The Coordination and Design of Point-Nonpoint Trading Programs and Agri-Environmental Policies

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Agricultural agencies have long offered agri-environmental payments that are inadequate to achieve water quality goals, and many state water quality agencies are considering point-nonpoint trading to achieve the needed pollution reductions. This analysis considers both targeted and nontargeted agri-environmental payment schemes, along with a trading program which is not spatially targeted. The degree of improved performance among these policies is found to depend on whether the programs are coordinated or not, whether double-dipping (i.e., when farmers are paid twice—once by each program—to undertake particular pollution control actions) is allowed, and whether the agri-environmental payments are targeted. Under coordination, efficiency gains only occur with double-dipping, so that both programs jointly influence farmers' marginal decisions. Without coordination, double-dipping may increase or decrease efficiency, depending on how the agri-environmental policy is targeted. Finally, double-dipping may not solely benefit farmers, but can result in a transfer of agricultural subsidies to point sources.

Key Words: environmental subsidies, environmental targeting, nonpoint source pollution, policy coordination, tradeable pollution permits, water quality

Reducing water pollution from agricultural production has become a leading priority of federal and state agencies charged with water quality protection [U.S. Department of Agriculture (USDA) and U.S. Environmental Protection Agency (USEPA), 1998]. Water quality assessments show agriculture to be a major, but largely unregulated cause of water pollution problems. It has become clear that achieving water quality goals in many regions of the nation will require significant reductions in pollution loads from agricultural sources (USEPA, 2000). Moreover, reliance on point source pollution controls to achieve water quality improvements in regions where agriculture and other nonpoint sources are significant contributors to water quality impairments increases the costs of environmental protection by precluding efficient allocation of control between point and nonpoint sources (Freeman, 1990).

Among the most important EPA initiatives to address agriculture's contributions to water quality problems now is the Total Maximum Daily Load (TMDL) program, which requires states to develop and implement watershed-based plans for surface waters that do not meet water quality standards. Essentially, the TMDL program requires states to identify the total pollution load reductions needed to achieve water quality standards in impaired waters, and to allocate the reductions among point and nonpoint sources (USEPA, 2004). Achieving standards will require agricultural nonpoint source pollution control in watersheds where agriculture is a significant cause of water quality impairments.

An important feature of EPA's TMDL program is that it provides states with substantial freedom for selecting policy instruments. Pollution trading is emerging as an appealing option. The growing

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dissatisfaction with traditional command-and-control approaches has led to substantial interest by state authorities and the EPA in the use of incentive-based mechanisms, with particular interest in pollution trading between point and nonpoint sources, as a means to strengthen nonpoint pollution controls and to enhance the coordination of point and nonpoint source controls (Elmore, Jaksch, and Downing, 1985; USEPA, 2001; USDA and USEPA, 1998; Great Lakes Trading Network, 2000; Faeth, 2000). Several fully implemented and pilot trading programs have been in operation for some time (Horan, 2001), and in January 2003, the USEPA announced rules for a national program, with funding for 11 pilot programs throughout the nation (USEPA, 2003).

In "textbook" form, point-nonpoint trading works as follows. Each pollution source is allocated pollution permits defining the permit holder's allowable emissions (for point sources) or estimated emissions (for nonpoint sources). Through trades, polluters can adjust their allowances by buying permits from or selling permits to other permit holders subject to rules governing trades (i.e., a trading ratio which defines how many nonpoint source permits trade for one point source permit). The gains from trade correspond to reductions in overall pollution control costs, as trading encourages a least-cost allocation of load reductions among sources.

It should be noted that U.S. trading programs have evolved with enforceable regulations applicable to point sources, while agriculture's participation is made voluntary by essentially giving agricultural sources a presumptive right to pollute at historical levels (Hoag and Hughes-Popp, 1997). This allocation of rights should not affect the ability to attain particular environmental outcomes efficiently, provided the total number of permits and the trading rules are defined appropriately. For instance, trading may not improve the environmental outcome if point source permit numbers remain fixed when trading is implemented, but the outcome would improve if the trading program was implemented along with a reduction in point source permits. In that case, additional controls on both sources could emerge. Making trading voluntary for agricultural sources does affect the distribution of costs and benefits. Farmers who participate would be paid to do so, and thus trading represents a source of potential income to them. Point sources are the sole purchasers of permits under such a setup, but their costs are still less than what they would incur if they undertook all abatement responsibilities.

Theoretical and numerical analysis of point-nonpoint trading has focused largely on the design of the optimal trading ratio between point and nonpoint sources (e.g., Shortle, 1987; Malik, Letson, and Crutchfield, 1993; Horan, 2001). Without exception, studies of trading examine program design and the gains from trade independent of the presence and design of other policies that have an impact on nonpoint pollution loads—trading is examined as a stand-alone approach to achieving pollution reductions. This assumption is limiting in practice. Given the various federal and state agri-environmental payment programs offered to subsidize soil and water conservation, and water quality protection practices, trading program development occurs in a context in which there are existing, although not notably effective, policies (Ribaudo, 2001). Farmers' incentives to participate in point-nonpoint trading programs, and the effects of such programs on resource allocation, will be affected by simultaneous opportunities to participate in agri-environmental payment programs. And the converse is true.

Accordingly, questions emerge about the benefits of coordinating the two types of programs, and also the design of policy parameters—whether or not coordination occurs. Presumably the environment and society are better off with coordination, but is agriculture better off? Also, how is the allocation of controls across watersheds affected by coordination?

To address these questions, the design of policy parameters must be considered. We focus significantly on two particular policy choices. The first is whether or not agricultural sources can collect payments from two programs for the same environmentally beneficial activity, referred to here as "double-dipping." This choice is of particular relevance because farmer participation has been voluntary for both USDA programs and point-nonpoint trading programs. If double-dipping is allowed, then each program cross-subsidizes the other, and farmers' marginal abatement costs are reduced relative to what would emerge under either program individually. If double-dipping is not allowed, then one program essentially taxes the ability of the other to produce environmental gains. For instance, a farmer might first enroll in the USDA program where some level of abatement effort is subsidized. The farmer can then participate in the trading program, but only for *additional* units of abatement effort. These additional units of effort will be more costly to purchase than the *initial* units if abatement costs are increasing in abatement effort. The ability to double-dip could therefore affect participation, and hence program performance and distributional outcomes.

The second policy choice examined in this analysis is the trading ratio. This choice will be affected by whether coordination occurs and whether double-dipping is allowed.

We begin with a conceptual model of agri-environmental payments and point-nonpoint trading. The conceptual model allows us to derive some interesting results about the merits of double-dipping and policy coordination, and implications for the design of trading ratios. This model does not indicate the potential magnitudes of gains from double-dipping or coordination, or the optimal trading ratio, the sensitivity of the trading ratio to the design of the agri-environmental payments, or nuances of the distributional outcomes. These are empirical issues. Insight about these issues can be gained only through numerical analysis. Results on these issues are presented using a simulation model of nitrogen pollution control in the Susquehanna River Basin (SRB), an important source of nutrient loads for the Chesapeake Bay. More will be said about the SRB and related nitrogen pollution problems below.

A Model of Point Source and Agricultural Nonpoint Source Pollution

Building on the model of Shortle, Horan, and Abler (1998), assume a particular resource (e.g., an estuary) is damaged by a single residual (e.g., nitrogen). Economic damages, D, are an increasing function of the ambient concentration of the residual, a, i.e., $D(a)$ with $D' > 0$. Ambient pollution depends on loadings from nonpoint sources, r_i $(i = 1, ..., n)$; point source discharges, e_k $(k = 1, ..., s)$; natural generation of the pollutant, ξ; and stochastic environmental variables that influence transport and fate, $\boldsymbol{\delta}$:

$$a = a(r_1, ..., r_n; e_1, ..., e_s; \xi, \boldsymbol{\delta}),$$
$$\partial a/\partial r_i, \partial a/\partial e_i \geq 0 \quad \forall i.$$

For heuristic purposes, denote point source (PS) polluters as firms that generate emissions (although many relevant point sources are not firms, but rather municipal wastewater treatment facilities), and denote nonpoint source (NPS) polluters as farms that generate pollution loadings. The term "farm" is used for heuristic purposes and because agriculture is the primary target of current U.S. point-nonpoint trading markets (Hoag and Hughes-Popp, 1997) due to agriculture's role as the leading contributor of polluted runoff.

