

Modeling state-level water quality management: the case of the Neuse River Basin

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Received 21 September 1998; received in revised form 29 January 1999; accepted 17 March 1999

Abstract

This research considers how the perceived costs of achieving water quality objectives are sensitive to three issues surrounding model structure and policy design. These issues include: (i) the extent of the regulated market, (ii) the responsibility of the regulated market for background pollution, and (iii) the use of alternative policy instruments. A large-scale process model is used to evaluate and compare the costs of nutrient reduction in the Neuse River Basin in North Carolina under various instruments, including a plan currently being considered by state regulators. The results emphasize the importance of flexibility in both model structure and policy design. © 2000 Elsevier Science B.V. All rights reserved.

JEL classification: Q0; Q2

Keywords: Water quality; Instrument choice; Non-point source pollution; Point source pollution; Incentive-based; Command and control

1. Introduction

Beginning with the 1987 Clean Water Act Amendments, federal water quality policy has increasingly offered states more flexibility in achieving pre-specified water quality standards. Indeed, an emphasis on ‘flexibility’ in the form of incentive-based (IB) systems and performance based standards is present in both

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Sections 101 and 302 in the Water Quality Amendments passed by the House (H.R. 961) during the 104th congress.¹ Yet while economists have argued for over two decades that flexible IB policies can meet environmental goals at lower cost than with mandated control measures, there have been very few real evaluations comparing these instruments, and even fewer evaluations—real or simulated—that have studied the features giving rise to cost differences.² This research intends to demonstrate the importance of incentives and flexibility on the costs of state-level water quality management while illustrating the sensitivity of these costs to particular decisions associated with a water quality model's design.³

Using a case study of nutrient control in North Carolina, three important conclusions emerge. First, while the results support the literature (see, e.g., Tietenberg, 1985) suggesting IB policies can offer potential large cost savings relative to command and control (CAC) tactics, this research illustrates that the magnitude and distribution of estimated costs can vary by more than 40% depending on what sources are targeted and the treatment of background residuals. Second, mixed systems that use a CAC approach in markets with potentially large transactions costs and an IB approach in markets with relatively low transactions costs and substantial control cost differences can achieve substantial cost savings. Since we rarely observe in practice the types of pure CAC or IB systems represented in the traditional literature, it useful to highlight the potential efficiency gains associated with such real world management plans as those considered in the Neuse River Basin in North Carolina. Third, since nutrient problems within a river basin are often a combination of point and non-point source pollution discharge, basinwide management strategies are likely to target both point and non-point source polluters. This research includes point and non-point source dischargers and illustrates that basinwide management strategies that target both types of sources offer greater potential cost savings than strategies targeting

¹ Congressional Research Service (1996).

² See, e.g., Dales (1968), Ayres and Kneese (1969), Kneese and Schultze (1975) for early discussions comparing incentive based instruments to command and control strategies. Tietenberg (1985) and Hahn and Hester (1989) discuss a reasonable number of empirical studies evaluating these two instruments. Both acknowledge, though, the simulated nature of these comparisons. Kling (1994) illustrates the sensitivity of costs to functional form choice.

³ While this research does indeed estimate the costs in a simulated fashion, much of the data and the policy objectives are not simulated. As will be discussed in more detail throughout the paper, this research uses data on actual soil characteristics within the Neuse River Basin; incorporates water quality parameters that are currently being used by the North Carolina Division of Water Quality (DWQ); includes factor prices from the Neuse River Basin as well as the recommended cropping practices as suggested to growers in the Neuse River Basin by the North Carolina Cooperative Extension Service; targets the same percentage reduction in estuarine nitrogen loadings as required in the Neuse River Nutrient Sensitive Waters Management Strategy; and includes the main sources—both point and non-point—that contribute the majority of the nitrogen loadings.

either source separately. Finally, these three conclusions suggest that a model's ability to evaluate and inform policy depend on the level of detail within the model and its flexibility to address alternative issues.

Section 2 develops a stylized model to help expose some potential and current extensions to the traditional approach to residuals management. After describing the particular application targeted—excessive nutrient discharge in the Neuse River Basin in North Carolina—and briefly introducing the model in Section 3, Section 4 provides a general comparison of the IB vs. CAC strategies under a variety of model specifications. Section 4 also presents a potential policy currently under consideration for controlling nutrients in the Neuse River and evaluates it along with three other plans. Section 5 discusses these results in the context of recent water quality legislation.

2. Theory

The flexibility and perhaps usefulness of a water quality model is in large part determined by its ability to address relevant control opportunities, physical realities of the environment, spatial detail, the extent of the market, and policy alternatives associated with a particular application. Before expounding on these elements, it may be useful to present a simple theoretical exercise to help illustrate how choices regarding these elements can influence the solutions intended to mimic the outcomes of alternative control strategies.⁴

This stylized model consists of regulating two farms whose nutrients affect downstream water quality. Assuming perfect competition and zero transactions costs, a least cost solution for a particular level of water quality is derived. The two farms are assumed to have abatement cost functions, $c_i(e_i)$, $i = \{a, b\}$, defined in terms of field-edge emissions, e_i , where $c'_i(e_i) \leq 0$ and $c''_i(e_i) \geq 0$. Water quality, Q , is represented by two receptor points, $Q = (q_m, q_n)$. The effect of a unit emission by farm i on water quality at receptor j is represented by the concentration coefficient, d_{ij} . The solution that meets predefined levels of water quality efficiently can be obtained by:

$$\min_e \sum_i c_i(e_i) \text{ subject to } \sum_i d_{ij} e_i \leq q_j \quad \text{for } i = a, b \text{ and } j = m, n \quad (1)$$

Following from the first order conditions, a familiar necessary condition for achieving the least cost solution is that the marginal costs of increasing water quality at each receptor be equalized across farms:

$$\gamma_m = [c'_a(e_a) - d_{an} \gamma_n] / d_{am} = [c'_b(e_b) - d_{bn} \gamma_n] / d_{bm} \quad (2)$$

where γ_m is the shadow price on water quality at monitoring point m . These marginal costs include the marginal costs of emissions, $c'_i(e_i)$, an adjustment for

⁴ The framework follows the approach presented by Montgomery (1972).

the effect of emissions on other locations, $d_{in}\gamma_n$, both weighted by monitoring point m 's transfer coefficient, d_{im} . A likely event, though, is that only one of the receptors will be binding (i.e., a corner solution), thereby leading to the well-known condition that the ratio of transfer coefficients equal the ratio of marginal abatement costs.

As illustrated by Montgomery (1972), an IB instrument given frictionless markets and zero transaction costs achieves the solution suggested in Eq. (2). Rarely will a CAC approach achieve such a solution since differences likely exist across agents in their control abilities. These differences, coupled with the rigidity of a CAC approach, result in solutions where the marginal costs of increasing environmental quality differ across firms. Specifying $\{\gamma_{a,m}^s, \gamma_{b,m}^s\}$ as the shadow prices on the water quality constraint m under a CAC strategy for farm a and b , and $\{e_a^s, e_b^s\}$ as their respective emissions, potential efficiency gains can be represented by shadow price differences as follows:

$$\gamma_{a,m}^s - \gamma_{b,m}^s = [c'_a(e_a^s) - d_{an}\gamma_n^s]/d_{am} - [c'_b(e_b^s) - d_{bn}\gamma_n^s]/d_{bm} \quad (3)$$

Indeed, it is these differences that represent the potential gains from implementing an IB instrument. That is, in order to realize any gain from implementing an IB system, the marginal costs of abatement must differ across firms. The greater these differences, the greater the incentives to trade.⁵

Eq. (3) raises a number of interesting issues regarding the development of a model to address today's water quality concerns in addition to traditional concerns. First, conventional analyses focus primarily on methods that reduce emissions per unit output—i.e., end-of-pipe treatment strategies—as evidenced by cost functions of emissions alone.⁶ Assuming separability between output and emissions may limit the number of control options confronting agents as well as some viable and potentially more realistic control strategies. While Kling (1994) has recognized this issue, it is seldom acknowledged and its significance rarely illustrated.⁷ Second, environmental differences under the traditional approach are most often confined to differences in the in-stream transport coefficient, d_{ij} , which provides

⁵ That is, of course, unless the amount of control required is so great that all sources must control as much as is economically possible (Tietenberg, 1985, pp. 45–47).

