# The Chesapeake Bay and the Control of NOx Emissions: A Policy Analysis

Alan Krupnick, Virginia McConnell, David Austin, Matt Cannon, Terrell Stoessell, and Brian Morton

Discussion Paper 98-46

August 1998

RESOURCES FOR THE FUTURE

1616 P Street, NW
Washington, DC 20036
Telephone 202-328-5000
Fax 202-939-3460

# The Chesapeake Bay and the Control of NOx Emissions: A Policy Analysis

Alan Krupnick, Virginia McConnell, David Austin,
Matt Cannon, Terrell Stoessell, and Brian Morton

**Abstract**

Nitrogen oxide emissions not only affect air quality but have recently been found to be an important source of nitrate pollution in the Chesapeake Bay. This analysis examines the costs, emissions, source-specific and location-specific allocations of NOx emissions reductions and the ancillary ozone related health benefits under a range of policy scenarios. The paper includes analysis of three separate policies. The first is a detailed analysis of the effect on nitrate loadings to the Bay of command and control policies specified in the Clean Air Act and as part of the OTAG process. The second is a comparison of alternative scenarios for reducing NOx emissions that meet nitrate loading goals, with or without concern for reducing ozone concentrations and the health effects they cause. The third is a comparison of alternative approaches to allocate NOx emissions to meet NOx reduction and ozone exposure goals while capturing the ancillary effect on nitrate loadings. This last analysis focuses on the stake the Bay jurisdictions have in the outcome of negotiations over NOx trading programs being developed by EPA for reducing ozone in the Eastern U.S. With the primary focus on the Chesapeake Bay jurisdiction, all three analyses integrate the ancillary ozone benefits of policies to reduce nitrate pollution, including examination of how these ancillary benefits change under alternative meteorological episodes, and explore lower cost alternatives to current regulatory programs in both qualitative and quantitative terms.

We find that the Chesapeake Bay benefits from efforts to reduce NOx emissions to meet the ambient air quality standard for ozone. Airborne NOx emission reductions slated to occur under the Clean Air Act in the Bay airshed will reduce nitrate loadings to the Bay by about 27 percent of the baseline airborne levels. The additional controls of NOx contemplated in what we term the OTAG scenario is estimated to result in an additional 20 percent reduction from this baseline. However, the paper's analysis of possible least cost options shows that the costs of obtaining such reductions can be significantly reduced by rearranging the allocation of emissions reductions to take advantage of source-type and locational considerations. In addition, we find that adding consideration of ancillary ozone-related health benefits to the picture does not alter any qualitative conclusions. Quantitatively, unless a link between ozone and mortality risk is assumed, the benefits are too small to affect the cost-saving allocations of NOx reductions. If the case for such a link can be made, the results change dramatically, with large overall increases in NOx reductions and a relative shift in controls to non-Bay states and utility sources. These specific effects are sensitive to the source-receptor coefficients linking NOx to ozone, however.

Our analyses also suggest that the Bay jurisdictions have a stake in the outcome of the NOx trading debate -- that some trading designs can lead to better outcomes for these jurisdictions than others. Nevertheless, a common feature of cost-savings policies is that they both rearrange emissions reductions and, in the aggregate, reduce emissions less than a command and control system. Thus, some trading regimes result in significantly smaller loadings reductions (up to 25 percent smaller) than the command and control approach.

Key Words: Chesapeake Bay, cost effectiveness, air pollution

JEL Classification Numbers: Q20, Q25, Q28

## Acknowledgments

The authors would like to acknowledge Erica Laich and her staff at E. H. Pechan Associates for providing 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 NOx emissions to ozone and to Paul for his sound judgment and counsel on the numerous assumptions undergirding an analysis like ours. Within RFF, we had excellent initial research assistance from Gar Ragland and invaluable discussions with Dallas Burtraw and Karen Palmer. 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 and the EPA's Acid Rain Office. Will Smith provided enormous moral, logistical, and intellectual support as well. Moral and intellectual support also came from the Chesapeake Bay Program Office, particularly Rick Batiuk and Julie Thomas. Finally, we would like to thank the attendees at our two project briefings for their interest and comments -- many of those mentioned above and other EPA staff as well as John Sherwell from Maryland's Department of Natural Resources and Mike Uhart from NOAA.

## Table of Contents



## List of Figures and Tables

# THE CHESAPEAKE BAY AND THE CONTROL OF NOX EMISSIONS: A POLICY ANALYSIS

Alan Krupnick, Virginia McConnell, David Austin,
Matt Cannon, Terrell Stoessell, and Brian Morton *

## I. INTRODUCTION

After years of worsening water quality in the Chesapeake Bay caused primarily by eutrophication from excessive nutrient inflows, the Governors of Maryland, Virginia, and Pennsylvania, and the Mayor of D.C. signed the Chesapeake Bay Agreement in 1987.[1] This agreement required 40 percent reductions in the controllable fraction of nutrients reaching the Chesapeake Bay by the year 2000. Municipal treatment plants and agricultural runoff were thought to be the primary sources of such nutrients. By 1992, bans on phosphorus in detergents and improvements in municipal sewage treatment, as well as some voluntary controls on farm runoff and land use restrictions helped bring about a 20 percent reduction in phosphorus loadings, but only a 4 percent reduction in nitrates. The relatively recent finding that emissions of nitrogen oxides (NOx) from the air are a major source of nutrient enrichment in the Chesapeake Bay, comprising anywhere from 20 percent to 35 percent of the Bay's controllable nitrate loads (Dennis, 1997), sparked major interest in pursuing NOx emissions reductions for Bay improvement.

During the same period, NOx emissions in the eastern U.S. have become the focus of efforts to meet ambient air quality goals. The Clean Air Act Amendments of 1990 (CAAA) initiated several NOx emissions reduction policies to meet both the ambient ozone standard (because NOx is an ozone precursor along with volatile organic compounds (VOCs)), and as part of the program to reduce acid deposition of sulfur dioxide ($SO_2$) and NOx emissions from utilities. In recognition of the importance of long-range transport of ozone in the eastern U.S. and the crucial role played by NOx in ozone formation, the Ozone Transport Assessment Group (OTAG) and the subsequent "Proposed NOx Trading Guidelines" issued by EPA supported the regional trading of NOx emissions to reduce ozone cost-effectively. The recent EPA rules tightening the ozone standard and setting a new fine particle standard

---

* Alan Krupnick, Senior Fellow and Acting Division Director, Quality of the Environment Division, Resources for the Future; Virginia McConnell, Senior Fellow, and David Austin, Fellow, Resources for the Future; Matt Cannon, Research Associate and Terrell Stoessell, Research Assistant, Resources for the Future; Brian Morton, EC/R Inc.

1 Chesapeake Executive Council, 1987.

will put further focus on NOx emissions reductions.[2] The possibility that the Chesapeake Bay may be the recipient of these "costless" nitrate loading reductions[3] as a result of both the 1990 Clean Air Act and the new EPA ambient air quality standards suggests that these benefits should be quantified.

This paper takes several perspectives on the issue of nitrate loadings to the Bay coming from airborne sources. It examines the costs, emissions, source-specific and location-specific allocations of NOx emissions reductions, and the ancillary ozone-related health benefits associated with NOx emissions reductions under a range of policy *scenarios*. These include *command and control* scenarios -- those applicable to states responding to the Clean Air Act Amendments of 1990 and to recent suggestion for further NOx emissions reductions made by OTAG. They also include scenarios designed either for meeting nitrate loading reduction targets or ozone exposure reduction targets in a cost-effective manner. The former target is examined as well in a model that permits ancillary ozone-related health benefits to be factored into the abatement allocation decision.

Specifically, this paper addresses three different issues:

*Policy Analysis 1. The Impact of Clean Air Act Regulations and OTAG Controls*. We examine the impact of NOx emissions reductions, which are anticipated to occur under existing air regulations targeted toward meeting ambient ozone standards, but which will also reduce nitrate loading to the Bay. Two separate "command and control" scenarios are considered: a Clean Air Act (CAA) scenario and additional mandated controls anticipated as part of what we term the OTAG scenario (one aspect of which is the reduction of NOx by electric utilities to meet a 0.15lb NOx/mmBTU performance standard; see below). We examine the impact of these regulations on nitrate loading and give estimates of both gross costs and costs net of the ozone-related health benefits.

*Policy Analysis 2: Comparison of the OTAG Command and Control Scenario to Other Scenarios for Achieving Nitrate Loadings Reductions*. This analysis takes the CAA scenario as the starting point and compares the outcomes from the OTAG command and control scenario to various alternatives:

---

[2] The implications of NOx emissions reductions for fine particle concentrations will be examined in a subsequent analysis. Though only 3-6% of the inventory of fine particles in the eastern U.S. is thought to be nitrates (formed through the conversion of NOx to nitrates in the presence of ammonia in the air), with a far higher fraction (30-40%) thought to be sulfates (from SO2 emissions), the health benefits of NOx control realized through the nitrate channel are estimated by EPA to outweigh the health benefits realized through the effect of NOx emissions reductions on ozone concentrations.

The most recent report from the Interagency Monitoring of Protected Visual Environments (IMPROVE) suggests a different composition for the inventory. For the Washington DC area, fine particles of nitrates are 15.1% in the spring, 5.6% in the summer, 12.8% in the fall, and 21.5% in the winter. Sulfates make up 46.6% of the concentrations in the spring, 60.4% in the summer, 42.7% in the fall, and 32% in the winter. While DC pollution is not necessarily representative of the entire Chesapeake Bay airshed, these numbers could be used as an upper limit of what to expect from fine particles in the immediate Chesapeake Bay region (Sisler, 1996).

[3] Costless from the point of view of regulatory activity associated with the Bay.

*a*. To the extent that the Chesapeake Bay policy community has an interest in focusing efforts for NOx emissions reductions on improvements in the Bay, it would be of interest to know the least cost way of getting nitrate loadings reductions and how this would compare to using the command and control (CAC) approach. Alternatively, the Bay community might want to include the maximization of ozone reduction benefits that stem from NOx emissions reductions into its objectives, having this factor as well as nitrate loading goals influence the allocation of NOx emissions reductions.

*b*. In another scenario, we examine the case in which no further NOx emissions reductions would be forthcoming in the Chesapeake Bay airshed after those specified in the CAA scenario and that, under these conditions, the Bay jurisdictions decide to "go it alone." That is, the Bay community on its own might focus on cleaning up the Bay, seeking NOx emissions reductions in the Bay jurisdictions for reducing nitrate loading. Or the community might want to focus on both nitrate loading and ozone benefit objectives. Given what they might otherwise have to do in an OTAG scenario, how would such other, more local policies compare?

*Policy Analysis 3. National NOx Policy and the Bay Community's Stake***.** EPA's efforts to design a NOx trading program for the eastern U.S. have implications for the jurisdictions in the Chesapeake Bay Agreement. This analysis examines whether there are some design choices that should be favored or opposed by the Bay jurisdictions, according to the effect these choices have on nitrate loading, the health of the population living in the Bay jurisdictions, and the abatement costs to NOx sources located in the Bay jurisdictions. This analysis, by focusing on a 13-state domain including DC, also provides implications for the overall design of a NOx trading system.

## II. PREVIOUS RESEARCH

An initial attempt to quantify the reduction in nitrate loading that might be realized from the Clean Air Act and some additional policy initiatives was made by Pechan et al. (1996). Pechan began with a 2005 emissions inventory developed as input to the EPA's Regional Oxidant Model (ROM) and this baseline was reduced according to assumptions associated with the implementation of the technology-based elements of the 1990 CAAA and some further reductions associated with new initiatives in discussion at that time.[4] The cost

[4] The OTC-Low Emission Vehicle petition and a possible requirement that electric utilities reduce NOx emissions to the lesser of 0.15 lbs NOx/million Btu boiler heat input or 85% reduction and other large point sources meet the same rate reduction or 70% reduction, whichever is lower. Scenario "C2" applied these restrictions to the Bay jurisdictions alone. Scenario "E" applied them to the entire Bay airshed, defined in Dennis (1996) as the geographic area whose NOx emissions account for 75% of the RADM model's predictions of Bay watershed deposition.

functions relating NOx emissions reductions to costs for pollution controls were estimated for electric utilities, other point sources, and mobile sources, taken from Pechan's Emission Reduction and Cost Analysis Model for NOx (ERCAM-NOx). The effects on nitrate loading were examined by pairing source-receptor coefficients that convert NOx emissions to nitrate deposition by Bay sub-basin (taken from runs of the Regional Acid Deposition Model (RADM)) to coefficients from the Chesapeake Bay Watershed Model, that in turn estimate the share of deposition by sub-basin to reach the Bay as nitrate loading.

Pechan (1996) found that the Clean Air Act applied to NOx sources in the Bay airshed states (Delaware, D.C., Indiana, Kentucky, Maryland, Michigan, New Jersey, New York, North Carolina, Ohio, Pennsylvania, Tennessee, Virginia, and West Virginia) resulted in reductions of 11.6 million pounds of nitrate loading at an average cost per pound reduced of $123. A scenario that examined additional controls in the Ozone Transport Region brought about a further 5.6 million pounds of reductions, raising the average costs to $147 per pound. Utility reductions were by far the cheapest, at $95/pound, versus $329 per pound from Low Emission Vehicles (LEVs) and $466 per pound from non-utility point sources.

Because NOx emissions reductions within the Bay jurisdictions are more productive for reducing nitrate loading to the Bay than are reductions from further away, the cost per pound for the Clean Air Act scenario applied *only* to the Bay jurisdictions is $75, with utility reductions in NOx the most cost-effective. A state like Kentucky was found to reduce a pound of nitrates, other things equal, at almost six times the cost of reducing a pound in the state of Maryland.[5] Overall, however, reductions in NOx emissions were found to be cost-competitive for reducing nitrate loading when compared with at least some forms of water-based loading reduction costs, even for air sources within the entire airshed. For instance, urban non-point source reductions were estimated to cost $143 per pound nitrates, although for some agricultural practices (such as low-till, which utilizes less fertilizers than standard tillage) costs drop to about $7 per pound (compared to the lowest estimate for any airborne source-state combination of $39 for Maryland utilities).

There are a number of ways the Pechan analysis can be improved, updated and extended. First, the Pechan analysis attributes all costs of air NOx emissions reductions to nitrate loading declines in the Chesapeake Bay. However, most of these reductions have been done or will be undertaken to improve air pollution and the associated health benefits, so attributing all of the costs to nitrate reductions in the Bay overstates the costs. At a minimum, if the costs of reducing nitrate loading to the Bay are attributed to the Bay cleanup, at the least the ancillary benefits from ambient ozone or other air quality improvements should first be subtracted from these costs. We show in the theoretical section below how and why the air quality improvements should be netted out of the costs of nutrient reduction. This netting out

[5] For utilities only, Maryland utilities meet the 0.15 lb/mmBtu restriction at a cost of $1,200 per ton NOx while Kentucky utilities meet it at $1,100. In terms of nitrate loadings, Maryland utilities face a cost of $39 per pound while Kentucky's utilities register at $254 per pound.

could not only significantly lower the average cost of nitrate loadings reductions from the air but will also be likely to alter the geographic and source-specific cost per pound comparisons.

Second, the analysis only considers existing command and control measures for reducing NOx. It does not examine alternative economic incentive control options that might achieve the same goals, but at lower overall costs. This may be important in the future, as more stringent policies continue to push the additional costs of control higher. Also, in light of the support for NOx trading from OTAG participants and the EPA, it would be useful to also consider the level and distribution of costs per pound associated with a tradable NOx permit scheme. This could be compared to a tradable ambient nitrate loading permit scheme. The latter is equivalent under certain conditions to a least cost allocation of NOx emissions reductions to meet any given nitrate loading reduction goal.

Finally, the Pechan analysis used emissions and cost estimates which Pechan has since updated, as part of its work in developing the OTAG inventory. Thus, the purpose of our analysis is to extend the initial Pechan analysis by examining the impacts and costs of both command and control and economic incentive (or lower cost) scenarios for reducing nitrate loading and NOx emissions, capturing the joint health benefits associated with ancillary reductions in ozone**,** and to do so using more up-to-date emissions and cost data.

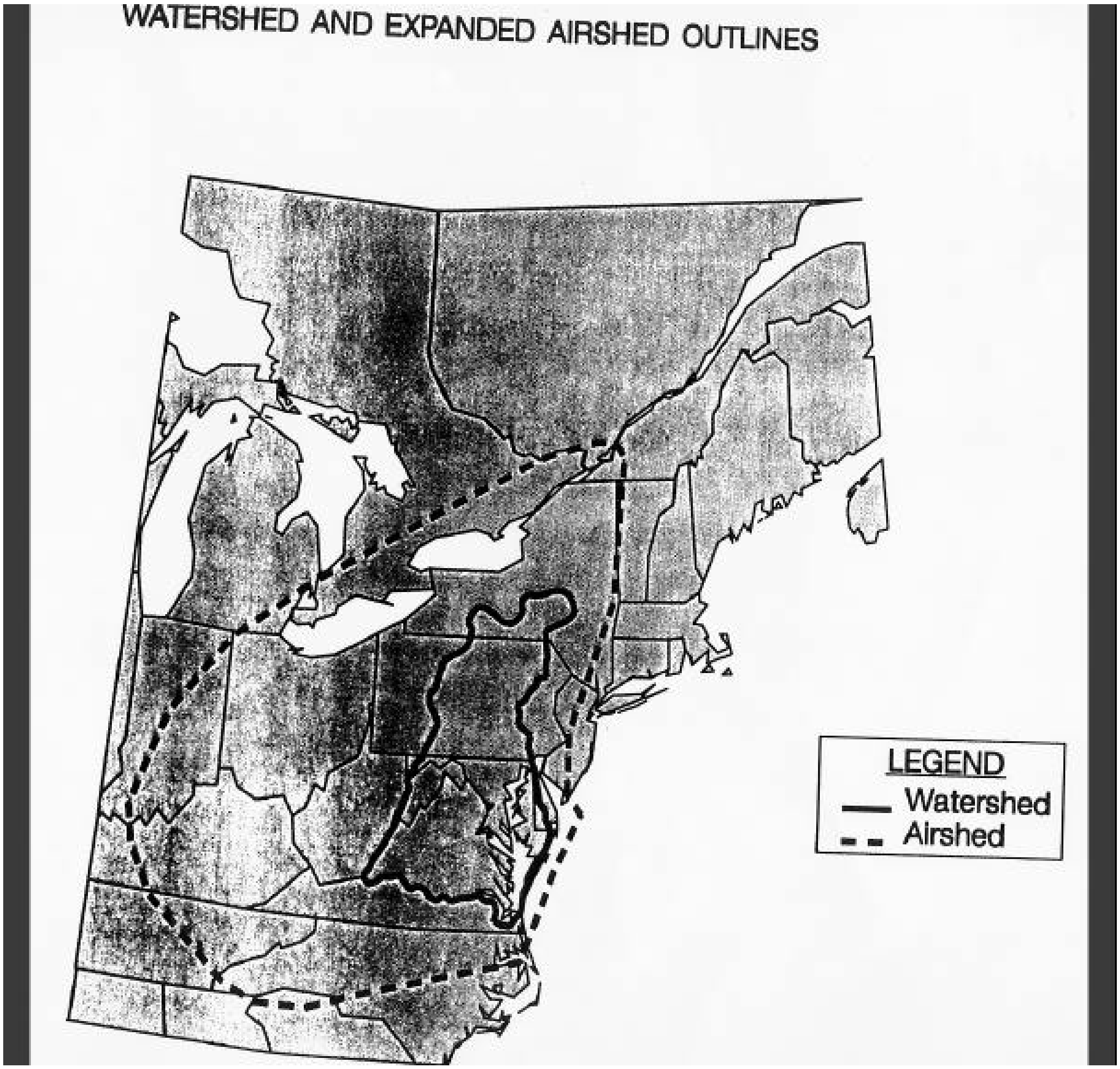
## III. DOMAINS AND SCENARIO DEFINITIONS

### A. Domains

The NOx emissions sources included in the analysis are located in the Chesapeake Bay airshed, as defined in Dennis (1996) using the Regional Atmospheric Deposition Model (RADM). This domain is shown in Figure 1 (along with the Chesapeake Bay watershed). Nitrogen oxides are emitted from sources in the airshed and transformed to nitrates. A portion of the nitrates reaches the watershed, is deposited to the land surface, of which a small fraction is released to the streams and rivers and the resulting nitrates are transported to the Chesapeake Bay. Some deposition is also directly to the Bay itself. Dennis shows that nearly every area in the U.S. east of the Mississippi River contributes something to the airborne nitrate deposition to the Bay watershed. But many regions have contributions too small to warrant inclusion in a major reduction effort. To focus on those regions that have the greatest impact, Dennis identified the areas that account for 75 percent of the watershed deposition. Based on this analysis, we included all or part of 13 states and the District of Columbia in our Chesapeake Bay source domain.