Emissions are observable and nonstochastic.[1] Loadings cannot be observed directly (at an acceptable cost) because they are diffuse and, via stochastic variations in environmental drivers (e.g., weather), they are also stochastic. The relation for site i is $r_i = r_i(\mathbf{x}_i, \mathbf{v}_i)$, where $\mathbf{x}_i$ is an $\{m \times 1\}$ vector of variable inputs (with jth element x_{ij}), and $\mathbf{v}_i$ represents site-specific, stochastic environmental variables. Accordingly, farms can only influence the distribution of their loadings through their input choices. These choices would include both standard production decisions (e.g., land allocated to particular crops, fertilizer and pesticide use, tillage practices) and also practices undertaken specifically to control pollution (e.g., buffer strips).

The costs of reducing emissions have been treated extensively in the literature (see Baumol and Oates, 1988; Hanley, Shogren, and White, 1997). Conventionally, costs are treated as an increasing function of the level of emissions reduction, or abatement. Following this convention, we define the kth firm's expected pollution control costs to be a function of abatement, denoted $e_{k0} - e_k$, where e_{k0} is some base level of emissions. Given that e_{k0} is a constant, notation is simplified by defining abatement costs as a function of emissions $c_{ek}(e_k)$, $(c'_{ek} < 0, c''_{ek} \geq 0)$. A deterministic abatement cost function of this type is inappropriate for farms due to the diffuse and stochastic features of loadings (Shortle and Dunn, 1986; Shortle, 1990). Instead, control costs are defined as a function of the input choices farmers make to influence the distribution of their loadings, i.e., $c_{ri}(\mathbf{x}_i)$.[2]

An ex ante efficient allocation of pollution control efforts minimizes the sum of private control costs and expected damage costs:[3]

[1] Point emissions are in fact often measured with error and subject to stochastic influences. Nevertheless, this treatment is standard and helps to contrast the typical theoretical treatment of point sources with nonpoint sources.

[2] This cost of pollution control activities is by definition the reduction in economic returns the farmer would incur relative to the case where the farmer is maximizing economic returns without pollution controls (Freeman, 1993), i.e., $c_{ri}(\mathbf{x}_i) = \pi^u_{ri} - \pi_{ri}(\mathbf{x}_i)$, where π^u_{ri} represents maximized economic returns in the unregulated environment, and $\pi_{ri}(\mathbf{x}_i)$ denotes the economic returns to the ith farm, restricted on the vector of input choices, $\mathbf{x}$.

[3] With ex ante efficiency, pollution allocations and policies to achieve these allocations are deemed efficient in expectation, *before* the realization of stochastic variables. In contrast, ex post efficiency would require allocations to be efficient *after* the realization of stochastic variables. Ex ante efficient allocations are not likely to be ex post efficient, but ex ante efficiency is the correct criterion under expected utility theory when decisions must be made prior to stochastic events.

$$\underset{e_k, x_{ij}}{\text{Min}}\ TSC = \sum_{i=1}^{n} c_{ri}(\mathbf{x}_i) + \sum_{k=1}^{s} c_{ek}(e_k) + E\{D(a)\}.$$

The necessary conditions for an efficient outcome reduce to

$$(1)\qquad -\frac{\partial c_{ri}}{\partial x_{ij}} = E\left\{D'(a)\frac{\partial a}{\partial r_i}\frac{\partial r_i}{\partial x_{ij}}\right\}\ \forall\, i, j$$

and

$$(2)\qquad -\frac{\partial c_{ek}}{\partial e_k} = E\left\{D'(a)\frac{\partial a}{\partial e_k}\right\}\ \forall\, k.$$

Condition (1) states that the marginal private benefits from using polluting inputs (i.e., the marginal reduction in control costs) should equal the associated marginal expected damages, and the marginal private costs of using abating inputs should equal the associated expected marginal external benefits (i.e., the expected marginal reduction in damages). Condition (2) states that the marginal private benefits of emissions (i.e., the marginal reduction in control costs) should equal the marginal expected damages of emissions [see Shortle and Abler (1997) for a more detailed description of these well-known conditions]. We now turn to the policy mechanisms used by the two agencies.

Agricultural Conservation Policies

Agricultural conservation policies are modeled as input-based subsidies which are utilized to reduce loadings.[4] Basing these subsidies on inputs is consistent with existing agri-environmental programs, such as the Environmental Quality Incentives Program (EQIP) and Conservation Security Program (CSP), in which payments are based on nutrient management or other production and/or land use practices (Claassen et al., 2001).[5]

Input subsidies are defined as:

$$\max\left\{s_{ij}(x_{ij}^U - x_{ij}), 0\right\},$$

where s_{ij} is the per unit subsidy rate applied to the use of input j by farm i, and x_{ij}^U is the farmer's cost-minimizing (baseline) level of input j in the unregulated outcome [i.e., consistent with setting $(\partial c_{ri})/(\partial x_{ij}) = 0, \forall\, i, j$]. If $s_{ij} > 0$, then the subsidy is to reduce the use of polluting inputs. If $s_{ij} < 0$, then the subsidy is to increase the use of abating inputs. Ex ante efficient rates would be farm-specific and set equal to the right-hand side of equation (1), evaluated at the optimum. While we allow for the subsidy rate to be farm-specific, in practice rates may be applied uniformly at least on a regional basis to reduce transactions costs.

A Point-Nonpoint Trading Market

Point-nonpoint trading is fundamentally different from "textbook" tradeable discharge permit markets involving only point sources (Shortle, 1987; Malik, Letson, and Crutchfield, 1993). Two fundamental design issues or questions arise when nonpoint sources are added to the mix. One question is: What is the appropriate legal entitlement, or property right, that would be transferred through trades between point and nonpoint sources? In conventional permit markets, such as those for SO_2 permits, pollution permits define allowable emissions for the permit holder, and firms can adjust their allowances by buying from or selling to other permit holders, subject to rules governing trades. However, actual loadings cannot be directly traded because they cannot be accurately monitored at reasonable cost and they have a significant random component (Letson, 1992; Malik, Letson, and Crutchfield, 1993; Shortle, 1987). Accordingly, an alternative basis for nonpoint trades is required. The option we consider entails trading changes in emissions for changes in estimated or expected loadings. In this case, data on agricultural land uses, and geophysical and climatic factors are input into models (e.g., SWAT or AGNPS) which estimate loads. Existing point-nonpoint trading programs are of this emissions-for-expected loadings type (Hoag and Hughes-Popp, 1997; Malik, Letson, and Crutchfield, 1993; Shortle and Abler, 1997).[6]

[4] Agricultural conservation policies come in a variety of forms to address the various linkages between agriculture and the environment, such as nonpoint pollution, soil erosion, and the provision of wildlife habitat and rural amenities. For the present study, we ignore the multifunctionalilty of agriculture and instead focus on policies designed to confront a single problem—nonpoint pollution.

[5] As pointed out by an anonymous reviewer, 60% of EQIP dollars go to livestock operations, which are considered point sources for National Pollutant Discharge Elimination System purposes. For simplicity, we are ignoring livestock in the current model and focusing instead on industrial point sources that do not receive agricultural subsidies. The Tar-Pamlico trading program provides one precedent of a point-nonpoint trading program where there is a differentiation between industrial point sources and livestock point sources, i.e., a trading ratio of 3:1 between cropland and industrial point sources and a ratio of 2:1 between livestock and industrial point sources (Hoag and Hughes-Popp, 1997).

[6] Plausible alternative bases include management practices (e.g., BMPs) that influence nonpoint loads. See Horan, Shortle, and Abler (2002) and Horan et al. (2002) for analyses of trades involving changes in input use.

A second question arising when nonpoint sources are included is: At what rate should trades occur? Emissions and expected loads are imperfect substitutes. Consequently, water quality goals may not be achieved or maintained if emissions and expected loads are traded one-for-one. A potential solution is to require that trades occur according to a "trading ratio" which defines the required reduction in expected loads for a one-unit increase in emissions (e.g., Malik, Letson, and Crutchfield, 1993; Shortle, 1987). Trading ratios are used in practice (see Horan, 2001, table 1), although there is no evidence to suggest the chosen ratios promote efficiency.