⁶ See, e.g., comparisons of IB vs. CAC strategies in the works of Montgomery (1972), Tietenberg (1973) and Kwerel (1976).

⁷ Both Rubin and Kling (1993) and Kling (1994) cite the case for reducing automobile emissions and suggest that the assumption of a fixed output level seems unrealistic. Rather, they suggest, it seems more tenable that manufacturers would be more likely to change their mix of vehicles to meet emissions standards. A number of recent studies focusing on non-point source pollution allow agents to change output mix to meet reduction restrictions (see, e.g., Johnson et al., 1991; Mapp et al., 1994; Helfand and House, 1995). Yet none of the non-point source literature examines the significance of this 'non-treatment' control option.

the spatial link across sources. Modeling non-point source pollution problems requires additional attention to both the physical realities (i.e., local conditions) associated with the transport of non-point source pollution through and over different soil types and the environmental influences on production and control technologies.⁸

Broader issues of concern relate to background residuals, the boundary of analysis, and finally potential policy instruments.⁹ Background residuals from unregulated sources can influence both the composition of control responsibility across a basin, as well as the level of required reduction to achieve the targeted levels of water quality. Additionally, where one draws the boundary of analysis (e.g., farm-level, county-level, basin-level, or state-level) can have serious implications as to the composition of sources, not to mention how output is defined.

Most analyses of regulatory systems compare subjecting all sources to an IB instrument to subjecting all sources to a CAC strategy. Yet it is likely that the transaction costs of implementing an IB system, or even a CAC strategy, differ across sources and in some instances these differences may be extremely large for one group relative to another group. Thus requiring all sources to participate in an IB system or a CAC strategy may not be cost-effective. Since actual applications differentiate between sources, subjecting some to a CAC strategy and others to an IB system, evaluating such ‘mixed’ systems may be quite informative and useful.¹⁰

To provide an accurate depiction of the cost of basinwide nutrient management, then, requires models flexible enough to acknowledge these issues and detailed enough to capture the factors likely to be responsible for heterogeneity across agents nutrient control abilities. The remainder of this research will focus on the issues associated with who to regulate and their nutrient reduction responsibility, as well as provide a comparison of various policy alternatives since these are issues confronting the North Carolina Division of Water Quality (DWQ) in targeting nutrient reduction in the Neuse River Basin.

⁸ Recent work by Weinberg et al. (1993) and Helfand and House (1995) account for the effects of soil type on nutrient runoff. Such ‘ground truth’ considerations can be found in the materials and energy balance approach used in early studies by Resources for the Future (RFF). See, e.g., Spofford et al. (1975), and their work on residuals management in the Delaware River Basin.

⁹ In addition to these concerns, the potential efficiency gains from implementing an IB system relative to a CAC strategy often rest on assumptions such as a perfectly competitive market and zero transactions costs. Hahn (1989) illustrates how relaxing the perfectly competitive market assumption can influence the conclusion one can draw about the relative efficiency across these instruments. Furthermore, transaction costs from implementing a pure IB system to control non-point source pollution are likely to be extremely large due to the difficulty of monitoring and enforcing this diffuse type of pollutant.

¹⁰ For a more detailed theoretical extension of the traditional approach to modeling potential efficiency gains that focuses on mixed systems, see Schwabe and Smith (1998).

3. Modeling nutrient control in the Neuse River Basin

3.1. *The Neuse River Basin: background and trouble signs*

The Neuse River Basin, with over 3000 stream miles, extends from the northwestern boundary of North Carolina contiguous with Virginia to the Pamlico Sound in New Bern (see Fig. 1). While this drainage basin is largely rural, it does receive nitrogen and phosphorus loadings from industrial, urban, and agricultural activities. Over the last 40 years, this region has experienced a doubling of the population, a near five-fold increase in the number of business establishments, and a 50% increase in the production of its major crops. Estimates of the annual loadings of nitrogen and phosphorus to the Neuse River suggest that these years have seen over a 30% increase in loadings (Stanely, 1988).

The abundance of nutrient loadings in recent years has led to low dissolved oxygen levels, and extensive blue-green algal blooms during the summer months. In 1988, nutrient loadings reached such a level throughout the Neuse River as to warrant a basinwide Nutrient Sensitive Waters (NSW) classification. During the summer of 1995, an unusually high level of precipitation, coupled with two major swine waste spills and an already nutrient-laden river basin resulted in conditions leading to fish kills involving over 11 million fish and huge algal blooms that rendered the Neuse river useless for recreation (Burkholder, 1995). In addition to the nearly anoxic conditions that caused plant and marine life to suffocate, considerable evidence also has been accumulated indicating the presence of toxic dinoflagellates (*Pfiesteria piscicida*), organisms that can kill fish and have caused adverse respiratory health effects on humans under laboratory conditions (Burkholder, 1995).

3.2. *State responses*

The evolution of North Carolina's water quality policy outlined in Table 1 highlights the 'extent of the market' issue, and the increasing role of flexibility in state-level policy design. Pre-1996, the regulated market consisted entirely of point sources. This table indicates that as the planning process evolved voluntary reduction requirements for non-point sources were dropped in favor of technology specific standards. Such a change is logical for two reasons. First, continued reduction requirements from point sources, prior to the 1996 proposed legislation, resulted in non-point sources overtaking point sources as the major contributor to our nation's water quality problems. Second, point sources have been required to reduce their discharges since some of the earliest water quality legislation, and given that we typically observe a positive relationship between marginal costs of reduction and the intensity of reduction requirements, non-point sources are the likely low cost alternative.

