Cost-Effective NO_x Control in the Eastern United States

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with Matt Cannon, Terrell Stoessell, and
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Resources for the Future
1616 P Street, NW
Washington, D.C. 20036
Telephone: 202–328–5000
Fax: 202–939–3460
Internet: http://www.rff.org

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Abstract

Reducing nitrogen oxide (NO_x) emissions in the eastern United States has become the focus of efforts to meet ozone air quality goals and will be useful for reducing particulate matter (PM) concentrations in the future. This paper addresses many aspects of the debate over the appropriate approach for obtaining reductions in NO_x emissions from point sources beyond those called for in the Clean Air Act Amendments of 1990. Data on NO_x control technologies and their associated costs, spatial models linking NO_x emissions and air quality, and benefit estimates of the health effects of changes in ozone and PM concentrations are combined to allow an analysis of alternative policies in thirteen states in the eastern United States. The first part of the study examines the cost and other consequences of a command-and-control approach embodied in the Environmental Protection Agency's (EPA) NO_x SIP call, which envisions large reductions in NO_x from electric utilities and other point sources. These results are compared to the alternative policy of ton-for-ton NO_x emissions trading, similar to that proposed by the EPA for utilities. We find that emission reduction targets can be met at roughly 50% cost savings under a trading program when there are no transaction costs.

The paper examines a number of alternative economic incentive policies that have the potential to improve upon the utility NO_x trading plan proposed by EPA, including incorporation of other point sources in the trading program, incorporation of *ancillary* PM benefits to ozone reductions in the trading program, and trading on the basis of ozone exposures that incorporates the spatial impact of emissions on ozone levels. For the latter analysis, we examine spatially differentiated permit systems for reducing ozone exposures under different and uncertain meteorological conditions, including an empirical analysis of the trade-off between the reliability (or degree of certainty) of meeting ozone exposure reduction targets and the cost of NO_x control. Finally, several policies that combine costs and health benefits from both ozone and PM reductions are compared to command-and-control and single-pollutant trading policies. The first of these is a full multipollutant trading system that achieves a health benefit goal, with the interpollutant trading ratios governed by the ratio of unit health benefits of ozone and PM. Then, a model that maximizes aggregate benefits from both ozone and PM exposure reductions net of the costs of NO_x controls is estimated.

EPA's program appears to be reasonably cost-effective compared to all of the other more complex trading programs we examined. It may even be considered an optimal policy that maximizes net aggregate benefits if the high estimate of benefits is used in which mortality risk is linked to ozone exposure. Without this controversial assumption, however, we find that EPA's NO_x reduction target is far too large.

Acknowledgements

The authors would like to acknowledge Erica Leich and her staff at E. H. Pechan Associates for providing much of the data on abatement options, cost, and emissions for this project. We would also like to thank Paul Guthrie and his staff at SAI for their work in providing source-receptor coefficients linking NO_x emissions to ozone and particulate concentrations and to Paul for his sound judgment and counsel on the numerous assumptions undergirding this analysis. We would like to acknowledge our sponsors: EPA's Office of Policy, Planning, and Evaluation, particularly Willard Smith and also Bob Noland, as well as EPA's Chesapeake Bay Program Office's Air Subcommittee. Finally, we thank Terry Dinan of the Congressional Budget Office for helpful comments at the AERE Workshop on Market-Based Instruments for Environmental Protection, in Boston, July 18-20, 1999.

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I. Introduction

Reducing nitrogen oxide (NO_x) emissions in the eastern United States has become the focus of efforts to meet ambient air quality goals for ozone. The Clean Air Act (CAA) Amendments of 1990 initiated a number of emissions-reduction policies designed to meet the ambient ozone standard.[2] Beyond this, several regional stakeholder groups, as well as the U.S. Environmental Protection Agency (EPA), called for large further cuts in NO_x emissions from electric utilities and for the regional trading of NO_x emissions to obtain these cuts cost-effectively.[3] A recent tightening of the ambient ozone standard served to put even more emphasis on NO_x reductions accompanied by an NO_x trading program.[4]

Receiving less policy attention have been the reduction in concentrations of fine particulate matter (PM) that accompany reduced NO_x emissions. This lack of attention is due partly to the small fraction of total PM made up of nitrates (which are created from NO_x emissions), particularly in the eastern United States, and partly to the lack of monitoring for fine PM, of which nitrates are a part. This situation is changing, however. EPA's recent setting of a fine particulate National Ambient Air Quality Standard (NAAQS) and the monitoring to go along with it has served to draw added attention to NO_x emissions-reduction strategies and their impact on nitrates and, therefore, fine PM.[5]

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[1] The authors would like to acknowledge support for this research from U.S. EPA's Policy Office and the Chesapeake Bay Program. The authors are affiliated with Resources for the Future. Virginia McConnell is also professor of economics at the University of Maryland, Baltimore.

[2] NO_x and volatile organic compounds are both precursors to the formation of ambient ozone. The primary National Ambient Air Quality Standard for ozone is not being met in many eastern U.S. states.

[3] The stakeholder groups include the Ozone Transport Commission (OTC) and the Ozone Transport Assessment Group (OTAG). These deliberations led to EPA's "NO_x SIP Call" (U.S. EPA 1997a) and the "Proposed NO_x Trading Guidelines" (U.S. EPA 1998).

[4] On May 14, 1999, the District of Columbia Court of Appeals remanded the new ozone standard and the fine particulate standard to EPA for another try. Because EPA's NO_x SIP call had been tied to the new ozone standard, at the end of May the same court stayed EPA's NO_x SIP call. On June 14, 1999, EPA announced its plans to issue a new plan for reducing NO_x. The new plan attempted to address issues cited in lawsuits filed by many northeastern states, who claimed that they could not attain the ozone standards without NO_x reductions in neighboring states (www.epa.gov/airlinks).

[5] In addition, EPA's Federal Advisory Act Subcommittee on Implementation of the Ozone and PM NAAQS addressed squarely the integration of policies to address these two pollutants (EPA 1998).

This paper addresses many aspects of the debate over the appropriate approach for obtaining cuts in NO_x emissions in the eastern United States, beyond those called for in the Clean Air Act Amendments of 1990. We start by examining the cost and other consequences of a command-and-control approach embodied in EPA's NO_x SIP call, which envisions large reductions in NO_x from electric utilities and other point sources. We then examine the consequences of NO_x trading, as proposed in the ton-for-ton emissions trading plan suggested by EPA. One difference between our approach and EPA's is that we focus on a 12-state (plus the District of Columbia) region, while EPA's plan covered 22 states.[6] However, the 12-state plan appears to be more in line with EPA's recent thinking about how to design an East Coast NO_x trading program (see www.epa.gov/airlinks).

Our paper examines alternative policies that may improve upon the NO_x trading plan for utilities proposed by EPA. First, EPA limited its plan to utilities. We search for greater efficiencies by expanding the program to other point sources.

Second, because ozone is a spatially variable pollutant, it is likely that a system of spatially differentiated trading, where permits to emit a ton are traded on the basis of their ozone impacts, would outperform a simple ton-for-ton trading system. We examine spatially differentiated permit systems for reducing ozone levels under different and uncertain meteorological conditions.[7]

Third, ozone creation is a stochastic process, depending fundamentally on weather patterns and conditions. We take these factors into account by empirically characterizing the trade-off between the reliability (or degree of certainty) of meeting ozone reduction targets and the cost of NO_x control.

Fourth, in controlling NO_x, both fine PM and ozone are reduced. We examine the consequences of treating PM reductions as *ancillary* to ozone reductions (that is, as a "free" health benefit of an ozone reduction policy) and of netting these benefits against costs in the search for a cost-effective NO_x reduction allocation to meet NO_x reductions or ozone-exposure reduction targets.

Finally, in several more integrated analyses, we first estimate the allocation of NO_x reduction for a multipollutant trading system, with the interpollutant trading ratios governed by the ratio of unit health benefits of ozone and fine PM. We then estimate the degree (and allocation) of NO_x reductions that maximize the net joint benefits of such reductions.

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[6] The 22-state plan originally envisioned by EPA has been analyzed by Dorris and others (1999). That analysis looks at NO_x emissions and ozone changes under several different regulatory and trading policies. It includes neither estimates of dollar benefits for ozone and PM reductions, nor an analysis of the costs of uncertainty, which our study does.

[7] It is important to note that the analysis of alternative trading or least-cost policies rely on available engineering data about the costs of possible alternative technologies for utilities and other point sources. One advantage of an NO_x trading policy, however, is that it would induce sources to search for the least costly options, some of which may include fuel switching for electric utilities and new technologies that may not be part of the decision set included in this analysis. To the extent that this is true, the cost savings presented here would be an underestimate of the savings that would be obtained under actual implementation.

II. Theoretical Issues

In this section, we briefly outline the various policy designs in general terms. In our model, control of NO_x emissions affects two different air pollutants: ambient ozone and PM levels.[8] Efficient spatial control of NO_x emissions when all benefits and costs are known requires that net benefits be maximized:

$$\max_{r_j}\left\{ \left(o\sum_{j=1}^{J}\sum^{I} S_{ij}r_j + v\sum_{j=1}^{J}\sum^{I} V_{ij}r_j\right) - \sum_{j=1}^{J} c_j(r_j, \bar{K}) \right\}, \tag{1}$$

where

o = dollar benefits of unit change in ozone concentration

v = dollar benefits of unit change in PM concentration

r_j = emissions reduction at source j, with a total of J sources

S_{ij} = source-receptor matrix mapping emissions of NO_x at source j to concentrations of ozone at receptor i

V_{ij} = source-receptor matrix mapping emissions of NO_x at source j to concentrations of PM at receptor i, with a total of I receptors

c_j = cost of controlling NO_x at source j

$\bar{K}$ = existing stock of capital

However, actual regulatory policies often result in outcomes that are far from fully efficient. Past policies have been effectively command and control, specifying either technology or proportionate rollbacks in NO_x emissions. This type of regulation has been shown by many analyses (such as Tietenberg 1985) to result in outcomes that do not come close to being cost-effective, even for the control of only one pollutant, such as ozone. On the other hand, when there are no transaction costs, trading NO_x emissions ton for ton will minimize the cost of achieving a target level of NO_x emissions reductions. Following Tietenberg (1985):

$$\min_{r_j} C(R) = \min_{r_j} \sum_{j=1}^{J} c_j(r_j), \tag{2}$$

subject to

$$-\overline{E} + \sum_{j=1}^{J} r_j \geq 0 \tag{2A}$$

[8] In another paper, we include cross-media impacts of NO_x emissions, examining their impact on both air and water pollution (increased nutrient levels).

where the notation is the same as Eq. 1 and where $\overline{E}$ is the aggregate NO_x reduction target and *C(R)* is total costs. Source trading can be limited to utilities or extended in our model below to include other point sources and mobile sources.