We now consider an emissions-for-expected loadings trading system, given that conservation programs are already in place, and making the common assumption that the permit market is perfectly competitive (e.g., Montgomery, 1972; Hanley, Shogren, and White, 1997). The discussion begins with some details of the market and an examination of how firm/farm behavior will occur in a perfectly competitive trading equilibrium. This behavior is then used to examine the optimal choice of policy parameters for this market.

There are two categories of permits: emissions permits ($\hat{e}$) and loadings permits ($\hat{r}$). Firms must have a combination of both types at least equal to their emissions, and farms must have a combination at least equal to their expected loadings. We assume 1:1 trading of permits within source categories, with trading ratios applicable for trades between categories. The cross-category trading ratio is denoted t—i.e., $t = |d\hat{r}/d\hat{e}|$.

The restriction of 1:1 trading within categories is analogous to many existing trading systems and allows us to focus on trading between rather than within source categories. However, it also implies certain inefficiencies. Specifically, 1:1 trading within a source category means emissions (expected loadings) by a firm (farm) close to the estuary are considered perfect substitutes for emissions (expected loadings) by a firm (farm) located very far away from the estuary, even though a smaller proportion of pollutants generated farther away actually reaches the estuary. Efficiency is reduced if pollutants are treated as perfect substitutes when they have differential marginal environmental impacts (McGartland and Oates, 1985; Montgomery, 1972; Tietenberg, 1995).[7]

[7] A trading system could be developed that allows firms to exploit differences in relative environmental impacts, but it would be highly complex and would involve a different trading ratio for each potential trade (Shortle and Abler, 1997).

Denote the market price of expected loadings permits as p_r, and the price of emissions permits as p_e. Firm k will choose emissions levels, emissions permit holdings ($\hat{e}_k$), and expected loadings permit holdings ($\hat{r}_k$) to minimize costs,

$$J_k = c_{ek}(e_k) + p_e[\hat{e}_k - \hat{e}_{k0}] + p_r[\hat{r}_k - \hat{r}_{k0}],$$

subject to the constraint that its total emissions are less than its permit holdings, $e_k \le \hat{e}_k + (1/t)\hat{r}_k$, where $\hat{e}_{k0}$ and $\hat{r}_{k0}$ are initial emissions and expected loadings permits held by firm k, respectively, and $(1/t)\hat{r}_k$ represents the emissions it is permitted to generate based on its holdings of expected loadings permits.

Assuming the emissions constraint is satisfied as an equality and, without loss of generality, assuming $\hat{r}_{k0} = 0$, $\hat{e}_k$ can be eliminated as a choice variable. The resulting first-order conditions are:

$$\text{(3)} \qquad \frac{\partial J_k}{\partial e_k} = \frac{\partial c_{ek}}{\partial e_k} + p_e = 0$$

and

$$\text{(4)} \qquad \frac{\partial J_k}{\partial \hat{r}_k} = -(1/t)p_e + p_r \gtreqless 0.$$

Given indifference between emissions and expected loadings permits at the margin in a competitive market equilibrium, (3) is satisfied as an equality, implying $t = p_e/p_r$. Using this relation and substituting the permit constraint into the objective function, we have $J_k = c_{ek}(e_k) + p_e[e_k - \hat{e}_{k0}]$.

Similarly, farms will choose inputs, expected loadings permit holdings ($\hat{r}_i$), and emissions permit holdings ($\hat{e}_i$) to minimize costs. Assuming, as in existing trading programs, that farms do not initially hold emissions permits, and following the methods used above for firms, it is easy to show farm i's net costs can be written as:

$$V_i = c_{ri}(\mathbf{x}_i) - \sum_{i=1}^{m} s_{ij}[x_{ij}^U - x_{ij}] - p_r[\tilde{r}_{i0} - E\{r_i(\mathbf{x}_i)\}],$$

where $\tilde{r}_{i0}$ represents the number of permits the farmer is allowed to sell to firms. The number of permits the farmer is allowed to sell may differ from the farm's initial endowment of permits ($\hat{r}_{i0}$), depending on whether double-dipping is allowed—that is, if the farmer can receive payments from both the conservation and trading programs to undertake the same activities. Note, the farmer's net costs will be negative if the farmer voluntarily participates in either of the programs.

If double-dipping is allowed, then the number of permits available for sale equals the farm's endowment, $\tilde{r}_{i0} = \hat{r}_{i0}$. The number of permits the farmer has available for sale is essentially exogenous in this case. We follow the convention of existing programs and define farm i's initial endowment of permits as the expected loadings the farm would produce in the absence of the trading program (i.e., the unregulated equilibrium), $\hat{r}_{i0} = E\{r_i(\mathbf{x}_i^u)\}$.

In existing and planned point-nonpoint trading programs which include agricultural sources, farms are not required to have permits. Instead, farms have an implicit, initial right to pollute, which is consistent with having permits equal to unregulated expected loadings levels. Trading occurs as farms contract with firms to reduce expected loadings in exchange for a fee.[8] The ith farmer's minimization problem is defined by V_i after substituting in the relation $\tilde{r}_{i0} = \hat{r}_{i0}$. The resulting first-order conditions are:

$$(5)\quad \frac{\partial V_i}{\partial x_{ij}} = \frac{\partial c_{ri}}{\partial x_{ij}} + s_{ij} + p_r E\left\{\frac{\partial r_i}{\partial x_{ij}}\right\} = 0 \quad \forall\, i, j.$$

If double-dipping is not allowed, then a farmer cannot sell permits for reductions for which he/she chooses to accept payment under the conservation program, and vice versa. Assuming conservation program choices are made first, a farmer has a choice in allocating his/her initial permit endowment between conservation payments and permit sales: the number of permits available for sale to point sources is endogenous and depends on the degree to which farmers are willing to accept conservation payments.

Define $x_{ij}^U - x_{ij}^c \geq 0$ to be the reduction in input use funded by the conservation program, with the corresponding expected reduction of nonpoint loads being $\hat{r}_{i0} - E\{r_i(\mathbf{x}_i^c)\}$. In this case, the farmer is only allowed to sell $\tilde{r}_{i0} = E\{r_i(\mathbf{x}_i^c)\}$ permits to point sources under the trading program. Farm i's net costs are rewritten as:

$$V_i = c_{ri}(\mathbf{x}_i^c - \tilde{\mathbf{x}}_i) - \sum_{i=1}^{m} s_{ij}\left[x_{ij}^U - x_{ij}^c\right] - p_r\left[E\{r_i(\mathbf{x}_i^c)\} - E\{r_i(\mathbf{x}_i^c - \tilde{\mathbf{x}}_i)\}\right],$$

where $\tilde{x}_{ij} = x_{ij}^c - x_{ij} \geq 0$ defines the additional reduction due to trading. Under this specification, the farmer chooses both $\mathbf{x}_i^c$ and $\tilde{\mathbf{x}}_i$ to minimize costs.

The first-order conditions (after substituting $x_{ij} = x_{ij}^c - \tilde{x}_{ij}$) are:

$$(6)\quad \frac{\partial V_i}{\partial x_{ij}^c} = \frac{\partial c_{ri}}{\partial x_{ij}} + s_{ij} - p_r\left[E\left\{\frac{\partial r_i(\mathbf{x}_i^c)}{\partial x_{ij}^c}\right\} - E\left\{\frac{\partial r_i(\mathbf{x}_i)}{\partial x_{ij}}\right\}\right] = 0 \quad \forall\, i, j$$

and

$$(7)\quad \frac{\partial V_i}{\partial \tilde{x}_{ij}} = -\frac{\partial c_{ri}}{\partial x_{ij}} - p_r E\left\{\frac{\partial r_i(\mathbf{x}_i)}{\partial x_{ij}}\right\} = 0 \quad \forall\, i, j.$$

Condition (7) is the standard condition for the type of trading programs commonly analyzed in the absence of conservation programs (Horan et al., 2001). Using (7), condition (6) reduces to:

$$(8)\quad s_{ij} = p_r E\left\{\frac{\partial r_i(\mathbf{x}_i^c)}{\partial x_{ij}^c}\right\} \quad \forall\, i, j.$$

Condition (8) is an arbitrage condition that states the marginal value of reducing x_{ij} under the conservation program should equal the marginal reduction in revenues from permit sales due to a reduction in the farm's initial permit allocation.