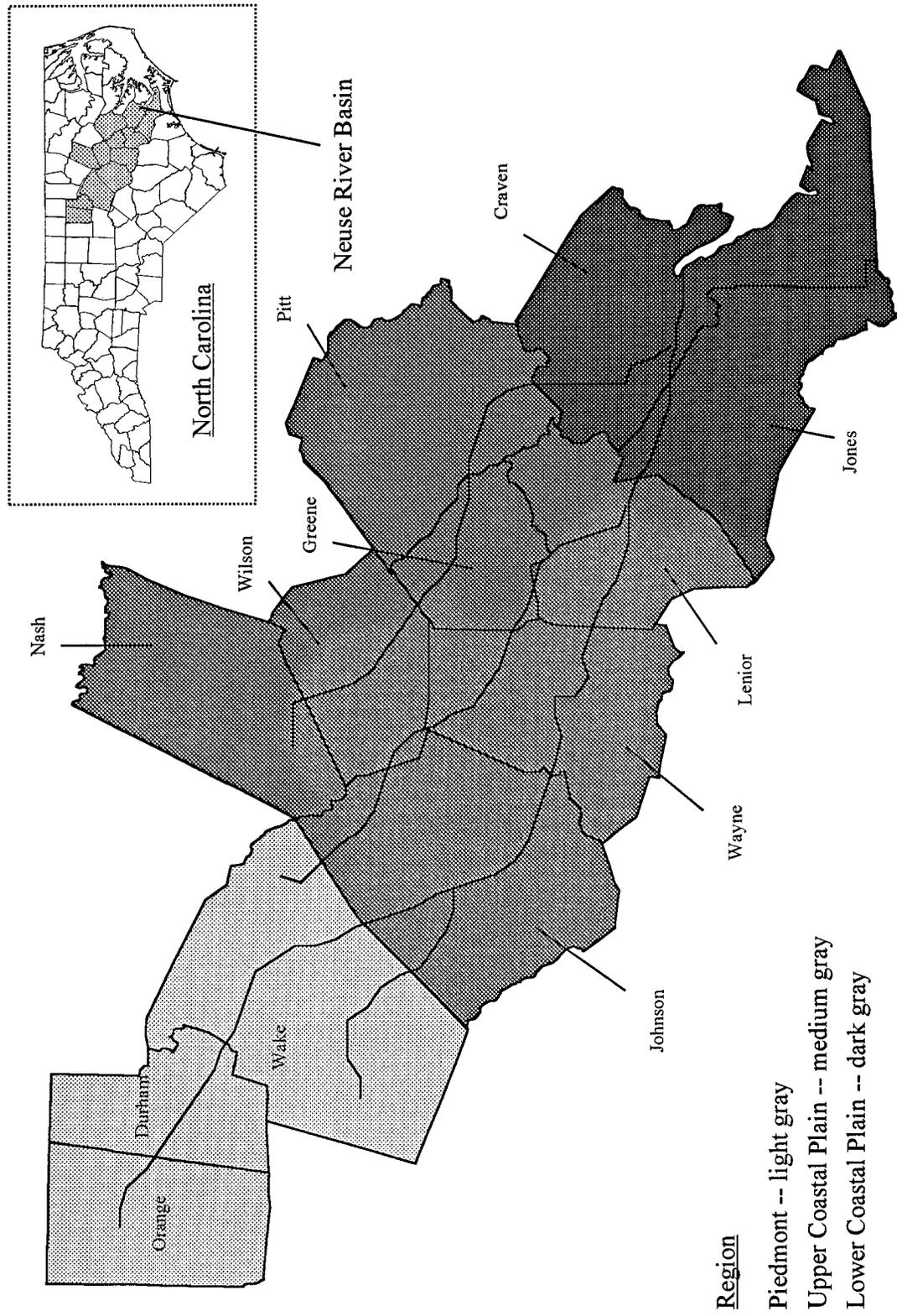


Fig. 1. Counties in the Neuse River Basin.

Table 1
Evolution of (proposed) source nitrogen reduction requirements pre- and post-1996

Source	Pre-1996	May 1996 version	July 1997 version
Requirements for non-point sources	No requirements	Fifty-foot vegetative buffers along all 'blue line' streams	Two alternatives for riparian buffers: (i) three-zone forested buffer, and (ii) vegetative buffer. Provisions are made when other options that achieve equivalent water quality protection are implemented. For instance, buffer width can be reduced if a nutrient management plan OR water management plan is in place and buffer requirements can be eliminated if BOTH nutrient and water management plans are in place
Requirements for point sources	Plant specific requirements. No set basin-wide standards for dischargers.	Either join a coalition and participate in a nutrient trading program to accomplish an overall 30% reduction, OR meet a total nitrogen load based on 6 mg/l and 1995 permitted flows. New and expanding dischargers have to meet a 6 mg/l limit.	Either join a coalition and participate in a nutrient trading program to accomplish an overall 30% reduction, OR meet a total nitrogen load based on 3.5 mg/l and 1995 permitted flows. New and expanding dischargers have to meet a 3.5 mg/l limit.

Table 1 also illustrates two choices confronting state regulatory agencies—(i) who to regulate, and (ii) how to regulate them—both of which will likely influence the distribution and efficiency of control costs. Both the 1996 and 1997 versions allow point sources the opportunity to join coalitions and participate in trading programs. Furthermore, the May 1996 version required non-point sources with land adjacent to 'blue line' streams install or maintain 50 foot buffers.¹¹ As the planning process evolved alternatives to installing buffer strips were added, as is indicated in the July 1997 version.

3.3. Neuse River Estuary Nutrient Analysis Model (NERNAM)

To evaluate alternative management strategies for reducing nutrient loadings in the Neuse River a mathematical programming model was developed that uses a

¹¹ 'Blue line' streams are streams that appear as blue lines on the versions of USGS 1:24 000 scale topography maps in areas of both new development and agriculture.

structural process design and a nutrient balance approach.¹² Production is represented as a set of discrete production activities with unit activity vectors. The constraint set includes limits on input availability, output requirements, quality requirements, and continuity conditions. This model is represented mathematically by the following familiar linear programming problem (for cost minimization):

$$\min \mathbf{c}^T \mathbf{x}_i \quad \text{subject to: } \mathbf{A} \mathbf{x} \leq b \text{ and } \mathbf{x} \geq 0 \quad (4)$$

where \mathbf{x} = vector of activity levels (n by 1), \mathbf{c} = corresponding price vector for activities \mathbf{x} (n by 1), \mathbf{A} = structural process matrix for constraint set (n by n), and b = limits on constraint set (n by 1). The structural process matrix consists of over 731 columns.

The specific approach to assemble the elements in the \mathbf{A} matrix follows the logic developed by Russell (1973) and is illustrated in Table 2. The solution process is comprised of various production and/or residual-influencing blocks, represented by the columns of \mathbf{A} . These are subject to constraints such as input availability or continuity conditions, represented by the rows of \mathbf{A} . The blocks can be separated into the major nutrient influencing activities, represented by the headings in Table 2. They are the activities involving production activities (X), field transport (F), control technologies (T), land transport (L), stream transport (S), and residuals discharge (D). Together, these blocks account for the main influences on the nutrient. The nutrient balance approach is achieved through imposing continuity conditions on the rows of the \mathbf{A} matrix. Each block treats the output from the previous block as an input. This unit input is subject to each block's residual-influencing process while maintaining continuity (i.e., mass balance). The residual is accounted for explicitly by treatment, some type of recycling, or through discharge into the environment. Other constraints, such as those associated with input availability, output, and environmental quality may also be imposed through the use of additional rows.

Another attribute of the structural process model is its ability to allow alternative activities for nutrient reduction. As suggested in Table 2, each grower has the option of various production activities, ($X1_i, X2_i$) and various control strategies ($T1_i, T2_i$). Furthermore, heterogeneous characteristics of the environment across growers are captured in two ways. First, by allowing for differences in the uptake and transport coefficients across agents, (F_i, L_i, S_i). Second, by allowing for different unit production coefficients within the production activities vector, or different unit control coefficients within the control strategies vector, given the same technologies.

¹² For a more detailed description of the model, see the work of Schwabe (1996). The structural process model design similar to those used in early RFF process models (Russell, 1973; Russell and Vaughn, 1974) was chosen for its ability to capture the responses of residuals management behavior from direct and indirect influences. It should be emphasized that the focus here, in contrast to the RFF work, is only on how model design affects policy analysis.

Table 2: Structural Process Model

	Production Activities				Field Transport		Control Technologies		Land Transport		Stream Transport		Residuals Discharge		Right-hand Side	
	X1 _a	X2 _a	X1 _b	X2 _b	F _a	F _b	T1 _a	T2 _a	T1 _b	T2 _b	L _a	L _b	S _a	S _b	D ₁ D ₂ D ₃ D ₄	B
Yield	120	115	130	120												= Bu.
input 1	i1 _{x1a}	.33	i1 _{x1b}	i1 _{x2b}												= 37
input 2	i2 _{x2a}	.60	i2 _{x1b}	i2 _{x2b}												= 69
...												= ...
nitrogen	n _{x1a}	1.2	n _{x1b}	n _{x2b}												= 138
...												= ...
input i	i _{x1a}	.5	i _{x1b}	i _{x2b}												= 57.5
uptake	u _{x1a}	.8	u _{x1b}	u _{x2b}												≥ 0
potential	1-u _{x1a}	.2	1-u _{x1b}	1-u _{x2b}	1	1										≥ 0
F reduce					.4	f _b									1	≥ 0
F delivery					.6	1-f _b	1	1	1	1						≥ 0
C reduce							c _{t1a}	.7	c _{t1b}	c _{t2b}					1	≥ 0
C delivery							1-c _{t1a}	.3	1-c _{t1b}	1-c _{t2b}	1	1				≥ 0
L reduce										.5	e _b				1	≥ 0
L delivery										.5	1-e _b	1	1			≥ 0
S reduce												.3	s _b		1	≥ 0
S delivery												.7	1-s _b			Q

