runoff for the sample farms. Therefore, changing the soil hydrological group, which affected only soluble runoff, had less effect on nitrogen runoff than did other parameters, which affected sediment-adsorbed nitrogen runoff. The same number and the same farms are targeted with the change in the soil hydrological groups, while for all other characteristics, the number of farms targeted increases.

Efforts to target low-cost farms for pollution reduction must consider spatial variability within farms as well as across farms in order to accurately measure each farm's average cost of reducing ni-

trogen runoff. Further research is needed to identify other parameters that vary within farms and are correlated with the farms' costs of reducing pollution.

Managerial and Institutional Characteristics

The model assumes rational, profit-maximizing behavior, but farmers vary in their knowledge and objectives related to pollution-control practices. For example, some farmers overapply nutrients because they lack knowledge about plant nutrient sources and requirements (Norris and Shabman 1992; NRC 1993). Other institutional constraints, such as the need to fill marketing quotas and limited borrowing capacity, may also limit adoption of NPS pollution control. Surveys could determine farm management attitudes, information barriers, and institutional barriers to controlling NPS pollution in order to refine strategies based on spatial information. These characteristics could be used to identify potential candidates for point/nonpoint source pollution trading, for educational programs and subsidies, or for control. However, this information is expensive to collect and subject to rapid change compared with soil and topographical attributes. A better strategy may be first to identify areas with relatively low costs of reducing pollution without consideration of management or institutional characteristics and then to survey only these farms to determine if other barriers exist. These results would allow tailored education, technical assistance, and other subsidies to be targeted to farms that have the highest potential payoff in

terms of reduced costs of water quality protection (NRC 1993).

Transaction Costs

The sensitivity of the value of information to transaction costs was analyzed by increasing per-farm transaction costs by 100%. The increase in per-farm transaction costs reduces the number of farms targeted from 93 to 79. Total control costs for the targeted standard are \$876,355 (\$667,795 compliance costs plus \$208,560 transactions costs). Total control costs for the uniform standard are \$3,229,599 (\$2,902,065 compliance costs plus \$327,534 transactions costs). The value of information is \$2,353,244 (\$3,229,599 − \$876,355), an increase of \$58,298, (2.5%) relative to the baseline. The sensitivity index is +0.025 (2.5/100). The value of spatial information increases slightly because the targeted standard has fewer farms on which standards must be contracted and enforced so total transaction costs increase by a smaller amount compared with the uniform standard.

The low sensitivity index implies that farmers' compliance costs are more important than transaction costs in selecting nitrogen runoff control policies. However, litigation costs were not considered in this study. More research is needed to determine how litigation costs would be affected by policies that are targeted based on spatial information. Targeting could increase farmers' perception of unfairness in the way the control burden is spread and could therefore increase litigation. However, spatially based targeting also provides a rationale for asking some farmers to bear more of the burden because they have lower costs of compliance or because they contribute more pollution. This rationale may make a targeted policy more defensible in court than in uniform application of the policy. Targeting also reduces litigation potential by reducing the number of people affected by pollution control programs.

Transaction and compliance costs could be reduced through participatory approaches, which involve farmers in decisions about how a policy is implemented. Farmers have specialized knowledge about their operations, which can be used to estimate the site-specific costs and effectiveness of best management practices. Water quality agency personnel can provide information to farmers about the site-specific impacts of agricultural practices on pollution potential. The information exchange between farmers and the agency may result in strategies that are tailored to the unique characteristics of each farm in order to achieve water quality goals at lower cost. By lowering farmers'

costs and giving farmers a voice in implementation of strategies, participatory approaches may make farmers more likely to implement pollution control strategies, thereby lowering enforcement costs.

Consumer Impacts

Spatial targeting reduces the number of producers impacted by water· quality programs. Targeting could limit increases in food and fiber costs resulting from environmental policies and save consumers money compared with an untargeted NPS control policy. Such potential savings should be considered in determining future expenditures on spatial information.