We could also consider the case where the farmer participates in the trading program first, a strategy which would be followed if this yields greater net benefits. The net cost function in this case is:

$$V_i = c_{ri}(\mathbf{x}_i) - \sum_{i=1}^{m} s_{ij}\left[x_{ij}^T - x_{ij}\right] - p_r\left[\hat{r}_{i0} - E\{r_i(\mathbf{x}_i^T)\}\right],$$

where $x_{ij}^U - x_{ij}^T \geq 0$ defines the input use reduction due to the trading program. However, such a situation is not likely to arise if the conservation incentives are sufficiently small relative to those of the trading program. This is because the trading program will lead to controls that raise farmers' marginal control costs to levels whereby the costs associated with additional controls may not be balanced by the conservation subsidies. We assume this is the case for simplicity and because agricultural pollution remains the largest source of water quality impairments even given existing conservation payments (USEPA, 2000), suggesting existing payments

[8] Providing farmers with initial permits equal to their unregulated estimated loadings does not affect the efficiency of the outcome as long as this does not preclude the total number of permits from being set at an efficient level, and as long as there is a sufficient mix of farmers so that no farmer gains market power from his/her initial permit allocation. Also, note that the qualitative results of our model would still hold if farmers' initial permits were set equal to some level of regulated estimated loadings, so long as farmers remain suppliers of permits.

are not set at levels that are providing substantial improvements.

Coordinated Program Design and Economic Efficiency

We now consider the design of conservation and trading programs assuming farmers participate in both programs.[9] First, suppose there is no double-dipping, in which case condition (7) holds. Without double-dipping, the conservation program does not affect farmers' incentives for the last unit of pollution controlled.[10] Coordination is therefore of no real consequence in this case. This is unimportant if the trading program can bring about an efficient outcome, which, by comparison of condition (7) to the ex ante efficiency condition (1), only occurs if the following condition holds:

$$(9) \quad p_r = E\left\{D'(a^*)\frac{\partial a^*}{\partial r_i}\frac{\partial r_i^*}{\partial x_{ij}}\right\}E\left\{\frac{\partial r_i^*}{\partial x_{ij}}\right\}^{-1} \quad \forall\, i, j,$$

where the superscript asterisks (*) indicate the variables are evaluated at their efficient values. Condition (9) is in general overdetermined in $\{n \times m\}$ equations in one unknown—no single value of p_r ensures this condition will be satisfied $\forall\, i, j$ (Shortle and Dunn, 1986; Tietenberg, 1995). This is because each input choice by each farm has a unique marginal impact on expected damages, but a single permit price does not provide differential incentives for farmers to consider these different impacts [see Horan, Shortle, and Abler (1998) and Shortle and Dunn (1986) for more on incentives applied to mean-based measures of environmental performance]. This result means efficiency cannot be obtained when double-dipping is not allowed. The inability to provide more efficient incentives through policy coordination may therefore be an important limitation of the no double-dipping case.

Now suppose double-dipping is allowed, in which case condition (5) holds. Comparing condition (5) with the ex ante efficiency condition (1), farms will operate efficiently if

$$(10) \quad s_{ij} + p_r E\left\{\frac{\partial r_i}{\partial x_{ij}}\right\} = E\left\{D'(a^*)\frac{\partial a^*}{\partial r_i}\frac{\partial r_i^*}{\partial x_{ij}}\right\} \quad \forall\, i, j.$$

With double-dipping, farmers' incentives for the final unit of pollution control [the right-hand side of equation (10)] depend on both the subsidy and the trading market. Equation (10) therefore implies the conservation subsidies could in principle be set at farm-specific levels to account for each farmer's unique marginal damage impacts to ensure an efficient outcome for farms, regardless of the value of p_r. [Recall, the problem in the no double-dipping case was that p_r could not be set at farm-specific rates to account for differential farm impacts; see equation (9).]

This means there is some flexibility in choosing the trading market policy parameters (t and the number of point source permits) given that the subsidy can always be adjusted to yield the efficient outcome. If a small trading ratio is chosen and/or if total point source permits $\hat{e}$ (where $\hat{e} = \Sigma_{k=1}^{s} \hat{e}_k$) are limited so that p_r is sufficiently large in equilibrium, then most of the reductions are made due to the trading program (since point sources have more incentives to purchase nonpoint permits when t and $\hat{e}$ are small) and the conservation subsidies can be set at relatively small levels to fine-tune the distribution. The result is that farms get paid twice for the same actions, yet efficiency is promoted at a relatively low cost to taxpayers. Of course, it remains an issue as to whether conservation subsidies can be set at farm-specific rates without the government incurring excessive transactions costs.

Having said the trading parameters $\hat{e}$ and t can be chosen to make p_r sufficiently large, there has been no discussion of how such choices will affect the efficiency of point source pollution control efforts. But by considering the dual to the trading problem, we can see there is flexibility to make such choices. A primal approach would be to directly choose t and $\hat{e}$, whereas the dual approach is to choose the permit prices p_e and p_r. Viewed in this manner, p_r can be set sufficiently large to reduce government payments, and p_e can be chosen to promote efficiency among point sources. Firms will operate efficiently if p_e is set at the following level:

$$(11) \quad p_e^* = E\left\{D'(a^*)\frac{\partial a^*}{\partial e_k}\right\} \quad \forall\, k.$$

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[9] While an input subsidy program can be designed efficiently (Horan, Shortle, and Abler, 1999), in which case there would be no need to include farms in the trading program, there is no evidence that USDA payments come close to efficient levels in practice. Because of this and because our focus is on the combined impacts of these programs, we ignore the degenerate case and only focus on farmer participation in both programs.

[10] If farmers participated in the trading program first, then the conservation subsidies and not the trading program would affect farmers' incentives for the last unit of pollution controlled.

However, if firms' emissions have differential marginal ambient impacts (note that efficiency does not require otherwise), then condition (11) is overdetermined in s equations in one unknown—there is no single value of p_e that ensures this condition will be satisfied $\forall$ k. Consequently, the trading system cannot be first-best for firms because a unique permit price does not provide firms with differential incentives to consider the differential marginal impacts of their emissions (Tietenberg, 1995). This inefficiency in the emissions market can affect the level of control deemed optimal among farms. But it does not affect the basic result that coordination can increase the efficiency of nonpoint pollution control when double-dipping is allowed.

Uncoordinated, Second-Best Program Design

We now consider the more common scenario that programs are not well coordinated. Assuming conservation policies are in place first, then trading programs must be designed with this in mind (i.e., the s_{ij}'s are taken as given). The solution can only be second-best, due to the limitations of trading as described above. Using a dual approach, the trading program is optimally designed by taking as given the farms' input and firms' emissions demand functions resulting from the firms'/farms' first-order conditions, and then choosing permit prices optimally.

First, consider the case of no double-dipping. Conditions (5) and (7) indicate the trading program provides the same incentives as a trading program implemented in the absence of conservation programs. Specifically, the last unit of nonpoint pollution control is based on the incentives provided by the trading program—not the subsidy program. The second-best permit prices, and hence trading ratio, are therefore the same as those derived in Horan et al. (2001) [see equation (12) below with s_{ij} set equal to zero]. The only difference is that the aggregate number of permits must be increased to offset the reduction in loads due to the conservation program.[11] Because the conservation program does not influence the last unit of farmer controls when double-dipping is not allowed, a well-targeted conservation program in this instance neither improves efficiency nor increases farmer controls.