Obj. Fcn. cX1_a cX2_a cX1_b cX2_b cT1_a cT2_a cT1_b cT2_b * * * * * * * * * * *

where: uptake-plant requirement; potential - potential nutrient after accounting for uptake; F reduce - reduction from field independent of control technology; F delivery -potential nutrient runoff independent of control technology; C reduce - reduction from control technology; C delivery - runoff after accounting for control technology; L reduce - reduction from transport; S reduce - reduction from stream transport; S delivery - nutrient reaching the estuary; Q - water quality constraint; cX's are vectors representing the unit costs per activity; * - indicate that effluent charges (prices) may be applied to the transport activities. Note - the numbers presented above correspond to example given in text. In this example, X2_a represents the production activity of producing corn under conventional tillage for grower a, T2_a represents the controlled drainage, the other blocks being defined above.

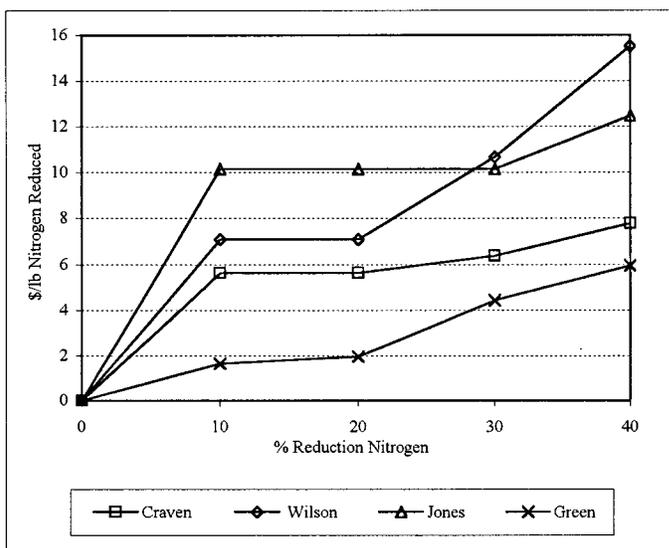
An example may help to illustrate this approach, albeit somewhat abridged. Suppose farm *a* chooses production activity $X2_a$ (e.g., corn with conventional till) and control technology $T2_a$ (e.g., water control structures). To produce 115 bushels of corn, farmer *a* applies 138 pounds of nitrogen. The inputs are specified in units per bushel. Multiplying 115 times 1.2 gives the required nitrogen. At maturity, suppose every bushel of corn requires 0.8 pounds of nitrogen. Thus, 92 pounds are taken up by the plant, thereby leaving 46 pounds of nitrogen as potential field runoff. We will call this section of the matrix the first block. The residual input into this block is 138 pounds of nitrogen, while the residual output from this block is 46 pounds. This 46 pounds of nitrogen is now the input into the next block, field transport. For each pound of nitrogen on the field, 60% has the potential to exit as runoff, the remaining being lost to groundwater recharge or denitrification. The input into the control technology block is 27.6 pounds, of which technology $T2_b$ reduces 70% allowing 8.28 pounds to traverse the field to the stream. During this transport, 50% is reduced and 50%, or 4.14 pounds, reaches the stream edge.

Now consider the last block, stream transport, considers the physical and biological decay that nutrients undergo from the time they enter the stream until the time they reach the estuary. Furthermore, assume that for every pound of nitrogen entering the stream, only 70% reach the estuary. As mentioned above, the input into this block is 4.14 pounds, the emerging nutrient delivery output, 2.9. If the constraint on nutrient delivery is 3 pounds, then the farmer need invoke no other strategy. The input availability requirements and added constraints are listed in column B. To summarize, the columns represent the blocks—production activities, control strategies, and transport media—while the rows represent input requirements and continuity conditions. Finally, Table 2 assumes the objective function from the above example is $(c_{x2b}X2_b + c_{t2b}T2_b)$, given that there are no fee associated with residual discharge.

NERNAM includes three major sources of nutrient loadings—cropping activities, wastewater treatment plants (WWTP), and swine operations. Cropping activities include growing corn, cotton, and soybeans. These three crops account for approximately 70% of the total planted acreage in the basin. Counties are treated as multi-product farms with each county's acreage aggregated by crop. Acreage production costs and crop yield differ by region—the Piedmont, the Upper Coastal Plain, and the Lower Coastal Plain.¹³ The natural differences in soil types and average farm size across these regions, both of which influence production costs

¹³ A budget generator is used to estimate the costs of each cropping system. The budget generator is an accounting program that generates estimates of variable and fixed costs per acre of cropland. The input data and expected yield estimates are based on recommendations from crop science specialists. This particular budget generator, developed at Oklahoma State University, is used by the NCSU Cooperative Extension Service to aid growers in recommended inputs and expected costs per acre.

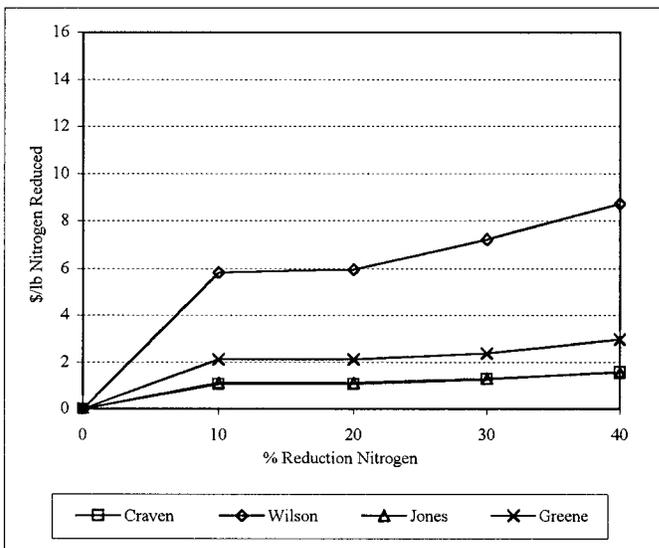
Table 3
Unit costs for selected counties under alternative levels of environmental aggregation^a



Nutrient				
Reduction	Craven	Wilson	Jones	Greene
10%	\$ 5.63	\$ 7.09	\$ 10.16	\$ 1.64
20%	\$ 5.63	\$ 7.09	\$ 10.16	\$ 1.96
30%	\$ 6.37	\$ 10.66	\$ 10.16	\$ 4.44
40%	\$ 7.79	\$ 15.53	\$ 12.48	\$ 5.95

(i) Accounting for Differences in Soil Across Counties

Table 3 (continued)

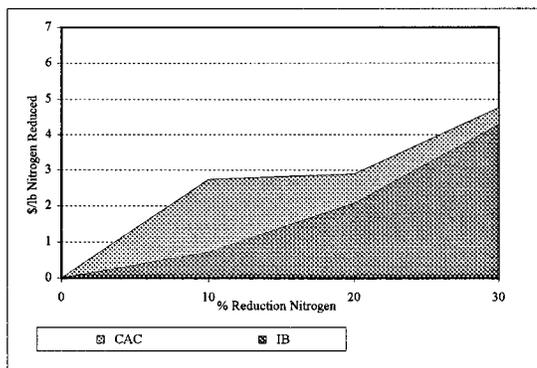


Nutrient				
Reduction	Craven	Wilson	Jones	Greene
10%	\$1.06	\$5.83	\$1.12	\$2.12
20%	\$1.06	\$5.96	\$1.12	\$2.12
30%	\$1.26	\$7.23	\$1.30	\$2.38
40%	\$1.58	\$8.73	\$1.56	\$2.97

(ii) Homogenizing Soil Across Sample Counties

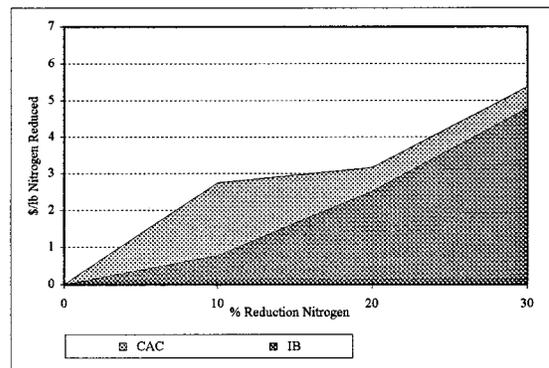
^aReductions are achieved by requiring each county reduce estuarine loadings by requirements as measured from baseline estuarine loadings.