Because our analysis includes the effects of NOx emissions reductions on ambient ozone and on the health of people living in areas where ozone concentrations change, the domain for estimating these effects includes the source domain and any additional areas in the U.S. likely to experience significant ozone changes as a result of NOx emissions reductions in the source domain. These additional areas include Connecticut, Long Island, Massachusetts, and Rhode Island.

# Figure 1


WATERSHED AND EXPANDED AIRSHED OUTLINES
LEGEND
Watershed
Airshed

## B. Scenarios

Each scenarios is defined with emissions control relative to a baseline. We first discuss the possible baselines.

Baselines

The baseline for the CAA scenario is 1990 emissions projected to 2005 using Bureau of Economic Analysis (BEA) and other projections of activity levels. This baseline does not account for emissions reductions mandated in the CAAA of 1990. The baseline for all other analyses is the NOx emissions, abatement measures, and activity levels projected by Pechan & Associates for 2005 under the Clean Air Act Amendments of 1990. Pechan (1996) defines the CAA scenario by applying growth and control factors to the Interim Inventory (EPA, 1993). These control measures include: Reasonably Available Control Technology (RACT)-level NOx controls on major point sources in ozone nonattainment areas, Title IV (Acid Rain) NOx emissions reductions on steam-electric utilities, enhanced inspection and maintenance (I&M) programs in the more polluted ozone nonattainment areas (under Title b), federal Tier I vehicle emissions standards (under Title II)), 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.

Scenarios

There are three sets of scenarios that follow the major policy analyses presented in the paper. The first set are the command and control (CAC) scenarios; the second set are alternative scenarios that have the same goal of Bay nitrate loading reduction as the CAC scenarios, which we term the least-cost and emissions trading scenarios. The final scenarios deal with the design of possible NOx emission trading policies for meeting ozone reduction goals in the eastern U.S. We describe each of these briefly below.

*1. Command and control scenarios*

There are two command and control scenarios. The first is basically the Pechan-defined Clean Air Act scenario, referred to here as CAA. The second (OTAG) includes higher levels of control identified as part of the OTAG process. These were 0.15 lbs NOx/mmBTU of boiler heat input, or 85 percent NOx removal for utilities and 70 percent NOx removal for non-utility point sources, whichever is less. EPA subsequently used approximately the same standards to undergird its State Implementation Plan (SIP) call to the states (FR, 1997).

Mobile sources were virtually ignored in the OTAG process. However, the original Chesapeake Bay Cost Allocation Study (Pechan, 1996) examined the effects of implementing a Low Emissions Vehicle Program in the Ozone Transport Commission Region (the Northeastern States). We require LEVs in all areas in our OTAG scenario.

*2. Alternative scenarios for achieving Bay nitrate loading reductions*

There are a number of scenarios in this part of the analysis, all of which take as their goal nitrate loading reductions that are equivalent to those resulting from the OTAG scenario.

*2a. Minimum costs for the entire Airshed*

- We examine first the allocation of NOx emissions reductions among sources and regions that would provide the lowest cost way of meeting the nitrate loading reduction target, attributing all of the costs to nitrate loading reductions.
- An extension of that scenario is to include the health benefits realized through the impact of NOx emissions reductions on ozone concentrations as part of the cost minimization algorithm. We examine both "mid" and "high" estimates of those ozone benefits (see below).
- Another scenario examines the allocation of reductions if the costs of NOx emissions reductions were minimized, while still achieving the same nitrate loading reduction goal. In this case, the allocation of NOx emissions proceeds by equalizing marginal costs per unit of NOx emissions reductions rather than the marginal costs per unit of nitrate loading reductions. One can think of this scenario as a trading system, like that of the $S0_2$ allowance market, where emissions trade on a one-to-one basis. Such a system will be less efficient than an ambient system where the effects of a ton of NOx on loadings are different depending on the location and type of source. However, there is experience with such a system and at least the perception that an emissions trading system has lower education and transactions costs than an ambient permit system.

*2b. Bay states undertake low cost policies independently*

- Finally, we include scenarios that focus on nitrate loading reductions only from the Bay jurisdictions,

The least-cost scenarios can also be thought of as similar to ambient trading programs. For instance, one can envision a perfectly operating allowance trading market where the currency of the trade is nitrate loadings to the Bay and a cap is set on nitrate loadings equal to that predicted by the OTAG scenario. This type of scenario has been shown to yield equivalent results to the least cost allocation of emission reductions to an ambient goal, in this case, in terms of nitrate loadings (Montgomery, 1972).

This system implies that sources whose NOx emissions have differential effects on nitrate loadings will trade their NOx emissions at ratios different than one. For instance, assume an electric utility in Ohio buys nitrate loading permits from an electric utility in Maryland, but the Ohio utility's NOx emissions have only half the effect on Bay loadings as that of the Maryland utility. Then, the Ohio utility could increase its NOx emissions by twice the reduction in NOx emissions from the Maryland utility, without changing loadings to the Bay. Thus, the trading ratio in terms of NOx emissions between these sources would be set at 2:1.

All of these scenarios are to some extent hypothetical, since air pollution policies are unlikely to focus primarily on Bay clean-up goals. They are informative, however, in that they help identify the lowest cost sources of additional reductions.

Note also that, in both the CAC and the alternative scenarios for meeting nitrate loading goals, NOx emissions reductions yield benefits from reduced ozone exposures that may be considered ancillary to the goal of improving the Bay. All analyses that involve meeting nitrate targets include estimates of these ancillary benefits. Because of the wide confidence intervals associated with health benefits estimates, we also perform some sensitivity analyses with 5 percent and 95 percent confidence interval estimates of benefits per ppb ozone-person-day (see below for a full discussion about how these benefits are estimated). In addition, the ozone benefits from NOx emissions reductions can be included directly in the NOx emissions allocation decision, with the effect that the allocation seeks to meet the nitrate loading reduction goal while minimizing the *net* costs of NOx control, which is the gross costs minus the ancillary ozone benefits (see the next section for the formal model).

*3. Alternative scenarios for reducing ozone exposures*

In this part of the analysis, we are interested in whether some options for NOx trading for ozone improvements are more or less favorable to the Bay jurisdictions than other options. Along the way, we examine the implications for abatement costs across the airshed states.

First, we consider the OTAG regulatory allocation of NOx emissions reductions and compare it to various systems of emissions trading, including EPA's plan that limits trading on a one-to-one emissions basis and only between utilities. We then examine variations on EPA's emissions trading plan, adding additional source types to trading and considering plans that would trade emissions at ratios guided by the effect of a source at a given location on ozone exposures to the population in a receptor region. The Bay jurisdictions' interests are described in terms of nitrate loading reductions, ozone exposure benefits, and abatement costs to Bay jurisdiction sources. We then address questions about the sensitivity of the results to alternative meteorological regimes. Finally, we examine a zonal trading plan where the Bay jurisdictions define their own trading zone in which more attention can be given to nitrate loading reductions.

In contrast to the scenarios for nitrate improvements, these scenarios have direct policy-relevance to the shape of a future NOx trading plan. No government officials (but some academics) are discussing the possibility of trading against an ozone exposure reduction goal.

All analyses provide estimates of the annual costs and benefits for each scenario compared to some baseline for the year 2005, expressed in 1990 dollars.

## IV. THE CONCEPTUAL MODEL

This section presents the conceptual model for allocating NOx emissions reductions cost-effectively to meet a nitrate loading reduction goal and, separately, an ozone exposure reduction goal. In addition, it specifies how the former goal can be met while simultaneously accounting in the optimization for the ancillary ozone-related health benefits associated with NOx emissions reductions.

Adopting notation used in Tietenberg (1985), in the least-cost model we seek to minimize total control costs over $J$ sources of nitrogen oxides,[6] given that we must reduce total Bay nitrate loading to some pre-established limits or targets $\overline{W_i}$ at each of $I$ measurement locations. If loadings are uniformly mixed, as we assume for the Chesapeake Bay, there will be only a single location and one limit $\overline{W}$, but for generality we assume multiple locations.

Letting $c_j(r_j)$ represent the cost of reducing NOx emissions at source $j$ by the amount $r_j$ -- from some baseline emissions level $\overline{e}_j$ -- and letting the function $L_{ji}(\cdot)$ map NOx emissions at source $j$ into Bay loadings at site $i$, the objective is simply to minimize the aggregate cost of those reductions across all sources, subject to achieving the water quality targets. The objective is:

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

*subject to*

$$-\overline{W_i} + \sum_{j=1}^{J} L_{ji}\left(\overline{e}_j - r_j\right) \leq 0 \tag{2}$$

That is, the regulator seeks to find the least expensive way of assuring that loadings $L_{ji}$ of the emissions remaining after controls are installed, $\left(\overline{e}_j - r_j\right)$, are no greater than the imposed water quality target $\overline{W_i}$. This model is easily adaptable to the case of meeting some target $\overline{X}$ for aggregate ozone exposures per "ozone season."[7] In that case, $\overline{X}$ would substitute for $\overline{W_i}$ in (2), and a set of source-receptor coefficients $d_{jk}$ mapping emissions at source j to ozone exposures at air receptor location k would replace the loadings matrix elements $L_{ji}$.[8]

---

[6] It is an open question how those J sources are selected. An extension of our work would be to derive the optimal extent of a regulatory region. This would involve trading off between regulator power, which presumably is greater the fewer sources that must be controlled, and costs of environmental control--as extending the regulatory domain could lower costs by creating increased gains from trading over a larger market, as well as possibly bringing large emissions sources under the control of the regulator.

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

[8] Because the ozone target is an aggregate one, the source-receptor coefficients $d_{jk}$ would be summed both over sources j and receptors k, and there would be a single standard.

In either case, total costs, summed over all emissions sources, are given by *C(R)*. To achieve an optimum given that reductions $r_j$ must be non-negative -- as must be the LaGrange multiplier $\lambda_i$ on the environmental target in (2), -- the Kuhn-Tucker conditions (see, e.g., Varian, 1992) must be satisfied. When the Bay loadings serve as the constraint, the key first-order condition for cost minimization, which we explain below, is:

$$\frac{\partial c_j}{\partial r_j} - \sum_{i=1}^{I} \lambda_i L_{ji} \geq 0 \qquad (3)$$

Cost minimization will not be achieved unless a "complementary slackness" condition is also satisfied, guaranteeing that for every source that optimally reduces its NOx emissions *(i.e.,* for which $r_j$>0), condition (3) is met with equality. In other words, at the optimum point, all sources are controlled to where their marginal control costs equal the sum of the shadow prices (the marginal cost of the last unit of pollution reduction) $\lambda_i$ of the environmental targets, weighted by the loadings $L_{ji}$ of their NOx emissions at each Bay receptor. For the single receptor (or uniform mixing) case, (3) implies that *sources are controlled to where the ratios of their marginal control costs, $MC_j$, to their Bay loading factors $L_j$ are all equal:*

$$\frac{MC_j}{L_j} = \lambda = \frac{MC_{j'}}{L_{j'}}$$

*for any sources j and j'*. Sources with marginal costs too high to achieve this would not reduce their emissions.[9]

This outcome can be achieved by regulatory fiat, but it has been shown elsewhere that a market-based, tradable permit approach, where permits for emissions, producing total loadings of $\overline{W_i}$ at each receptor *i*, can achieve the same outcome. (See Montgomery (1972) for initial development of a trading model in this context, and Krupnick et al. (1983) for refinements.) The same is true for the case of controlling emissions to meet an ozone exposure reduction target.[10] In a competitive trading market, equilibrium permit prices of $P_i = \lambda_i$ would arise at each (Bay or air) receptor site and, subject to the qualifications in Krupnick et al. (1983), would achieve the cost-minimizing outcome without regulatory intervention (beyond issuing the correct number of permits and setting the correct trading

[9] For the case of an aggregate ozone exposure reduction goal, there would be only a single shadow price . Each source's loading coefficient Lj in this expression would be replaced by the source's contribution $\sum_{k=1}^{K} D_j$ to total ozone exposures.

[10] In our least-cost simulations, we will take from the command-and-control case the ozone exposure reductions implied, and treat these reductions as the goal.

ratios). That is, separate markets would exist for permits specific to each receptor.[11] Or said another way, trading ratios would need to be set for trades between any two regions. The initial allocation of the permits need have no effect on the outcome. Under this scheme, low-cost sources would reduce emissions beyond what they are permitted to emit, and would sell their excess permits at price $P_i$ to high-cost sources.

The control of NOx sources to achieve water quality goals will inevitably also create benefits from improved air quality. These "ancillary" air benefits must be accounted for if emissions controls are to achieve the water quality target in a socially optimal manner. Here the problem is one of social welfare maximization, rather than cost minimization, given an imposed water quality target. That is, we seek to maximize ancillary air benefits $B_a$ net of the costs of achieving the target:

$$\max_{r_j} B_a\left(\sum_{j=1}^{J}\sum_{k=1}^{K} d_{jk}\cdot r_j\right) - \sum_{j=1}^{J} c_j(r) \qquad (4)$$

subject again to constraint (2). Here $d_{jk}$ is a source-receptor matrix mapping emissions at source $j$ to concentrations at (air) receptor $k$; and $c_j$ and $r_j$ are as before. (Note the substitution of "concentrations" for "loadings" when the constraint switches from water quality to air quality.) The first-order condition for maximization of (4) which is analogous to the first order condition (3) is:

$$\frac{\partial B_a}{\partial r_j}\sum_{k=1}^{K} d_{jk} - \frac{\partial c_j}{\partial r_j} + \sum_{i=1}^{I} \lambda_i L_{ji} \le 0 \qquad (5)$$

As before, complementary slackness implies that 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. This can be seen by rearranging terms in (5):

$$MC_j - MB_{a_j} = \sum_{i=1}^{I} \lambda_i L_{ji}.$$

($MB_{a_j}$ is the marginal ancillary benefits, $\left(\partial B_j / \partial r_j\right) * \sum_{k=1}^{K} d_{jk}$, from controlling source *j.)*

Where uniform mixing is assumed, this condition simplifies to

$$\frac{MC_j - MB_{a_j}}{L_j} = \lambda = \frac{MC_{j'} - MB_{a_{j'}}}{L_{j'}} \qquad (6)$$

---

[11] If different source types affect pollution transport (or exposures) differently, e.g., utilities with tall stacks create more ozone downwind than mobile sources, the source-receptor matrices would differ by source type as well as location. This implies that trading ratios in a marketable permit system, to maximize the gains from trade, should also differ by source types involved in the trade, whether or not the trade crosses regional boundaries.

for any two sources *j* and *j'*.

Three important results are implied by this equation. First, rather than controlling all sources to where their marginal control costs equal $L_j \lambda$ , the shadow price of the constraint weighted by their individual loading factors, here sources which produce air benefits are controlled more than they otherwise would be, as an increasing function of those air benefits. Second, accounting for the air benefits implies a lower shadow price $\lambda$ than before, one that reflects the true social cost of meeting the water-quality target. Finally, and perhaps most important, because the air benefits are external to the Bay clean-up, a trading market for Bay nitrates would not achieve the optimal outcome unless a mechanism were implemented to internalize each source's ancillary benefits in the abatement decision. The straightforward system for the *gross* cost-minimization model would not work here.

## V. DATA

### A. Emissions and Costs

The basic database for emissions and costs in the analysis which follows is contained and generated in a model developed by E. H. Pechan and Associates for EPA, called the Emission Reduction and Cost Analysis Model for Oxides of Nitrogen (ERCAM-NOx). This model is supplemented by VOC emissions data for mobile sources (needed for projecting ozone effects) and a few cost revisions to the E.H. Pechan cost assumptions.

ERCAM-NOx (1996) contains 1990 baseline NOx emissions for utilities, other point sources, mobile sources, and other area and nonroad sources and projects these emissions forward to 2005 using state-level, 2-digit Bureau of Economic Analysis earnings indicators and population projections. Cost and associated emission reduction estimates were developed for a variety of "engineering" abatement options for each source-type.[12] 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. Finally, specific abatement options by source are identified that would be in place in 2005 in response to requirements set by the 1990 Clean Air Act and its implementation as specified in a state's Implementation Plan.

Interested readers should refer to Pechan (1996) for details. Note that mobile source emissions are estimated at the county level by using county-specific data as input to the MOBILE5 Model, EPA's current generation vehicle emissions inventory model. The county level data include ambient temperature, the type of Inspection and Maintenance (I&M) program in place or forecast, and the type of reformulated gasoline to be used.

Developing reasonable estimates of the costs of mobile source abatement technologies for 2005 is difficult. Technological and political changes make the evolution of particular controls, such as LEVs and even enhanced I&M programs, challenging to forecast. The current Pechan database relies on state-by-state forecasts for the Clean Air Act scenario, but some of

[12] Facilities covering about 23% of the emissions from non-utility point sources have no abatement options in the Pechan database.

those forecasts are already proving to be inaccurate. In addition, the emission reduction forecasts in the Pechan database for certain mobile source strategies are assumed by EPA's MOBILE inventory model to be quite large. For example, the combination of LEV vehicles and enhanced I/M reduces NOx emissions by more than 25 times the reduction from a basic I/M program. These large forecast emissions reductions for the enhanced I/M alternatives come from the MOBILE Model which we think may be overstating the reduction potential of Enhanced I/M (Harrington, McConnell and Cannon, 1998). However, in this analysis, we use the Pechan data for mobile sources including the most recent cost estimates used by E.H. Pechan. In future work, we will use our own estimates of costs and emission reduction under different mobile source strategies and compare these to the Pechan estimates. Appendix 3 describes the methods for predicting cost in more detail. Finally, because mobile sources emit both NOx and VOCs, both of which contribute to the formation of ozone, we use the average of the NOx and VOC reductions as the "NOx emissions reduction" for calculating ozone benefits.

A particularly challenging issue in determining the cost of control from both point and mobile sources is how to count the costs of moving from one technology to another. If, for example, technology A is in place in the CAA scenario, but technology B would be used in the OTAG scenario, what are the costs of the B technology, given that A is already in place? There are a number of situations in which the costs are easy to determine -- if the B treatment can clearly be added on to the technology in place, then the costs of moving to B are just the costs of B. An example for utility sources is controls for wall-fired coal boilers. Low-NOx burners can be installed alone or in combination with overfire air (the "B" technology) to reduce NOx. The difference in cost between the two is the cost of the overfire air technology. For mobile sources, if reformulated gas is added to I&M which is already in place, the added costs are just the additional cost of the reformulated gas.

There are, however, many situations where the new controls require a switch from one technology to another, rather than an add-on. In the case of mobile sources, switching from one I&M type to another, or in the case of utilities, wall-fired coal boilers can change from low-NOx burners to selective non-catalytic reduction. In these situations we assume that the capital cost of the dropped technology is not recoverable for point sources and that 25 percent of the capital cost is recoverable from I&M programs. We also assume that the O&M costs of the dropped technology are fully recoverable.