Now consider the case of double-dipping. Conditions (1) and (4) define the input and emissions demand functions $x_i(p_r)$ and $e_k(p_e)$, which are substituted into the objective function:

$$\underset{p_e, p_r}{\text{Min}}\ TSC = \sum_{i=1}^{n} c_{ri}(\mathbf{x}_i(p_r)) + \sum_{k=1}^{s} c_{ek}(e_k(p_e)) + E\{D(a(p_r, p_e))\}.$$

Using the dual approach, the first-order necessary conditions for an interior solution can be used in combination with relations (3) and (7) to provide the following expressions for optimal permit prices:

$$(12)\quad p_r^* = \sum_{i=1}^{n}\sum_{j=1}^{m}\left[E\left\{D'(a^*)\frac{\partial a^*}{\partial r_i}\frac{\partial r_i^*}{\partial x_{ij}}\right\} - s_{ij}\right]\kappa_{ij}^* \times \left[\sum_{i=1}^{n}\sum_{j=1}^{m}E\left\{\frac{\partial r_i^*}{\partial x_{ij}}\right\}\kappa_{ij}^*\right]^{-1}$$

and

$$(13)\quad p_e^* = \sum_{k=1}^{s}E\left\{D'(a^*)\frac{\partial a^*}{\partial e_k}\right\}\eta_k^*,$$

where

$$\kappa_{ij}^* = \frac{(\partial x_{ij}^*/\partial p_r)}{\sum_{i=1}^{n}\sum_{j=1}^{m}(\partial x_{ij}^*/\partial p_r)}, \quad \eta_k^* = \frac{(\partial e_k^*/\partial p_e)}{\sum_{k=1}^{s}(\partial e_k^*/\partial p_e)},$$

and r_i^* and a^* are functions of e_k^* and x_{ij}^*, which are the solutions to (3), (7), the first-order conditions for p_e and p_r, and the zero profit conditions defining entry and exit, $V_n = 0$ and $J_s = 0$.[12]

Interpreting κ_{ij} as a weight (since $\Sigma_{i=1}^{n}\Sigma_{j=1}^{m}\kappa_{ij} = 1$), the numerator of the expression for p_r^* is the expected marginal social cost of input use in excess of existing conservation subsidy rates, averaged across all farms and inputs. The denominator is the expected marginal contribution of input use in loadings production, averaged across all farms and inputs. The averaging of impacts across all farms in equation (12) is a consequence of the restriction of 1:1 trading within the nonpoint source category, and the averaging of impacts across all inputs is due to the use of a loadings-based instrument rather than

[11] Another difference is that if conservation programs induce substantial nonpoint controls, then there will be little, if any, room left for trading between point and nonpoint sources (since the initial allocation of nonpoint permits is based on post-conservation expected loads). This outcome is highly unlikely, however, given the historical size of agricultural payments.

[12] The regulatory agency will not be able to induce optimal entry and exit when only permits are used, and the zero profit conditions are different than the conditions the regulatory agency would optimally choose.

an input-based instrument for nonpoint sources. As a result, the second-best price (p_r^*) does not give farms incentives to exploit differences in their relative marginal environmental impacts as would a differentiated price system (i.e., one that emerges from having differentiated trading ratios).[13]

Interpreting η_k as a weight (since $\Sigma_{k=1}^{s} \eta_k = 1$), the emissions permit price (p_e^*) equals the expected marginal social cost of emissions, averaged across firms. The averaging of impacts across firms is again a consequence of the restriction of 1:1 trading within the point source category and implies cost-increasing inefficiencies in the allocation of pollution control efforts. The inefficiencies occur because p_e^* does not give firms incentives to exploit differences in their relative marginal environmental impacts, as a differentiated price system would.

As discussed above, the second-best trading ratio is simply $t = p_e^*/p_r^*$. A ratio of t = 1 implies indifference at the margin between the source of pollution reduction. Ratios greater than one imply a high cost of agricultural controls relative to firms' controls, and thus a preference for emissions reductions at the margin. The reverse is true for ratios of less than one. Clearly, the impact of the conservation subsidies is to increase the trading ratio, making it more expensive for firms to purchase expected loadings reductions. This is because fewer nonpoint reductions are needed after the impact of the subsidies is taken into account. The impact on the total number of permits ($\hat{e}$) is ambiguous, as are the efficiency implications relative to the case with no double-dipping.

It is not possible to analytically determine how better targeting of input subsidies might affect the second-best trading ratio in the double-dipping scenario. Intuitively, however, it might be expected that a greater reliance on input subsidies would be desired relative to trading when input subsidies are better targeted, which could result in a larger trading ratio to discourage point source purchases of farm loadings. We form this expectation because targeted input subsidies have the potential to be more efficient than a trading program involving 1:1 trading within source categories and a uniform trading ratio. Indeed, efficiently targeted input subsidies might be expected to completely crowd out trading, although, as described above, we do not view this case as realistic. In contrast, a greater reliance on trading might be desired relative to the use of input subsidies when input subsidies are not targeted, resulting in a smaller trading ratio to encourage trading.

These expectations result from the fact that the trading program, while nontargeted, is performance-based. We might therefore expect that trading provides better incentives for farmers to gauge the impacts of their choices on performance than do the nonperformance-based, nontargeted input subsidies. These and other issues are examined further through the use of a numerical example.

A Numerical Example

To gain further insight about the implications of agri-environmental programs for the design of trading programs, the gains from coordination, and distributional outcomes, we present results from a model of nitrogen pollution control from point sources and agricultural nonpoint sources in the Susquehanna River Basin (SRB) in Pennsylvania. The Susquehanna is the 16th largest river in the United States, comprises about 43% of the Chesapeake Bay watershed, and provides 50% of the fresh water entering the Bay (Susquehanna River Basin Commission, 1998). About 76% of the Susquehanna's 27.5 thousand square mile watershed is located in Pennsylvania, occupying approximately 45% of the state, with smaller portions found in Maryland and New York [Chesapeake Bay Program (CBP), 2004].

Nutrient pollution is a major problem in the Bay, and nitrogen is a major concern which has been the focus of pollution control efforts over the past several decades (CBP, 2004). The SRB is the major source of nutrients entering the Bay, and agriculture is by far the leading source of nitrogen (CBP, 2001; Pennsylvania Department of Environmental Protection, 1996). Accordingly, reducing nitrogen loads from agriculture as well as other point and nonpoint sources is a major objective of state and federal agencies with responsibilities for the Bay (CBP, 2004). Analogous to other regions of the nation, point sources of nitrogen in the SRB have been subject to stringent regulation, while agricultural controls have largely been voluntary. And as in other regions, there is growing interest in the use of trading as a means to allocate nitrogen reductions from agriculture between point and nonpoint (CBP, 2004).

Our Susquehanna River Basin model is implemented at a highly aggregate level [as opposed to

[13] The degree to which this creates inefficiencies depends on the degree of heterogeneity of marginal impacts and on correlations between key environmental and cost relationships. Also, the second-best price, p_r^*, also does not give farms incentives to exploit differences in risk effects among inputs. These issues are not discussed in further detail here, but interested readers are referred to Horan et al. (2001).

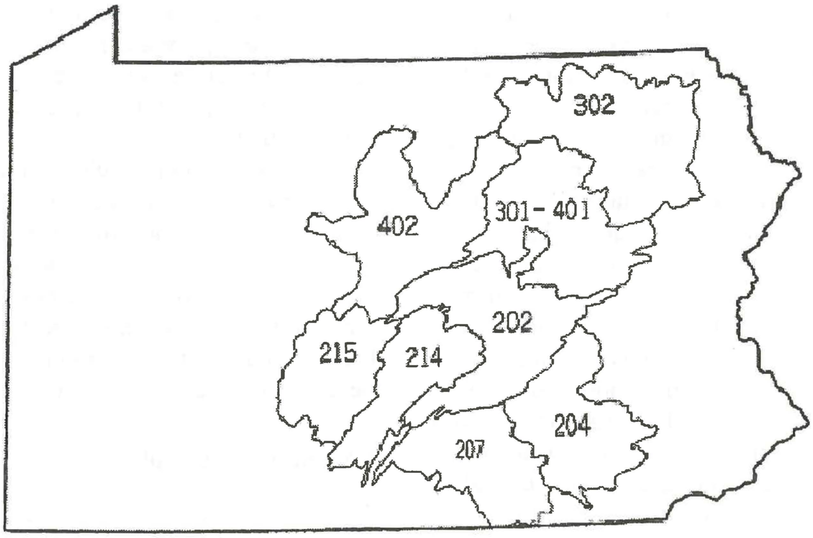

Figure 1. Susquehanna River Basin (SRB) watersheds in Pennsylvania

modeling all 9,726 farms growing corn in the SRB (Abler et al., 2002)] to capture essential features of the economic problem without being overly cumbersome or costly to construct and compute. This analysis considers eight SRB sub-watersheds (or aggregations thereof) as defined by the Pennsylvania Department of Environmental Protection for water quality reporting and planning activities in the SRB (see figure 1). For each sub-watershed, we develop aggregate models of (*a*) agricultural pollution control costs for each region, (*b*) point source pollution control costs for each region, (*c*) nutrient delivery from each region to the Chesapeake Bay, and (*d*) the economic damage costs of ambient nutrient pollution in the Bay.