Table 4
Basinwide unit costs under alternative markets and loading responsibility: CAC vs. IB



% Reduction	Unit Costs	
	CAC	IB
10	\$2.74	\$0.72
20	\$2.89	\$2.09
30	\$4.76	\$4.29

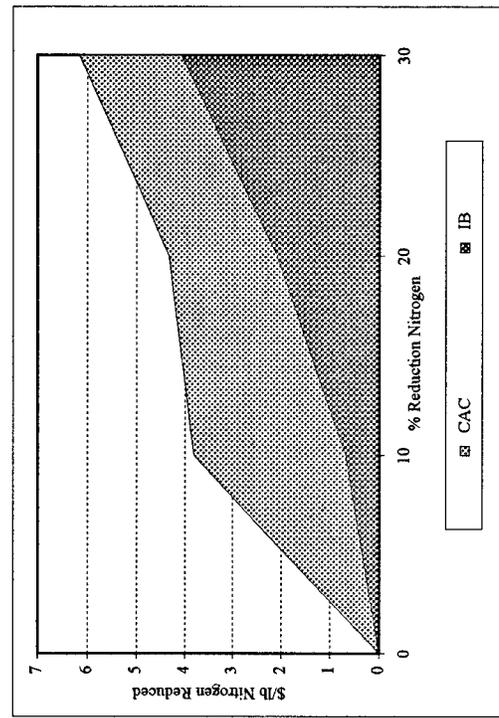
(i) Cropping Activities



% Reduction	Unit Costs	
	CAC	IB
10	\$2.74	\$0.78
20	\$3.17	\$2.52
30	\$5.37	\$4.79

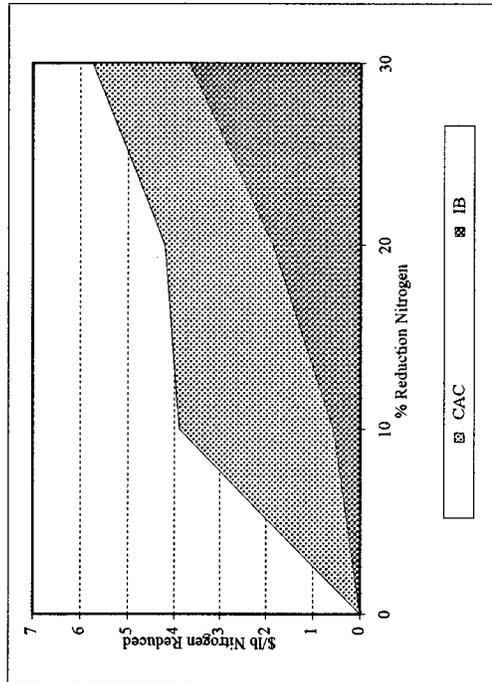
(ii) Cropping Activities and Swine Loadings

Table 4 (continued)



% Reduction	Unit Costs	
	CAC	IB
10	\$3.81	\$0.69
20	\$4.33	\$2.08
30	\$6.15	\$4.07

(iv) Complete Model



% Reduction	Unit Costs	
	CAC	IB
10	\$3.88	\$0.58
20	\$4.21	\$1.87
30	\$5.74	\$3.67

(iii) Cropping Activities and WWTP

Baseline Solutions		Control Costs for 30% Reduction: CAC (dollars)	
	Total Baseline Cost (dollars)	Total Baseline Nitrogen (pounds)	30% Reduction: CAC (dollars)
(i) Cropping Activities Model	\$157,272,876	3,623,590	\$5,176,000
(ii) Cropping Activities and Swine Loadings	\$157,272,876	3,965,403	\$6,392,000
(iii) Cropping Activities and WWTP	\$157,272,876	4,642,714	\$7,991,000
(iv) Complete Model	\$157,272,876	4,984,527	\$9,206,000

and residual generation, support this grouping. Since characteristics of the soil influence nutrient runoff, three land indices are developed based on characteristics of each county's soil and are used to adjust potential runoff estimates. Three control practices are available to reduce nutrient runoff—controlled drainage, conservation tillage, and vegetative filter strips. A control technology's ability to reduce runoff also depends on characteristics of the land, and thus each technology's unit cost and reduction effectiveness estimates are adjusted by the three land indices.¹⁴

The model includes 18 WWTPs. Secondary and tertiary treatment options are available to each plant where reductions are feasible. Secondary treatment achieves end-of-pipe nitrogen concentration limits of 6 mg/l, while tertiary treatment achieves nitrogen concentration limits of 3.5 mg/l. Finally, nitrogen loadings from swine operations are included by their location without control options. The role of swine operations in this research is to account for background pollution.

To evaluate how the costs of nutrient reduction and, in turn, the evaluation of policy is impacted by decisions concerning the extent of the market and background residuals, four submodels are developed. The first, submodel A, serves as a baseline and includes only cropping activities within the basin. Each county is treated as a cost-minimizing farm producing corn, cotton, and soybeans under one of two tillage strategies and three alternative nutrient control strategies.¹⁵ Holding output (county crop bushels) constant, control costs are estimated by restricting the level of allowable estuarine nitrogen loadings.¹⁶

Submodel A can be used to make a variety of points. For instance, Table 3 illustrates how the ability to reduce estuarine nitrogen loadings can differ across farms in the Neuse River Basin for a sample of four counties. Unit cost estimates are derived through placing consecutive restrictions (10–40%) on each county's baseline estuarine nitrogen loadings. The differences in the unit control costs across these four counties capture graphically the theoretical differences in shadow prices on the water quality constraints, $\gamma_{a,m}^s - \gamma_{b,m}^s$, presented in Eq. (3). Indeed, the relative differences across each curve provide an indication of the potential efficiency gains at the margin from implementing an IB instrument vs. a CAC strategy. While both graphs focus on cropping activities alone and do not allow for a redistribution of crop output as a means of control, they differ in their treatment

¹⁴ Cost and effectiveness estimates for each control strategy were estimated based on North Carolina Cost-Share Program Best Management Practices cost estimates from 1985–1994, and expert advice from soil scientists and biological engineers at North Carolina State University. See the work by Schwabe (1996) for a more detail.

¹⁵ It should be emphasized that the output constraint for cropping activities limits the viable control options typically available to farms, such as changing output mix. The sensitivity of control costs to this restriction is illustrated by Schwabe (1999).

¹⁶ Baseline nutrient loadings for all models presented here were estimated by running the model with a constraint on output (county crop bushels), yet with no restrictions on estuarine nitrogen loadings.

of the environment. Graph (i) accounts for differences in soil characteristics across counties while graph (ii) does not.¹⁷ Comparisons across graphs suggest that characteristics of the environment can influence both the level of unit control costs and the relative differences in unit control costs.¹⁸

Submodel B includes both cropping activities and WWTP discharge within the basin and evaluates issues surrounding the extent of the regulated market.¹⁹ Submodel C includes cropping activities and swine operations and evaluates the effects of additional loadings from unregulated sources. In this submodel, no additional control options are introduced beyond that available under submodel A. Baseline nitrogen loadings increase with the inclusion of swine loadings, and thus so do the reduction responsibilities of the corn, cotton, or soybeans producers in achieving a 30% reduction. Finally, the complete model includes all three sources of nutrient loadings and evaluates the potential efficiency of alternative policy instruments.