Changing Economic Conditions

Changes in commodity prices, input costs, agricultural policies, land ownership patterns, zoning laws, and other economic conditions may affect the optimal spatial distribution of NPS pollution control. Well-designed spatial decision support systems (Negahban et al. 1994; Covington et al. 1988) allow economic parameters to be varied and merged with spatial data to analyze public policies. For example, in the case of pollution-reduction trading between point and/or nonpoint pollution sources, prices of pollution permits would have to be monitored to ensure that no single permit buyer or seller in the watershed dominated the market and kept prices artificially high or low.

NPS Pollution Processes

NPS pollution policies centered on spatial information rely not only on data such as field slope, cover crops, and distance to streams, but also on simulation models that can link changes in farm practices or farming systems to ultimate changes in environmental outcomes. The information base is growing but currently has many gaps. Furthermore, research is needed that links pollution performance standards to desired outcomes with respect to human, animal, and ecosystem health. Such information, if validated, should reduce the potential litigation of targeted NPS pollution control strategies as well as improve their cost effectiveness.

Conclusions and Implications

Achieving water quality protection objectives in the next century will be challenged by increasing public desire for clean water, decreasing govern-

ment budgets, and public ambivalence about command and control approaches. New technologies are increasing the supply of spatial information that can be used to lower the costs of nonpoint source pollution prevention and control policies. The demand for spatial information for NPS pollution control will depend upon the design of NPS pollution policies. This study estimated the value of spatial information in reducing NPS control costs, which include both compliance and transaction costs, for a case study of dairy farms in the Lower Susquehanna watershed.

Costs of a uniform and a targeted performance standard designed to achieve a 40% reduction in nitrogen runoff were compared. The types of practices used to achieve the reduction were similar in the uniform and targeted standards except that they were carried out on fewer farms in the targeted standard. Farms reduce runoff by strip-cropping, manure storage, manure incorporation, and shifting timing of manure application. Targeting based on spatial information reduced compliance costs by nearly 80% compared with the uniform standard. Transaction costs were also reduced by targeting because fewer farms required contracting and enforcement of the performance standard. Total control costs were 75% lower with targeting.

The basic elements of a flexible, cost-effective agroenvironmental policy appear to be *clear* environmental quality objectives (e.g., performance-based standards); *targeted* policy resources to priority concerns, areas, and farms; *flexible incentives* (positive or negative) for farmers to achieve objectives; and *tailored* assistance and information to those farmers who must change their farming practices and/or farming systems to obtain environmental quality objectives (Batie and Arcenas 1997). The information and research needs of such an ideal targeted NPS pollution control policy are impressive. Clearly, both a research agenda and policy implications are found in the limited use of performance standards, missing knowledge about linkages of pollution source and impact, and lack of knowledge about adoption behavior (Batie and Ervin 1997). There is, however, considerable evidence addressing needed changes in farm practices to reduce runoff and leaching rates from an individual farm. This knowledge can help determine, in an approximate manner, what farms and what regions have the potential to be the more important sources of pollution (NRC 1993; U.S. Congress/OTA 1995; USDA/ERS 1994).

Research similar to this case study can assist in identifying which parameters are best correlated with improved environmental outcomes at low cost. Spatial information can help determine which

farms should be targeted to receive tailored assistance and/or regulatory attention. While regulatory authorities may justifiably be concerned about information costs and gaps, in the future information costs are likely to decline as spatial information technology improves and is more widely adopted. Market mechanisms such as nonpoint/nonpoint or point/nonpoint source trading could reduce some of the public administrative costs of achieving water quality objectives. Also, the pursuit of a targeted NPS pollution control policy based on spatial information will induce more research attention on informational gaps.

Further research is needed on the use of spatial information to reduce NPS pollution control costs. However, there is enough preliminary evidence to suggest that targeting NPS pollution control policies using spatial information has the potential for considerable cost-savings in achieving water quality goals.

References

Batie, S.S., and A. Arcenas. 1997. "Farmer-Led Environmental Management: Applying Lessons from Business-Led Environmental Management." Unpublished manuscript. Department of Agricultural Economics, Michigan State University, East Lansing, Michigan.