Farm Costs and Loadings

Nonpoint loads are assumed to be the result of corn production (which also utilizes most manure produced from livestock operations in the region), modeled as a two-level, constant elasticity of scale technology that exhibits constant returns to scale at both levels (Sato, 1967). Following prior work based on this approach (e.g., Kawagoe, Otsuka, and Hayami, 1985; Thirtle, 1985; Binswanger, 1974), aggregate production in the *i*th region, denoted y_i, is a function of a composite biological input, x_{iB}, and a composite mechanical input, x_{iM}, i.e.:

$$y_i = A_i\left(\alpha_i x_{iB}^{\rho_i} + (1 - \alpha_i)x_{iM}^{\rho_i}\right)^{1/\rho_i},$$

where A_i and α_i are parameters, and $\rho_i = (\sigma_i - 1)/\sigma_i$, where σ_i is the elasticity of substitution between the biological and mechanical inputs. Similarly, x_{iB} is produced using land, x_{i1}, and fertilizer, x_{i2}:

$$x_{iB} = K_i\left(\beta_i x_{i1}^{\gamma_i} + (1 - \beta_i)(\mu_i x_{i2})^{\gamma_i}\right)^{1/\gamma_i},$$

where K_i and β_i are parameters, μ_i is the proportion of nitrogen taken up by the plant, and $\gamma_i = (\sigma_{Bi} - 1)/\sigma_{Bi}$, with $\sigma_{Bi} = s_{Bi}\sigma_{iL,N} + s_{Mi}\sigma_i$, where s_j ($j = B, M$) is the cost share of the *j*th input in production, and $\sigma_{iL,N}$ is the elasticity of substitution between x_{i1} and x_{i2}. Because nitrogen is more or less a fixed proportion of fertilizer, x_{i2} is denoted as nitrogen. The mechanical input is produced using capital and labor. Assuming the prices of these inputs are fixed, there is no reason to further decompose x_{iM} into its constituent parts, as capital and labor will be used in fixed proportions.

The price of corn, denoted p, is fixed and does not vary within the watershed. The same is true of the prices of nitrogen, w_2, and the mechanical input, w_M. Land supply is defined regionally and takes a constant elasticity form,

$$L_i = b_i w_{i1}^{\eta_i},$$

where b_i is a parameter, w_{i1} is the price of land, and η_i is the elasticity of supply. Land supply reflects the opportunity cost of this input, which differs in each region. Given this specification, agricultural sector control costs (i.e., foregone profits and landowner surplus) are:

$$c_{ri}(\mathbf{x}_i) = py_i(\mathbf{x}_i^U) - \sum_{j\neq 1} w_{ij}x_{ij}^U - \int_0^{x_{i1}^U} w_{i1}(v)\,dv - \left[py_i(\mathbf{x}_i) - \sum_{j\neq 1} w_{ij}x_{ij} - \int_0^{x_{i1}} w_{i1}(v)\,dv \right].$$

To calibrate the economic model in a realistic fashion, we use cost share and production share data for corn production in eight sub-watersheds of the Susquehanna River Basin (SRB) in Pennsylvania (USDA, 2000; Pennsylvania Agricultural Statistics Service, 1998). For the parameters σ_i, $\sigma_{iL,N}$, μ_i, and η_i, we adopt the mean of estimates reported in the literature ($\sigma_i = 0.5$, $\sigma_{iL,N} = 1.25$, $\mu_i = 0.7$, and $\eta_i = 0.3$). See Horan, Shortle, and Abler (2002) for particular references and distributions of values reported in the literature.

Region i's loadings function is of the form

$$r_i = r_{1i}P_i + r_{2i}P_i(1-\mu_i)x_{i2} + r_{3i}\frac{[P_i(1-\mu_i)x_{i2}]^2}{x_{i1}},$$

where P_i is precipitation, and r_{qi} ($q = 1, 2, 3$) are parameters. Increases in nitrogen have a positive impact on loadings ($\partial r_i/\partial x_{i2} > 0$), and so the subsidy and trading programs optimally increase the cost of nitrogen use. In contrast, increases in land use (holding nitrogen use constant) have a negative impact on loadings ($\partial r_i/\partial x_{i1} < 0$), and so the subsidy and trading programs optimally decrease the cost of land. The loadings functions are calibrated from SRB data (Nizeyimana et al., 1997) and from research results used to develop Total Maximum Daily Load (TMDL) recommendations for Pennsylvania. Precipitation is taken to be stochastic in our simulation model and is assumed to be gamma distributed with a mean and variance based on precipitation data for the regions. Another source of uncertainty sometimes mentioned in the nonpoint case is uncertainty regarding the effectiveness of farm controls (i.e., in terms of our model, the coefficients r_{1i} and r_{2i} would be uncertain) (Malik, Letson, and Crutchfield, 1993). This uncertainty may result from a lack of experience with farm controls, but over time it would presumably diminish. While this study does not model this type of uncertainty explicitly, we acknowledge it could affect policy variables in ways that allocate greater control to point sources (Malik, Letson, and Crutchfield, 1993).

Firm Costs

Firms' abatement cost functions are of the constant elasticity form

$$c_i(e_i) = z_i e_i^{\zeta_i} + F_i,$$

where $z_i > 0$ is a parameter, $\zeta_i < 0$ is the elasticity of costs with respect to emissions, and F_i is a fixed cost. To ensure realism, the parameters are calibrated using data from a Susquehanna River Basin Commission report (Edwards and Stoe, 1998). The report provides base-level emissions (abatement) for the most important point sources of nitrogen in the SRB (primarily wastewater treatment facilities), as well as costs for adopting various nutrient control technologies (e.g., three-stage annual treatment and five-stage annual treatment), and the emissions levels for each source under these technologies. Data for these technologies and emissions levels were aggregated to the regional level from individual sources. Under this calibration, point sources incur some control costs even before the trading program is implemented, for instance in response to previous pollution control legislation targeted only at them. Our interests in their costs under a trading program will largely focus around their increased costs relative to what they incurred prior to the trading program.

Nutrient Delivery

Loadings and emissions are defined as nutrients entering into the sub-watershed in which they originate. However, only a fraction of the loadings or emissions generated from each region is delivered to become part of the ambient pollution concentration in the estuary, which is the chief area of concern for policy purposes. Delivery generally depends on deterministic factors such as distance from the estuary and topography, and also stochastic processes such as weather (see Smith, Schwarz, and Alexander, 1997). The proportion of the emissions/loads that is delivered is modeled as a stochastic delivery coefficient, φ_i, the realized value of which differs by region and by source type within a particular region. Total delivered loads are therefore represented by

$$a = \sum_{i=1}^{n} \varphi_i r_i + \sum_{k=1}^{n} \varphi_k e_k.$$

This relation represents a first-order approximation to the actual delivery process, which is thought to be reasonable in many cases (Roth and Jury, 1993).

The delivery coefficients are taken to be gamma distributed, with the variances derived from results of the U.S. Geological Survey SPARROW model for the SRB (Smith, Schwarz, and Alexander, 1997).

Table 1. Delivery Coefficient Distributions: Low and High Heterogeneity Models

SRB Watershed[a]	Mean Delivery Coefficients (for point and nonpoint sources): Low Heterogeneity (SRB estimates)	Mean Delivery Coefficients (for point and nonpoint sources): High Heterogeneity (random estimates)	Standard Deviation of Delivery Coefficients (both models)
202	0.611	0.91	0.070
204	0.660	0.60	0.070
207	0.560	0.05	0.137
214	0.710	0.50	0.110
215	0.731	0.70	0.114
301–401	0.581	0.30	0.160
302	0.684	0.90	0.126
402	0.626	0.02	0.068

[a] See Pennsylvania map, figure 1, showing the eight Susquehanna River Basin sub-watersheds.

Two sets of means were used, however, to examine the role of delivery coefficient heterogeneity across regions. Without significant regional heterogeneity in mean delivery coefficients, the restriction of 1:1 trading within source categories might be expected to result in few, if any, inefficiencies because there are few benefits from exploiting differences in the marginal impacts of each source's emissions/loadings. If this is so, the result would be that the economic and environmental performance of the economically efficient program and all the second-best programs (i.e., targeted and nontargeted subsidy programs, with or without double-dipping) will tend to converge. Alternatively, greater divergence in results might be expected when there is significant regional heterogeneity of mean delivery coefficients.