### B. NOx to Nitrate Source-Receptor Coefficients

Outputs from runs of EPA's Regional Acidic Deposition Model (RADM) and the Chesapeake Bay Program's Watershed Model (CBWM)[13] are linked in Pechan (1996) to

---

[13] The CBWM is used to relate a kg of nitrate deposition by basin to nitrate loadings actually reaching the Chesapeake Bay. The CBWM (Phase III) (Linker, et al., 1993) simulates the land and water movement of nutrients from the Bay sub-basins to the Bay for a wide variety of nutrient sources, including airborne sources. The modeling domain covers the 64,000 square miles of the Chesapeake Bay's drainage system. This model, calibrated to weather conditions and water flows over the 1984-87 period, estimates total annual loads to the Bay of 324 million pounds of nitrates, of which 27%, or 105 million pounds, come from airborne sources.

convert NOx emissions at a source location (either substate, state or region) and for a source type to deposition in one of fifteen Chesapeake Bay Basins (in kg of wet plus dry nitrates per hectare per year), and deposition in these basins to nitrate loadings to the Chesapeake Bay (in pounds per year). Table 1 provides load to emission ratios by state. As a point of departure, our estimate of loadings from the point and mobile sources in our domain (which omits area and non-road sources and is for much lower NOx emissions than implied by the CBWM scenarios) is about 66 million pounds for the base year.

**Table 1. Load to Emission Ratios (lbs Nitrates / tons NOx)**

| | Source | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **State/City** | **All** | | **Utility** | | **Mobile** | | **Major Points** | |
| Atlanta | | | | | | | 2.40 | U,N,M |
| Delaware | 32.61 | N | 34.56 | U | 30.31 | M | | |
| Detroit | | | | | | | 6.31 | U,N,M |
| DC | 23.22 | N | 18.93 | U | 20.77 | M | | |
| Indiana | | | 4.42 | U,N,M | | | | |
| Kentucky | | | 4.42 | U,N,M | | | | |
| Maryland | 32.61 | N | 34.56 | U | 30.31 | M | | |
| New Jersey | 4.90 | U,N,M | | | | | | |
| New York | 17.72 | U,N,M | | | | | | |
| North Carolina | 7.57 | U,N,M | | | | | | |
| Ohio | 8.06 | U,N,M | | | | | | |
| Pennsylvania | 19.10 | N | 19.84 | U | 19.71 | M | | |
| East Pennsylvania | 21.34 | U,N,M | | | | | | |
| West Pennsylvania | 17.72 | U,N,M | | | | | | |
| Tennessee | 5.59 | U,N,M | | | | | | |
| West Virginia | | | 13.90 | U,N,M | | | | |
| Virginia | 23.22 | N | 18.93 | U | 20.77 | M | | |

**Codes:**

U: This ratio was used for utility sources.
N: This ratio was used for non-utility point sources.
M: This ratio was used for mobile sources.

## C. NOx to Ozone Source-Receptor Coefficients

One of the major areas of potential air quality benefits associated with reductions in NOx emissions is health effects associated with reduced ozone exposure. To model these health effects we linked NOx emission changes at point sources and NOx plus VOC emission changes for mobile sources to ozone changes at receptor areas. These linkages were made by running an ozone simulation model, the Urban Airshed Model (UAM-V), in what can be called an "attribution mode" (Guthrie and Krupnick, 1998) for three "typical" ozone episodes. The attribution mode involves developing a source-receptor matrix linking a source type/region's NOx emissions to a change in ozone in each receptor region for a given set of meteorological conditions. This is obtained by running the model for baseline emissions and then perturbing each source region's NOx emissions in turn and 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.

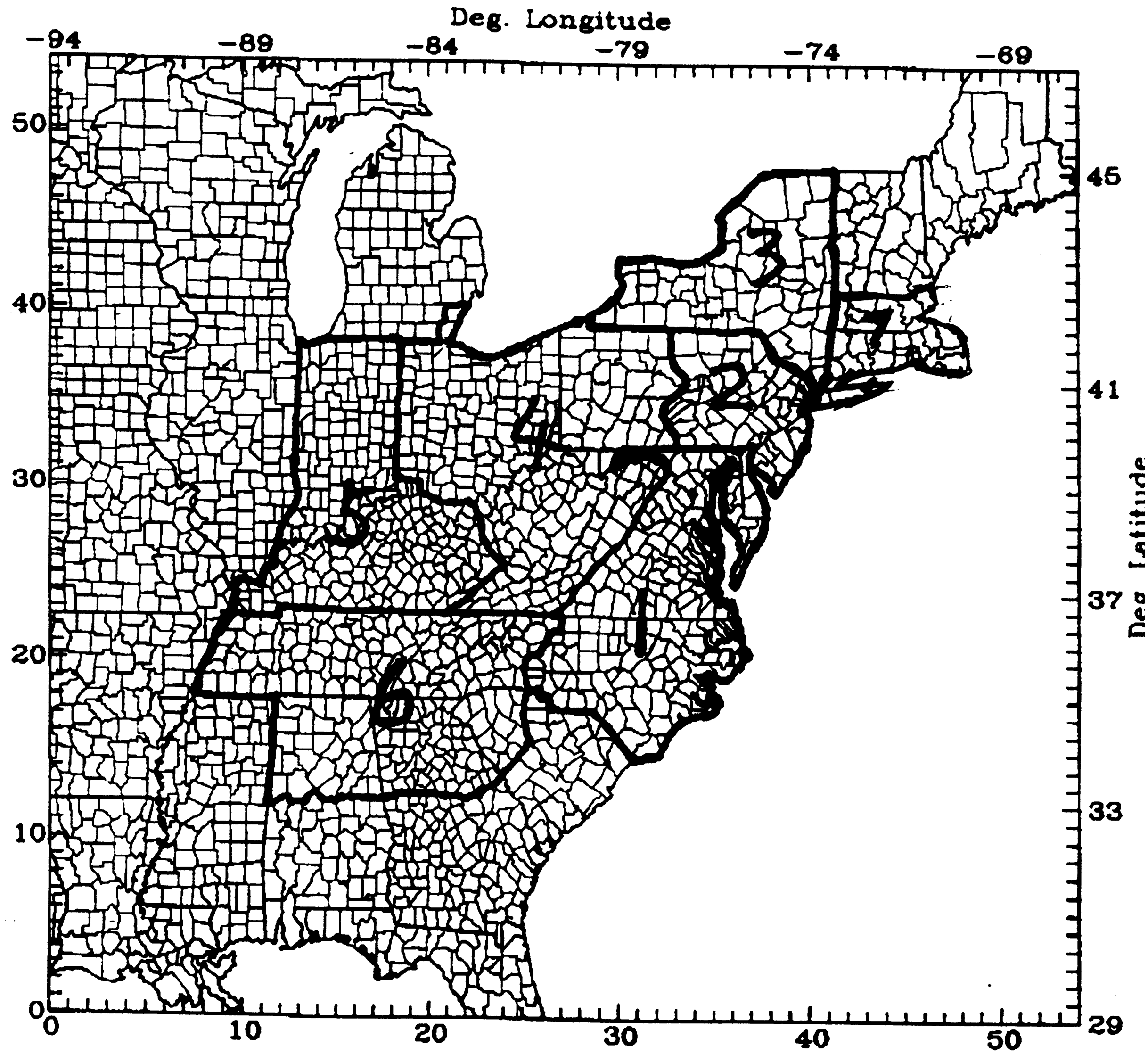
This approach may be starkly contrasted to the standard approach for using air quality models: the scenario mode. In this mode, a baseline set of NOx emissions is fed into this model which produces ozone concentrations. Then another set of NOx emissions is used, this time related to a particular control scenario applied over the study domain. The resulting changes in ozone are attributed to the NOx emissions reductions over the domain. However, no attributions are made to specific source regions or sources. In addition, these analyses usually are developed to examine nonattainment issues. Hence, the meteorological inputs to the model are taken from multi-day episodes of extremely high ozone concentrations.

This scenario mode has two major deficiencies for estimating ancillary benefits in a least-cost modeling framework such as ours. First, the restriction of episodes to extreme violations of ambient standards is at odds with the needs of a benefits analysis that attempts to measure annual or seasonal impacts. The former is performed to help judge when abatement plans are sufficient to reach attainment. A benefits analysis is conducted to total the benefits of season-long ozone reductions. Because it is generally agreed that there are no thresholds in the effects of ozone on health, and that the concentration-response functions are linear, ozone reductions from any baseline concentration generate benefits. Focusing only on extreme events, which, by definition are rare, would vastly underestimate the ancillary benefits of summer long ozone reductions. Rather, "typical" ozone episodes should be modeled to capture health benefits over the ozone season.

Second, the scenario mode is completely impractical for finding a least-cost allocation of NOx abatement. In this mode, hundreds, if not thousands, of trial and error runs would be needed to identify a combination of abatement by source and region that attains the goal at a minimum cost. On the other hand, if source-receptor coefficients linking NOx emissions at a source-type and region to ozone concentrations at receptors can be specified, an optimization model can be utilized, and the optimization takes a matter of seconds. Of course, source-receptor coefficients assume linearity or that the impacts of emissions changes are the same regardless of the level of emissions. This assumption needs to be tested and we are in the process of testing it now.

To use the UAM-V in "attribution mode," we defined six source regions (labeled 1 through 6 on Figure 2) and seven receptor regions (the same 6 source regions plus region 7: New York City, Long Island, Connecticut, Rhode Island and Massachusetts). These 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 follow state lines as much as possible. We also needed a small number of such regions to minimize the number of UAM-V runs required to develop the matrices. Finally, receptor region 7 was added, even though this area is not a source of Chesapeake Bay nitrates, to capture ozone-related effects of NOx emissions reductions within the source regions,

**Figure 2**


Deg. Longitude
−94
−89
−84
−79
−74
−69
50
40
30
20
10
0
0
10
20
30
40
50
45
41
37
33
29
Deg. Latitude
1
2
3
4
5
6
7

particularly New York and the Midwest. The resulting set of regions is by no means the only possible choice, and the degree of distinction between regions will obviously vary with weather conditions.

The overall approach to developing a particular "row" of a source-receptor matrix is to perturb emissions from the sources in one source region (with inventories taken from ROM inventories supplied by E. H. Pechan and prepared by SAI) and calculate the changes in daily one hour peak[14] ozone concentrations at each grid cell in the receptor regions and then compute either an unweighted or population-weighted average ozone change over the grid cells in a given receptor area, and then do the same thing for each source region in turn. This approach (which is detailed, along with all the results) is described in Guthrie, Mansell, and Gao (1998) and is a simplification of that used by Rao, Mount, and Dorris (1997).

We distinguish between S-Rs for NOx emitted at high altitudes, which would be applicable to major non-utility sources and utilities, and S-Rs for mobile sources, which perturb low-level NOx and anthropogenic VOC emissions by an equal percentage change. NOx emissions from all point sources in a region were reduced by 70 percent of the 2005 baseline in the UAM-V runs for generating the point source S-R matrices. This figure is within the range of expected NOx emissions reductions from point sources modeled within the OTAG process and underlies EPA's recent proposed SIP call. For mobile sources, we reduced NOx and anthropogenic VOC emissions by 20 percent from the 2005 baseline. This is roughly in the range of expected reductions from mobile source controls discussed by EPA and in the OTAG process.[15]

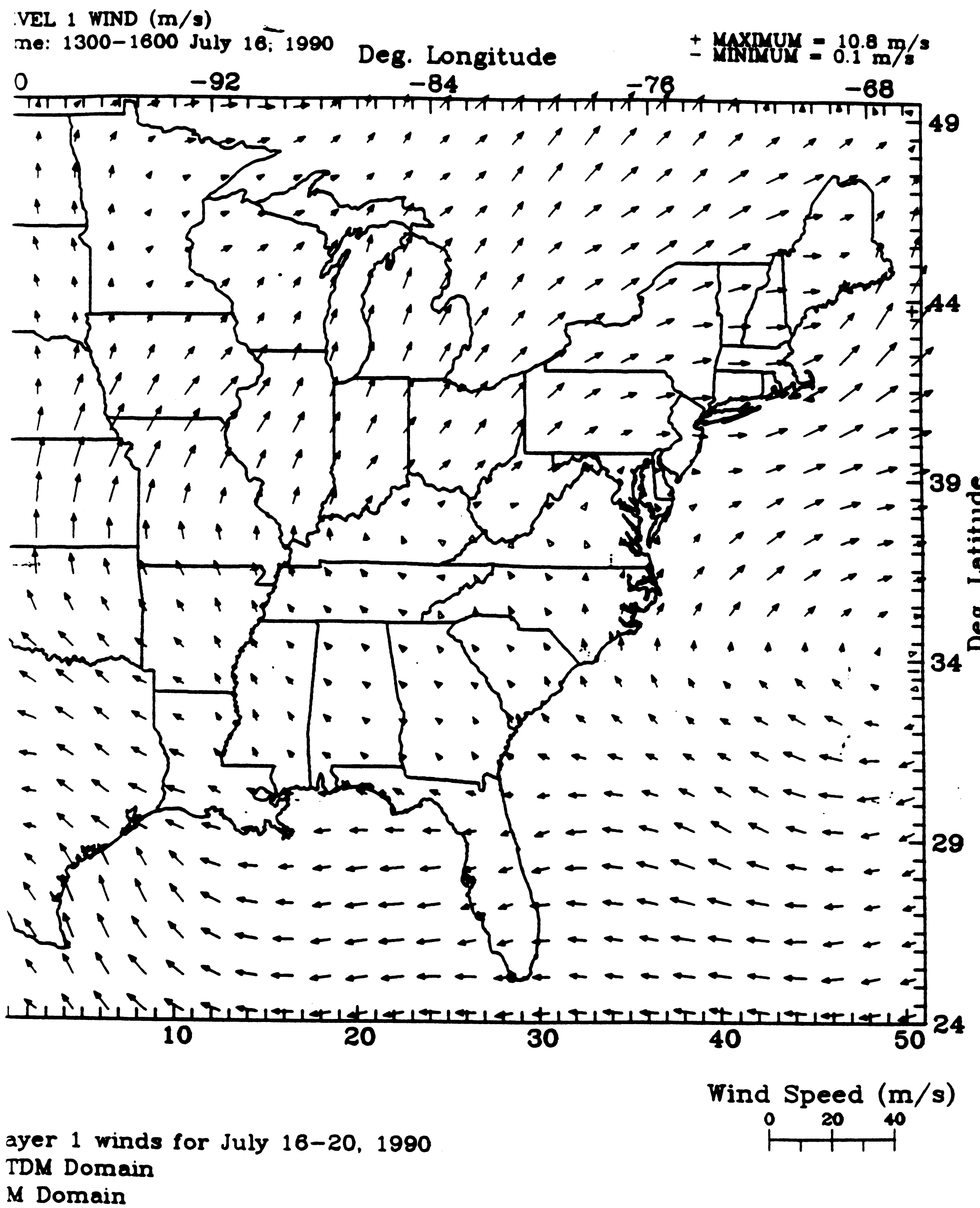
To derive the ozone source-receptor matrices, we decided to run the UAM-V for five-day episodes and average the results for the last two days of each episode for each region. The first three days are ignored to give the UAM-V a chance to work through the initial conditions. Data availability dictated choosing episodes from 1990.[16] We observed that there are three roughly defined ozone patterns which are generally of interest. They are characterized by broad but distinct areas of elevated ozone concentrations centered approximately over the Midwest, the Northeast corridor and the Southeast. Figure 3 is the average wind pattern for the 5-day Northeast episode we used. The episodes reflect in part the physical distributions of precursor emissions and characteristic weather patterns, and in

[14] Most of the concentration-health response functions use either this measure or an average over a longer period.

[15] Originally, a significant analysis of the non-linearities associated with the S-R coefficients was expected. Thcse nonlinearities can arise from sensitivity to the ozone baseline or to the magnitude of the NOx (and VOC) changes. Funding limitations precluded these tests. Note, however, that the statistical analysis by Rao, Mount, and Dorris (1997) provides evidence that at a relatively high level of spatial aggregation, a linear model fits the micro simulation results as well as any other.

[16] The needed information consists of the output fields from a mesoscale meteorological model constrained by observations. Most of the datasets which have been prepared for past ozone simulations fall into the worst-case category, and/or do not cover the domain of interest, which is quite large by ozone modeling standards. The only comprehensive dataset available which appears to be suitably prepared is the Interagency Working Group on Air Quality Modeling (IWAQM) dataset prepared for the year 1990. Thc entire year was processed.

**Figure 3**



.VEL 1 WIND (m/s)
me: 1300–1600 July 16, 1990
Deg. Longitude
+ MAXIMUM = 10.8 m/s
– MINIMUM = 0.1 m/s
0
–92
–84
–76
–68
49
44
39
34
29
24
Deg. Latitude
10
20
30
40
50
Wind Speed (m/s)
0
20
40
ayer 1 winds for July 16–20, 1990
TDM Domain
M Domain

part the areas where exposure to elevated ozone concentrations has been identified as an air quality issue. 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 et al., 1997). This correspondence is important because the NESCAUM report identified windfields for the eastern U.S. that correspond to the top 20 percent of daily maximum ozone readings within each of the three regions, in turn.[17,18]

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 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 observations are not evenly distributed and may have missed the actual peak.

All told, we developed unweighted and population-weighted S-R coefficient matrices for six source and seven receptor regions, for each of three ozone episodes, for both point sources and mobile sources, summing to 12 matrices developed from 37 runs of UAM-V. Table 2 presents the population weighted NOx to ozone S-R matrices along with "trading ratios" (see below). Population data used to weight the ozone changes for areas within each receptor region are taken from BEA 2005 projections by state.

To summarize the results, we consider first the average episodes. These are computed by calculating a simple average of the population weighted two-day average source-receptor coefficients for each of the three episodes, by source type and source-receptor combination. We find:

(i) For both point and mobile sources: the effect of a region's NOx emissions on the ozone exposure of its own population (the on-diagonal coefficients) is generally larger than the transported effects, except New York NOx emissions affect ozone in New England more than ozone in New York; indeed, transport effects are

---

[17] The 20th percentile ozone concentrations are quite low: 70 ppb, 72, ppb, and 67 ppb for the NE, MW, and SE, 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 like ours are the most important target, are representing most of the 20 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.

[18] The criteria for selection of candidate intervals were: 1) a relatively localized ozone peak (i.e., distinguishable region of elevated ozone concentrations) in one of the regions of interest (northeast, midwest and southeast), 2) a pattern duration of three days minimum, with five days preferable, 3) ozone concentrations above 80 ppb over a broad region, with a preference for higher peak concentrations, and 4) preference for more strongly-defined regional-scale wind flows, similar to those identified in the NESCAUM report. Six initial candidate intervals were identified which were then narrowed to three. The selection criteria are necessarily somewhat subjective and the selected set of intervals is certainly not unique.

relatively small in most cases, particularly from emissions in Tennessee and the South.

(ii) In particular, midwestern utility NOx emissions change ozone locally about three times more per ton of NOx 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.

(iii) There is evidence of a NOx "scavenging" effect. NOx scavenging is a physical process by which reduced (increased) NOx emissions can *increase (reduce)* the production of ambient ozone. Often, it occurs close to tall stacks emitting NOx but the UAM-V model shows it occurring over large areas. In particular, we find that reduced NOx emissions actually increase ozone *on net* for mobile source NOx emissions from E. Pa.–N.J. on itself and on New England's ozone. The "New England" effect counts ozone changes above Long Island and New York City . This effect appears for each episode type.

Looking across episodes:

(i) 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 the three specific episodes.

(ii) One difference of interest is that for the classic northeast episode (i.e., a Bermuda High), transport of emissions in the South and Midwest are more important. For instance, Midwestern point source emissions affect New England, New York City, and E. PA-NJ as much or more than these emissions affect the Midwest.