Focusing only on one pollutant, ozone, an alternative to simple NO_x trading is suggested by the fact that emissions of NO_x have a different impact on ambient ozone concentration and therefore on health impacts, depending on location. We examine another possible, more targeted, optimization goal, which is to minimize the costs of meeting an aggregate ozone-exposure reduction target (i.e., where population exposed as well as concentration change is accounted for). The cost minimization model above is easily adaptable to the case of meeting a target $\overline{X}$ for aggregate ozone exposures per "ozone season."[9] In this case, $\overline{X}$ would substitute for $\overline{E}$ in the constraint (Eq. 2A), and the set of source-receptor coefficients (S_{ij}) mapping emissions at source region *j* to ozone exposures at air receptor location *i* would be needed to convert NO_x reductions, r_j , to ozone reductions:

$$-\overline{X} + \sum_{j=1}^{J} \sum^{I} S_{ij}(w) r_j \geq 0 \qquad (3)$$

The links between emissions and ozone exposures as captured by the source-receptor coefficients, S_{ij}, will vary as weather patterns, *w*, change over the ozone season. This implies that trading ratios should change as weather patterns change over the season. But the frequency with which those patterns occur is fundamentally stochastic. Changing trading ratios during the season in response to actual or anticipated changes in the weather is clearly not feasible. The best policymakers can do is establish trading ratios and set the aggregate ozone target for the summer season. Aggregate summertime ozone exposure levels will then depend on the actual weather patterns.

We can examine the trade-off between greater controls and greater reliability in achieving a given ozone exposure target. The source-receptor coefficient matrix, S_{ij}, is a random variable with mean coefficients equivalent to some average episode. The actual ratios at which sources can trade are fixed and are referred to as S_{ij}*s. Given the trading ratios and the target level of ozone exposures in each receptor region, T_j, pollution sources decide how much NO_x emissions to reduce, r_i, in each region. Given the trading ratios, emissions-reduction choices, and random weather patterns, there is some probability that the ozone target will be met:

$$\Pr[f(S_{ij}, S_{ij}^*, r_i) \leq T_j] = \alpha \qquad (4)$$

Greater controls, r_i, will result in a higher probability of meeting the target level of ozone. In Section IV, we examine empirically the costs of achieving greater certainty in attaining the ozone goals.

The focus above has been on attaining ozone goals in the cost-minimizing policy formulations above. The ozone policy could also take account of the joint or ancillary reduction in PM that occurs as ozone policies are implemented. Taking account of these PM benefits in the maximization algorithm will change the allocation of controls among sources. To account for the "ancillary" PM benefits, we

[9] This is the sum of ozone parts-per-billion × person-days over the entire airshed and ozone season.

maximize social welfare, given the ozone target. That is, we seek to maximize PM benefits, B_p, net of the costs of achieving the ozone standard:

$$\max_{r_j} B_p\left(v\sum_{j=1}^{J}\sum_{i=1}^{I}V_{ij}\cdot r_j\right)-\sum_{j=1}^{J}c_j\left(r_j\right) \tag{5}$$

subject again to constraint (Eq. 3). Here V_{ij} is a source-receptor matrix mapping NO_x emissions at source j to PM concentrations at (air) receptor i; c_j, and r_j are as before. The optimal point of control occurs when marginal control costs *net of marginal ancillary benefits* are equal to the weighted shadow prices of the constraints:

$$\frac{MC_j - MB_{a_j}}{S_j} = \lambda = \frac{MC_{j'} - MB_{a_{j'}}}{S_{j'}} \tag{6}$$

for any two sources j and j'.

Eq. 6 implies that, in order to maximize the ancillary PM benefits net of NO_x control costs while still attaining the ozone exposure standard in each region, the marginal control costs net of the marginal benefit of the associated PM reductions must be equal across sources. Accounting for the effects of PM reductions means that sources that produce PM benefits are controlled more than they otherwise would be, as an increasing function of those air benefits. Also, accounting for the additional PM air benefits implies a lower shadow price, λ, than before, one that reflects the true social cost of the ozone target.

We note that we are making no claim that optimizations of the kinds described here reflect emissions trading or other particular policies. To estimate comparative costs of particular policies requires examining transaction costs, indivisibilities, monopoly issues, and a host of implementation issues that are beyond the scope of this paper. Nevertheless, we will use the term "trading" as synonymous with least-cost optimization for this analysis.

III. Methods and Data

The model includes estimates of the cost of NO_x control from major point sources, including utilities and other nonutility sources, as well as measures of the benefits from control of both ozone and PM. The benefits side of the model links emissions to concentrations of ozone and PM to population exposures and to health benefits. Figure 1 shows the domain of the analysis and the seven different regions [source regions (1-6) and receptor regions (1-7)] of the eastern United States covered in the analysis.[10] In conjunction with Figure 1, Table 1 translates the seven source-receptor regions into the abbreviated designations used throughout the paper. The data underlying this model are so extensive that we will summarize only the most important aspects for the results presented.[11]

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[10]The original six regions cover the Chesapeake Bay airshed and, coincidentally, also cover the region recently targeted for further NO_x controls on electric utilities by EPA. We added a seventh receptor region—New England—to capture health effects on this region.

[11] For more detail about the data and sources, see Krupnick and others (1998) or contact the authors.

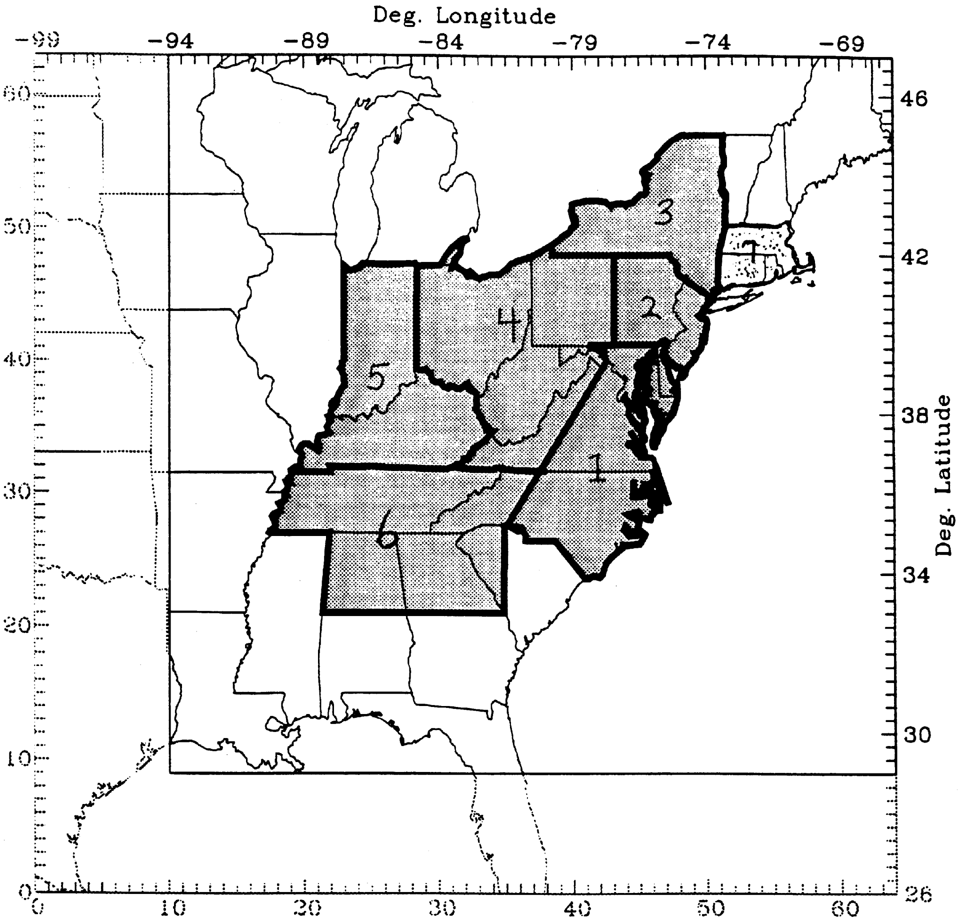

Figure 1. Modeling Domain.

Table 1. Region to State Correspondence.

Region	*Abbreviated Designation*	*States corresponding to Region*
1	MD-VA	Maryland, eastern Virginia, Delaware, eastern North Carolina
2	E.PA-NJ	Eastern Pennsylvania, New Jersey
3	NY	New York (except Long Island)
4	OH-W.PA-WV	Ohio, western Pennsylvania, West Virginia, western Virginia, southeastern Michigan
5	IND-KY	Indiana, Kentucky
6	TN-South	Tennessee, northern Mississippi, northern Georgia, northeastern South Carolina
7	New England	Massachusetts, Connecticut, Rhode Island, Long Island

Emissions and Control Cost Estimates

Baseline. The baseline for the analysis is projected NO_x emissions from activity levels forecast for the year 2007 under the regulations specified by the Clean Air Act Amendments of 1990.[12] The database for our analysis includes 599 utilities and 9,448 point sources. The controls assumed to be in place include: Reasonably Available Control Technology (RACT)-level NO_x controls on major point sources in ozone nonattainment areas, Title IV (Acid Rain) NO_x emissions reductions on steam-electric utilities, and stringent controls on projected new major point sources in ozone nonattainment areas (assumed to be selective catalytic reduction) required under EPA's New Source Review requirements.