We therefore explore two alternative models: high heterogeneity and low heterogeneity. Mean delivery coefficients for the low heterogeneity model are taken from SPARROW results for the SRB, while mean delivery coefficients for the high heterogeneity model are chosen randomly to ensure significant heterogeneity (see table 1). Heterogeneity can be measured by the coefficient of variation of mean delivery coefficients across regions (i.e., standard deviation of mean delivery divided by average mean delivery across regions). The coefficient of variation is 0.09 in the low heterogeneity model, and 0.68 in the high heterogeneity model.

Economic Damages

Economic damages from pollution are a second-order approximation of actual damages, which is taken to be an increasing, convex function of a, $D(a) = d_1 a + d_2 a^2$. Smith (1992) relates groundwater damages to the net benefits of agriculture. We use such a relation to calibrate damages, setting initial expected damages equal to 22.5% of unregulated, private agricultural net benefits, and by choosing an elasticity of damages equal to 1.6. The percentage used to set initial expected damages is slightly larger than values reported by Smith for groundwater damages, but we are considering damages from both point and nonpoint sources (whereas Smith only considered damages by agriculture). The elasticity guarantees convex damages, and hence social risk aversion with respect to pollution. This will bias optimal trading ratios downward (Shortle, 1987; Malik, Letson, and Crutchfield, 1993). This feature provides a nice contrast with our double-dipping model in which we expect larger trading ratios.

The Simulation Experiment

We examine the efficient outcome, as defined by equations (1) and (2) (equivalent to optimal coordination), and four uncoordinated trading/conservation subsidy scenarios: (*a*) targeted subsidies, no double-dipping; (*b*) targeted subsidies, double-dipping; (*c*) nontargeted subsidies, no double-dipping; and (*d*) nontargeted subsidies, double-dipping. In each case, the choices of trading program parameters, $\hat{e}$ and t, depend on the level of the conservation subsidies. It is assumed conservation subsidy rates for the uncoordinated scenarios are chosen to minimize the following weighted version of total social costs:

$$WTSC = \sum_{i=1}^{n} c_{ri}(\mathbf{x}_i) + \sum_{k=1}^{s} c_{ek}(e_k) + \psi E\{D(a)\},$$

Table 2. Simulation Results for High and Low Heterogeneity Models ($\psi = 0.33$)

Scenario	[1] Efficiency Gain	[2] Farm Resource Costs	[3] Firm Resource Costs	[4] Farm Subsidy Receipts	[5] Farm Permit Sales Revenues	[6] Trading Ratio	[7] Total Permits
Efficient Outcome	100.0	100.0	100.0	—	—	—	—
Low Heterogeneity (LH):							
▸ No double-dipping: targeted	99.9	98.8	100.2	14.3	201.8	0.90	95.9
▸ Double-dipping: targeted	100.0	98.8	100.2	85.5	142.8	1.45	75.4
▸ No double-dipping: nontargeted	99.9	98.8	100.2	11.3	194.7	0.90	95.1
▸ Double-dipping: nontargeted	96.2	104.1	100.2	56.9	168.8	1.16	74.4
High Heterogeneity (HH):							
▸ No double-dipping: targeted	84.0	91.0	106.2	13.9	200.0	0.94	95.7
▸ Double-dipping: targeted	93.0	86.7	105.4	91.9	136.3	1.59	76.6
▸ No double-dipping: nontargeted	84.0	91.0	106.2	11.7	194.1	0.94	95.1
▸ Double-dipping: nontargeted	81.0	96.2	106.2	56.6	168.4	1.22	74.5

where $\psi < 1$ is a weight on expected damages. This form of the objective function is chosen for two reasons.

First, with expected damages receiving less weight than private costs, the resulting subsidy rates will be too low. This in turn leaves room for gains from trading. This setup is also consistent with a belief that current conservation payment rates are not set at levels which can yield substantial environmental gains, possibly reflecting a low weight placed on environmental performance relative to farm income and other considerations [see Rausser and Foster (1991) for a discussion of rent-seeking in agri-environmental programs].[14]

Second, placing at least some weight on expected damages allows subsidy rates in the targeted conservation subsidy scenarios to be differentiated across regions to reflect, at least to some extent, differential marginal environmental impacts across regions. Subsidy rates are applied uniformly across regions in the nontargeted subsidy scenarios. Note that because the uncoordinated conservation policy objective is consistent with the trading authority's objective to minimize *TSC*, the gains from coordination will be less than if the objectives were conflicting.

[14] An alternative approach is for the USDA to minimize total social costs subject to a budget constraint. If USDA had a fixed budget constraint for subsidy programs, then the problem would turn into a game between the USDA, farmers, and the EPA: USDA would have to set its subsidy rates conditional on expectations about farmer responses to both the conservation and trading programs in order to estimate whether the budget constraint would be satisfied. This would introduce a strategic component into the problem that would be interesting to analyze in a separate study but is beyond the scope of the current paper.

The results are reported in table 2 for the high heterogeneity and low heterogeneity models, with $\psi = 0.33$ in each case. All monetary results are presented as indices, which are used due to our interest in the relative (as opposed to absolute) performance of the trading programs and also to overcome some scaling effects. The first column of results reports an efficiency gain index. Efficiency gain (*EG*) for a particular scenario is calculated as the percentage reduction in expected social costs (*TSC*) relative to the baseline data consisting of an unregulated equilibrium for farms and some prior degree of controls for firms, i.e., $EG = (TSC^B - TSC^s)/TSC^B$, where TSC^s represents expected social costs in scenario s, and TSC^B represents baseline expected social costs. The maximum potential efficiency gain occurs in the efficient outcome. We therefore divide each scenario's efficiency gain by that of the efficient outcome; hence, the efficiency gain index (EGI) for the efficient outcome is 100. EGI values below 100 represent the degree to which the potential efficiency gains have been achieved.

Columns [2] and [3] of table 2 present pollution control costs to farms and firms, not including any transfers due to subsidy receipts or permit sales/purchases, i.e., the real resource costs associated with pollution control. In each case, the values are indices, with the numerator being control costs for the scenario under consideration and the denominator being the efficient level of control costs. Values greater than 100 imply over-control relative to the efficient level, while values less than 100 imply under-control.

Columns [4] and [5] report results for farm subsidy receipts and farm permit sales revenues. In each case, the values are indices, with the numerator being farm receipts/revenues for the scenario under consideration and the denominator being farm control costs for the scenario under consideration. Values represent the percentage of control costs that are covered by the receipts/revenues, with values greater than 100 indicating the receipts/revenues more than cover control costs, and values less than 100 indicating control costs are not covered. These two columns are additive, and in each case the sum is greater than 100, which must be the case for farms to voluntarily reduce their pollution under these two programs.

The final two columns in table 2 represent optimal choices of the trading program parameters for each scenario. The trading ratio (column [6]) is presented in absolute terms, not as an index. Total permits (column [7]) are represented as an index, with the numerator being the total permits (denominated in terms of emissions) for the scenario, and the denominator being total initial emissions and loads (denominated in terms of emissions, using the optimal trading ratio for the scenario at hand to convert loads to emissions).

The overall pattern of results is the same for the high heterogeneity (HH) and low heterogeneity (LH) models, with program efficiency differences being more pronounced in the high heterogeneity model [recall, the double-dipping (DD) and no double-dipping (NDD) scenarios would both be efficient if there were no heterogeneity in delivery coefficients].

First, consider the targeted scenarios. In both models, the DD scenario produced greater efficiency gains than the NDD scenario, although the degree of difference depends largely on the heterogeneity of delivery coefficients. The DD scenario is more efficient because this scenario provides farmers with targeted incentives, where the final unit of pollution controlled depends on the incentives provided by both the subsidies and the permit market. In contrast, the NDD scenario does not provide targeted incentives because the final unit of control is governed only by incentives created by the permit market. Better targeting of farm controls under DD results in smaller expected damages relative to NDD. Cost savings due to a more efficient allocation of controls makes up the rest of the difference in economic net benefits between the two scenarios. This cost savings effect is easiest to observe in the HH model, where better targeting under DD results in lower control costs for both farms and firms—with farms incurring proportionately lower costs relative to NDD. The improved targeting of farm controls is also evidenced by fewer permits under the DD scenario. This is in spite of the fact that, under the NDD scenario, farms have initial permit allocations equal to their pre-trade but post-conservation expected loads.