Table 4 provides a simple illustration of how basinwide unit cost and control cost estimates differ under these submodels. Similar to Table 3, costs are estimated with successive restrictions on baseline loadings. The CAC strategy requires each source reduce its baseline estuarine loadings by the pre-specified percentage.²⁰ Alternatively, the IB system places the constraint on total basinwide estuarine loadings, thereby allowing the model to assign reduction responsibility to the least cost abater until the reduction requirement is met. As expected, for any particular submodel the unit costs are greater under the CAC strategy. If we assume that the potential efficiency gains from implementing the IB instrument vs. the CAC strategy can be represented by the difference between the unit cost curves, our treatment of who counts and their responsibility has several implications.²¹ First, comparing solutions under *cropping activities* and *cropping activities and WWTPs* to *cropping activities and swine loadings* and the *complete model*, respectively, we observe a slight increase in unit costs and decrease in potential efficiency gains when sources are given added responsibility for background pollution. Second, when additional nutrient sources with nutrient reduction capabilities are included (i.e., WWTPs), unit costs and potential efficiency gains increase quite dramatically.

¹⁷ The sensitivity of both cost and relative cost savings estimates under alternative spatial assumptions (i.e., transfer coefficients) is illustrated by Schwabe (1998).

¹⁸ For a more detailed analysis of how alternative levels of environmental aggregation influence cost estimates and efficiency gains, see the work of Schwabe (1998).

¹⁹ For WWTPs, only control costs were included. Each plant was treated as a cost-minimizing firm with baseline estuarine nitrogen loading estimates based on their 1995 level.

²⁰ While most CAC strategies we typically observe in actual policy consist of specifying technology requirements, the representation assumed here is common in the environmental economics literature.

²¹ While rental rates on land are not included in these costs, all other capital and equipment costs are amortized over their expected lives to allow us to compare yearly cost estimates.

Table 4 also provides a simple comparison of the control costs for a 30% reduction in nitrogen loadings under each of the submodels. In all four cases, the 30% reduction was achieved under a CAC strategy. Even though the reduction is given in percentage terms suggesting the absolute level of nutrient reduction differs across submodels, these comparisons imply that the costs of reduction can vary greatly depending on the extent and the responsibility of the regulated market. Indeed, the differences in these potential efficiency gains across regulated markets underscores the importance of acknowledging that *who* and *what* we decide ‘counts’ can have serious implications on what we may ultimately define as the efficient solution. Furthermore, the ability to illustrate how costs differ across sources under alternative specifications may be important to interpreting the implications of aggregate goals.

4. Analysis of the Neuse River Basin Plan and other alternatives

We now consider an actual policy decision confronting the state of North Carolina and show how the complete model would have informed the policy process. Under the proposed Neuse River Basin Plan developed by North Carolina’s DWQ, both point and non-point sources are to be regulated.²² Regulated non-point sources include cropping activities, animal operations, stormwater sewage runoff from municipalities, and stormwater runoff from developing land areas. Regulated point sources include the 31 existing industrial and municipal WWTP in the Neuse River Basin with discharge greater than or equal to 0.5 million gallons of discharge per day.

As Table 1 illustrated, the May 1996 version of the draft proposed by DWQ contained three central requirements.²³ To evaluate the costs of achieving these three requirements, the following assumptions are made. Point sources are divided into two groups consisting of the Lower Neuse River Basin Association (LNBA) members and non-LNBA members.²⁴ The LNBA members are required, as a

²² The Neuse River Basin Plan differs from most other point/non-point source plans in that non-point source regulation is mandatory. In contrast, the Tar-Pamlico River Basin, a northern contiguous basin to the Neuse River Basin, has implemented a point/non-point source trading program with voluntary non-point source participation.

²³ At the time of this research, the proposed plan is in draft form and continues to be revised. The version of the plan this research evaluates is the proposed May 23rd, 1996 plan. During a period for which the Neuse River Basin Nutrient Management Plan was open for public comment, and in response to the May 1996 plan, the author, V. Kerry Smith, and James Easley presented the partial system to the North Carolina DWQ in November of 1996. While the July 1997 Version and the Partial System are similar, the July Version includes some combination BMPs contingencies that go beyond the scope of this research.

²⁴ At this time, the LNBA consisted of 11 members.

group, to achieve a 30% reduction in estuarine loadings from their group's aggregate baseline nitrogen loadings. This requirement essentially mimics an IB system. The non-LNBA members are required to implement the secondary treatment technology unless they already achieve a 6 mg/l concentration level. This requirement mimics a CAC approach in that the effluent standard is based on WWTPs implementing at minimum a secondary treatment technology. For cropland owners, a 50-foot buffer on both sides of all perennial and intermittent streams in the Neuse River Basin amounts to implementing approximately 26 000 acres of buffers (North Carolina Division of Water Quality, 1997), clearly a technology-based CAC strategy.

Table 5 lists the costs and nutrient loadings estimated by the model for the May 1996 version of the Neuse River Basin Plan. Also included in Table 5 are the costs of achieving the same percentage reduction in nitrogen loadings under three other regulatory alternatives, including:

- (a) *Pure CAC*—All sources are subject to a CAC strategy.
- (b) *Pure IB*—All sources participate in an IB system.
- (c) *Partial System*—All counties (farms) are subject to a CAC strategy, while all WWTPs participate in an IB system.

Under all of the policy scenarios the baseline loadings are nearly equivalent thus facilitating cost comparisons for the 30% reduction requirement as measured from baseline estuarine loadings. The Pure CAC is the most restrictive. This format is similar to how most past empirical analyses of alternative instruments represent CAC strategies. It should be emphasized that the Pure CAC is not a technology-based standard but rather an effluent-based standard that gives farms the opportunity to implement their least-cost control strategy. While this effluent-based standard is more flexible than the technology-based standard, the effluent-based standard is likely to incur monitoring and enforcement costs similar to that under

Table 5
Costs for 30% nutrient reduction in the Neuse River under alternative policies (1994 \$)^a

Sources	Pure CAC System	Neuse River Plan	Partial System	Pure IB System
Non-point Source	US\$6,391,660	US\$6,924,211	US\$6,391,660	US\$3,912,900
Total				
Point Source Total	US\$2,814,908	US\$2,153,197	US\$807,304	US\$2,167,024
Basinwide Total ^b	US\$9,206,568	US\$9,077,408 (99%)	US\$7,198,964 (78%)	US\$6,079,924 (66%)

^aThese are annual costs and do not include land rents. All solutions are estimated holding each county's output of corn, cotton, and soybeans constant. Swine loadings are included in each solution, with the additional loading reduction responsibility under the Pure CAC strategy and partial system assigned to cropping activities. Total estuarine loadings under each policy scenario are essentially equivalent thereby facilitating these comparisons.

^bThe number in parentheses are the control costs as a % of the CAC control costs.

an IB system.²⁵ On the opposite end of the spectrum is the Pure IB. This is the alternative most models provide to CAC strategies when comparing various policies for pollution control. In between these two extremes are the actual plan and the Partial System (which is very similar to the July 1997 version in Table 1). The CAC strategy under the Partial System requires all agents meet the effluent standard (measured at the estuary), but allows the use of control options in addition vegetative filter strips.