CAC Policy Assumptions. The command-and-control (CAC) policy used here assumes control requirements on sources that mirror EPA's NO_x SIP call. Electric utilities will be required to reduce NO_x emissions to the lesser of 0.15 lbs NO_x per million Btu boiler heat input or 85% reduction; other large point sources must meet the same rate reduction or 70% reduction, whichever is lower.

Emission Rates and Costs of Control. The basic database for emissions and costs is from a model by E.H. Pechan and Associates for EPA, called the Emission Reduction and Cost Analysis Model for Oxides of Nitrogen (ERCAM-NO_x) (Pechan 1997). Cost and associated emissions-reduction estimates were developed for a variety of "engineering" abatement options for each source type.[13] Each point source is assigned a unique vector of abatement costs based on the size of that source's boiler and associated economies of scale.

A particularly challenging issue in estimating the cost of control is determining the marginal costs of increased abatement, given the discrete technologies in the database. In some cases, the technologies are physically additive—that is, the additional costs are just the cost of the added technology. In others, increased abatement is obtained by, in effect, replacing one technology with another, such as changing from low-NO_x burners to selective noncatalytic reduction. We assume for most sources that the capital cost of the dropped technology is not recoverable but operation and maintenance costs are recoverable.

NO_x to Ozone Source-Receptor Coefficients

To examine the consequences of policies recognizing the dependence of ozone concentrations on the locations of the source of NO_x and the populations affected, we developed source-receptor coefficients linking NO_x emissions at source area *j* to population-weighted ozone changes at receptor area *j*.[14] Population data used to weight the ozone changes for areas within each receptor region are

[12] Projected in Pechan & Associates 1997, which defines the CAA scenario by applying growth and control factors to the Interim Inventory (EPA 1993).

[13] We note that facilities covering about 23% of the emissions from nonutility point sources have no abatement options in the Pechan database.

[14] Our source regions are small enough to experience no gross differences in winds within the region, are based on physical distinctions affecting meteorology (separated by the Appalachian Mountains, for example), and, otherwise, follow state lines as much as possible. We defined a small number of such regions to minimize the number of UAM-V runs required to develop the source-receptor matrices. The resulting set of regions is by no

taken from Bureau of Economic Affairs projections by state. These coefficients were developed by running an ozone simulation model, the Urban Airshed Model (UAM-V), in what can be called an "attribution mode"[15] (Guthrie and Krupnick 1998) for three "typical" five-day ozone episodes using emissions and meteorological information from 1990.[16] The episodes are characterized by broad but distinct areas of elevated ozone concentrations centered approximately over the Midwest, the Northeast corridor, and the Southeast. Figure 2 shows the average wind pattern for our five-day Northeast episode.[17]

In contrast to OTAG and the work of Dorris and others (1999),[18] we have chosen typical rather than extreme ozone episodes. Extreme episodes were used in these other studies to help judge when abatement plans are sufficient to reach attainment. In our analysis, we are interested in annual or seasonal effects for which typical episodes are more appropriate. Focusing only on extreme events, which by definition are rare, would vastly underestimate the ancillary benefits of summer-long ozone

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means the only possible choice, and the degree of distinction between regions will obviously vary with weather conditions.

[15] The attribution mode involves developing a source-receptor matrix for each episode linking a source type/region's NO_x emissions to a change in one-hour peak ozone concentrations at each grid cell in the receptor regions. This is obtained by running the model for baseline emissions and then, for each region in turn, perturbing the source region's NO_x emissions (by source type), attributing the change in ozone over the grid to the source type/region, and summing over the grids (in our case using population weights) contained within each receptor region. NO_x emissions from all point sources in a region were reduced by 70% of the 2007 baseline in the UAM-V runs for generating the point-source S-R matrices. This figure is within the range of expected NO_x emissions reductions from point sources modeled within the OTAG process and underlies EPA's recent proposed SIP call. We assume stable source-receptor coefficients to the size of the emissions change and the background ozone concentration—assumptions being tested and to be reported on in later work. This approach, along with some results, is described by Guthrie and others (1998) and is a simplification of that used by Rao and others (1997).

[16] The UAM-V was run for five-day episodes, and the results averaged for the last two days of each episode for each region. The first three days are ignored to give the UAM-V a chance to work through the initial conditions. Data availability dictated choosing episodes from 1990. These patterns correspond (under extreme conditions) to the OTAG modeling episodes and, more generally, to the patterns identified in the recent Northeast States for Coordinated Air Use Management (NESCAUM) report (Miller and others 1997). This correspondence is important because the NESCAUM report identified windfields for the eastern United States that correspond to the top 20% of daily maximum ozone readings within each of the three regions, in turn. The 20th percentile ozone concentrations are quite low: 70 parts per billion (ppb), 72 ppb, and 67 ppb for the Northeast, Midwest, and Southeast, respectively. These concentrations are low because of the inclusion of rural monitors. Because of this, we feel that urban concentrations, which for a health benefits analysis such as ours are the most important target, are representing most of the 20th percentile readings. Thus, by choosing intervals that generally mimic these 20th percentile readings, we believe that we will reasonably capture typical urban summer day windfields when ozone is high enough to be problematic.

[17] The source-receptor relationships among regions are shown in Appendix B for all episodes.

[18]Dorris and colleagues use three meteorological episodes occurring in 1995, 1991, and 1988 to examine the effects of NO_x reductions on ozone formation in the eastern United States. These were years in which ozone levels were particularly high. The episodes chosen from this study occurred in 1990, a good year for ozone levels.

reductions, while any simple scaling of such events to capture benefits over an ozone season would overestimate such benefits.

A few caveats about these choices are in order. We recognize that 1990 was a "good" ozone year and that, therefore, our resulting source-receptor matrix may underestimate typical changes in ozone concentrations attributable to sources. Further, the large-scale features of the ozone patterns are likely to be well represented, but small-scale features should be interpreted with caution. In addition, the absolute maximum ozone concentration recorded may not capture the actual maximum for a particular day, since the ambient concentrations are not evenly distributed and the monitors may have missed the actual peak.

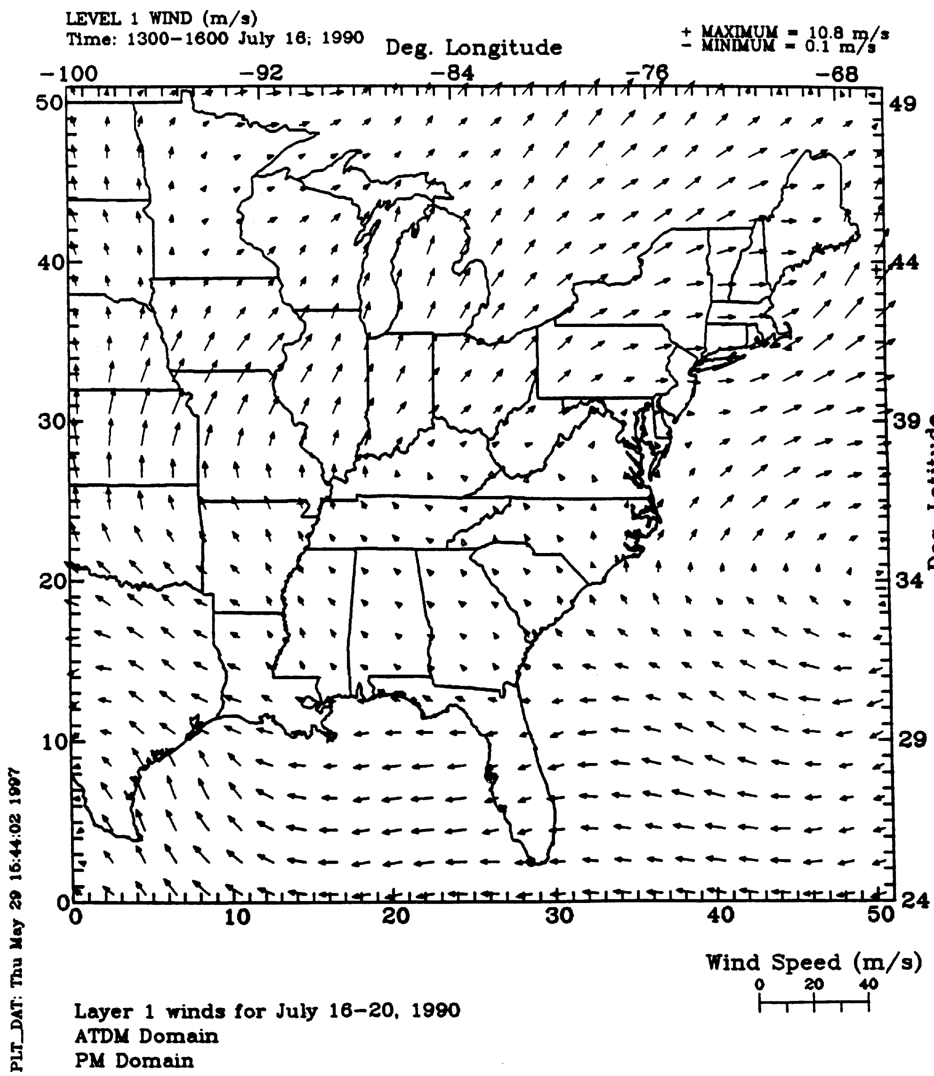

Figure 2. Wind Directions and Magnitudes for Northeast Episode.

All told, we developed unweighted and population-weighted source-receptor coefficient matrices for six source and seven receptor regions, for each of three ozone episodes, for utility and nonutility point sources and mobile sources, summing to 12 matrices developed from 37 runs of UAM-V. The following analysis does not use the mobile source matrices, so these will not be discussed further. The source-receptor coefficients for all episodes are shown in appendix B.