Batie, S.S., and D.E. Ervin. 1997. "Flexible Incentives for Environmental Management: A Typology." Paper presented at conference, Flexible Incentives for Adoption of Environmental Technologies in Agriculture, June 8–9, Gainesville, Fla.

Bohm, P., and C.S. Russell. 1985. "Comparative Analysis of Alternative Policy Instruments." In *Handbook of Natural Resource and Energy Economics*, ed. A.V. Kneese and J.L. Sweeney. New York: North-Holland.

Bosch, D.J., C.L. Carpentier, and R. Heimlich. 1995. "A Farm Model for Evaluating Nonpoint Source Pollution Abatement Programs." SP-94-03, Department of Agricultural and Applied Economics. Blacksburg: Virginia Polytechnic Institute and State University. February.

Brooke, A., D. Kendrick, and A. Meeraus. 1992. *GAMS: A User's Guide, Release 2.25.* Washington, D.C.: World Bank.

Camacho, R. 1992. "Chesapeake Bay Program Nutrient Reduction Strategy Reevaluation Report No. 8: Financial Cost Effectiveness of Point and Nonpoint Source Nutrient Reduction Technologies in the Chesapeake Bay Basin." ICPRB Report no. 94-4. Washington D.C.: Interstate Commission on the Potomac River Basin. December.

Carpentier, C.L. 1996. "Value of Information for Targeting Agro-pollution Control: A Case Study of the Lower Susquehanna Watershed." Ph.D. diss. Department of Agricultural and Applied Economics. Blacksburg: Virginia Polytechnic Institute and State University.

Chesapeake Bay Program. 1994. *Trends in Phosphorus, Nitrogen, Secchi Depth, and Dissolved Oxygen in Chesapeake*

Bay, 1984 to 1992. CBP/TRS 115/94. Washington, D.C.: U.S. Environmental Protection Agency. August.

———. 1996. *State of the Chesapeake Bay, 1995*. Annapolis, Md.: Chesapeake Bay Program Office.

Covington, W.W., D.B. Wood, D.L. Young, D.P. Dykstra, and L.D. Garrett. 1988. "TEAMS: A Decision Support System for Multiresource Management." *Journal of Forestry* 86: 25–33.

DeCoursey, D.G. 1985. "Mathematical Models for Nonpoint Water Pollution Control" *Journal of Soil and Water Conservation* 40:508–13.

Ford, S.A. 1992. "1991 Pennsylvania Dairy Farm Business Analysis." Extension Circular. University Park: Pennsylvania State University, Cooperative Extension.

Heatwole, C.D., V.O. Shanholtz, E.R. Yagow, and E.R. Collins, Jr. 1987. "Targeting Animal Waste Pollution with Virginia GIS." Paper no. 87-4049. Saint Joseph, Mich.: American Society of Agricultural Engineers.

Krier, J.E., and W.D. Montgomery. 1973. "Resource Allocation, Information Cost and the Form of Government Interventions." *Natural Resources Journal* 13:89–105.

Lee, W.F., M.D. Boehlje, A.G. Nelson, and W.G. Murry. 1980. *Agricultural Finance*. Ames: Iowa State University Press.

Malik, A.S., B.A. Larson, and M. Ribaudo. 1994. "Economic Incentives for Agricultural Nonpoint Source Pollution Control." *Water Resources Bulletin* 30:471–79.

National Research Council (NRC), Board on Agriculture. 1993. *Soil and Water Quality: An Agenda for Agriculture*. Washington, D.C.: National Academy Press.

Negahban, B., C.B. Moss, J.W. Jones, J. Zhang, W.D. Boggess, and K.L. Campbell. 1994. "Optimal Field Management for Regional Water Quality Planning." ASAE Paper no. 94-3553. Presented at the International Winter Meeting sponsored by the American Society of Agricultural Engineers, St. Joseph, Mich.