Evaluating the actual plan and Partial System provide more informative and realistic estimates than either the Pure CAC or Pure IB. Cost comparisons suggest that the Neuse River Basin Plan achieves approximately US\$130 000 in potential efficiency gains over the Pure CAC. These gains arise because some point sources, namely LNBA members, are allowed to participate in an IB system. This system permits a low cost agent to assume some of the reduction responsibility for a high cost agent. Yet compared with the Pure IB, the Neuse River Basin Plan experiences a potential loss in efficiency of approximately US\$3 million annually. These losses in efficiency result from an across county uniform standard requiring non-point sources implement vegetative filter strips. Such a solution ignores the county-to-county variations in run-off due to soil, topographical, and environmental conditions that differentially affect the unit cost and reduction effectiveness of the various control strategies.

Finally, the Partial System that subjects county-level farms to a (effluent-based) CAC strategy and allows all point sources the opportunity to participate in an IB system realizes approximately US\$1.9 million annually in potential efficiency gains over the Neuse River Basin Plan. While there are gains from allowing counties the flexibility to choose their management strategy (thereby eliminating ineffective control measures), the greatest gains arise from allowing all point sources the opportunity to participate in a pure IB system.²⁶ In each scenario the control costs for points sources are less than those for non-point sources, yet the relative differences can range from US\$1.7 million (Pure IB) to US\$5.6 million (Partial). Point source responsibility as a percentage of control costs range from 36% (Pure IB) to 11% (Partial).

²⁵ Due to the paucity of data, transactions costs including monitoring and enforcement costs were not estimated. Research by Hahn (1989), Hahn and Hester (1989), Tietenberg (1990), and Stavins (1995) all suggest that transactions costs can limit the actual cost-effectiveness of IB systems. Research by Kerr and Mare (1997) suggests transactions costs can produce efficiency losses of approximately 10%.

²⁶ Whether the Partial System is more cost effective than the Neuse River Plan depends on the magnitude of transaction costs incurred under each strategy. For point sources, it is likely that the transaction cost differences under each strategy will not differ greatly. For counties, and under a CAC strategy, transaction cost differences between these two alternatives will arise from differences in the costs of monitoring whether growers implement the required vegetative buffer strips (the Neuse River Plan), or the required strategies under the Partial System.

Table 6
Costs of a 30% reduction in estuarine nutrient loadings by county under alternative policy instruments (per unit costs in parentheses)^a

County	Neuse River Basin (1994 \$)	Pure CAC System (1994 \$)	Pure IB System (1994 \$)	Partial System (1994 \$)
Orange	US\$85,800 (US\$312.73)	US\$121,441 (US\$174.54)	--	US\$121,441 (US\$174.54)
Durham	US\$5,850 (US\$84.87)	US\$8,520 (US\$92.58)	--	US\$8,520 (US\$92.68)
Wake	US\$1,941,425 (US\$7.36)	US\$1,333,088 (US\$8.08)	US\$1,841,028 (US\$6.32)	US\$781,840 (US\$5.74)
Johnston	US\$1,019,081 (US\$14.82)	US\$930,676 (US\$6.66)	US\$436,101 (US\$3.90)	US\$728,380 (US\$5.77)
Nash	US\$482,890 (US\$27.09)	US\$239,960 (US\$7.41)	US\$180,670 (US\$5.99)	US\$239,960 (US\$7.41)
Wilson	US\$692,643 (US\$18.60)	US\$691,937 (US\$12.37)	US\$213,600 (US\$7.09)	US\$436,740 (US\$11.63)
Wayne	US\$1,061,416 (US\$3.17)	US\$951,698 (US\$3.32)	US\$564,126 (US\$1.94)	US\$964,206 (US\$2.92)
Greene	US\$590,570 (US\$3.87)	US\$724,420 (US\$5.11)	US\$833,440 (US\$5.43)	US\$724,420 (US\$5.11)
Lenoir	US\$1,001,137 (US\$4.62)	US\$2,102,315 (US\$8.47)	US\$1,313,460 (US\$4.73)	US\$1,060,228 (US\$4.30)
Pitt	US\$962,860 (US\$4.67)	US\$504,870 (US\$2.49)	US\$9,370 (US\$0.06)	US\$504,870 (US\$2.49)
Jones	US\$556,570 (US\$8.14)	US\$671,580 (US\$11.43)	--	US\$671,580 (US\$11.43)
Craven	US\$677,166 (US\$5.64)	US\$926,063 (US\$5.69)	US\$688,129 (US\$4.35)	US\$956,779 (US\$5.26)

^a Per unit costs are in \$ per pound of nitrogen reduction measured at the estuary.

Whether these outcomes over-estimate or under-estimate the actual outcome depend on a number of issues, one largely being the assumption of zero transactions costs. Obviously, this assumption alone likely leads to an overestimate of the potential efficiency gains from the more flexible IB instruments when compared with the technology-based CAC strategy. Yet, if the estimated cost savings are large, the likelihood that transaction costs render the IB instrument the less cost-effective strategy diminishes. Furthermore, since the partial strategy regulates non-point sources with a CAC strategy, albeit an effluent-based standard, transaction costs are greatly reduced.²⁷

Table 6 highlights another element in the design of water quality models for policy usefulness—the ability to acknowledge spatial detail. The control costs of achieving a 30% reduction for each county are presented along with the county-specific control costs for point sources. The distribution of costs varies quite extensively, ranging across counties with a minimum variation of US\$2670 (Durham County) to a maximum variation of US\$1.16 million (Wake County). Efficiency arguments aside, this suggests that alternative policies will impact some communities more than others. An obvious concern raised from these distributional implications is their potential impact on small communities—communities that may not have the tax base to support or the expertise to manage such programs. Indeed, this is a central issue in the current water legislation dealing with the transition from federal grant's based assistance to a program of grants to capitalize State Water Pollution Control Revolving Funds.²⁸ These comparisons illustrate both the different efficiency and distributional impacts possible under alternative policies. It should be noted, though, that in addition to influencing the distribution of costs across countries, the choice of policy instrument will likely influence the distribution of the benefits as well. These benefits will predominately fall on the downstream counties (Smith et al., 1998).

Table 6 also presents the cost per pound of nitrogen reduction by county. As expected, the cost per pound varies quite dramatically across counties. On average, the cost per unit across counties is lowest under the pure IB system (US\$39.81), followed by the partial system (US\$35.40), the pure CAC (28.18) and finally the Neuse River Basin Plan (US\$41.30). In terms of the median cost per county, the

²⁷ A characteristic of the model setup that would suggest that the outcome could underestimate the potential efficiency gains from the IB system is that trading only occurs across counties, not within counties.

²⁸ Implicit in Table 6 is the assumption that the citizens and firms within that county absorb all costs generated within a county, whether public or private. While this is likely for publicly owned treatment works, private compliance costs are more likely be borne by the firms affected, and then passed on to their customers, employees, and stockholders, any of which may or may not be county residents. The distributional effects still arise, though, if we were to simply account for publicly owned treatment works.

IB system has the lowest (US\$4.12), followed by the partial system (US\$5.76), with both the CAC strategy and the Neuse River Basin Plan having a median cost per unit across counties of US\$7.75. Finally, differences in the standard deviation across these counties by plan are quite dramatic. The standard deviation of unit costs across counties under the IB system is US\$2.76, while under both the partial system and the pure CAC very similar—US\$52.67 and US\$52.31, respectively. The standard deviation under the Neuse River Basin Plan is greatest among the plans at US\$88.45. These estimates emphasize the importance of acknowledging differences across agents in their ability to reduce nutrient loadings. In particular, those strategies that do acknowledge differences are likely to achieve the required reductions with lower costs at the margin than those that do not. Furthermore, and as illustrated above, the more rigid the strategy the more likely the unit costs of control will differ across agents.