To summarize the results of the UAM-V analyses based on the average episode we find:

- The effect of a region's NO_x emissions on the ozone exposure of its own population is generally larger than the transported effects (except NO_x emissions in New York affect ozone in New England more than ozone in New York).
- Transport effects are relatively small in most cases, particularly from emissions in Tennessee and the South.
- Midwestern utility NO_x emissions change ozone locally about three times more per ton of NO_x than in either New York or New England and far less than the effect New York emissions have on New England ozone. Midwestern utilities have a greater effect on Maryland and Virginia ozone than on that of New England and New York.

Looking across episodes we find the following:

- The size of the coefficients is somewhat affected by episode type, although the general points made above with respect to the average episode apply to all three specific episodes.
- One difference of interest is that for the classic northeast episode (i.e., a Bermuda High), transport of emissions in the South and Midwest to northern and eastern regions are greater than in other episodes. For example, Midwestern point-source emissions affect New England, New York City, and E.PA-NJ as much or more than these emissions affect the Midwest in the Northeast episode.

One relevant way to summarize these results is in terms of what are called "trading ratios." A trading ratio measures the *total* effect of a ton of NO_x emitted in one region divided by the total effect of a ton of NO_x emitted in another region. Effects are measured by ozone exposures (in ppb × person-days) and are computed by multiplying the ozone coefficients in one column by the respective receptor region populations and summing the products. If one were to develop an NO_x trading program based on *ozone exposure effects,* these ratios would define the terms of trade. Table 2 shows the trading ratios for different episodes. The ratios are computed with the MD-VA region as the numeraire. This means, for instance, that reducing a ton of NO_x from point sources in NY results in 25% more exposures over the domain than doing the same thing in MD-VA.

The most striking result for the average episode is the uniformity of trading ratios, with the exception of those for OH-W.PA-WV and, to a lesser extent, NY. The higher ratios for these regions imply that a ton of NO_x reduction in these areas is worth more (in terms of exposure reductions) than a ton elsewhere. Given the source-receptor coefficients, the large effect of the OH-W.PA-WV emissions on population in the same region (coupled with a large population) drives this result. (See Appendix B for the source-receptor coefficients.)

Table 2. Point-Source Trading Ratios for Ozone, by Episode.

Episode	*Point Sources by Region*					
Type	*MD-VA*	*E.PA-NJ*	*NY*	*OH-W.PA-WV*	*IND-KY*	*TN-South*
Average	1.0	1.06	1.25	1.51	1.01	1.00
NE	1.0 (1.30)[a]	0.66	0.89	1.24	0.92	0.96
MW	1.0 (1.16)[a]	1.61	0.97	1.27	0.79	0.50
SE	1.0 (0.55)[a]	0.86	2.68	2.68	1.68	2.16

Notes: Numbers indicate total ozone exposures (ppb × person-days) in an entire region from a ton of NO_x emitted in one region relative to total ozone exposures from a ton emitted in the MD-VA region. Regions are defined in Table 1. [a]Ratio of the coefficient from the Northeast, Midwest, or Southeast episode to that from the average episode.

This uniformity is not so evident across episodes. For instance, for the Midwest episode, emissions reductions in E.PA-NJ are very valuable, those in the south are relatively unproductive in reducing ozone exposures, and the OH-W.PA-WV effect is still evident. In the Southeast episode, emissions in New York and the region to the south and west (OH-W.PA-WV) have a large impact on ozone levels, 2.68 times higher than the effect of emissions in MD-VA. The E.PA-NJ region has the smallest impact in this episode.

NO_x to Fine PM Source-Receptor Coefficients

We used similar methods to link NO_x emissions at source area *i* to population-weighted nitrate PM concentrations at receptor area *j*. These linkages were made by using multiple runs of a major EPA-supported fine PM simulation model, the Regulatory Modeling System for Aerosols and Deposition (REMSAD). One matrix showing this linkages was produced, representing a full year's meteorology. Source and receptor regions were defined identically to those for ozone.[19] The source-receptor coefficients for PM are shown in Appendix B.

The population-weighted source-receptor coefficients are converted to trading ratios as in the ozone example above and shown in Table 3. We find that a ton of NO_x reduces nitrate PM concentrations over the domain by far more if emitted in E.PA-NJ than anywhere else and that the least nitrate reductions come from NO_x reduced in MD-VA and TN-South. Thus, policies designed to cost-effectively reduce nitrates should provide added incentives for NO_x reductions in the E.PA-NJ region.

Table 3. Trading Ratios Based on NO_x to Nitrate Concentrations.

MD-VA	*E.PA-NJ*	*NY*	*OH-W.PA-WV*	*IND-KY*	*TN-South*
1.0	2.8	1.4	1.6	1.5	1.0

Note: Ratios are based on total nitrate PM exposures ($\mu g/m^3$ × person-days) in the entire region from a ton of NO_x emitted in one region relative to total exposures from a ton emitted in the MD-VA region. Regions are defined in Table 1.

[19] Sulfur dioxide emissions were held constant throughout and assumed to be at levels consistent with full implementation of Title IV.

Health Benefits Model

The population-weighted source-receptor matrices linking NO_x emissions to changes in ozone or PM nitrate concentrations are multiplied by regional population and estimates of the health damages per person per unit change in ozone or nitrates to estimate ancillary health benefits per day for NO_x emissions reductions. These daily estimates are aggregated to either the ozone season or the full year, depending on the pollutant. Ozone benefits occur only over the ozone season, but PM benefits accrue all year.

The per-person health damage estimates are taken from a recently updated version of the RFF Health Benefits Model, which is embedded in the Tracking and Analysis Framework, a Monte Carlo-based system for simulation modeling of complex and uncertain relationships (Bloyd and others 1996). The model includes a comprehensive set of epidemiological concentration-response functions and value functions taken from the economics literature and EPA reports, such as the Regulatory Impact Analysis for Particulates and Ozone (U.S. EPA 1997b).

The benefit estimates are highly sensitive to the treatment of mortality risks.[20] In line with the general reluctance of EPA and others to ascribe mortality benefits to ozone (EPA 1997b), mortality risks are assumed to be zero over 90% of the probability distribution, with 10% of the distribution determined by results from Ito and Thurston (1996) (a study generally on the high side of the studies showing a relationship between ozone and mortality risk).

The linkage between particulate nitrates and mortality risks is based on the study by Pope and others (1995), following EPA's cost-benefit analysis of the 1990 Clean Air Act Amendments (U.S. EPA 1999). Nitrates can be considered equally potent as sulfates, equally potent as fine PM (PM2.5), or equally potent as PM10[21], with potency diminishing as one moves through this series. We generated unit value estimates (for mortality risks only) under the first and third assumptions to bound the mortality estimates.

The benefit estimates are also highly sensitive to the way in which mortality risk reductions are valued (see Krupnick and others 1999 for a full discussion of the options). Values in the literature range from implicit values of a statistical life (VSL) of $4.8 million to $70,000, depending on the conceptual and empirical basis for the estimates. The traditional approach, based on labor market and accidental transport death studies, yields an estimate of $4.8 million when uncorrected for advanced age of the people who appear to be at risk from air pollution. When corrected using the Jones-Lee and others (1985) study, the VSL drops to $3.2 million. The latter figure is used for the high (95th percentile) estimates of mortality reduction benefits for ozone (mean and 5th percentile estimates of mortality risk reductions being zero).

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[20] See Lee and others (1995) for a discussion of the epidemiological literature linking ozone exposure to mortality risks and the decision to treat these effects as described in the text. EPA omits ozone-mortality functions in their quantitative analysis because of the large uncertainties over whether such a relationship exists (U.S. EPA 1999, 5-4). The report notes that the PM-mortality function that is included may be counting some of the ozone effect to the extent ozone and PM concentrations are collinear.

[21] PM2.5 includes particles with diameter up to 2.5 microns; PM10 includes particles with diameter up to 10 microns.

For the purpose of this analysis, we focus on two sets of unit value estimates, termed "Mid" and "High." The Mid scenario represents the combination of assumptions that, in our judgment, best reflect the consensus state of the science, whereas the High scenario is intended to push the assumptions one can make to the limit. This is done because, as will be seen below, the least-cost analyses are fairly insensitive to the size of the benefits. By the same logic, we do not present scenarios with "Low" benefit estimates. The differences between the Mid and High scenarios are determined by the following assumptions: the potency of NO_x PM, the weather episode for ozone, the VSL, and the chosen range of estimates from the Monte Carlo model. For the Mid scenario, NO_x has the potency of PM10, the ozone weather episode is Northeast, VSL is $3.2 million, and mean estimates are chosen from the Monte Carlo model. The resulting Mid values are a $4.90 unit estimate for ozone and a $58 unit estimate for nitrate PM. Units are in 1990 dollars/person in entire population per 0.01 ppm ozone or per $\mu g/m^3$ nitrates experienced over the year, respectively. For the High scenario, NO_x has the potency of sulfates, the ozone weather episode remains Northeast, VSL is $4.8 million, and the 95th percentile unit values are chosen from the Monte Carlo model. As discussed above, this 95th percentile for ozone includes ozone mortality risk. The resulting High values are $75.20 for ozone and $239 for nitrate PM. See Appendix B2 for further details.

Benefits per Ton of NO_x Reduction

The discussions above need to be tied together to provide a more complete picture of the effect of a ton of NO_x reduction in the source areas on regionwide health benefits through the ozone "pathway" and the nitrate "pathway." There are two factors to consider. The first is the rates of transforming NO_x emissions to ozone and to PM nitrates. The second factor is the potency of ozone and PM nitrates in terms of their effect on health risks. Because the source-receptor coefficients for ozone are significantly "larger" than those for nitrates (see Appendix B), the benefits per ton of NO_x reduction operating through the ozone link are larger than those operating through the nitrate link, even though nitrates are more "potent" than ozone.[22]

Table 4 provides the unit benefit estimate in dollars per ton of NO_x reduction for the Mid and High sets of assumptions.

We use the data on air benefits and the costs of NO_x control to do the economic analyses described in Section II for the 11-state region. We first examine the proposed regulatory policies, which we call the CAC policy. We then compare that to a variety of trading options and finally to an overall comparison of the costs and benefits of controlling NO_x in the region.