One relevant way to summarize these results is in terms of what are termed "trading ratios." A trading ratio measures the *total* effect of a ton of NOx emitted in one region divided by the total effect of a ton of NOx emitted in another region. Effects are measured by ozone exposures (in ppb-person/day) 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 a NOx trading program based on *ozone exposure effects,* these ratios would define the terms of trade. Table 3 provides the trading ratios, for point and mobile sources separately, by episode. The ratios are computed with the MD-VA region point sources as the numeraire. This means, for instance, that reducing a ton of NOx from point sources in New York results in 25 percent more benefits over the domain than doing the same thing in Maryland-Virginia and that a ton of NOx reduced by mobile sources in New York results in 22 percent more benefits than a ton of NOx reduced from point sources in Maryland-Virginia.

**Table 2. Population weighted NOx to ozone S-R matrices, by source type and episode.**

| | | | | | | |
|---|---|---|---|---|---|---|
| ***Episode Type*** | | | | | | |
| **Receptor Region** | | | | **Point Sources** | | |
| ***Average*** | 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 |
| NYC-LI-CT-RI-MA | 0.0439 | 0.6789 | 2.8298 | 0.5555 | 0.1248 | 0.0090 |
| | | | | **Mobile Sources** | | |
| ***Average*** | MD-VA | E.PA-NJ | NY | OH-W.PA-WV | IND-KY | TN-South |
| MD-VA | 5.2740 | 0.3377 | 0.0354 | 1.0916 | 0.1610 | 0.4616 |
| E.PA-NJ | 1.1404 | -1.2920 | 0.6132 | 1.1448 | 0.1992 | 0.0080 |
| NY | 0.0138 | 0.1129 | 3.4902 | 0.8826 | 0.4715 | 0.0360 |
| OH-W.PA-WV | 0.1098 | 0.0039 | 0.0012 | 2.5714 | 1.4044 | 0.2791 |
| IND-KY | 0.0045 | 0.0003 | 0.0006 | 0.1892 | 3.0977 | 0.7483 |
| TN-South | 0.4299 | 0.0001 | 0.0001 | 0.2268 | 0.4153 | 10.6284 |
| NYC-LI-CT-RI-MA | 0.1864 | -2.4698 | 2.3052 | 0.6968 | 0.1596 | 0.0058 |
| | | | | **Point Sources** | | |
| ***NE*** | 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 |
| NYC-LI-CT-RI-MA | 0.0645 | 0.4334 | 2.9909 | 1.3254 | 0.2822 | 0.0269 |
| | | | | **Mobile Sources** | | |
| ***NE*** | MD-VA | E.PA-NJ | NY | OH-W.PA-WV | IND-KY | TN-South |
| MD-VA | 8.2390 | 0.0148 | 0.0003 | 0.8901 | 0.0908 | 0.3873 |
| E.PA-NJ | 2.4177 | 0.9929 | 0.0299 | 1.6367 | 0.1829 | 0.0235 |
| NY | 0.0124 | 0.4299 | 2.6215 | 1.0231 | 0.8299 | 0.1079 |
| OH-W.PA-WV | 0.0636 | -0.0020 | 0.0021 | 2.3365 | 2.0353 | 0.7631 |
| IND-KY | 0.0025 | 0.0000 | 0.0002 | 0.0353 | 2.9681 | 2.0939 |
| TN-South | 0.0315 | -0.0001 | 0.0005 | -0.0020 | -0.0034 | 6.6103 |
| NYC-LI-CT-RI-MA | 0.3741 | -2.7392 | 2.0058 | 1.2002 | 0.2253 | 0.0179 |

| | Point Sources | | | | | |
|---|---|---|---|---|---|---|
| ***MW*** | 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 |
| NYC-LI-CT-RI-MA | 0.0672 | 1.7536 | 2.1528 | 0.1710 | 0.0069 | 0.0001 |
| | **Mobile Sources** | | | | | |
| ***MW*** | MD-VA | E.PA-NJ | NY | OH-W.PA-WV | IND-KY | TN-South |
| MD-VA | 2.4496 | 0.7874 | 0.0984 | 0.6255 | 0.0215 | 0.1011 |
| E.PA-NJ | 0.9301 | -6.2273 | 0.3566 | 0.9082 | 0.0492 | 0.0002 |
| NY | 0.0292 | -0.4361 | 5.7489 | 1.0913 | 0.1520 | -0.0001 |
| OH-W.PA-WV | 0.2686 | 0.0135 | -0.0002 | 2.8128 | 0.8243 | 0.0124 |
| IND-KY | 0.0105 | 0.0002 | 0.0003 | 0.3507 | 3.7389 | 0.0653 |
| TN-South | 1.1772 | 0.0008 | 0.0000 | 0.3569 | 0.8386 | 8.8940 |
| NYC-LI-CT-RI-MA | 0.1855 | -3.0201 | 1.7938 | 0.4494 | 0.0263 | -0.0001 |
| | **Point Sources** | | | | | |
| ***SE*** | 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 |
| NYC-LI-CT-RI-MA | -0.0001 | -0.1504 | 3.3457 | 0.1702 | 0.0854 | 0.0000 |
| | **Mobile Sources** | | | | | |
| ***SE*** | MD-VA | E.PA-NJ | NY | OH-W.PA-WV | IND-KY | TN-South |
| MD-VA | 5.1333 | 0.2111 | 0.0076 | 1.7591 | 0.3708 | 0.8963 |
| E.PA-NJ | 0.0734 | 1.3585 | 1.4531 | 0.8894 | 0.3656 | 0.0002 |
| NY | 0.0000 | 0.3449 | 2.1004 | 0.5335 | 0.4325 | 0.0001 |
| OH-W.PA-WV | -0.0029 | 0.0001 | 0.0017 | 2.5649 | 1.3537 | 0.0617 |
| IND-KY | 0.0005 | 0.0008 | 0.0014 | 0.1814 | 2.5860 | 0.0855 |
| TN-South | 0.0809 | -0.0004 | -0.0003 | 0.3256 | 0.4106 | 16.3809 |
| NYC-LI-CT-RI-MA | -0.0004 | -1.6502 | 3.1159 | 0.4406 | 0.2270 | -0.0003 |

**Table 3. Trading Ratios, by source type and episode. (MD-VA Point Sources = 1)**

| | Point Sources | | | | | |
|---|---|---|---|---|---|---|
| **Episode Type** | MD-VA | E.PA-NJ | NY | OH-W.PA-WV | IND-KY | TN-South |
| Average | 1 | 1.06 | 1.25 | 1.51 | 1.01 | 1.00 |
| NE | 1 | 0.66 | 0.89 | 1.24 | 0.92 | 0.96 |
| MW | 1 | 1.61 | 0.97 | 1.27 | 0.79 | 0.50 |
| SE | 1 | 0.86 | 2.68 | 2.68 | 1.68 | 2.16 |
| | **Mobile Sources** | | | | | |
| Average | 1.92 | -0.94 | 1.22 | 1.71 | 1.11 | 2.63 |
| NE | 2.33 | -0.36 | 0.67 | 1.40 | 0.94 | 1.60 |
| MW | 1.13 | -2.04 | 1.10 | 1.38 | 0.84 | 1.71 |
| SE | 2.62 | -0.01 | 2.77 | 3.17 | 2.09 | 7.06 |

The most striking result for the average episode and point sources is the uniformity of trading ratios for point sources, with the exception of the OH-W.PA-WV and, to a lesser extent, New York. The higher ratios for these regions imply that a ton of NOx emissions reduction from point sources in these areas is worth more (in terms of exposure reductions) than a ton elsewhere. Given the S-R coefficients, the high effect of the OH-W.PA-WV on its own population (coupled with high population) drives this result, in part.

This uniformity is not so evident across episodes. For instance, for the midwest episode, 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.

Finally, the most striking result for mobile sources is that a ton of NOx emissions reduction from mobile sources reduces more ozone exposures than a ton of point source NOx emissions reductions, particularly for MD-VA. This is not surprising as the NOx emissions from mobile sources are emitted where the people are. So a population-based exposure measure is likely to favor mobile source reductions more than an area-weighted measure.

In terms of their trading ratios, there is greater nonuniformity across regions for mobile sources, partly because of the scavenging effect observed for E.PA-NJ. Reductions in the South and the Midwest are particularly productive.

## D. Ozone Benefits

The population-weighted source-receptor matrices linking NOx emissions to changes in ozone are multiplied by regional population data and estimates of the health damages per person per unit change in ozone to estimate ancillary health benefits of NOx emissions reductions.

The estimates of health damages per person per annual change in ozone concentrations are taken from Krupnick and Burtraw (1997), Table 4. The estimates are derived from a Monte Carlo simulation model, including a comprehensive set of epidemiological concentration-response functions and values taken from the economics literature.[19] The benefit estimates are

[19] This model is derived from the Tracking and Analysis Framework, which was developed on behalf of the National Acid Precipitation Assessment Program (Bloyd, 1996).

highly sensitive to the treatment of mortality risks. In line with the general reluctance of EPA and others to ascribe mortality benefits to ozone (EPA, 1997), mortality risks are assumed to be zero over 90 percent of the probability distribution, with 10 percent of the distribution determined by two studies in the literature showing positive effects. The value of reducing such risks is set at $3.2 million (rather than the consensus accidental death VSL of $4-5 million), to account for the disproportional effects on older people and their lower than average willingness-to-pay (WTP) (as determined in Jones-Lee et al., 1985). Table 4 provides the damage per person per unit concentration estimates used in this study (converted to 1990 dollars from 1989 dollars, to match the cost estimates) and Table 5 provides the ancillary benefit estimates applied to each source-type per ton NOx reduced by source region. We assume that ozone benefits are only registered during the ozone season (153 days per year). The estimates assume an average episode, in terms of relating NOx emissions to ozone concentrations.

**Table 4. Annual Unit Damages for Health Effects from Ozone**
(1990 dollars/person/0.01 ppm ozone)

| 5% | Mean | 95% |
|---|---|---|
| $3.16 | $7.38 | $86.43 |

These damage estimates can be interpreted in the following way. The central estimate of $7.38 for ozone is the mean WTP of an average person for a reduction in ozone of 0.01ppm[20] averaged over the year. The mean WTP is based on the expected values of the S-R coefficients and the valuation coefficients and an aggregation algorithm designed to eliminate double-counting of health effects.

**Table 5. Monetary Benefits over the Study Domain per ton of NOx Emitted, by Source Type and Region. Average Episode. (1989 dollars).**

| **Point Sources** | | | | | | |
|---|---|---|---|---|---|---|
| | **Source Regions** | | | | | |
| | MD-VA | E. PA-N.J. | NY | OH-W. PA-WV | IND-KY | TN-South |
| 5th percentile | 61 | 64 | 75 | 91 | 61 | 61 |
| Mean | 141 | 150 | 176 | 213 | 143 | 141 |
| 95th percentile | 1656 | 1761 | 2062 | 2501 | 1671 | 1656 |

| **Mobile Sources** | | | | | | |
|---|---|---|---|---|---|---|
| | **Source Regions** | | | | | |
| | MD-VA | E. PA-N.J. | NY | OH-W. PA-WV | IND-KY | TN-South |
| 5th percentile | 116 | -57 | 74 | 104 | 67 | 160 |
| Mean | 271 | -133 | 172 | 242 | 157 | 373 |
| 95th percentile | 3180 | -1563 | 2020 | 2833 | 1839 | 4368 |

20 The current daily peak 1-hour standard is 0.12 ppm, the new 8-hour averaged standard is 0.08 ppm.

## VI. RESULTS OF THE POLICY ANALYSIS

Here we present the results of the three major policy analyses outlined in the Introduction. First, we examine the outcomes under the planned Clean Air Act and OTAG regulatory (CAC) programs. We refer to the Clean Air Act scenario as the CAA scenario and to the additional OTAG controls as the OTAG scenario. In the second part of the analysis, we compare the CAC policies to various hypothetical least-cost outcomes as a guide to future policy decisions. Finally, we examine possible NOx trading rules for meeting future ozone goals, and their impact on the Bay.

### Policy Analysis 1: The Impact of Clean Air Act Regulations and OTAG Controls

Baseline Emissions

Tables 6 and 7 below show the forecast baseline or initial levels of NOx emissions and Bay nitrate loads resulting from air sources before either the CAA or OTAG regulatory scenarios are implemented. All forecasts are for annual activity in the year 2005. Ohio and then Pennsylvania contribute the largest share of NOx emissions, but Pennsylvania emissions have by far the largest impact on Bay nitrate loadings, with Maryland, Virginia, Ohio, New York and West Virginia emissions also contributing significantly to nitrate loadings.

Table 7 shows how the different NOx emissions sources account for both total NOx emissions and the impact on nitrate loads in the baseline case. In the Bay jurisdictions alone, mobile sources and utilities account for approximately equal shares of NOx emissions, while in the Airshed, utilities account for substantially more than mobile sources. Nitrate loadings are proportionate to emissions by source category in the Bay states and mobile sources contribute proportionately more to Bay nitrate loadings than their share of emissions. (These results parallel those of the Pechan Analysis (1996), though there are some differences because the emissions inventories have been updated).

Impact of the Clean Air Act Scenario

The results of the Clean Air Act requirements are shown in the top section of both Tables 8 and 9. Table 8 shows that reductions in NOx emissions due to controls under the Clean Air Act result in reductions in nitrate loadings to the Bay of 17.5 million pounds a year from the airshed -- about 27 percent of the airborne nitrate load -- with the Bay jurisdictions accounting for about half of the total. Although utilities are responsible for about half of the baseline airborne nitrate loadings in the airshed (Table 7 above), they account for almost 80 percent of the loadings reductions in the region. Under the Clean Air Act controls, the percentages are a little lower for the Bay jurisdictions alone, but the finding that utilities are required to reduce emissions by a relatively large percentage holds for the smaller Bay region as well. Examination of NOx emissions reductions in both regions (Table 8) makes it clear that utilities have been heavily targeted for NOx emissions reductions under Clean Air Act mandates.

**Table 6. No-Control Baseline NOx Emissions and Nitrate Loads. Year 2005**

| State | Emissions (thousand tons/year) | % of total emissions | Nitrate Load (thousand lbs/year) | % of total load |
|---|---|---|---|---|
| **Bay jurisdictions** | | | | |
| DC | 13 | 0.2% | 281 | 0.4% |
| Maryland | 283 | 5.4% | 9,145 | 13.8% |
| Pennsylvania | 781 | 14.9% | 15,459 | 23.4% |
| Virginia | 380 | 7.2% | 7,810 | 11.8% |
| **Bay total** | | 27.7% | | 49.4% |
| **Other Airshed States** | | | | |
| Delaware | 63 | 1.2% | 2,047 | 3.1% |
| Indiana | 350 | 6.7% | 1,546 | 2.3% |
| Kentucky | 444 | 8.5% | 1,962 | 3.0% |
| Michigan | 353 | 6.7% | 2,226 | 3.4% |
| New Jersey | 250 | 4.8% | 1,224 | 1.8% |
| New York | 379 | 7.2% | 6,719 | 10.2% |
| North Carolina | 339 | 6.5% | 2,569 | 3.9% |
| Ohio | 985 | 18.8% | 7,937 | 12.0% |
| Tennessee | 167 | 3.2% | 933 | 1.4% |
| West Virginia | 455 | 8.7% | 6,328 | 9.6% |
| Total Airshed | 5,242 | | 66,186 | |

**Table 7. Share of NOx Emissions and Nitrate Load Reductions by Source Category. No Control Baseline**

| | **Bay jurisdictions** | | **Airshed** | |
|---|---|---|---|---|
| | % emissions by source | % nitrate loadings by source | % emissions by source | % nitrate loadings by source |
| Mobile Sources | 45% | 46% | 37% | 40% |
| Utilities | 44% | 44% | 53% | 50% |
| Non-Utilities | 11% | 10% | 10% | 10% |

| Scenario | Bay jurisdictions: NOx Emissions Reductions (000 tons) | Bay jurisdictions: Nitrate Load Reductions (000 lbs) | Bay jurisdictions: Cost (M$) | Bay jurisdictions: Mean Benefit (M$) | Bay jurisdictions: High benefits (M$) | Airshed: NOx Emissions Reductions (000 tons) | Airshed: Nitrate Load Reductions (000 lbs) | Airshed: Cost (M$) | Airshed: Mean Benefits (M$) | Airshed: High Benefits (M$) |
|---|---|---|---|---|---|---|---|---|---|---|
| CAA | 357 | 8,099 | 417 | 33 | 387 | 1,462 | 17,543 | 1,185 | 125 | 1,468 |
| Mobile Sources | 15% | 17% | 244 | 6 | 67 | 9% | 11% | 513 | 7 | 88 |
| Utility | 72% | 71% | 67 | 24 | 279 | 83% | 79% | 423 | 108 | 1,265 |
| Non-Utility | 13% | 12% | 105 | 3 | 41 | 8% | 10% | 248 | 10 | 115 |
| OTAG | 310 | 7,049 | 517 | 26 | 307 | 1,024 | 13,669 | 1,725 | 84 | 985 |
| Mobile Sources | 13% | 13% | 96 | 2 | 25 | 9% | 10% | 286 | 4 | 46 |
| Utility | 85% | 85% | 356 | 23 | 276 | 84% | 85% | 1,114 | 75 | 876 |
| Non-Utility | 2% | 2% | 65 | 1 | 6 | 7% | 5% | 326 | 5 | 63 |

**Table 8 - Annual Nitrate Load and NOx Emissions Reductions, Benefits and Costs ($1990) . Year 2005**

**Table 9 - Gross and Net Costs per pound of Nitrate Reduced**

| Scenario | Bay jurisdictions: Gross Cost/ton of NOx reduced | Bay jurisdictions: Gross Cost/lb of nitrate reduced | Bay jurisdictions: Cost/lb net of mean ozone benefits | Bay jurisdictions: Cost/lb net of high ozone benefits | Airshed: Gross Cost/ton of NOx reduced | Airshed: Gross Cost/lb of nitrate reduced | Airshed: Cost/lb net of mean ozone benefits | Airshed: Cost/lb net of high ozone benefits |
|---|---|---|---|---|---|---|---|---|
| CAA | 1,168 | 51 | 47 | 4 | 811 | 67 | 60 | -16 |
| Mobile Sources | 4,436 | 126 | 123 | 91 | 4,039 | 265 | 261 | 220 |
| Utility | 262 | 5 | 3 | -15 | 349 | 31 | 23 | -61 |
| Non-Utility | 2,283 | 57 | 55 | 34 | 2,000 | 136 | 130 | 73 |
| OTAG | 1,668 | 73 | 70 | 30 | 1,685 | 126 | 120 | 54 |
| Mobile Sources | 2,462 | 102 | 100 | 76 | 2,979 | 206 | 203 | 172 |
| Utility | 1,348 | 60 | 56 | 13 | 1,298 | 96 | 90 | 21 |
| Non-Utility | 9,286 | 435 | 432 | 393 | 4,657 | 465 | 458 | 376 |

The costs of obtaining reductions in nitrate loadings resulting from regulations that are going to be undertaken to improve air quality can be considered as "free" to the Bay. These reductions would occur at no additional cost to the Bay. It is nonetheless interesting to know what the costs of these air reductions are. The top part of Table 8 shows the costs for the Clean Air Act reductions. These costs are estimated to be about $400 million for the Bay jurisdictions annually and $1.185 billion annually for the airshed by the year 2005. Mobile source reductions appear to have the highest costs in both the Bay jurisdictions and the entire airshed. Utilities are the most cost-effective source of NOx emissions reductions, with low costs and the greatest emission reductions.

The benefits of improvements in ozone resulting from NOx emissions reductions for the different regions are also shown at the top of Table 8 for the CAA scenario.[21] The mean value of benefits is small, less than 10 percent of the costs in the Bay and a slightly higher percentage of costs in the airshed. However, the "high" estimate of benefits is greater than the costs, meaning that net costs are negative in both the Bay and the airshed. The high estimate of benefits is much higher than the mean benefit estimate because it includes potential mortality effects from NOx emissions. The different benefits estimates are discussed at length in section V.D.