Norris, P.E., and L.A. Shabman. 1992. "Economic and Environmental Considerations for Nitrogen Management in the Mid-Atlantic Coastal Plain." *American Journal of Alternative Agriculture* 7:148–56.

Novotny, V., and G. Chesters. 1981. *Handbook of Nonpoint Source Pollution Sources and Management*. New York: Van Nostrand Reinhold Environmental Engineering Series.

Pennsylvania Cooperative Extension Service. 1992. "Pennsylvania State Farm Enterprise Budgets." Department of Agricultural Economics and Rural Sociology. University Park: Pennsylvania State University. April.

Rader, D.N. 1994. "Nutrient Trading as a Management Option: The Tar-Pamlico Experiment." In *Economic Issues Associated with Nutrient Management Policy: Proceedings of a Regional Workshop*, ed. P.E. Norris and L.E. Danielson. Sponsored by Southern Regional Information Exchange Group-10. SRIEG-10 no. 32; SRDC Publication no. 180. Mississippi State: Mississippi State University. March.

Ritter, W.F. 1990. *Manual for Economic and Pollution Evalu-ation of Livestock Manure Management Systems*. Harrisburg: Pennsylvania Department of Environmental Resources, Bureau of Soil and Water. March.

Serotkin, N., (ed.) 1993. *Penn State Agronomy Guide*. University Park: Penn State Cooperative Extension Service.

Shanholtz, V.O., and N. Zhang. 1988. *Supplementary Information of Virginia Geographic Information System*. Special Report, VirGIS 88-21. Blacksburg: VirGIS Laboratory, Department of Agricultural Engineering, Virginia Polytechnic Institute and State University.

Thomann, R.V., J.R. Collier, A. Butt, E. Casman, and L.C. Linker. 1994. *Response of the Chesapeake Bay Water Quality Model to Loading Scenarios*. CBP/TRS 101/94. Modeling Subcommittee of the Chesapeake Bay Program. Washington, D.C.: U.S. Environmental Protection Agency. April.

Tietenberg, T.H. 1998. *Environmental Economics and Policy*. 2d ed. New York: Addison-Wesley Educational Publishers.

U.S. Congress, Office of Technology Assessment (OTA). 1995. *Targeting Environmental Priorities in Agriculture: Reforming Program Strategies*. OTA-ENV-640. Washington, D.C.: U.S. Government Printing Office. September.

U.S. Department of Agriculture (USDA); Economic Research Service (ERS). 1994. *Agricultural Resources and Environmental Indicators*. Agricultural Handbook 705. Washington, D.C.: U.S. Government Printing Office. December.

———. 1996. "Agricultural Outlook Supplement." Washington, D.C. April.

U.S. Department of Agriculture (USDA), Soil Conservation Service (SCS). 1986. *Urban Hydrology for Small Watersheds*. Washington, D.C.

———. 1991. *Pennsylvania Technical Guide*. Harrisburg Penna.

———. n.d. National Resources Inventory Database. Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 1992. *Incentive Analysis for Clean Water Act Reauthorization: Point Source/Nonpoint Source Trading for Nutrient Discharge Reductions*. Washington, D.C.: Office of Water and Policy, Planning, and Evaluation.

Virginia Chesapeake Bay Program. 1993. *Virginia's Tributary Strategies*. Richmond: Virginia Department of Environmental Quality. April.

Williams, J.R., C.A. Jones, J.R. Kiniry, and D.A. Spanel. 1989. "The EPIC Crop Growth Model." *Transactions of the ASAE* 32:497–511.

Wischmeier, W.H., and D.D. Smith. 1978. *Predicting Rainfall Losses—A Guide to Conservation Planning*. Agriculture Handbook no. 537. Washington, D.C.: U.S. Department of Agriculture.

Yagow, E.R., V.O. Shanholtz, J.W. Kleene, S. Mostaghimi, and J.M. Flagg. 1990. "Annual Estimation of Nitrogen in Agricultural Runoff." ASAE Paper no. 90-2054. St. Joseph, Michigan: American Society of Agriculture Engineers.