The rich database underlying our model generates a large number of possible scenarios for any given comparison between policies. There are two types of point sources (electricity generators and nonutility point sources); two criteria pollutants reduced when NO_x emissions are reduced (ozone and PM as nitrates); for ozone, four source-receptor matrices covering three different meteorological

[22] The EPA's cost-benefit analysis of the Clean Air Act Amendments of 1990 (U.S. EPA 1999, Appendix D) provides coefficients reflecting the mortality risk reductions and other health improvements per unit ozone and PM2.5 or PM10 change. These coefficients are about 10 times larger per unit PM ($\mu g/m^3$) than per unit ozone (ppb).

episodes (Northeast, Midwest, and Southeast) and the average of these; and Mid and High assumption sets for computing health benefits of ozone and PM reductions. Below, we present the most compelling set of these various combinations. For instance, we emphasize utility-only scenarios, because this matches EPA's plan for trading and present "all point-source scenarios" because of the expanded opportunities for cost-cutting compared to the utility-only case.

Table 4. Benefits for Ozone and PM Matter (as Nitrates) per Ton of NO_x Reduction by Source Region, in $1990.

Benefits/Ton		*MD-VA*	*E.PA-NJ*	*NY*	*OH-W.PA-WV*	*IND-KY*	*TN-South*
Mid	Ozone[a]	306	201	271	379	283	294
	PM[b]	36	103	52	59	54	35
High	Ozone[a]	4,712	3,100	4,171	5,835	4,349	4,520
	PM[b]	150	425	215	245	222	145

Notes: See the text for definitions of Mid and High assumptions. For ozone, the Northeast episode was used. Regions are defined in Table 1. [a]Benefits calculated over the ozone season. [b]Benefits calculated over the year.

IV. Results

CAC versus NO_x Trading

Table 5 summarizes the results of the CAC policy and a simple NO_x trading policy (mimicked by finding the least-cost NO_x reduction allocation to attain the aggregate NO_x reduction under the CAC policy).[23] For the CAC policy, the costs for the utilities-only case are just over $1 billion a year, with health benefits related to ozone and PM reductions ranging from $140 million to $1.7 billion per year. Considering both utility and nonutility point sources (all point-source trading), the costs are just under $1.4 billion a year and benefits range from about $200 million to just under $2.4 billion, depending on whether the Mid or High estimates of benefits are used. Utilities bear, by far, the largest share of the costs of NO_x control under the CAC policy.

Table 5 also shows that utility-only NO_x trading has the potential to achieve the same emissions reduction as the CAC policy at almost one-half the cost, assuming transaction costs are similar under the two policies. Although the costs for utilities fall by almost one-half to only $621 million a year, ozone and PM benefits are similar under the two policies because NO_x reductions in the aggregate are the same.

Figure 3 shows the marginal cost curve for NO_x controls on utilities for the entire region. Additional control costs are fairly insensitive to additional emissions reductions up to a point, and then rise sharply. NO_x reductions from utilities under the emissions trading policy discussed here, 740 million

[23] See Appendix A for details on the optimization model.

tons per year, are still in the elastic portion of the marginal cost curve and well below the elbow after which marginal costs rise rapidly.

Table 5. Results of CAC and NO_x Emissions Trading.

	Utilities Only		*All Point Sources*	
	CAC	*NO_x Trading (ton for ton)*	*CAC*	*NO_x Trading (ton for ton)*
Total abatement cost (millions $/yr)[b]	1,141	621	1,369	680
Utilities			1,141	411
Other point sources			228	269
NO_x emissions reduction				
Tons/yr (thousands)	741	740[a]	1,022	1,026[a]
Tons/ozone season (thousands)	333	339	451	451
Air pollution reduction benefits (millions $/yr)[b]				
Total Mid benefits	142	142	195	198
Ozone, average episode	101	101	136	138
PM	41	41	59	60
Total High benefits	1,720	1,729	2,336	2,365
Ozone, average episode	1,551	1,559	2,094	2,118
PM	169	170	242	247
Shadow price ($/ton NO_x)		4,646		3,226

Notes: CAC is command and control; [a]NO_x emissions reductions are not exactly the same under CAC and NO_x trading because of lumpiness in technology. [b]1990 dollars.

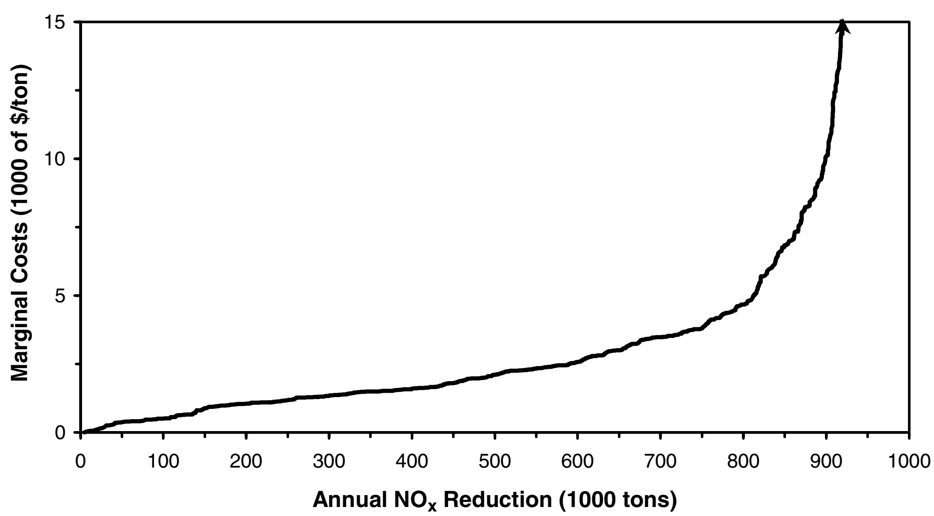

Figure 3. Gross Marginal Cost per Ton NO_x Reduction, Utilities Only.

Not only are the aggregate cost savings large under utility-only NO_x trading, but all regions stand to benefit from moving away from CAC (Figure 4a). Cost savings are highest in New Jersey and are, in general, somewhat higher in the southern part of the region. Figure 4b shows the change in emissions reductions for NO_x trading compared to CAC. This map reveals how sources might trade under an emissions trading program. If permits were distributed to plants on the basis of the emissions reductions under CAC, then, on average, plants in the Midwest (Indiana, Ohio and Kentucky) and plants in the northern part of the trading region (New York, Pennsylvania, and New Jersey) would be likely to sell permits (increase emissions reduction), and plants in the southern part of the region (Maryland, Virginia, North Carolina, and West Virginia) would buy permits.

For all point-source NO_x trading, Table 5 shows that costs are again reduced by about 50% from the CAC policy. What is more interesting is that costs are only slightly higher for controlling all point sources than they were in the utilities-only trading case, while emissions reductions of NO_x are almost 50% larger! This occurs because there are many low-cost point-source options available in the trading case. The share of costs paid by utilities falls dramatically when other sources are included in the optimization, and the shadow price per ton of NO_x removed falls from $4,646 per ton for the utilities-only case to $3,226 per ton when all point sources are included.[24] The implication is that there is potential to lower costs by bringing other sources into trading policies as long as the transactions costs of doing so are not large.

Emission Trading versus Ozone Exposure Trading

An alternative to simple ton-for-ton emissions trading is to allow sources to trade on the basis of the ozone exposures over the ozone season rather than on the basis of NO_x emissions over the year. The former approach captures the fact that sources reducing NO_x emissions adjacent to large population centers will have a larger impact on health than those that reduce equivalent emissions in more remote locations. The impact of sources in each region on ozone exposures in other regions, or the ozone exposure trading ratios, appeared in Table 2 above. In theory, costs can be saved through exposure trading to attain the aggregate ozone exposure reduction during the ozone season found in the NO_x trading case by shifting controls to relatively low-cost sources that have an impact on large surrounding populations. The disadvantage with basing trading on exposures is that each source would have to trade at a different ratio with sources in each of the other regions. Such a trading policy is likely to have much higher transaction costs than does emissions trading.

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[24] Studies of least-cost control strategies for reducing NO_x by Dorris and others (1999) and by New York State Energy Research and Development Authority (1999) find similar results: trading policies would tend to bring in more low-cost industrial sources and would result in fewer utility controls compared to a CAC policy. However, these studies differ from this one in a number of ways, including the handling of ozone constraints and in the number of sectors and pollutants included in the analysis. Those studies included analysis of mobile sources, while this one does not. This study includes analysis of uncertainty and of both ozone and PM benefits, and those studies do not measure benefits and have only an ozone constraint.

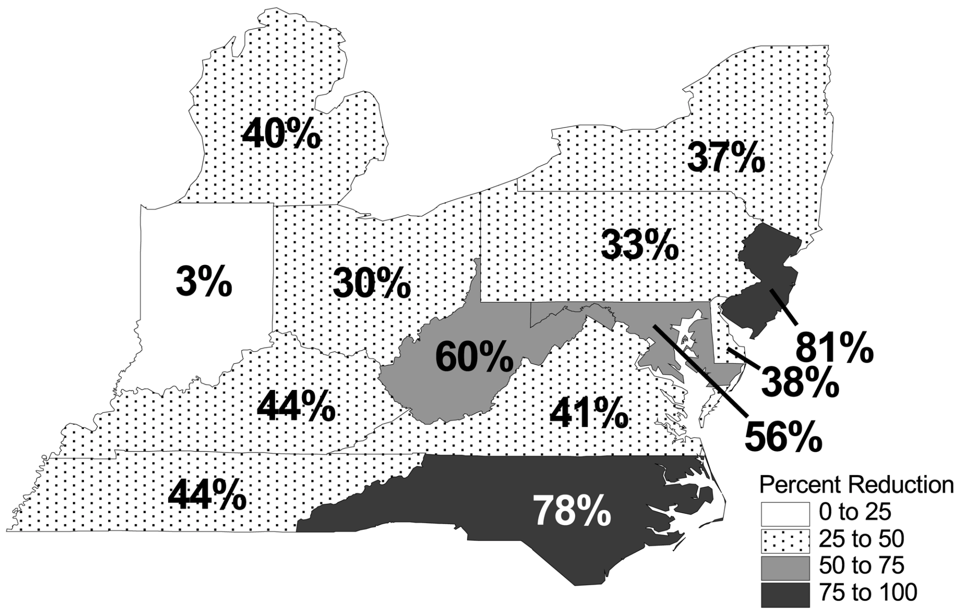

Figure 4a. Percentage Reduction in Gross Cost with NO_x Trading, Compared to CAC, for Utilities Only.