5. Conclusion

Water quality policy is presenting policy makers with a greater variety of instruments for implementation and an expanding ‘market’ to potentially regulate. With this in mind, three general conclusions can be drawn from the comparisons above. First, as water quality policy becomes more flexible, models need to be more flexible in terms of acknowledging all the relevant sources that contribute to the problem, as well as evaluate more realistic and viable policy alternatives than the traditional pure IB or CAC strategy. Second, as policy focuses more on non-point sources for pollution reduction, the greater is the need to capture the detail within the environment that may influence control cost differences. Finally, mixed strategies that more closely mimic actual management strategies can achieve appreciable cost savings compared to the traditional CAC approach.

Concern over nutrient control in the Neuse River Basin provides a nice application for illustrating both these conclusions and the importance of model detail and flexibility to informing current policy. Issues confronting DWQ included what sources to regulate; what controls should be required; the reduction responsibility of each source; and if a trading system is allowed, who should be permitted to participate. The May 1996 proposed Neuse River Basin plan targeted WWTPs and cropping activities as the major sources of reduction. The plan implemented a CAC style standard requiring all farms install buffer strips. It specified that WWTPs in the LNBA were allowed to participate in a trading system, while non-members were to meet technology-based effluent standards.

Using *NERNAM* to evaluate a sequence of plans that were intended to achieve the desired level of nutrient reduction provided both expected and surprising results. Of the plans evaluated the Pure IB system, represented by the least cost solution, was the most efficient, followed by the Partial System, the proposed plan,

and then the Pure CAC strategy. While the efficiency ordering of the Pure IB system, the proposed plan, and the Pure CAC strategy was expected, the degree of inefficiency of the proposed plan was surprising—a mere 1% cost savings vs. the Pure CAC strategy. The model structure allowed us to identify how the distribution of costs varied across these two instruments, indicating that non-point source costs increased under the proposed plan and point source costs decreased. These patterns captured two shortcomings of the proposed plan. First, requiring all non-point sources install buffer strips overlooks differences in local conditions that influence the effectiveness of buffer strips. This requirement is very inefficient, and hence, costly. A less inefficient strategy, as suggested by a comparison with the mixed system, is to allow each source the option to choose among buffer strips, conservation tillage, or controlled drainage in order to meet the required level of reduction. Second, limiting participation in trading schemes limits the potential efficiency of this IB system, as illustrated by comparisons with the Pure System, as well as the Partial System.

Federal water quality policy is increasingly allowing states more flexibility in designing strategies to achieve the federal standards. This research presents some modest findings that illustrate that the costs of achieving those standards can vary appreciably by altering the mix of who is regulated, their reduction responsibility, and finally, how they are regulated. Additionally, policies that target both point and non-point sources and include a mixed strategy approach can offer regulators a useful and potentially attractive option to strategies that focus on sources separately and/or use pure systems.

Acknowledgements

The author would like to thank James E. Easley, V. Kerry Smith, and Harold Winter for helpful comments on earlier drafts of this paper. This research was partially supported by the North Carolina Agricultural Research Service at North Carolina State University.

References

- Ayres, R.U., Kneese, A.V., 1969. Production, consumption, and externalities. *American Economic Review* 59, 282–297.
- Burkholder, J., 1995. North Carolina Nutrient Summit, Raleigh, NC.
- Congressional Research Service, 1996. Water Quality: Implementing the Clean Water Act, January 17.
- Dales, J., 1968. *Pollution, Property, and Prices*. University of Toronto, Toronto.
- Hahn, R.W., 1989. Economic prescriptions for environmental problems: how the patient followed the doctor's orders. *The Journal of Economic Perspectives* 3 (2), 95–114.

- Hahn, R.W., Hester, G.L., 1989. Marketable permits: lessons for theory and practice. *Ecology Law Quarterly* 16, 361–406.
- Helfand, G.E., House, B.W., 1995. Regulating nonpoint source pollution under heterogenous conditions. *American Journal of Agricultural Economics* 77, 1024–1032.
- Johnson, S.L., Adams, R.W., Perry, G.M., 1991. The on-farm costs of reducing groundwater pollution. *American Journal of Agricultural Economics* 73, 1063–1073.
- Kerr, S., Mare, D., 1997. Transactions Costs and Tradeable Permits Markets: The United States Lead Phasedown, Eighth Annual Conference of AERE, Tilburg, Netherlands, June.
- Kling, C.L., 1994. Emission trading vs. rigid regulation in the control of vehicle emissions. *Land Economics* 70, 174–188.
- Kneese, A.V., Schultze, C.L., 1975. *Pollution, Prices, and Public Policy*. The Brookings Institute, Washington, DC.
- Kwerel, E., 1976. To tell the truth: imperfect information and optimal pollution control. *Review of Economic and Statistics*, August, 595–602.
- Mapp, H.P., Bernardo, D.S., Sabbagh, G.J., Geleta, S., Watkins, K.B., 1994. Economics and environmental impacts of limiting nitrogen use to protect water quality: A stochastic regional analysis. *American Journal of Agricultural Economics* 76, 889–903.
- Montgomery, W.D., 1972. Markets in licenses and efficient pollution control programs. *Journal of Economic Theory* 5, 395–418.
- North Carolina Division of Water Quality, 1997. Neuse River Nutrient Sensitive Waters Management Strategy. Draft Proposal, July.
- Rubin, J., Kling, C., 1993. An emission saved is an emission earned: an empirical study of emission banking for light-duty vehicle manufacturers. *Journal of Environmental Economics and Management* 25, 257–274.
- Russell, C.S., 1973. *Residuals Management In Industry*. John Hopkins Univ. Press, Baltimore, MD.
- Russell, C.S., Vaughn, W.J., 1974. A linear programming model of residuals management for integrated iron and steel production. *Journal of Environmental Economics and Management* 1, 17–42.
- Schwabe, K.A., 1996. *Source Heterogeneity, Resource Characteristics, and the Performance of Marketable Permits*. Unpublished PhD thesis, North Carolina State University, Raleigh, NC.
- Schwabe, K.A., 1998. *The Effects of Output Specification and Heterogeneity on Modeling Nonpoint Source Pollution Regulation*. Unpublished paper under revision, Ohio University Working Paper Series.
- Schwabe, K.A., 1999. *The Effects of Separability on Incentive-Based Instrument Performance*. *Economics Letters*, forthcoming.
- Schwabe, K.A., Smith, V.K., 1998. *Have Incentive Based Policies Been Oversold? The Case of Non-point Source Pollution in the Neuse River Basin*. Unpublished paper under revision, Center for Environmental and Resource Economics, Duke University.
- Smith, V.K., Schwabe, K.A., Mansfield, C., 1998. Does nature limit environmental federalism? In: Schwab, R. (Ed.), *Environmental Economics and Public Policy: Essays in Honor of Wallace E. Oates*. Edward Elgar, Cheltenham, UK.
- Spofford Jr., W.O., Russell, C.S., Kelly, R., 1975. Operational problems in large scale residuals management models. In: Mills, E.S., National Bureau of Economic Research, *Economic Analysis Of Environmental Problems*. Columbia Univ. Press, New York.
- Stanely, D.W., 1988. Historical Trends In Nutrient loading to the Neuse River Estuary, North Carolina. In: Lyke, W.L., Hoban, T.J. (Eds.), *Proc. Symp. Coastal Water Resources*. AWRA Technical Publication Series TPS-88-1. American Water Resources Association, Bethesda, MD.
- Stavins, R.N., 1995. Transactions costs and tradable permits. *Journal of Environmental Economics and Management* 29, 133–148.
- Tietenberg, T.H., 1973. Specific taxes and the control of pollution: A general equilibrium analysis. *The Quarterly Journal of Economics* 87, 503–522.

- Tietenberg, T.H., 1985. Emissions Trading: An Exercise In Reforming Pollution Policy. Resources for the Future, Wash., DC.
- Tietenberg, T.H., 1990. Economic instruments for environmental regulation. *Oxford Review of Economic Policy* 6, 17–33.
- Weinberg, M., Kling, C., Wilen, J., 1993. Water markets and water quality. *American Journal of Agricultural Economics* 75, 278–291.