Table 9 examines the cost-effectiveness of the CAA controls from both the Bay states and the entire airshed. The first columns for each region attribute all of the costs first to NOx emissions reductions and then to reduction of nitrate loadings to the Bay. The Bay jurisdictions are able to more cost-effectively control NOx to reduce nitrate loadings to the Bay compared to the entire airshed. The cost-effectiveness is $51/pound for the Bay jurisdictions and $67/pound for the region as a whole. However, it is not the case that Bay jurisdiction's control of NOx emissions are more cost-effective than those in the airshed as a whole. The Bay states reduce NOx emissions at an average cost per ton of $1,168 compared to only $810 for the airshed. Rather, the proximity of the Bay states to the Bay makes them more cost-effectiveness in reducing nitrate loading.

In terms of sources, Table 9 shows that in both the Bay region and the airshed, the gross costs per pound of nitrate reduced are dramatically lower for utilities than for the mobile sources under CAA controls. It makes sense, then, that the Clean Air Act targeted utilities so strongly. It is also noteworthy that the cost per ton of NOx reduced for utilities is much lower for the Bay jurisdictions than for the airshed as a whole.

Table 9 also shows cost-effectiveness results when costs are attributed to both the reduction of nitrate loads in the Bay and ambient ozone levels. The benefits of ancillary ozone improvements are netted out of the costs before the cost per ton is calculated. "High" and "mid" measures of benefits are used. As Table 8 shows, the mean benefits are quite small and have little impact on the cost-effectiveness results. However, again, the high estimate of benefits, which includes potential mortality effects of NOx emissions, has a large impact on the cost-effectiveness results. The "high" benefits outweigh the costs for the airshed region as

[21] See discussion of ozone benefit estimation in Section V.

a whole and costs per ton are negative $16. The reason for this seems to be that the benefits from the utility sector are large enough to outweigh total costs for the region.

Another important point about the ozone benefits is that ton for ton, mobile sources result in lower benefits than do utilities: $59/ton for mobile sources versus $89/ton for utilities. This occurs in spite of generally higher source-receptor coefficients for mobile sources (which emit NOx where the people are) primarily because emission reductions from mobile sources in E.PA-NJ have a strong ozone-enhancing effect (termed NOx scavenging).[22] If the benefit per ton estimates are examined for a particular state that does not have scavenging (such as Maryland), mobile sources are far more productive in creating health benefits: $161/ton versus $78/ton for mobile and utility sources, respectively.

Impact of the additional OTAG controls

The lower sections of Tables 8 and 9 show the results of additional controls being considered for controlling NOx emissions as part of the OTAG scenario. Under this scenario, nitrate loadings fall by 13.7 million pounds in the entire airshed -- or another 35 percent from the 48.5 million pound CAA level -- and NOx emissions are reduced by one million tons annually. About half of these improvements in nitrate reductions come from Bay jurisdictions, but this Bay region accounts for only about one third of the NOx emissions. As under the CAA scenario above, NOx emissions reductions in the Bay jurisdictions have a larger impact on nitrate loads to the Bay than NOx emissions reductions in other parts of the airshed.

Even more of the reductions under the OTAG scenario come through utilities (85 percent) compared with the Clean Air Act controls (Table 8). And, mobile sources make up a smaller share in both the Bay jurisdictions and in the region as a whole: 13 percent in the Bay jurisdictions and 10 percent in the airshed, compared to 17 percent and 11 percent under the Clean Air Act controls.[23]

However, from Table 8, it is clear that the costs of control change from the CAA case. Emission reductions are lower and the costs of achieving them are significantly higher under the OTAG case; in other words, the marginal costs of greater reductions rise with tighter controls, as we would expect. There are also some changes in the sources of reductions from the base to CAA case. The costs of controlling utility sources under OTAG are much higher, and the mobile source controls are lower. The latter occurs because of the assumptions within the Pechan data that the additional costs of LEV vehicles will be relatively low.[24] Also, note that the ozone benefits are smaller for both mean and high benefits because NOx emissions reductions are smaller than they were under the CAA scenario.

Table 9 shows again what we would expect for the stricter controls in the OTAG scenario relative to the CAA scenario -- that the cost-effectiveness of NOx emissions

---

[22] See Table 2 above for S-R coefficients.

[23] Recall that under the OTAG scenario, mobile source controls include only the introduction of Low Emission vehicles.

[24] Note that there are no LEVs in the CAA scenario but that LEVs are mandated in the OTAG scenario.

reductions worsens for the OTAG scenario, climbing for the airshed from $811/ton under the CAA scenario to $1,685/ton for the OTAG scenario. For the Bay jurisdictions, whose costs were quite a bit higher than those for the airshed as a whole in the CAA scenario, costs increase less dramatically, to the point where average NOX cost-effectiveness is no different for the Bay and the region ($1,668/ton for the Bay and $1,685/ton for the region).

From the perspective of the Bay community, it is cost-effectiveness in terms of nitrate loading reductions that is most important. These results are shown in Table 9. Costs for the airshed also climb in terms of this measure under the OTAG scenario (from $67/pound for the CAA scenario to $126/pound for the OTAG scenario). But the Bay jurisdictions widen their advantage as a cost-effective source of nitrate loading reductions, being 42 percent cheaper per pound reduced versus 24 percent cheaper per pound in the CAA scenario. Thus, the Bay jurisdictions are an attractive target for further airborne nitrate loading reductions.

Which sources within the Bay jurisdictions are the most attractive targets? For the CAA scenario, utilities were the most cost-effective (using gross costs), at $5/pound, followed by non-utility point sources at $57/pound, and mobile sources at $ 126/pound. For the OTAG scenario, there are some dramatic changes. Utilities are still the most cost-effective source, at $60/pound, but mobile source costs actually fall to $102/pound. The decrease in costs for mobile sources is, as we discussed above, primarily because LEV vehicles are assumed to have relatively low cost, at $76 per vehicle, and because emissions reductions from LEVs are large especially in conjunction with enhanced I&M (see Appendix 3). Further regulation of non-utility point sources in the Bay jurisdictions appears not to be cost-effective, at $435/pound reduction.

Tables 10 and 11 show the impact of the CAA and OTAG scenarios across states, for both Bay jurisdictions and airshed states. Table 10 shows percentage reductions in both NOx emissions and nitrate loads to the Bay for each state that will result from successively more stringent regulations. The first column shows the percentage change in NOx from the baseline to the CAA, and the second column show the percent change from the CAA scenario to the OTAG scenario. The percentage variation differs by state, and some states doing relatively more under the OTAG scenario, and some less. The two states with the greatest baseline amount of NOx, Pennsylvania and Ohio, will reduce by a relatively large percentage, but some states will reduce by even higher percentages. Looking at the fifth and sixth columns of Table 10 we see that the percentage reductions in nitrate loadings are roughly similar to the percentage reduction in NOx emissions reductions.

However, the share of emission reductions among states does seem to change over time and vary with the pollutant under examination. Under the OTAG scenario, the share of NOx emissions reductions required for the Bay jurisdictions rises, and only a few of the other airshed states increase their share of the total reductions compared to the CAA case (the share of reductions for the Bay jurisdictions rises from 24 percent to 31 percent from the CAA to OTAG cases).

**Table 10. State Contributions of NOx Emissions and Nitrate Loading to the Bay from Air Sources**

| State | NOx Emissions Reductions | | | | Nitrate Loading Reduction | | | |
|---|---|---|---|---|---|---|---|---|
| | Percent reduction in NOx Emissions (%) | | State Share of total NOx Emissions Reductions | | Percent reduction in Nitrate Loadings (%) | | State share of total Nitrate Loading Reduction | |
| **Bay Jurisdictions** | **CAA** | **OTAG** | **CAA** | **OTAG** | **CAA** | **OTAG** | **CAA** | **OTAG** |
| DC | 23 | 20 | 0 | 0 | 23 | 19 | 0 | 0 |
| Maryland | 27 | 33 | 5 | 7 | 28 | 34 | 15 | 17 |
| Pennsylvania | 30 | 32 | 16 | 17 | 30 | 32 | 27 | 25 |
| Virginia | 11 | 20 | 3 | 7 | 10 | 18 | 5 | 9 |
| **Total** | | | 24 | 31 | | | 47 | 51 |
| **Other Airshed States** | | | | | | | | |
| Delaware | 27 | 30 | 1 | 1 | 28 | 32 | 3 | 3 |
| Indiana | 33 | 22 | 8 | 5 | 33 | 22 | 3 | 2 |
| Kentucky | 34 | 35 | 10 | 10 | 34 | 35 | 4 | 3 |
| Michigan | 31 | 16 | 7 | 4 | 31 | 16 | 4 | 2 |
| New Jersey | 27 | 16 | 5 | 3 | 27 | 17 | 2 | 1 |
| New York | 17 | 21 | 4 | 7 | 17 | 21 | 7 | 9 |
| North Carolina | 21 | 25 | 5 | 7 | 21 | 26 | 3 | 4 |
| Ohio | 32 | 24 | 22 | 16 | 32 | 24 | 14 | 9 |
| Tennessee | 17 | 40 | 2 | 5 | 17 | 40 | 1 | 2 |
| West Virginia | 36 | 44 | 11 | 13 | 36 | 44 | 13 | 13 |
| **Total Airshed** | | | 100% | 100% | | | 100% | 100% |

**Table 11. Costs of NOx Emissions and Nitrate Loading Reduction**
**(Assume gross costs unless stated otherwise)**

| State | Cost of NOx Emissions Reductions (millions of dollars) | | Cost-Effectiveness of Nitrate Loading Reductions | | | Cost Effectiveness of NOx Emissions Reductions ($/ton) | |
|---|---|---|---|---|---|---|---|
| | CAA | OTAG | CAA | OTAG | | CAA | OTAG |
| | | | $/lb | $/lb | $net/lb (high benefits) | | |
| **Bay Jurisdictions** | | | | | | | |
| DC | 12 | 5 | 174 | 122 | 73 | 3,833 | 2,500 |
| Maryland | 123 | 96 | 48 | 43 | 14 | 1,596 | 1,433 |
| Pennsylvania | 190 | 301 | 40 | 87 | 35 | 800 | 1,728 |
| Virginia | 93 | 115 | 117 | 89 | 43 | 2,323 | 1,721 |
| **Other Airshed States** | | | | | | | |
| Delaware | 21 | 37 | 36 | 80 | 54 | 1,224 | 2,657 |
| Indiana | 90 | 97 | 177 | 421 | 251 | 785 | 1,871 |
| Kentucky | 63 | 151 | 94 | 337 | 143 | 416 | 1,495 |
| Michigan | 19 | 86 | 27 | 342 | 163 | 171 | 2,148 |
| New Jersey | 184 | 57 | 549 | 388 | 408 | 2,703 | 1,900 |
| New York | 98 | 109 | 84 | 91 | 38 | 1,506 | 1,627 |
| North Carolina | 28 | 121 | 52 | 234 | 131 | 400 | 1,782 |
| Ohio | 214 | 258 | 84 | 201 | 59 | 678 | 1,620 |
| Tennessee | 12 | 105 | 76 | 340 | 202 | 425 | 1,900 |
| West Virginia | 40 | 188 | 17 | 106 | 19 | 241 | 1,471 |
| **Total Airshed** | 1,187 | 1,726 | 1,575 | 2,881 | 1,633 | 17,101 | 25,853 |

However, the Bay states contribute a much greater share of the nitrate loading reductions and this share increases from 47 percent to 51 percent under the OTAG scenario. This is to be expected since the Bay states are adjacent to the Bay; but it can also be looked at the other way, that about half of the air contributions come from states that are not directly adjacent to the Bay, such as New York, Ohio, and West Virginia. And, there are also some important differences in how total nitrate loading reductions are distributed across states under the two scenarios. For example Pennsylvania contributes almost 3 times as much nitrate loading reduction in the OTAG scenario as Virginia, even though their initial nitrate loadings to the Bay were only about double Virginia's.

Table 11 shows the total costs of NOx emissions reductions and two cost-effectiveness measures by state. As we would expect, total costs are the highest for the states which have the largest emissions, or which are making the most reductions, such as Pennsylvania, Ohio and West Virginia. However, what is most interesting in Table 11 is that the gross costs per pound of nitrate load reduced are much lower in the Bay jurisdictions (except D.C.) than most of the other states. In the OTAG case, Maryland's cost per pound nitrate load reduced ($43) are much lower than for any other state -- about half that of the other Bay states. Costs per pound reduced are usually higher in the OTAG case, but not always -- Virginia and New Jersey actually show costs per pound falling. The costs per ton of NOx reduced vary a great deal across states. The variation is less under OTAG and costs are higher everywhere except in Virginia and D.C.

Finally, including "high" benefits in the cost-effectiveness calculation results in a large variation in cost/pound nitrate loadings reductions across the states. The net costs per pound for New Jersey sources is more than 10 times the costs in the Bay jurisdictions. However, the relative distribution of cost-effectiveness between states is not changed much. Although the cost-effectiveness estimates including the high ozone benefits are much lower, the relative ranking of the states does not appear to change much. This is because the source-receptor relationships linking NOx emissions to ozone changes over the airshed are similar across states.

### Policy Analysis 2: Comparison of the CAC Scenarios to Scenarios for Achieving Nitrate Loading Reductions

In this section we examine results for the entire airshed and then present details of the changes in technology choices for the various scenarios, and the effect of different ozone episodes on the least-cost results when ozone benefits are included in the optimization. Then, we turn to results for runs with the Bay jurisdictions alone.

#### Policy Analysis 2.a: Airshed Analysis

In this section, we examine a series of potential least cost control policies that, as the name suggests, achieve some target emissions reduction at the lowest possible cost. Suppose that, like those interested in achieving the goals of the Clean Air Act, the Chesapeake Bay community also has a claim on NOx emissions reductions. The Bay policy community might want to focus states' efforts entirely on cleaning up the Bay and, therefore, would want to know the least cost way of getting nitrate loading reductions and compare these to the OTAG scenario. These least cost results can provide information about how policies to reduce nitrate loads to the Bay can be better targeted.

There are a number of different ways of looking at possible least-cost ways of reducing nitrate loadings to the Bay, all of which result in nitrate loading reductions equivalent to that from the OTAG scenario examined above. With the controls mandated by that scenario, nitrate loads to the Bay are forecast to fall by 13.67 million pounds a year (Table 8). We compare this OTAG scenario to three least-cost scenarios. This analysis minimizes the costs of reducing nitrate loadings across all of the states in the airshed. We examine the situation where the controls are limited only to the Bay jurisdictions in a separate section below.

The three least cost scenarios are:

*Scenario i*. Gross Least cost. The least cost way of achieving the nitrate loading reduction goal, minimizing gross abatement costs.

*Scenario ii.* Net Least cost; minimizing net costs instead of gross costs. This scenario is like (i). above but seeks a NOx emissions reduction allocation that meets the nitrate goal cost-effectively in light of the additional objective of maximizing health benefits from ozone reductions.

*Scenario iii*. Emission trading. The least cost solution for meeting the nitrate loading target may be too complex to implement because it requires each air source to trade emissions at different ratios with every other source.[25] As an alternative we examine a different incentive scenario, called emissions trading, where emissions are exchanged or traded ton for ton. It embodies the idea that a "least-cost" allocation of NOx abatement activity would take place, but using emissions rather than loadings as a unit of account. Such allocations are altered until the same nitrate loading reduction goal is met.

*Scenario i: Least cost*

Table 12 compares the least cost scenario for getting 13.7 million pounds of nitrate loading reductions to the OTAG scenario. The least cost allocation of reductions (minimizing gross costs, section I of Table 12) would attain this target at only one-third the cost of the OTAG scenario allocation, while resulting in NOx emissions reductions that are 80 percent of those under the OTAG allocation. This cost-savings arises out of differences in both the geographic and source-type allocations of NOx emissions reductions.

[25] The controversy over such trading in the OTAG debate is illustrative of the implementation difficulties for this approach.

| Table 12. Summary Results for OTAG and Least Cost Scenarios<br>Total Nitrate Reduction = 13.67 million lbs/yr for all Scenarios | | | | | | |
|---|---|---|---|---|---|---|
| | | Regions | | Sources | | |
| **OTAG** | **Total** | **Bay juris-dictions** | **Other Airshed States** | **Mobile Sources** | **Utilities** | **Non-utility Point Sources** |
| Share of Nitrate Loading Reduction (% of total) | 100% | 51% | 49% | 10% | 85% | 5% |
| NOx Emissions reductions (thousands of tons) | 1,024 | 310 | 714 | 96 | 858 | 70 |
| Cost ($millions) | $1,725 | $517 | $1,208 | $286 | $1,114 | $326 |
| Cost/pound nitrate loadings reduced | $126 | $73 | $182 | $206 | $96 | $465 |
| Net cost/pound nitrate loadings reduced (mean benefits) | $120 | | | | | |
| **I. Least Cost for Nitrate Loading Goal (minimize gross costs)** | | | | | | |
| Share of Nitrate Loading Reduction (% of total) | 100% | 62% | 38% | 26% | 66% | 8% |
| NOx Emissions reductions (thousands of tons) | 813 | 374 | 439 | 186 | 543 | 84 |
| Cost ($millions) | $581 | $371 | $210 | $90 | $440 | $51 |
| Cost/pound nitrate loadings reduced | $43 | $44 | $40 | $26 | $49 | $47 |
| Shadow Price ($/lb Nitrates) | $111 | | | | | |
| **II. Least Cost for Nitrate Loading Goal (minimize net costs, high benefits)** | | | | | | |
| Share of Nitrate Loading Reduction (% of total) | 100% | 57% | 43% | 23% | 69% | 8% |
| NOx Emissions reductions (thousands of tons) | 875 | 343 | 532 | 180 | 608 | 87 |
| Cost ($millions) | $655 | $331 | $324 | $94 | $504 | $56 |
| Ozone Benefits ($millions) | $921 | $377 | $543 | $193 | $640 | $89 |
| Net Costs ($millions) | -$266 | | | | | |
| Net cost/pound nitrate loading reduced (high benefits) | -$19 | -$8 | -$40 | -$31 | -$14 | -$31 |
| Shadow Price ($/lb Nitrates) | $44 | | | | | |
| **III. Emissions Trading (minimize net costs, high benefits)** | | | | | | |
| Share of Nitrate Loading Reduction (% of total) | 100% | 53% | 47% | 24% | 68% | 8% |
| NOx Emissions reductions (thousands of tons) | 988 | 324 | 663 | 214 | 680 | 93 |
| Cost ($millions) | $854 | $283 | $571 | $189 | $601 | $63 |
| Net Costs ($millions) | -$219 | | | | | |
| Net cost/pound reduced (high benefits) | -$16 | -$11 | -$22 | -$25 | -$12 | -$28 |
| Shadow Price ($/ton emissions) | $786 | | | | | |

Not surprisingly, the Bay jurisdictions take a larger share of the emission reductions in the least-cost scenario, 46 percent compared to 30 percent in the OTAG scenario (see Figure 4). In addition, the mobile source share in the airshed rises dramatically, from 10 percent in the OTAG regulatory scenario up to 26 percent in the least-cost scenario. And, although the breakdown by source type for the Bay states are not shown, the mobile source share goes from 12 percent in the OTAG scenario to 28 percent in the least cost scenario. These increases are at the expense of utility reductions. The shift away from utilities toward mobile sources may seem unlikely given the results presented earlier that the cost per pound of nitrate removed were much higher, on average, for mobile sources under the OTAG scenario. It must be, therefore, that although the average costs are high for mobile sources in the OTAG scenario, there are some mobile source options that, at the margin, are quite low. For instance, the average cost of reducing a pound of nitrate from mobile sources is $23, but the marginal cost of adding a LEV program to an existing high-enhanced I&M program is only $1/lb nitrate removed.[26]

Average costs per pound of nitrate reduced falls under the least cost scenario to $43/lb from $126/lb with OTAG. The distribution of the costs per pound reduced among states is shown in Figure 5. Costs per pound are actually negative in three states (even before accounting for ancillary benefits): Kentucky, the District of Columbia and New Jersey.