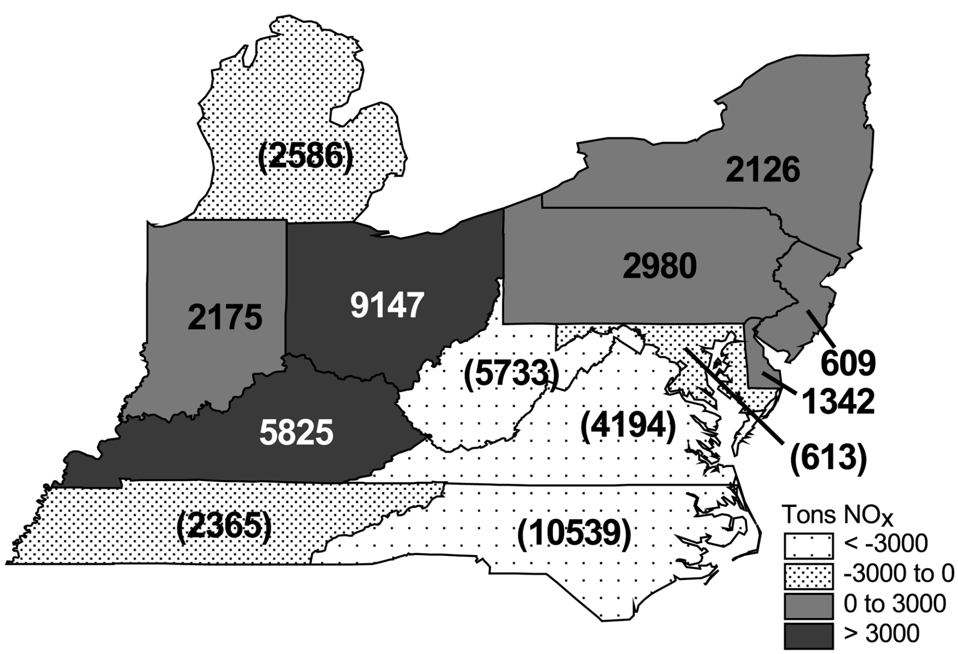

Figure 4b. Tons of NO_x Reduced (Increased) with NO_x Trading, Compared to CAC.

Moreover, agreeing on the appropriate trading ratios is complicated because meteorological conditions vary over the ozone season, causing different impacts on ozone levels and populations exposed in the various regions. Table 6 presents results of ozone exposure trading, assuming the ratios depend on each of the three different episodes, and then compares these results to those for NO_x emissions trading.

The reductions in costs for the utilities-only case from ozone exposure trading relative to emissions trading are, in the aggregate, very small. Costs fall by only 2–3% for most episodes as a result of ozone trading. One reason the cost changes are small is that most of the trading ratios for utilities (Table 2 above) are fairly close to 1.0. Spatial disparities in pollution effects *over the entire eastern region* are not large relative to cost differentials across sources.

The episode with the most disparity in trading ratios is the Southeast episode. (In this episode, the New York region and the Ohio–Western Pennsylvania–West Virginia region have the highest ratios relative to Maryland and Virginia (2.68 to 1.0). As expected, we do see the greatest cost savings with ozone trading for this episode—costs fall from $621 million to $597 million. This cost savings arises because, in the aggregate, NO_x emissions reductions are not as large. But, with lower NO_x emissions reductions comes lower PM exposure reductions, which offset even the minor savings that result from lower abatement costs. In short, there appears to be no clear benefit to a spatially differentiated trading system over a simple policy of allowing sources to trade emissions ton-for-ton.

Although this conclusion holds in the aggregate, the regional effects across these two types of policies are fairly different. Some states have costs that are as much as 50% greater (Delaware) under ozone exposure trading compared to emissions trading, and others have costs that are more than 50% lower (West Virginia). However, since we have no a priori reason to prefer one distribution of costs among

Table 6. Comparison of NO_x Trading and Ozone Exposure Trading.

	Utilities Only					*All Point Sources*		
		Ozone Exposure Trading by Episode					*Ozone Exposure Trading by Episode*	
	NO_x Trading	*Avg.*	*NE*	*SE*	*MW*	*NO_x Trading*	*Avg.*	*SE*
Ozone exposure reductions (millions of ppb × person-days per ozone season)	*	31.6	34.6	28.3	32.1	*	42.9	37.9
NO_x emissions reductions (thousand tons/year)	740	734	737	708	728	1,026	1,003	937
Costs (millions 1990$)	$621	$616	$618	$597	$608	$680	$666	$620
PM exposure reductions (millions of μ/m^3 × persons per year)	260	259	259	250	258	377	371	340

Note: NE is Northeast; SE is Southeast; and MW is Midwest. *Ozone exposures will vary depending on the meteorological episode.

regions to another, these distributional differences provide no basis for favoring ozone exposure trading over the administratively simpler emissions trading.

Finally, these results are basically unchanged when all point sources are included in the trading programs. We show the results for the average and Southeast episodes for ton-for-ton emissions trading and ozone exposure trading in Table 6. Particularly for the Southeast episode, there are some cost savings under ozone exposure trading. However, those savings are not large, and PM benefits are also lower.

Uncertainty Analysis

Because there does not appear to be a clear advantage to an ozone trading policy, we return to a deeper examination of NO_x trading. Ozone episodes through the ozone season are stochastic; thus, there is uncertainty about whether a particular ozone exposure target will actually be met. In this section, we examine the probability that a given ozone target will be met, given assumptions about the probability distribution of meteorological events. We then trace the cost function for achieving greater certainty in meeting the target. This illustrative analysis is for utilities only.

In the absence of other evidence, we model three weather patterns—the Northeast (NE), the Southeast (SE), and the Midwest (MW)—as occurring randomly throughout the ozone season. Hence, the source-receptor coefficients linking NO_x emissions in one region to ozone exposures in another are random variables with mean coefficients equivalent to those for the average episode.[25] We assume that the regulatory authorities target total ozone exposure levels in trying to achieve ozone standards. Because the source-receptor matrices are stochastic over the ozone season, there will be a probability distribution for such exposures for any given level of NO_x control.

Figure 5 shows the results of the uncertainty analysis. Each point on the graph is associated with a given level of total NO_x control. The vertical axis shows the costs of those controls and the horizontal axis shows the probability that a given level of ozone exposure reductions will be realized. The two lines apply to two different targets; the first is the ozone exposure reduction that would occur under controls required under CAC requirements when an average ozone episode prevails, and the second is a 5% tighter target. In each case, we have identified the point associated with total NO_x emissions reductions of 740,000 ton per year, the NO_x reductions under the CAC case. Points to the right show the costs and the probability of meeting or exceeding the ozone-exposure reduction target (termed "reliability") if emissions are reduced more than this amount (2% and 5% respectively), and points to the left show lower emission reductions (–2% and –5%).

For the CAC case, NO_x emissions reductions can be increased by a modest 2% with small added costs, resulting in a large increase in the probability of meeting the target (the probability increases from 62% to about 90%). To obtain even greater reliability, the costs rise more steeply. On the other hand, a 2% smaller NO_x reduction does not save much, but results in a much lower probability of

[25] We estimated this distribution by assigning each of the three episodes numbers with equal probability and drawing 20 times to get a weighted distribution over the season. We repeat this process 600 times to get the full distribution of possible weights.

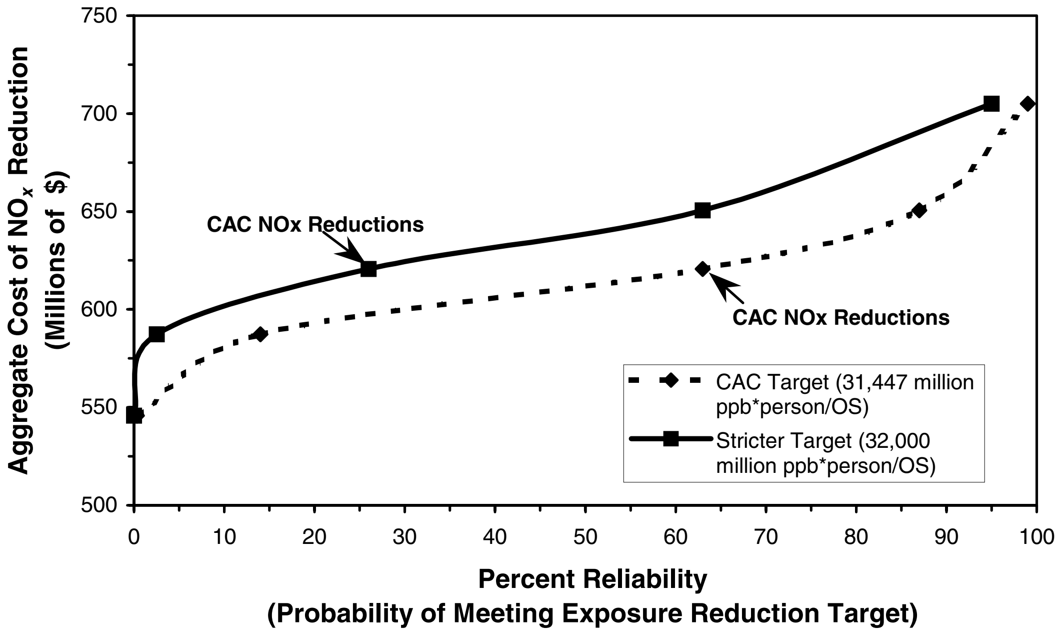

Figure 5. Aggregate Cost of NO_x Reductions versus Percentage Reliability, for Utilities Only.

meeting the target. It turns out that even though the trading ratios are different for different regions, sometimes by a factor larger than 2 (see Table 3 above), the distribution of ozone outcomes is fairly tight. This suggests that, in most situations, it is better to err on the high side of meeting emissions-reduction targets, since there may be net benefits to overcontrol. PM Trading Policies

We now examine how the outcomes of a trading policy geared to meeting PM-exposure reduction goals would differ from those of a similar policy geared to meeting ozone-exposure or NO_x emissions reduction goals. We have already seen that an ozone-exposure trading policy does not result in much lower costs than an NO_x trading policy delivering the identical ozone exposure reduction. Thus, here we see whether a PM-exposure trading policy, to attain PM exposure reductions arising from the NO_x trading policy, does any better.