Still focusing on targeted scenarios, consider the payments farmers receive to reduce their loads. Subsidy receipts are approximately six times larger under DD than under NDD in each model. This occurs because, under DD, the combination of the two programs induces farmers to provide more environmental benefits under DD, and the subsidies must pay for these additional benefits.[15] But even with substantially larger subsidy payments under DD, the total payments received by farmers are less than 6.5% larger under DD than NDD. This is because the trading program is optimally adjusted in response to the subsidy payments farmers receive for the same pollution reductions. Specifically, the trading ratio is less than one under the NDD scenario, as is consistent with theory given our specification for risk (Shortle, 1990; Malik, Letson, and Crutchfield, 1993; Horan et al., 2001). However, the trading ratio is greater than one under the DD scenario. The larger trading ratio under DD makes it more expensive for firms to purchase loadings reductions, reducing the equilibrium price farms receive for permits. But farms are willing to sell their permits for a lower price because their abatement efforts are also subsidized by USDA, yielding a targeted reduction in the opportunity cost of abatement by nonpoint sources.

The net result is that firms spend less on loadings reductions under the DD scenario. In fact, the larger total subsidy received by farmers under DD is almost entirely offset by reduced permit revenues, so that farmers have only a slight to moderate preference for DD. Firms, however, have a much stronger preference for the DD scenario. Their control costs are almost identical under DD and NDD, but their loadings permit expenditures are much larger under NDD. Specifically, after accounting for the different bases in the index values of permit sales, our findings indicate firms' expenditures on loadings permits are more than

[15] The difference in subsidies between DD and NDD is diminished as the weight on expected damages is reduced. For instance, if we were to reduce the weight on expected damages to 10% when calculating subsidy rates, then for the nontargeted programs DD would still be more efficient—but the difference in the subsidy index between DD and NDD is cut to approximately 11.2 (compared to a 45.6 difference when a 33% weight is used). The reduction is greater for the targeted case.

30% larger under NDD in the LH model and more than 54% larger under NDD in the HH model. Because farms are no better off from the larger subsidies they receive under DD, while firms are better off, we conclude that the extra subsidy payments which arise under DD are effectively transferred to firms under an optimally designed trading program.

Now consider the nontargeted scenario. The NDD scenario is more efficient than the DD scenario when the conservation policies are not targeted. The reason is, under the NDD scenario, the permit market provides performance-based incentives for the final unit of agricultural pollution control—that is, farmers have incentives to gauge the impacts of their choices on performance. In contrast, under the DD scenario, both (nontargeted) subsidies and the permit market provide incentives for the final unit of agricultural pollution control. Consequently, the permit market provides fewer incentives in the DD scenario because the subsidies make up the difference. This reduced reliance on the trading program to provide incentives reduces the efficiency of the DD scenario relative to the NDD scenario. This is because the trading program provides better incentives for farmers to gauge the impacts of their choices on performance than do the nonperformance-based, nontargeted input subsidies. So, whereas targeted input subsidies improve the relative efficiency of the NDD scenario (because targeted input subsidies are more efficient than trading when there is 1:1 trading within source categories and a uniform trading ratio), nontargeted subsidies reduce the relative efficiency of the NDD scenario.

Upon considering the payments farmers receive to reduce their loads in the nontargeted scenarios, subsidy receipts are approximately five times larger and permit sale revenues significantly lower under DD than under NDD in each model. This result is consistent with those of the targeted scenarios and for the same reasons. Also consistent with the targeted scenarios, and for essentially the same reasons, the total payments received by farmers are less than 6.5% larger under DD than NDD. As with the targeted scenarios, the net result is that firms spend less on loadings reductions under the DD scenario. Again, firms clearly prefer the DD scenario, although not as strongly as under the targeted scenarios. After accounting for the different bases in the index values of permit sales, findings show that firms' expenditures on loadings permits are only about 9.5% lower under DD in each model, as compared to a 30–50% difference in the targeted scenarios. Firms clearly benefit the most from DD when subsidies are targeted.

Based on the results reported in table 2, farms are slightly better off in each model under a nontargeted approach, regardless of whether double-dipping occurs. But the same is not so for firms. Firms are slightly better off under NDD when subsidies are not targeted, but they are significantly better off under DD when subsidies are targeted. Nontargeted DD ratios are lower than the targeted DD ratios, for roughly the same number of permits. This means firms face greater incentives to purchase nonpoint reductions in the nontargeted DD program relative to the targeted DD program. As described above, the reason is that the trading program provides better incentives for farmers to gauge the impacts of their choices on performance when input subsidies are not targeted, while the opposite is true when input subsidies are targeted. Accordingly, it makes economic sense to utilize trading to a greater degree when input subsidies are not targeted, and to utilize the input subsidies to a greater degree when they are targeted (implying less need for trading). The relative utilization of the two programs affects the degree of income transfer farmers receive from subsidies and trading, with firms funding a much larger proportion of farm controls under DD when subsidies are not targeted.

Finally, consider the impacts to taxpayers, which can be seen by their level of subsidy payments. Taxpayers are better off under NDD relative to DD, because under NDD taxpayers do not have to pay for loadings reductions that firms are willing to pay for. Taxpayers are also better off when subsidies are nontargeted. As described above, this is because the trading program is utilized to a greater extent when subsidies are not targeted, transferring more of the financial burden of paying for loadings reductions to firms.

Conclusions

Input-based payments to improve the environmental performance of agriculture have long been of interest and have been implemented in various forms by the federal and state governments (Ribaudo, 2001). In general, the water quality gains from these programs have not been large, as agriculture remains a leading cause of water quality impairments (Ribaudo, 2001). Recently, there has been growing interest in and use of pollution trading to reduce nonpoint source pollution from agriculture and other nonpoint sources of pollution, and to improve the efficiency of the allocation of pollution load reductions between point and nonpoint sources. While there has been much research on the design of stand-alone input-based incentives and point-nonpoint trading schemes (Shortle and Horan, 2002), the joint implementation of these mechanisms has received little formal attention.

In this analysis, we have studied a mixture of these policies. The mixture involves (*a*) input-based agri-environmental payments that are offered at levels which can produce some efficiency gains but are insufficient to achieve an efficient outcome, and (*b*) point-nonpoint trading. This mixture is analogous to the existing policy setting in which agricultural agencies are offering agri-environmental payments that are inadequate to achieve water quality goals, with state water quality agencies considering additional mechanisms to achieve the needed pollution reductions. We consider both targeted and nontargeted agri-environmental payment schemes, but have no spatial targeting of the trading program. The analysis examines both the coordinated and noncoordinated use of these policies. In the noncoordinated case, it is assumed the agri-environmental payments are already in place and the trading authority takes these into account when designing the trading program.

Policies administered by different agencies are not usually perfect substitutes, but rather may be complementary or competing. In the setting we examine, the combined performance of these policies, both for the coordinated and noncoordinated cases, is better than when each program is administered individually. Hence, these policies are complementary in nature. The degree of improved performance depends on whether the programs are coordinated or not, whether double-dipping is allowed, whether the agri-environmental payments are targeted, and whether there is enough heterogeneity in the system to take advantage of targeted payments.

Coordinating the mechanisms can only provide economic efficiency gains (relative to noncoordination of the jointly implemented policies) when the two programs jointly influence farmers' decisions about the last unit of pollution controlled. We demonstrate that this can only occur under double-dipping—i.e., when the farmer can be paid twice (once by each program) for undertaking a particular combination of pollution control actions.

When the programs are not coordinated and agri-environmental payments are lower than what is required for efficiency in a stand-alone setting, then double-dipping may either increase or decrease the efficiency of pollution control allocations, with the result depending on the targeting of the input payments. Double-dipping increases efficiency if the input-based policies are well-targeted. This is because, through double-dipping, the marginal incentives provided by the programs are combined, with farmers receiving targeted incentives which are closer to efficient levels than would occur under either program individually. Farmers and point sources are also better off under double-dipping in the well-targeted case. In fact, double-dipping results in a transfer of much of the agricultural subsidies to point sources, provided the trading program is optimally designed.

If the input-based policies are not well-targeted, then the trading program provides better incentives, since it is performance-based. In this case, efficiency is improved by prohibiting double-dipping so that farmers only face performance-based incentives for their marginal choices. Double-dipping may result in a substantially higher income transfer to farmers in this case, but at the expense of point sources.

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