Because a least-cost strategy involves fewer emission reductions than the OTAG strategy, it might be that there would be lower ozone benefits, thereby canceling out any cost savings. In fact, ozone benefits are so low in the mean-benefits estimate that this consideration is not an issue. We find that the mean-benefits in the least-cost scenario are $65 million, compared to $84 million in the command and control scenario, making net costs $516 in the least-cost scenario compared to $1,642 in the command and control scenario (not shown). The least cost scenario still yields net costs that are less than one-third of those for the command and control scenario. This implies, for Bay policy making, that from both an ozone and a Bay perspective there can be substantial savings from targeting low cost sources of nitrate reductions.

Another way of gauging the costs of this scenario is by examining the shadow price of nitrate reductions, which is $111 per pound. This means that the last unit of nitrates reduced to just meet the nitrate loadings reduction target costs $111/pound. Any further tightening of the nitrate loadings reductions target will cost at least this much per pound. This figure can be compared to the average cost, which is far lower at $23/pound.

*Scenario ii: Net Least cost; minimizing the net costs of nitrate loading reduction*

Table 12 shows the results for a stronger test of the impact of including ozone benefits in the analysis. Part II examines the scenario where the allocation is designed to minimize the

---

[26] This extreme example is due to assumptions made by Pechan at the direction of EPA. Maximum LEV credits are only assigned with high enhanced I&M programs. As a result, extremely high benefits, in terms of NOx and VOC reductions, are assigned when this coupling occurs (see Appendix 3 for more discussion of this issue).

**Figure 4. Difference in NOx Emissions by State:
Least Cost (LC) compared to
Command and Control (CAC) Scenario**

**(OTAG Nitrate Reduction = 13.7 million lbs/year)**

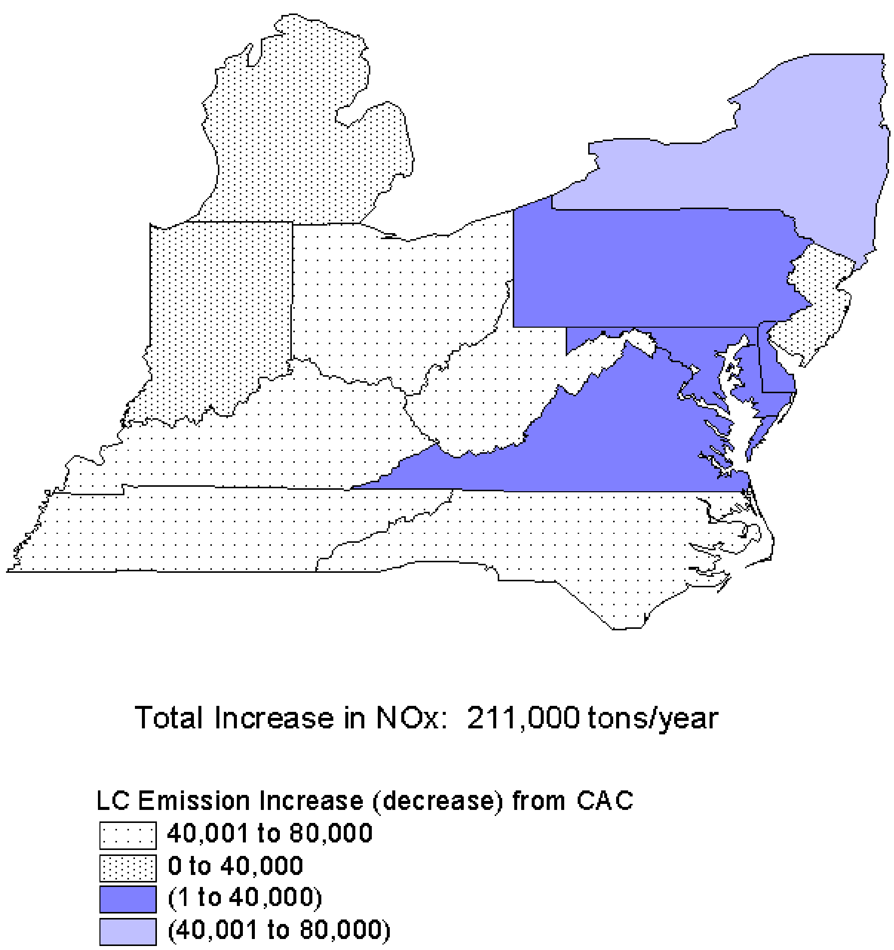
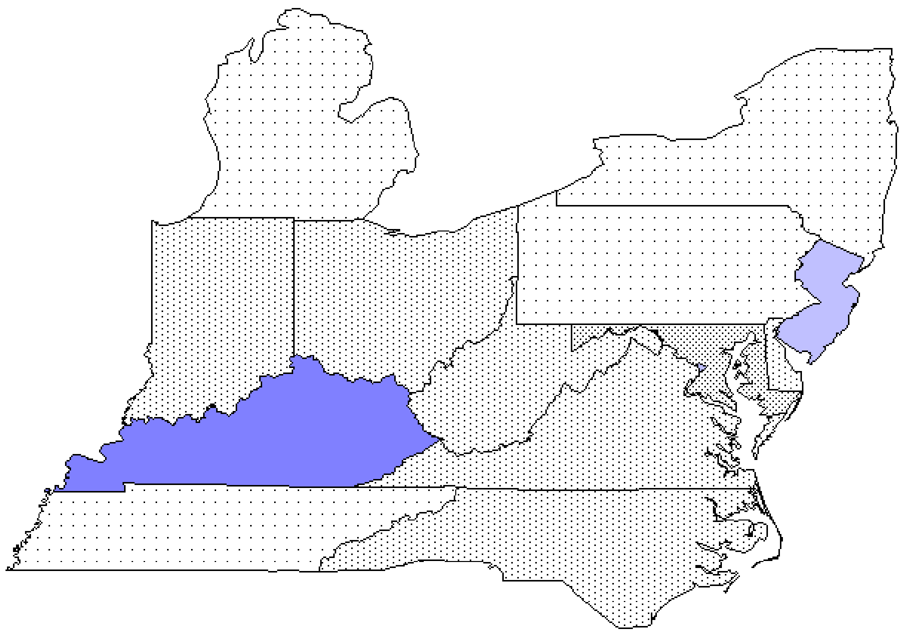


**Figure 5. Cost Effectiveness of Nitrate Reductions by State: Gross Least Cost Scenario for OTAG**



Average Cost per lb Nitrate reduced : $43

Positive (Negative) Cost / lb Nitrate Reduced

- 51 to 75
- 26 to 50
- 0 to 25
- (1 to 100)
- (101 to 600)

costs of reducing nitrate loads with the ozone benefits netted out. With the mean-benefits assumption, net costs are virtually unchanged when such an allocation is made (at $516 million) but there is some minor rearrangement of abatement (this run is not included in the table since there is so little impact).

To gain further insight, we redid the allocation using our "high" estimate unit value for ozone benefits. These results are shown in part II of Table 12. In this case, net costs are negative: $296 million in "net" benefits, with abatement costs of $655 million and health benefits of $921 million. Emission reductions are larger than in the original least-cost scenario: 875,000 tons versus 813,000 tons.

What about the differences in the allocation of abatement activity? Geographically, optimizing on net benefits instead of gross costs reduces the Bay jurisdictions' contributions to NOx emissions reductions for all source categories -- a 13 percent reduction for mobile sources and 6 percent and 7 percent reductions for utilities and point sources in the Bay, respectively. The lion's share of these differences can be accounted for by Pennsylvania, where, because of scavenging, NOx emissions reductions from mobile sources are found to *increase* ozone. Thus, to maximize ozone benefits requires lower reductions (or even increases) in NOx emissions from mobile sources in this state. (See Figures 6 and 7 for state-by-state details). For the airshed, NOx emissions reductions are 10 percent higher than they are in the gross cost scenario, because ozone reduction benefits are internalized in abatement option choices. These added emission reductions are primarily those from utilities in non--Bay jurisdictions. Mobile source emission reductions even fall slightly relative to the gross cost scenario.

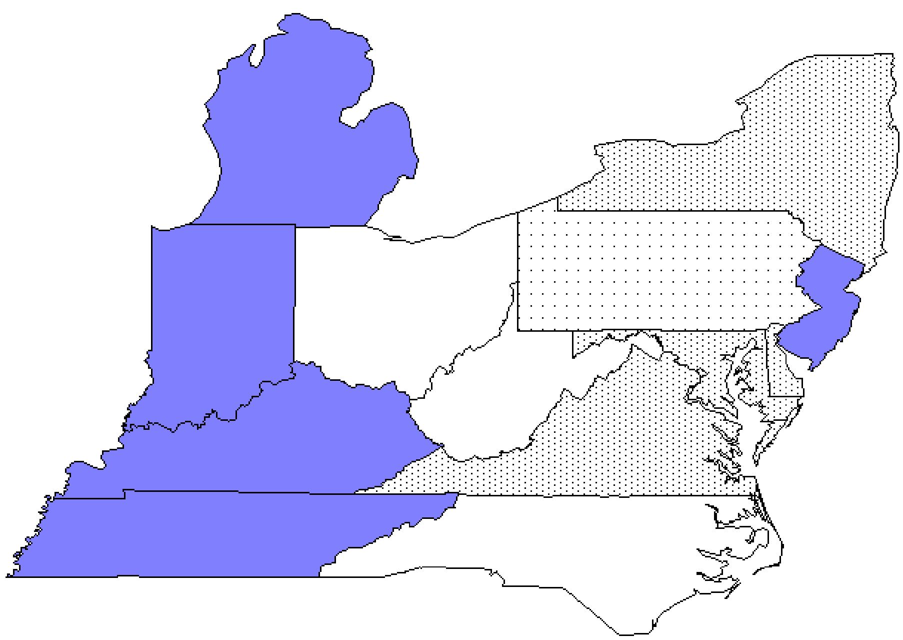
Finally, in Table 12 section II, note that the shadow price of nitrate reductions in this high benefit case is only $44/pound, much lower than the $111 shadow price when only gross costs are considered. This at first appears contrary to expectation because this scenario achieves greater NOx emission reduction which is likely to result in higher marginal costs. However, the associated marginal ozone benefits more than offset these costs.

*Scenario iii: Emissions trading*

This scenario achieves the target level of nitrate loading reductions but does so by allowing ton for ton trading among all NOx sources. Sources will therefore trade until the cost of removing a ton of NOx is equal among all sources. Section III of Table 12 shows results for the emissions trading scenario in which the high estimate of benefits is netted out from costs. Emission trading is not the lowest cost way of meeting the nitrate loading reduction goal (net costs are higher than those in the least-cost scenario (-$219 versus -$266 million), but still far lower than for the OTAG regulatory scenario ($740 million is costs of this scenario net of high benefits, see Table 8 above). Emissions trading is likely to be a more feasible way of achieving a lower cost solution, since tons of NOx can be traded one for one, making the transactions costs low. Under the true least cost scenario, sources might have to trade at different ratios, depending on the region and source type.

**Figure 6. Difference in NOx Emissions by State: Net Least Cost (with High Benefits) compared to Gross Least Cost Scenario**

**(OTAG Nitrate Reduction = 13.7 million lbs/year)**

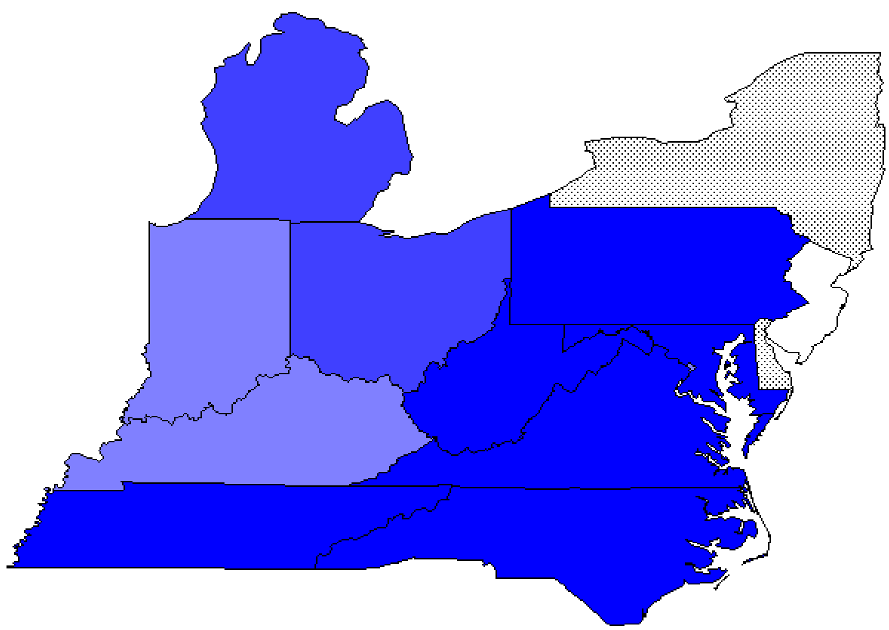


Total Decrease in NOx: 62,000 tons/year

Net LC Emission Increase (Decrease) from Gross LC

- 15,001 to 30,000
- 0 to 15,000
- (1 to 15,000)
- (15,001 to 30,000)

**Figure 7. Cost Effectiveness of Nitrate Reductions: Net Least Cost (with High Benefits) for OTAG**



Average Cost per lb Nitrate Reduced: $ -19

**Positive (Negative) Cost / lb Nitrate Reduced**

- 0 to 60
- (1 to 60)
- (61 to 120)
- (121 to 180)
- (181 to 560)

The costs of the emissions trading scenario are higher than the least-cost scenario because there are more emission reductions: 988 million tons versus only 875 million tons in the least-cost scenario (high benefits). Who must undertake the added emission reductions? It is not sources in the Bay jurisdictions, whose share of NOx emissions reductions falls from 39 percent of the total in the least-cost scenario to 33 percent of the total in the emissions trading scenario. In fact, most non-Bay jurisdiction increase their emissions and every Bay state decreases them. In fact, some Midwestern states like Tennessee and Michigan more than double their emission reductions of NOx under the emission trading scenario which includes high benefits.

In summary, the emission trading scenario may be more politically feasible, but is 18 percent more expensive than the least cost scenario, and shifts control away form the Bay states to the Midwest.

**Technology Choices by Scenario**

The above narrative has examined the effects of alternative scenarios on costs and NOx emissions reductions (and other factors) amongst the states and across source types. However, we have not yet discussed how technology choices change within any given source type between least cost and regulatory scenarios.

As shown in Table 13, turning first to mobile source details, the command and control scenario for the CAA leaves thirty-eight percent of the 84 mobile source regions without any mobile source emissions-reducing program. Low-enhanced I&M (20 percent) and high-enhanced I&M with reformulated gasoline (17.9 percent) are the most common CAA programs undertaken. These technology forecasts for mobile sources are based on state by state polling of what technologies were to be put in place as part of implementation of the CAA in each jurisdiction. Our analysis uses these full responses as the OTAG scenario, and the CAA choices for mobile sources were defined as the OTAG selections minus LEV programs. As a result, only the OTAG scenario includes LEV vehicle options. In the Pechan database, the LEV option can be used in only the Ozone Transport Region (OTR) or only Massachusetts and New York, or in the entire region. When LEV vehicles are chosen in combination with High Enhanced I/M very large emission reductions result (see Appendix 3 for more detail on this). Any of the technology options can be chosen under the least cost scenario, but the result is that there is much less variation in programs selected. 27 percent of regions do not undertake any emission reducing program, but all regions that do reduce emissions establish high-enhanced I&M or high- enhanced I&M with LEV.

**Table 13. Most common technologies in mobile source regions**
**(n=84)**

| | |
|---|---|
| **CAA** | |
| No policies | 38.1% |
| Low Enhanced I/M | 20.2% |
| High Enhanced I/M, with reformulated gas | 17.9% |
| High Enhanced I/M, without reformulated gas | 7.1% |
| Basic I/M, with reformulated gas | 4.8% |
| Low Enhanced I/M, with reformulated gas | 4.8% |
| | 92.9% |
| **OTAG** | |
| Min LEV, OTR states | 26.2% |
| Low Enhanced I/M, and Min LEV OTR states | 20.3% |
| High Enhanced I/M, reformulated gas, Max LEV OTR states | 17.9% |
| Min LEV, outside OTR states | 11.9% |
| High Enhanced I/M, Max LEV outside OTR | 6.0% |
| Basic I/M, and Min LEV outside OTR states | 3.6% |
| | 85.9% |
| **Least Cost** | |
| High Enhanced I/M, Max LEV | 69.0% |
| No policy | 27.4% |
| High Enhanced I/M | 3.6% |
| | 100% |

Considering details for the electric utility sector (Table 14), coal/wall boiler with low-NOx burner (LNB) is the most common technology (26.5 percent) used by the 656 utilities in the CAA command and control scenario. In the OTAG command and control scenario, utilities are most likely to use coal/wall boilers with selective non-catalytic reduction + selective catalytic reduction (SNCR+SCR) (23.9 percent) and coal/tangential boilers with SNCR+SCR (22.4 percent). In the least-cost allocation scenario, coal/wall boilers with SNCR+SCR are used less frequently (7.5 percent). Instead, the most common technology selected is coal/wall boiler with LNB+ overfire air (OFA) (18 percent). This technology is probably preferred because the addition of OFA *given* the existence of LNB may be more efficient than a switch to a different set of technologies, such as SNCR+SCR.

The most common technologies used by the 4,017 point sources are the same in the CAA and OTAG command and control and the least-cost scenarios. Industrial, Commercial, and Institutional (ICI) Boilers-residual oil with LNB and ICI boilers-coal/stoker with SNCR-Urea account for 32-38 percent of the technologies used in each scenario. A reason for the similarity between technologies used in the CAA and least-cost scenarios is that the cost minimization solution results in only 22 percent of point sources (as opposed to 55 percent of utilities) adopting a technology different from the CAA technology. Of the sources using technologies that differ from the CAA, ICI Boilers Natural Gas with OT+WI and Internal Combustion Engines-Gas with L-E are the most common technologies used.

**The Impact of Alternative Ozone Episodes on Model Results**

Three different ozone episodes (representing meteorological conditions during "typical" but not extreme high ozone conditions in the Northeast, in the Midwest, and in the Southeast) were specified (see section V.C). Table 15 shows the effects of different ozone episodes on the results for the least cost scenario that minimizes the costs net of high ozone benefits of achieving the 13.7 million pounds nitrate loading reduction goal. The Midwest and Southeast episodes are quite similar to each other and are somewhat different from the average of all episodes. They require lower NOx emissions reductions to meet the nitrate loadings target, which implies that the target will be met at a lower cost. The Northeast episode is quite different because reductions in NOx during this episode are so productive in reducing ozone, NOx reductions and costs are largest for these "NE" conditions.

For many regions, controlling utilities is more cost effective during the NE episode, so there is a shift toward more NOx emissions reduction from utilities and less reduction from mobile sources under this episode. This appears to push utility costs up more than it pushes mobile source costs down.