Table 7 provides the results for the PM-exposure trading scenario versus NO_x trading, for both utilities only and all point sources.

Table 7 shows that, in the utility-only case, the PM-exposure reduction target can be obtained somewhat more cheaply ($15 million) by PM exposure trading than by an NO_x trading policy. This is in contrast to a comparable ozone-exposure reduction policy that has almost trivially lower cost than the NO_x trading policy. This difference occurs because the source-receptor coefficients for PM are more spatially differentiated than those for ozone.

PM exposure trading results in more cost savings relative to NO_x trading when all point sources are permitted to trade. Costs fall by some 14% compared to only 2.5% in the utilities-only comparison. Interestingly, this comes at the "price" of only slightly smaller ozone exposure reductions (4%). In contrast to our earlier finding that ozone exposure trading provided virtually no cost savings relative to NO_x trading, PM exposure trading for all point sources provides significant cost savings over NO_x trading to achieve a PM exposure goal.

Policies That Include Joint Pollutants in the Optimization

So far, we have considered the effects of policies for controlling NO_x emissions, controlling ozone exposure, or controlling PM exposure, treating any reductions in the nonoptimized pollutant concentrations as ancillary. In this section, we examine whether considering ozone and PM reductions jointly alters the optimal allocation of NO_x reductions. If not, then policymakers have a "two-fer," a policy that cost-effectively meets ozone goals *and* meets PM goals. However, if the allocation differs between the two cases, then policymakers need to ask how the incentives of the ozone-exposure trading program can be altered to account for the PM effects.[26] The following analysis is done for utilities only, to illustrate the issue.

We consider the joint benefits from NO_x reductions in two ways. First, we redo the ozone cost-effectiveness analyses above but with annual PM benefits "netted out" of each plant's costs. This is a relatively limited way to alter the optimization problem and is done to see if PM health benefits are large enough (and differ enough across locations), relative to control costs, to alter the allocation of NO_x reductions.

Table 7. PM Exposure Trading compared with NO_x Trading for Utilities Only and for All Point Sources.

	Utilities only		*All Point Sources*	
	NO_x Trading (Avg.)	*PM Trading*	*NO_x Trading (Avg.)*	*PM Trading*
Reduction in:				
Ozone exposures (millions of ppb × person-days per ozone season)	31.6	31.2	42.9	37.9
PM exposures (millions of $\mu g/m^3$ × person-days per year)	260.1	260.1	377	340.4
NO_x emissions (thousand tons/year)	740	726	1,026	937
Costs (millions 1990$)	621	606	680	620

Table 8 compares the gross cost-effectiveness results to the "net" cost-effectiveness results for selected runs of the model for the utilities-only case. In general, even though PM benefits assumptions are set at High, total costs and other outputs of the model are only trivially different in the "gross" and "net" cost-effectiveness cases, with the allocation of costs across states altered in only

[26] In keeping with the historical prominence of the ozone context, we do *not* consider how *ancillary* ozone effects alter a basically PM-based policy.

minor ways.[27] This provides some evidence that the relative allocation of NO_x reductions under various types of trading policies for achieving ozone goals would not be substantially changed if PM benefits were taken into account. Policymakers can pursue strategies to reduce ozone that will also be likely to achieve PM goals cost-effectively.

The second approach we use to recognize jointness is to permit, in effect, multiple-pollutant, spatially differentiated trading. We set a monetary health benefit target based on ozone health benefits and then seek the allocation of NO_x reductions that meets that monetary target at least cost, allowing benefits to come from either ozone or PM reductions. This approach mimics an interpollutant trading policy. As shown in Table 9, again, cost savings over NO_x trading are minimal.

Table 8. Comparison of Net Cost-Effectiveness Policies and Gross Cost-Effectiveness Policies.

	Utilities Only			
	"Net of PM" NO_x Trading	*"Gross" NO_x trading*	*"Net of PM" Ozone Trading (Northeast)*	*"Gross" Ozone Trading (Northeast)*
Reduction in:				
Ozone exposures (millions of ppb × person-days per ozone season)	34.4	31.6	34.6	34.6
PM benefits, High estimate (millions 1990$)	170	170	170	170
NO_x emissions (thousand tons/year)	738	740	738	737
Costs (millions 1990$)	618	621	618	618

Table 9. NO_x Trading versus Interpollutant Trading for Utilities Only.

	NO_x Trading	*Interpollutant Trading*
Cost (millions of 1990$)	621	616
NO_x reductions (thousand tons/ozone season)	334	332
Higher costs	—	MI, NJ, OH, PA, WV
Lower costs	—	DE, KY, NY, NC, TN, VA

Note: Assumes Mid ozone benefits and High PM benefits.

[27] The same result holds when nonutility point sources are brought into the analysis.

Finally, we ask a very different question—a cost-benefit question about the optimal amount of reductions in NO_x emissions when both PM and ozone are taken into account, rather than a cost-effectiveness question about optimal allocation of NO_x reductions to meet a pollution target (see equation (1) in Section II above). As shown in Table 10, the optimal NO_x reduction is extremely sensitive to assumptions about the size of the unit benefits. What we term the Mid benefits assumption set is, in our judgment, the fairest reading of the standard literature. The Third column of Table 10 shows the results when benefits net of costs are maximized under these assumptions. Only 28 of 599 NO_x sources would be controlled, reducing NO_x by 85,000 tons, for costs of only $10 million and net benefits of $7 million. This compares to 428 sources controlled under the NO_x trading policy described above to meet the CAC reduction level of 740,000 tons per year at a cost of $621 million annually. Even when we "push" the PM risk and valuation assumptions to generate the largest PM benefits (the High-Mid assumption set), optimal emissions reductions only double (to 134,000 tons), while both costs and net benefits, roughly triple.

However, under the High-High assumption (that is, adding the ozone-mortality risk linkage), the optimal outcome turns out to be quite similar to the proposed NO_x trading policy. The optimal emissions reductions are even somewhat larger than would occur under that policy (761,000 if we are maximizing net benefits versus 740,000 under the NO_x trading). We find that costs are only slightly higher for the optimal policy ($669 million versus $621 million) and that there is not much difference in the distribution of costs among regions under the two policies. New Jersey's costs are a bit lower for the optimal policy, but most other states experience an increase in costs relative to NO_x trading. We conclude that if ozone is thought to have an effect on mortality risk, and if this effect is reasonably characterized by the Ito and Thurston study, and if the VSL is computed in the standard way, then EPA's NO_x reduction SIP call appears to be a good target, as does the trading program to attain it.

Table 10. CAC and NO_x Trading versus Optimal Programs for High, High-Mid, and Mid Benefit assumptions, for Utilities Only.

	CAC (High Benefits)	*NO_x Trading (High Benefits)*	*Mid Benefits*	*High-Mid Benefits*[a]	*High Benefits*
Number of sources controlling	599	428	28	68	439
Net benefits (millions 1990$)	732	1,255	7	26	1,261
Costs (millions 1990$)	1,140	621	1	27	669
Ozone benefits (millions 1990$)	1,703	1,706	13	21	1,756
PM benefits (millions 1990$)	169	170	5	33	174
NO_x reductions (thousand tons/yr)	741	740	85	134	761

Note: CAC is command and control. [a] The High-Mid Scenario combines the Mid and High scenarios, mixing the High PM assumptions with the Mid ozone assumptions.

V. Conclusions

The basic implication of our results is that EPA's utility-based NO_x trading policy is reasonably efficient compared with more sophisticated and harder to implement policies involving trading on the basis of spatially differentiated effects of NO_x on ozone, fine PM concentrations, or both. We find, for instance, that the ozone source-receptor coefficients linking emissions in one region to changes in air quality in another are not sufficiently large, irrespective of meteorological conditions, to offset differentials in the marginal costs of alternative abatement options both between and within utility sources. Therefore, the cost savings from departing from a one-to-one trading scheme are insignificant. The story for PM nitrates is similar. In addition, although the health benefits of reducing a $\mu g/m^3$ of fine PM concentrations exceed those of reducing a ppb of ozone, the source-receptor coefficients for PM nitrates are so small that the ozone benefits (per unit NO_x emissions) dominate.

Even more important, our cost-benefit analysis reveals that the aggregate utility emissions reductions sought by EPA are quite close to the optimal emissions reductions. This strong conclusion is tempered in two ways. The first is that this result *requires* the assumption that ozone exposures affect mortality risk. EPA has been reluctant to reach this conclusion on the basis of the few studies showing such an effect and the many showing no significant effect. Unless ozone affects mortality, the NO_x reductions required by EPA are far too large. This conclusion holds *irrespective of assumptions made about nitrates and health.* The second caveat is that the optimal *distribution* of NO_x reductions differs substantially from the one we find from our NO_x trading policy.

When we allow other point sources to be included in the NO_x trading analysis, we find that there are far larger cost savings relative to the CAC policy than under the utilities-only comparison and that the aggregate utility costs are lower. There are some low-cost, nonutility point sources, which suggests that EPA should consider broadening the trading program to incorporate those sources—if they can be included with relatively low transaction costs.

Another implication of our results concerns the trade-off between costs and the degree of certainty in attaining ozone-exposure reduction targets. We find that the probability of meeting an ozone target is very responsive to small changes in costs and that it is, therefore, advantageous to err on the side of overcontrol.

Our analysis has numerous limitations that argue for caution in interpreting the results. The most important is that the NO_x control options are limited to those provided by Pechan and Associates and, specifically, do not extend to fuel-switching options and to changes in activity levels, such as the quantity of electricity generated. Adding such options might alter the cost functions and result in ozone exposure trading having a larger cost-reducing effect. Another limitation is uncertainty over the source-receptor matrices. Testing of linearity assumptions regarding the ozone coefficients revealed them to be basically linear, but testing was limited. More important, these coefficients are most appropriate to typical, rather than extreme, ozone events. The size of these coefficients for extreme events would be larger. Finally, all of our "trading" cases are found through cost-minimization algorithms, implying that trading is frictionless. Issues of how such a trading program would be designed, the type and extent of transaction costs, and the effects of various constraints are critical for actual implementation of such a program. Examination of those issues and development of a bilateral trading model will be the subject of another paper.

Appendix A: The Optimization Method

This appendix describes the algorithm and mathematical approach used to find the least-cost allocation of NO_x emissions reductions to meet the given constraints. This algorithm determines a "convex hull," which is a marginal cost function containing only technologies not dominated by other technologies, in terms of costs and emissions reductions. The convex hull is used as the basis for determining the control technology for each source in the optimization process.

Determining the Convex Hull

Scenarios examining the costs of pollution reduction, given the fact that Clean Air Act (CAA) technologies were in place, used the 2007 CAA technologies as the starting point in the determination of the convex hulls. The appropriate cost and NO_x emissions reduction of every possible technology for each source was computed relative to the CAA starting point. All technologies with NO_x emissions reductions less than the NO_x emissions reductions of the starting point technology were dropped from the set of feasible options. Technologies having greater NO_x emissions reductions than the starting point technology but providing them at a lower cost, and thus having negative marginal costs, were included in the feasible set. These technologies were included to account for the possibility that the command-and-control scenario specified technologies that were less efficient than other available options. For the remaining technologies, the marginal cost of moving from one technology to another (the change in cost/change in NO_x emissions reductions) of all technologies relative to the starting point were calculated. The technology having the minimum slope was selected as the next point on the convex hull after the starting point. Information for this point (marginal cost, marginal NO_x emissions reduction, load-to-emission ratio, technology identifiers, and so on) was retained, and this point was used as the new reference point in determining the next point on the convex hull. Technologies having fewer NO_x emissions reductions than the new convex hull point are dominated by other technologies and therefore were dropped. The minimum slope from this point was determined, identifying the next point on the convex hull. The process was repeated until the technology options were exhausted for each source.

For a given source, cost and NO_x emissions-reduction information for the technologies on the convex hull was retained in the form of marginal costs and marginal NO_x emissions reductions. Total cost and NO_x emissions reductions can be derived from the marginal values.

For a given source, total cost of technology,

$$i = \sum_{j \leq i} c_j x_j$$

where c_j is the marginal cost of technology *j*, where the *j* technologies are less expensive than the *i* technology, and x_j is the marginal NO_x emissions reduction of technology *j*.

Total NO_x emissions reductions from the source for technology *i* is the summation of the emissions reductions from the less costly technologies, or

$$\sum_{j \leq i} x_j$$

When ancillary ozone benefits were incorporated into the analysis, both gross and net (gross minus ancillary benefits) marginal costs were calculated. The convex hull based on minimum gross marginal costs (retaining the net marginal cost information) was determined for use in cost minimization scenarios optimizing over gross costs, while the convex hull based on minimum net marginal cost (retaining the gross marginal cost information) was determined for cost minimization scenarios optimizing over net costs.

Optimization Process:

Most optimization scenarios minimized the total cost of NO_x emissions reductions:

$$\min \sum_{k,j} c_{kj} x_{kj}$$

where

k = index of sources,

j = index of abatement options on convex hull for a source ,

c = marginal cost of technology j given technology $j - 1$ for source k, and

x = marginal NO_x emissions reduction of technology j given technology $j - 1$ for source k.

The NO_x emissions reduction for a given source and technology option was constrained not to exceed the amount determined by the convex hull:

$$x_{kj} \le d_{kj} \forall k, j$$

where d is the marginal NO_x emissions reduction as determined by the convex hull.

Scenarios that explicitly accounted for a source's impact on ozone or PM included the constraint:

$$\sum_{k} a_k \left(\sum_{j} x_{kj} \right) \ge L$$

where

L = target level of ozone or PM reduction and

a = coefficient converting NO_x emissions into ozone.

Scenarios not accounting for sources' impacts on ozone or PM used the constraint:

$$\sum_{k,j} x_{kj} \ge E$$

where E equals the NO_x emissions-reduction constraint.

Appendix B

Appendix B1. Population-Weighted Source-Receptor Matrices for PM and Ozone

Table B1a. PM (μg/m³ per 1,000 tons NO_x)

	Source Region					
Receptor Region	*MD-VA*	*E.PA-NJ*	*NY*	*OH-W.PA-WV*	*IND-KY*	*TN-South*
MD-VA	0.0052	0.0014	0.0003	0.0030	0.0008	0.0007
E.PA-NJ	0.0032	0.0173	0.0022	0.0046	0.0006	0.0002
NY	0.0003	0.0014	0.0014	0.0006	0.0002	0.0000
OH-W.PA-WV	0.0004	0.0003	0.0003	0.0062	0.0048	0.0009
IND-KY	0.0000	0.0000	0.0001	0.0015	0.0143	0.0025
TN-South	0.0003	0.0000	0.0000	0.0005	0.0036	0.0097
New England	0.0022	0.0125	0.0124	0.0031	0.0003	0.0001

Table B1b. Ozone (Average Episode, ppb per 1,000 tons NO_x)

	Source Region					
Receptor Region	*MD-VA*	*E.PA-NJ*	*NY*	*OH-W.PA-WV*	*IND-KY*	*TN-South*
MD-VA	2.7320	0.3635	0.0516	1.3029	0.1463	0.1794
E.PA-NJ	0.4493	2.6331	0.5710	1.1665	0.1270	0.0120
NY	0.0026	0.5583	2.4430	0.6007	0.2132	0.0412
OH-W.PA-WV	0.0941	0.0077	0.0025	1.7857	1.3706	0.3025
IND-KY	0.0036	0.0002	0.0002	0.3099	2.7504	0.9347
TN-South	0.4417	0.0007	0.0002	0.2986	0.4982	3.3256
New England	0.0439	0.6789	2.8298	0.5555	0.1248	0.0090

Table B1c. Ozone (Northeast Episode, ppb per 1,000 tons NO_x)

	Source Region					
Receptor Region	*MD-VA*	*E.PA-NJ*	*NY*	*OH-W.PA-WV*	*IND-KY*	*TN-South*
MD-VA	3.7101	–0.0432	0.0006	0.7401	0.0413	0.1932
E.PA-NJ	0.9015	2.4772	0.1331	2.1351	0.2148	0.0360
NY	0.0016	0.6739	2.3239	0.6983	0.4390	0.1236
OH-W.PA-WV	0.0704	0.0024	0.0059	1.3116	2.4372	0.8406
IND-KY	0.0016	0.0001	0.0002	0.0088	2.2100	2.6808
TN-South	0.0316	0.0001	0.0001	–0.0011	–0.0002	2.5335
New England	0.0645	0.4334	2.9909	1.3254	0.2822	0.0269

Table B1d. Ozone (Midwest Episode, ppb per 1,000 tons NO_x)

	Source Region					
Receptor Region	*MD-VA*	*E.PA-NJ*	*NY*	*OH-W.PA-WV*	*IND-KY*	*TN-South*
MD-VA	2.5916	1.0121	0.1082	0.6232	0.0232	0.0479
E.PA-NJ	0.4078	3.8519	0.1784	0.5675	0.0141	0.0000
NY	0.0061	0.4729	3.8936	0.9633	0.0501	0.0001
OH-W.PA-WV	0.2116	0.0174	–0.0007	2.9821	0.7294	0.0098
IND-KY	0.0090	0.0003	0.0001	0.4196	3.0427	0.0566
TN-South	1.2003	0.0020	0.0002	0.3563	1.2619	2.5379
New England	0.0672	1.7536	2.1528	0.1710	0.0069	0.0001

Table B1e. Ozone (Southeast Episode, ppb per 1,000 tons NO_x)

	Source Region					
Receptor Region	*MD-VA*	*E.PA-NJ*	*NY*	*OH-W.PA-WV*	*IND-KY*	*TN-South*
MD-VA	1.8944	0.1214	0.0461	2.5454	0.3746	0.2970
E.PA-NJ	0.0386	1.5704	1.4014	0.7968	0.1521	0.0000
NY	0.0000	0.5280	1.1116	0.1407	0.1503	0.0000
OH-W.PA-WV	0.0002	0.0032	0.0022	1.0635	0.9453	0.0571
IND-KY	0.0004	0.0002	0.0004	0.5011	2.9984	0.0667
TN-South	0.0934	0.0000	0.0002	0.5406	0.2328	4.9055
New England	–0.0001	–0.1504	3.3457	0.1702	0.0854	0.0000

Appendix B2. Annual Unit Benefits for Health Effects from Ozone and PM2.5.

Table B2a. Mid and High Unit Benefits for Ozone (Seasonal)

	5%	*Mean*	*95%*
Morbidity	2.30	4.90	9.40
Mortality	0	0	65.80
Mid		$4.90	
High			$75.20

Table B2b. Mid Unit Benefits for Nitrate Particulates (Nitrates Potency of PM10, Mortality Only)

	5%	*Mean*	*95%*
Morbidity	3.50	6.00	8.80
Mortality (VSL Approach)	22.00	52.00	112.00
Mid		$58	

Table B2c. High Unit Benefits for Nitrate Particulates (Nitrates as Potent as Sulfates, Mortality Only)

	5%	*Mean*	*95%*
Morbidity	3.50	6.00	8.80
Mortality (VSL Approach)	40.00	102.00	230.00
High			$239

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