**Table 14. The Most Common Technologies in CAA, OTAG and Least Cost (OTAG) Scenarios. Utilities and Point Sources**

| Five most common technologies for utilities (n=656) | |
|---|---|
| **CAA** | |
| Utility Boiler-Coal/Wall: LNB | 26.5% |
| Utility Boiler-Coal Tangential: LNC1 | 21.5% |
| Gas Turbines-Natural Gas: None | 9.0% |
| Utility Boiler-Oil-Gas/Wall: LEA+BOOS | 7.6% |
| Utility Boiler-Oil/Gas/Tangential: None | 5.8% |
| | 70.4% |
| **OTAG** | |
| Utility Boiler-Coal/Wall: SNCR+SCR | 23.9% |
| Utility Boiler-Coal/Tangential SNCR+SCR | 22.4% |
| Gas Turbines-Natural Gas: None | 9.0% |
| Utility Boiler-Oil-Gas/Wall: LNB+SNCR | 6.0% |
| Utility Boiler -Coal/Vertical: Combustion Control | 5.5% |
| | 66.8% |
| **Least Cost for OTAG** | |
| Utility Boiler-Coal/Wall: LNB+OFA | 18.0% |
| Utility Boiler-Coal/Tangential LNC1 | 12.4% |
| Utility Boiler-Coal/Wall: SNCR+SCR | 7.5% |
| Gas Turbines-Natural Gas: None | 7.0% |
| Utility Boiler-Coal/Wall: LNB | 6.1% |
| | 50.9% |

| Five most common technologies for point sources (n=4,017) | |
|---|---|
| **CAA** | |
| ICI Boilers-Residual Oil: LNB | 18.2% |
| ICI Boilers-Coal/Stoker: SNCR-Urea | 14.7% |
| ICI Boilers-Natural Gas: LNB | 8.5% |
| ICI Boilers-Coal/Stoker: None | 8.1% |
| Internal Combustion Engines-Gas: L-E | 6.3% |
| | 55.8% |
| **OTAG** | |
| ICI Boilers-Residual Oil: LNB | 17.1% |
| ICI Boilers-Coal/Stoker: SNCR-Urea | 15.1% |
| ICI Boilers-Coal/Stoker: None | 7.8% |
| ICI Boilers-Natural Gas: LNB | 7.0% |
| Internal Combustion Engine-Gas: L-E (med speed) | 6.2% |
| | 53.2% |
| **Least Cost for OTAG** | |
| ICI Boilers-Residual Oil:LNB | 21.6% |
| ICI Boilers-Coal/Stoker: SNCR-Urea | 16.4% |
| Internal Combustion Engine-Gas: L-E | 10.3% |
| ICI Boilers-Coal/Stoker: None | 6.5% |
| ICI Boilers-Natural Gas: LNB | 5.7% |
| | 60.5% |

**Table 15. Effects of Different Ozone Episodes on Results From Net Least Cost Scenario[a]**

| **Least Cost for Nitrate Goal (minimize net costs, high benefits)** | **Average of all episodes** | **Northeast episode** | **Midwest episode** | **Southeast episode** |
|---|---|---|---|---|
| NOx Emissions reductions (millions of tons) | 875 | 900 | 857 | 865 |
| Gross Costs ($millions) | $655 | $701 | $645 | $666 |
| Ozone Benefits ($millions) | $921 | $1,093 | $834 | $802 |
| Net cost/pound nitrate loading reduced (high benefits) | -$19 | -$29 | -$14 | -$10 |
| Shadow price ($/lb nitrates) | $44 | $47 | $55 | $59 |

[a.] Minimize costs net of high ozone benefits to achieve 13.7 million pounds nitrate loading reduction goal.

**Policy Analysis 2.b: Bay Jurisdictions Analysis**

Another way to gain insight into policies the Chesapeake Bay jurisdictions might favor is to suppose that no NOx emissions reductions beyond those specified in the CAA case are to occur in the airshed as a result of EPA or state SIP policies. In this case, the Bay jurisdictions might wish to decide what further NOx emissions reductions from only the Bay jurisdictions are in their best interests. The Bay community might focus on cleaning up the Bay, seeking NOx emissions reductions within the Bay jurisdictions to meet nitrate loading reduction goals at least cost. Or the community might also want to count possible *ancillary* benefits from ozone reductions. Where should their efforts at cleaning up air sources be placed? Within what type of policy? On which sources?

As noted above, there are a number of different approaches to reduce nitrate loadings to the Bay. These are compared for equivalent nitrate loading reductions -- those registered by the OTAG scenario, but in this series of analyses applied to the Bay jurisdictions only. With the controls mandated by that scenario, nitrate loadings to the Bay are forecast to fall by 7 million pounds a year. As in the airshed analysis above, we compare this scenario to the three other scenarios. The analysis minimizes the costs of reducing nitrate loadings across the Bay jurisdictions and DC.

Table 16 compares the least cost approach and the emissions trading scenario to the OTAG scenario. We find that the choice of the least-cost and emissions trading scenario doesn't really matter, primarily because the nitrate source-receptor coefficients are not much different among the Bay jurisdictions. Savings are over 50 percent between either of these scenarios and the OTAG scenario. In particular, the least cost allocation of reductions would attain this goal at only 44 percent of the OTAG allocation, while resulting in NOx emissions reductions that are almost identical to OTAG) (310,000 tons for OTAG and 303,000 tons for the least-cost scenario). Note also that the marginal cost of obtaining the last unit of nitrate loading reductions to meet the nitrate goal is, at $91/pound, about three times the average cost of $32.

**Table 16. Summary Results from OTAG and Least Cost Scenarios for the Bay States**
**Total Nitrate Reduction = 7 million lbs/yr for all Scenarios**

| | Total | Regions | | | Sources | | |
|---|---|---|---|---|---|---|---|
| | | MD | VA | PA and DC | Mobile Sources | Utilities | Non-utility Point Sources |
| **OTAG** | | | | | | | |
| Share of Nitrate Loading Reduction (% of total) | 100% | 32% | 18% | 50% | 2% | 85% | 13% |
| NOx Emissions reductions (thousands of tons) | 310 | 67 | 67 | 176 | 39 | 264 | 7 |
| Cost ($millions) | $517 | $96 | $115 | $306 | $96 | $356 | 65 |
| Cost/pound nitrate loadings reduced | $73 | $42 | $88 | $87 | $102 | $60 | $434 |
| **I. Least Cost for Nitrate Goal (minimize gross costs)** | | | | | | | |
| Share of Nitrate Loading Reduction (% of total) | 100% | 34% | 20% | 45% | 20% | 74% | 6% |
| NOx Emissions Reductions (thousands of tons) | 303 | 71 | 72 | 160 | 59 | 223 | 21 |
| Cost ($millions) | $227 | $36 | $38 | $153 | - $26 | $229 | $24 |
| Cost/pound nitrate loadings reduced | $32 | $15 | $26 | $48 | - $19 | $44 | $48 |
| Net cost/pound reduced (high benefits) | -$4 | | | | | | |
| Shadow Price ($/lb Nitrates) | $91 | | | | | | |
| **II. Least Cost for Nitrate Goal (minimize net costs, high benefits)** | | | | | | | |
| Share of Nitrate Loading Reduction (% of total) | 100% | 33% | 18% | 48% | 23% | 71% | 6% |
| NOx Emissions Reductions (thousands of tons) | 307 | 70 | 66 | 172 | 71 | 216 | 20 |
| Gross Cost ($millions) | $245 | $32 | $36 | $177 | $4 | $219 | $22 |
| Ozone Benefits ($millions) | $332 | $64 | $68 | $200 | $80 | $233 | $19 |
| Gross cost/pound nitrate loadings reduced | $35 | $14 | $27 | $52 | $3 | $44 | $47 |
| Net cost/pound nitrate loadings reduced (high benefits) | - $12 | - $14 | -$25 | -$5 | -$48 | -$3 | $6 |
| Shadow Price ($/lb nitrates) | $38 | | | | | | |
| **III. Emissions Trading (minimize net costs, high benefits)** | | | | | | | |
| Share of Nitrate Loadings Reduction (% of total) | 100% | 27% | 22% | 50% | 26% | 68% | 6% |
| NOx Emissions reductions (thousands of tons) | 317 | 58 | 78 | 180 | 84 | 213 | 19 |
| Cost ($millions) | $264 | $11 | $60 | $193 | $32 | $210 | $21 |
| Gross cost/pound nitrate loadings reduced (High Benefits) | $37 | $6 | $38 | $55 | $18 | $44 | $47 |
| Net cost/pound nitrate loadings reduced (High Benefits) | -$12 | -$22 | -$14 | -$5 | -$37 | -$4 | $7 |
| Shadow Price ($/ton emissions) | $781 | | | | | | |

The state-by-state allocations of NOx emissions reductions across these scenarios are not much different, either. Maryland and Virginia reduce NOx more and Pennsylvania reduces NOx less in the least-cost scenarios relative to the OTAG scenario. Rather, the principal differences between the least-cost and OTAG scenarios are the greater reliance on mobile source controls and non-utility point source controls (with correspondingly less reliance on utility controls) in the least cost scenario, for each of the Bay jurisdictions.

For the Bay jurisdictions the mobile source share of nitrate loading reductions goes from just 2 percent in the OTAG scenario to 20 percent in the (gross) least cost scenario. These increases are mirrored by a fall in the utility share from 85 percent to 74 percent. The shift away from utilities toward mobile sources occurs because, as we noted above, there are mobile source control options *not mandated* in the OTAG scenario that deliver NOx emissions reductions at negative costs relative to the *mandated* option. Costs per pound of nitrate loadings reduced are therefore negative for mobile sources (-$19 per pound); they are about equal ($44 vs. $48 per pound) for utilities and non-utility point sources.

Table 16 also shows what happens when *high* ozone benefits are included in the least-cost decision on allocation of NOx emissions reductions. Net costs are negative: $87 million in "net" benefits, with abatement costs of $245 million and health benefits of $332 million. Compared to the "gross" least-cost scenario where ozone benefits are not included in the allocation decision, costs are a bit higher (the latter scenario showed costs of $227 million), although the total emission reductions are not much different (303,000 tons in the gross cost scenario and 307,000 in the net cost scenario). Note that the shadow price is still positive (i.e., the costs to society of meeting the nitrate reduction goal are positive at the margin at $38/pound), but negative on average (-$12).

What shifts in NOx abatement does concern for ozone benefits bring about? Virginia's NOx emissions reductions are smaller and Pennsylvania's are larger when there is the additional concern about ozone factored into the allocation decision (the net-least cost scenario). This is because of the greater ozone benefits that Pennsylvania's NOx emission reductions can bring compared to Virginia. There is less reliance on utility NOx emissions reductions in the "net" least-cost scenario (the utility share falls from 85 percent to 70 percent of the total reductions), with more reliance on mobile sources. Note that benefits exceed costs for each of the states and for mobile and utility source types but not for non-utility point sources.

Table 16 also shows that the emissions trading scenario, *in the aggregate and with "high" benefits*, is more or less equivalent in this instance to the "net" least-cost scenario (with *high* benefits), in the sense that average gross and net costs per nitrate loadings reduced are nearly identical. This result suggests that a simple one-for-one NOx trading system does relatively well at meeting nitrate loading goals cost-effectively. However, the allocation of NOx emissions reductions across states is somewhat different. Maryland's NOx emissions reductions are 83 percent of what they were in the least-cost scenario, while Virginia's are 18 percent larger. Across sources, mobile sources control much more with emissions trading than in the least-cost scenario. Thus, moving to an emissions trading system from a least-cost system results in some state and source reallocations. But these are minor compared to the differences between these two scenarios and the OTAG scenario.

**Policy Analysis 3: National Policy and the Bay Community's Stake**

The Bay jurisdictions have a stake in the outcome of the national debate over NOx controls to meet the ozone standards in the eastern U.S. This debate has been fed by the OTAG process, in which a wide variety of stakeholders in the eastern U.S. met over a two - year period to develop models and policy insights addressing the long-range ozone transport issue. The debate became more focused with EPA's recent issuance of a proposed SIP call, requiring specified NOx emissions reductions of each state, and of a NOx trading guidance document detailing EPA's plan for obtaining NOx emissions reductions cost-effectively (EPA, 1997; EPA, 1998).

The results presented in this section are designed to help the states affected by these EPA actions, particularly the Chesapeake Bay jurisdictions, to formulate their responses to EPA's NOx emissions reduction strategies. Accordingly, we have recast the problem examined above, that of minimizing NOx abatement costs (with or without netting out ancillary ozone-related health benefits) to meet nitrate loading reduction goals, to one in which NOx abatement costs are minimized for meeting either aggregate NOx emissions reduction targets -- in effect, the EPA emissions trading plan -- or ozone exposure reduction goals.[27] To address the Bay jurisdictions' particular concerns, we note the abatement costs and ozone benefits of alternative policy approaches as they affect the Bay jurisdictions as well as noting the nitrate loadings reductions that are an ancillary consequence of a NOx emissions reduction strategy to reduce ozone. It is important to note that our database is not identical to that which EPA used to undergird its SIP call and Trading Guidance Documents.[28] Thus, quantitative comparisons between EPA's SIP call and Guidance with our results are not advised. However, our results provide an indication of the possible consequences of this program relative to a no trading program and to a variety of variations on the trading theme.

---

[27] More specifically, the emissions constraint problem is defined as finding the allocation of ozone season NOx emissions that minimizes the *annual* costs of reducing these emissions subject to the constraint. This approach leads to a higher cost per ton removed than would be the case if we counted NOx emissions all year round or adjusted the cost estimates for each technology to seasonal costs. The Pechan database does not permit the latter to be done in a fully defensible way. The emissions constraint is computed from the Pechan database, which provides estimates of NOx emissions per day of the ozone season for all sources and their NOx control technologies. Nitrate loading reductions are tallied assuming that NOx emissions are reduced all year long. Nitrate reductions for all scenarios would be reduced by a scalar if we assumed that the NOx emissions controls were shut off in the non-ozone season.

The ozone exposure constraint problem is defined as finding the allocation of ozone season NOx emissions that minimizes the annual costs of reducing seasonal NOx to meet an ozone exposure constraint. This constraint is defined in terms of the sum of the products of reductions in ozone concentrations per person per day at each receptor, multiplied by the population exposed in each receptor and 153 days in the ozone season.

[28] Particular examples include: our domain is only 13 states plus D.C., EPA's is 22; the emissions and cost databases are different.

The NOx control policies we consider have basically four "margins," or design parameters: policy type (OTAG-CAC, one-for-one emissions trading, and ozone exposure trading); participants (some states may decide to form a trading zone with no trading across the zone); source type (EPA recommends only utilities in the program, but other options would add other point sources and even mobile sources); and the underlying S-R matrix used to measure the spatially differentiated effects of NOx on ozone.

There are three groups of results. The first examines variations on EPA's proposed NOx trading plan, such as the source types participating; the second examines the effects of using "trading ratios" to guide emissions trading between sources; and the third considers a "two-zone" option, in which the Bay jurisdictions "go it alone," trading amongst themselves but not with non-Bay jurisdictions (and vice-versa). We consider least-cost and emissions trading options in each group.

1. <u>The EPA's Proposed Trading Plan</u>

EPA's SIP call set emission reductions by utility, point, and mobile sources beyond CAA baselines, corresponding to our OTAG scenario. The agency then issued proposed guidance for how a NOx trading plan could be constructed that would help some sources meet these goals at reasonable costs. The "key" features of the plan (key in terms of the modeling capabilities we have) are that only utilities are in the trading program and emissions trade one-for-one between utilities, even though a ton of emissions in one area may do more or less damage than a ton of emissions in another area. This simplification mirrored that made for SO2 allowance trading in Title IV of the CAAA and was justified by EPA on grounds that introducing trading ratios other than one-to-one would make the trading system too complex.[29]

2. <u>EPA's Plan and the OTAG Scenario</u>

The most basic comparison is between the OTAG scenario and EPA's plan, given in columns (2) and (3) on Table 17. This comparison is made by using the utility NOx emissions from the OTAG scenario as the constraint for the cost minimization model. For EPA's plan, non-utility point and mobile sources are assumed to perform according to the OTAG scenario. The total cost of the OTAG scenario is $1.726 billion, as noted in VI.A above, where utilities shoulder $1.1 billion of this cost. With EPA's plan, in which the utility reductions can be obtained by inter-utility trading of emissions anywhere in the domain, utility costs fall about 13 percent to $969 million. Average cost- effectiveness, in terms of NOx emissions reductions by utilities falls from $2,788 to $2,425 per ton. That they don't fall more is probably because the performance standard is very tight, so there is little opportunity for gains from trade.

---

[29] EPA's "SIP call" actually requires utilities to meet a 0.15lb NOx/mmbtu standard and 70 percent reduction for large non-utility boilers, whichever is less. Our modeling assumes, in contrast to EPA, but more in line with the OTAG process, that utilities may meet the less stringent of the 0.15lb/mmbtu standard or 85 percent reduction and 70 percent reduction for the large non-utility boilers.

**Table 17. Consequences of Alternative NOx trading plans to Meet Aggregate NOx Emissions Reduction Targets**

| | OTAG | EPA Plan: Emissons Trading (ET) with Utilities only[1] | ET Adding Other Point Sources[26] | ET Adding Mobile Sources | Ozone Exposure Trading (OT) with Utilities only: Average Episode | OT with Utilities only: NE Episode | OT with all sources (6a) |
|---|---|---|---|---|---|---|---|
| **Airshed Abatement Costs ($ millions)** | 1,726 | 1,581 | 1,245 | 987 | 1,578 | 1,379 | 805 |
| **Airshed Utility Costs ($ millions)** | 1,114 | 969 | 888 | 728 | 966 | 767 | 436 |
| **Airshed Ozone Exposure Reductions** | 41,612 | 41,580 | 41,596 | 42,060 | 41,580 | 41,580 | 42,060 |
| **Bay jurisdictions Abatement Costs ($ millions)** | 517 | 472 | 412 | 327 | 473 | 403 | 246 |
| **Bay jurisdictions (MD-VA) Ozone Benefits ($millions)** | 22.9 | 21.5 | 23.0 | 22.4 | 19.5 | 16.6 | 24.2 |
| **Nitrate Loading Reductions (millions of lbs.)** | 13.7 | 12.7 | 13.3 | 13.9 | 11.6 | 11.6 | 11.0 |

[1] Sources excluded in this trading scenario are assumed to be under the OTAG plan.

Should the Bay jurisdictions support EPA's plan over the OTAG scenario? Abatement costs are lower, with all of this cost savings accrued by utilities. So, on a cost basis the plan deserves support. Yet, nitrate loadings to the Bay fall more with the command and control approach -- 13.7 million pounds a year versus 12.7 million pounds -- combined with higher ozone benefits than for EPA's plan. Thus, the Bay jurisdictions face a tradeoff between costs and nitrate loading reductions when choosing between these two options.

3. Variations on EPA's Plan

Would variations on EPA's Plan be better for the airshed and the Bay jurisdictions? We examine three variations: a plan that adds other point sources to the trading program, a plan that adds mobile sources to the previous plan, and a plan that introduces emissions trading at ratios dictated by runs of the Urban Airshed Model-V, which we term ozone exposure trading (OT).

*Adding Point Sources*. Allowing additional source-types to trade should reduce overall abatement costs of meeting the aggregate emission reductions target, but the effect on the Bay jurisdictions is not predictable, *a priori*. Column (3) shows that adding other point sources further lowers aggregate abatement costs from $1.581 billion with utility trading alone to $1.245 billion, lowering utility costs from $969 million to $888 million. From the Bay jurisdictions' perspective, this program has lower abatement costs *and* larger nitrate loading reductions, so it clearly dominates the utility-only trading plan.

*Adding Mobile Sources.* It will obviously be technologically difficult, if not impossible, to incorporate all mobile sources into a trading program in the foreseeable future. However, some states already have programs involving mobile source and point source trades, called Mobile Emission Reduction Credit Programs, and one can imagine trading programs involving fleet vehicles. In any event, modeling this type of program provides an estimate of potential benefits from incorporating this source type into trading.

The result is a rather dramatic reduction in airshed-wide abatement costs compared to the utility plus other point source trading program, with costs falling from $1.245 billion to $987 million and further cost savings to the utilities. Costs for the Bay jurisdictions fall dramatically as well when mobile sources are added because other regions foot the bill. A surprising finding is that nitrate loading reductions are actually greater with this plan (13.9 million pounds) than with any of the above approaches, including the OTAG scenario. Ozone benefits are about the same as with the other approaches. The combination of much lower abatement costs and greater nitrate loadings reductions makes this all-source trading scenario a possible winner.

*Departing From 1:1 Trades.* Section III made it clear that reductions in NOx emissions are more productive in reducing ozone exposures from some locations than from others, and that this "productivity" varies by episode type. We model several variants of a trading program to capture such effects. The first (column 6) returns to the utility-only program, where the trading ratios are taken from the source-receptor coefficients applicable to the average of the NE, MW, and SE episodes and the constraint is the ozone exposure reductions obtained from reducing emissions under EPA's planned program (column 2).

Surprisingly, the utility-only, average episode trading policy performs no better from the airshed perspective, and worse over our three measures from the Bay jurisdictions' perspective than EPA's one-for-one NOx trading plan. However, this result is not robust to the episode chosen. If, instead, trading ratios are defined by the northeast episode (column 7), then utility abatement costs to meet the identical standard ($767 million) are quite a bit lower then the EPA Plan ($969 million), with Bay jurisdiction costs falling as well. Cost-effectiveness in terms of NOx emissions reductions is now only $2,153.

The second variant is an all-source ozone exposure trading program. This program (column 7), in comparison to the all-source emissions trading program (column 5) saves another $180 million and reduces utility costs to $436 million. The Bay jurisdictions face a tradeoff between lower costs and higher ozone benefits on the one hand and lower nitrate loadings reductions on the other.

4. <u>Sensitivity to alternative ozone episodes to meet ozone exposure reduction targets</u>

A number of questions are raised by the above results showing how the consequences of using the NE or the average episode to define trading ratios are so different. On the one hand, the advantages of "spatially differentiated" permit trading systems are well known. Conceptual work by Tietenberg (1985) and others proves the efficiency benefits of designing tradable permit systems that take into account the differential location effects of emissions on concentrations. On the other hand, whether such a more complex system is "worth" the trouble is strictly an empirical issue. In addition, this literature is silent on the issue of how to establish such ratios in the presence of modeling uncertainty and meteorological variability. Clearly, the first step is to establish whether choosing alternative S-R coefficients for the trading ratios leads to wildly different results.

Table 18 shows the consequences of meeting ozone exposure targets by trading emissions at ratios dictated by the UAM-V for various meteorological conditions for both a utility-only and an all-source permit system. Columns (2) and (3) show the NE and average episode results again, but add results for an all-source program. Results using the other episode types are in columns (4) and (5).

Examining the utility-only program first, we see the low costs associated with the NE S-R coefficients, an indication of how productive NE meteorological conditions are for reducing ozone. The MW episode and the average episode are the most similar in their consequences. And defining trading ratios according to the SE episode results in the target not being reached, even with all utilities maximally reducing their NOx emissions.

From the Bay's perspective, we see the usual tradeoffs. Nitrate loading reductions are larger for the most costly scenario to the airshed (11.6 million pounds for the average episode compared to 10.4 million pounds for the NE episode). From a cost perspective, however, the Bay jurisdictions want the most efficient approach for the airshed, which is to trade according to the NE episode. Ozone benefits to the Bay are largest with the MW episode. Comparing these

**Table 18. Consequences of Alternative Trading Ratios for Meeting An Ozone Exposure Reduction Target (from CAC). Utilities only (all sources)**

| | Average Episode | NE Episode | MW Episode | SE Episode | CAC Average Episode | CAC NE Episode |
|---|---|---|---|---|---|---|
| **Utilities-only** | | | | | | |
| Airshed Abatement Costs ($millions) | 966 | 767 | 916 | Infeasible | 1114 | |
| Bay jurisdiction Abatement Costs ($millions) | 310 | 241 | 311 | I | 356 | |
| Bay jurisdiction (MD-VA) Ozone Benefits ($millions) | 19.5 | 16.6 | 26.2 | I | 24.8 | 27.2 |
| Nitrate loading reductions (millions of lbs.) | 11.6 | 10.4 | 11.4 | I | 11.6 | |
| **All Source-Types** | | | | | | |
| Airshed Abatement Costs ($millions) | 789 | 617 | 843 | 878 | 1726 | |
| Bay jurisdiction Abatement Costs ($millions) | 240 | 230 | 287 | 181 | 520 | |
| Bay jurisdiction (MD-VA) Ozone Benefits ($millions) | 24.2 | 26.7 | 30.8 | 10.6 | 27.5 | 31.6 |
| Nitrate loading reductions (millions of lbs.) | 10.9 | 10.0 | 11.8 | 11.1 | 13.7 | |

trading options to the corresponding OTAG scenario, we find lower costs and lower ozone benefits to the Bay jurisdictions with the trading options compared to the OTAG scenario. Thus, the standard trade-offs are much in evidence, with nitrate loadings not much of an issue, except for the relatively low loadings reductions for the NE episode.

Because mobile sources have different S-R coefficients than point sources, it would be no surprise to find the story somewhat different when all sources are considered. Indeed, while the NE episode is still the cheapest, this is followed by the average, then the MW, and then the SE episode, the last being feasible when all source types are in the program. The implications for the Bay are different as well. The most surprising finding is that even though trading according to ratios based on the SE episode is the most expensive for the airshed, it's the *cheapest* for the Bay jurisdictions, who do not participate very much given their low effects on ozone throughout the domain under the SE conditions.[30] Furthermore, nitrate loading reductions are actually larger with trading ratios from the MW episode than from the SE episode. Note how with the SE episode, ozone benefits within the Bay jurisdictions are very small. So, the Bay jurisdictions face particularly interesting tradeoffs in supporting one set of trading ratios over another.

What implications does this sensitivity of results to trading ratios have for tradable program design from the airshed perspective? If attainment is what matters (and we are NOT modeling attainment here), then, given that the NE episode is associated with most nonattainment (at least for the current standard), the trading ratios should be set for a NE episode to deliver the most ozone reductions when and where they are most needed. At the same time, if we are most concerned about maximizing the ozone reduction benefits per ton of NOx reduced (which is, of course, what we are ultimately concerned about), then with NOx emissions reductions being the most "productive" during NE conditions, the NE trading ratios should be adopted. Thus, in both situations, the NE ratios are the most promising. This is a lucky result, meaning that the program can be designed to take advantage of a win-win situation.

There is a downside to dealing with this meteorological variability and cost sensitivity by using the NE episode -- during a SE or MW episode, the exposure target probably won't be met. To illustrate the problem, we found the least-cost allocation of NOx emissions to meet the ozone exposure constraint using the NE episode S-R coefficients and then calculated ozone exposure reductions *given* the NOx emissions reductions allocation. These runs of the model show (Table 19) that the occurrence of a MW episode and a SE episode, respectively, when trading ratios are based on the NE episode, results in ozone exposure targets being missed in the airshed by 20 percent and 22 percent, respectively. The estimates for the Bay are also included. The MW episode can clearly cause the greatest losses to the Bay, if trading ratios are established on the basis of the NE episode.

---

[30] Note that trading ratios are administratively established and drive the NOx allocations. However, ozone benefits are related to *actual* meteorological conditions that will differ from any administratively determined ratios.

**Table 19. Loss in Benefits from a Mismatch between Trading Ratios and Episode Experienced.**

| | Domain | | Bay | |
|---|---|---|---|---|
| Episode Type | Ozone Exposures (billion person* ppb*day/ozone season) | % of NE | Ozone Exposures (billion person* ppb*day/ozone season) | % of NE |
| NE | 49.1 | - | 14.9 | - |
| SE | 38.4 | 78 | 17.4 | 85 |
| MW | 39.2 | 80 | 8.0 | 54 |
| AVE | 42.3 | 86 | 13.4 | 90 |

## 5. Two Zone Trading

Table 20 presents the results of dividing the domain into two trading zones--one with the Bay jurisdictions, the other with the other states. Such a program might be implemented if the Bay jurisdictions, in their concern about nitrate loadings, were uneasy about permitting NOx trades from sellers outside the area to buyers in it, although supportive of programs where the Bay jurisdictions are net sellers of NOx. While the Bay jurisdictions might be glad to "export" pollution to the non-Bay jurisdictions to help reduce nitrate loadings, the non-Bay jurisdictions are not likely to permit this one-way arrangement. So if the Bay jurisdictions pressed their plan, the only stable outcome is where the Bay jurisdictions meet their ozone reduction or emissions reduction targets alone, creating, in effect, two trading zones with no trade between them.

**Table 20. Single zone versus two-zone trading to Meet Ozone Exposure Targets. All source types**

| | 2 | | | 2a | | 2b |
|---|---|---|---|---|---|---|
| | **Domain-Wide Trading** | | | **Two Trading Zones** | | |
| | Bay | Non-Bay | Total | Bay | Non-Bay | Total |
| Abatement Costs ($millions/year) | 243.9 | 545.5 | 789.4 | 248.7 | 540.7 | 789.4 |
| Emission reductions (tons NOx/year) | 120.1 | 268.4 | 388.5 | 121.2 | 267.3 | 388.5 |
| Nitrate Loading Reductions (lbs Nitrate/year) | 5,529 | 5,369 | 10,898 | 5,668 | 5,369 | 11,037 |

Conceptually, a two-zone program can be expected to be more expensive, in the aggregate, than a one-zone program, but a comparison of other effects is ambiguous conceptually. For instance, if sources in the Bay "zone" have lower marginal costs than those outside this zone, then, in effect, there will be NOx emissions on net being traded from within to outside of the Bay zone. If sources within the Bay zone have higher marginal costs than those outside this zone, the trading will be in the reverse direction on net.

The results (in table 20) show that aggregate abatement costs are virtually identical in the one zone and two zone situations, indicating flat marginal cost curves around the implied control levels. However, in the two zone situation, the Bay's sources shoulder more of the abatement costs than in the one-zone situation and the sources in the non-Bay zone shoulder correspondingly less. Nitrate loading reductions are higher in the two zone situation because emission reductions are greater for sources in this zone. The same issue can be investigated for an emissions trading system, in this case EPA's plan, i.e., including utilities only, for one zone and for two zones. The results are identical qualitatively. So, the Bay jurisdictions face the tradeoff of higher abatement costs for greater nitrate loading reductions if they elect to form a separate trading zone.

## VII. CONCLUSIONS AND POLICY IMPLICATIONS

Airborne nitrates contribute from 20 to 30 percent of the nitrate loading to the Chesapeake Bay. Given the interest in policies for reducing NOx emissions to meet ambient air quality standards for ozone (and particulates), it is useful to examine how the parallel concern over nitrate deposition to the Bay affects the appropriate allocation of NOx emissions reductions among utilities, non-utility point sources, and mobile sources.

Our analyses of the issue are of three types. The first is a detailed analysis of the effect on nitrate loadings to the Bay of command and control (CAC) policies specified in the CAA and as part of the OTAG process. The second is a comparison of alternative scenarios for reducing NOx emissions that meet nitrate loading goals, with or without concern for reducing ozone concentrations and the health effects they cause. The third is a comparison of alternative approaches to allocate NOx emissions to meet ozone exposure goals while tallying the ancillary effect on nitrate loadings. It therefore focuses on the stake that the Bay jurisdictions have in the outcome of negotiations over NOx trading programs being developed by EPA for reducing ozone in the Eastern U.S.

This analysis is unique in several respects -- in its exploration of quantitative estimates of the lower cost alternatives to current regulatory programs, in its integration of ancillary benefits, in its focus on the Bay jurisdictions, and in its examination of these ancillary benefits for alternative meteorological episodes.

The most basic message to the Bay community is that the Chesapeake Bay obtains a bonus from efforts to reduce NOx emissions to meet the ambient air quality standard for ozone. In our analysis, airborne NOx emission reductions slated to occur under the Clean Air Act by 2005 in the 13 state (plus D.C.) domain will reduce nitrate loadings to the Bay by about 27 percent of the baseline airborne loadings of 66 million pounds per year. The additional control of NOx contemplated in our OTAG scenario (0.15lb NOx/mmBTU for utilities or 85 percent removal, whichever is less and 70 percent reductions for large point sources; LEVs for mobile sources) is estimated to result in an additional 20 percent reduction from this baseline. This bonus is not a panacea, however, because final decisions on the additional levels of NOx control have yet to be made, and, even if these estimates are accurate, the approximately 47 percent reduction in associated nitrate loadings constitutes only about a one-sixth reduction in the controllable nitrate loads from all sources in the Bay.

Clearly, the costs of obtaining such reductions can be significantly reduced by rearranging the allocation of emissions reductions to take advantage of source-type and locational considerations. This paper has not focused on the policies needed to obtain such savings but has pointed to the locations and technologies that have the greatest potential to achieve the targets in lower cost ways. In addition, we find that adding consideration of ancillary ozone-related health benefits to the picture does not alter any qualitative conclusions. Quantitatively, unless a link between ozone and mortality risk is assumed, the benefits are too small to affect the cost-saving allocations of NOx reductions. With such a link, NOx reductions are larger in total and shift to non-Bay states and towards utility sources. These specific effects are sensitive to the source-receptor coefficients linking NOx to ozone, however.

Our analyses also suggest that the Bay jurisdictions have a stake in the outcome of the NOx trading debate -- that some trading designs can lead to better outcomes for these jurisdictions than others. Nevertheless, a common feature of cost-savings policies is that they both rearrange emissions reductions and, in the aggregate, reduce emissions less than a command and control system. Thus, some trading regimes result in significantly smaller loadings reductions (up to 25 percent smaller) than the command and control approach.

### A. Regulatory Analysis (CAA and OTAG Scenarios)

Both in implementing CAA mandates and in going beyond such direct mandates to implementing controls suggested by our OTAG scenario, the Bay jurisdictions (Pennsylvania, Maryland, Virginia, and the District of Columbia) are a relatively inexpensive source of nitrate loading reductions, measured in $/nitrate loading terms. This result is due almost entirely to the locational advantage of these states, i.e., their NOx emissions reductions produce greater reductions in nitrate loadings to the Bay than NOx emissions reductions from more outlying states. However, in some cases, NOx emissions reductions from sources in the Bay jurisdictions are less cost-effective than reductions in other states when effectiveness is measured as $/ton of emissions. Utilities are responsible for the vast majority of the NOx and nitrate loading reductions.

An important extension of the analysis of current regulatory programs of this paper is to take into consideration both the air and water impacts of controls in the cost-effectiveness analyses. Clearly, not all of the costs of NOx emissions reductions that are undertaken primarily to improve air quality should be attributed to nitrate loading reduction in the Bay. We develop a method for netting the ancillary benefits from NOx emissions reductions costs to develop a more inclusive measure of net social costs of NOx emissions reduction policies. Are the above conclusions changed if some of these effects -- the ozone-related health benefits from NOx emissions reductions -- are added to the picture? In general, the conclusions are *not altered* when ozone-related health benefits are considered. With our best estimate of the health benefits per unit ozone exposure reductions, the health benefits are too small to matter.

However, there is significant controversy in estimating ozone-related health benefits, particularly concerning the effects of ozone on mortality risks. We have developed a "high" estimate of ozone benefits per person-ppb exposure change that incorporates effects on mortality -- a benefit category ignored in our "best" benefit estimates. However, even using this high estimate, the cost-effectiveness of NOx emissions reductions in the non-Bay jurisdictions versus the Bay jurisdictions is not much altered. This occurs because the S-R relationships linking NOx emissions to ozone changes over the ozone airshed are similar across states. Therefore, states like Kentucky and Maryland have a similar effect on aggregate ancillary health benefits, which does not alter the cost-effectiveness comparison based on net rather than gross cost terms. At the same time, each state's cost-effectiveness drops dramatically when the "high" benefit estimate is netted out of abatement cost. For the airhsed, the aggregate cost effectiveness falls from \$126/lb nitrate to \$54/lb nitrate loading when moving from the CAA to the OTAG scenario.

## B. Comparison of OTAG Regulatory Scenario to Alternative Scenarios for Meeting Nitrate Loading Targets

As shown in table 12, significant cost savings to meet nitrate loading reduction goals are potentially available by improving the targeting of NOx emissions reduction policies to take advantage of cost-differentials across sources and locational differences in the creation and transport of ozone and nitrate loadings to the Bay. The limit of such cost savings (under the least-cost scenario) is over \$1 billion annually (for utility, point source, and mobile sources) from the OTAG regulatory scenario costs of \$1.7 billion. These savings can be realized by shifting NOx abatement to mobile sources, such as through enhanced I&M,[31] relaxing the 0.15 lb NOx/mmBTU requirement on utilities and large point sources (such as through a trading program) and shifting abatement to sources of all types within the Bay jurisdictions.

In attempting to meet nitrate loading reduction goals while minimizing *net* costs (abatement costs minus ancillary "high" ozone-related health benefits) emission reductions and abatement costs are larger than those in the least-cost scenario. But, ozone benefits are far larger than they would be if the allocation of NOx were done to meet nitrate loading goals alone. At the same time, there is a important regional aspect to the comparison of NOx

---

[31] The sensitivity to this result needs to be explored in future work. There are very large emission reductions assumed to occur at very low cost under enhanced I/M in the MOBILE Model (see Appendix 3). These assumptions have recently been called into question (Harrington, McConnell and Cannon, 1998).

emissions reductions in this scenario to one without ancillary benefits considered. In particular, eastern Pennsylvania and New Jersey exhibit ozone scavenging in all of the ozone episodes we considered. This results in a "bias" against NOx emissions reductions for these states. For the NE (Bermuda high) episode, the bias is towards NOx emissions reductions in Ohio and New York.

While the least-cost scenario can provide a sense of the limits of cost savings over the OTAG scenarios, our emissions trading scenario provides, perhaps, a more realistic view of such savings. In this scenario, in effect, emissions are traded across regions and sources ton-for-ton, but the emissions cap is set (through a series of optimization runs) to meet the same nitrate loading goal for the Bay that was used for both the OTAG scenario and the least cost scenario described above. This scenario delivers substantial cost savings over the OTAG scenario, but costs and emission reductions are larger than they are in the least-cost scenario. The reductions of NOx shift much more away from the Bay jurisdictions to states in the Midwest.

Alternatively, if the emissions trading program to meet the nitrate loading goal is made more complex by accounting for ancillary ozone benefits (high benefits case) at the same time,[32] the nitrate loading goal can be met at much lower cost per ton than can be obtained from an emissions trading program alone.

Because it may be difficult to envision a trading program involving states outside the Bay partners in the Chesapeake Bay Agreement, we also examined the consequences of implementing a variety of policies to meet nitrate loading goals only within these Bay jurisdictions. The conclusions for this "Bay jurisdiction" analysis are qualitatively the same as those for the above "airshed" analysis, with the exception that the results of an emissions trading program look much more like those in the least-cost scenario. This occurs because the least-cost scenario takes locational differences in the NOx to nitrate loading relationships into account while the emissions trading scenario does not, and these relationships are very similar across the Bay jurisdictions but not across all states in the airshed.

Finally, we looked at the impact of different ozone episodes on the least cost allocations for achieving the nitrate loading reduction goal. The type of episode causes costs, emission reductions and allocation across sources to vary, but not dramatically. Of the three types of ozone episodes, the Northeast episode stands out as requiring the most NOx emissions reductions to meet the target. As a result, it is more costly to reach the nitrate loading goal under the Northeast episode than under the other episodes (about 10 percent more expensive). In addition, for many regions controlling utilities is more cost-effective during the Northeast episode, so if policy were to made based on those events there would be a shift toward more utility reductions and fewer mobile source reductions.

[32] One way to do this is through a tax on NOx emissions that internalizes the ancillary ozone benefits.

### C. The Bay Jurisdictions' Stake in NOx Trading for Reducing Ambient Ozone

This series of scenarios is motivated by EPA's recently issued proposed Guidance Document for a NOx trading program to help meet ambient ozone standards in the eastern U.S. It takes the point of view of the Bay jurisdictions, asking whether particular design features of this program deserve support from the Bay jurisdictions, in terms of the ancillary effect on nitrate loadings, ozone health benefits enjoyed by people living in the Bay jurisdictions, and costs to sources within the Bay jurisdictions. The results cannot be quantitatively compared to EPA's modeling efforts in support of the guidance document, or to modeling results produced by the OTAG process, however. Our domain is only 13 states plus D.C., while the EPA domain includes an additional eight states; we do not consider area sources, while EPA does; the emissions and cost database we used from Pechan is not identical to the database used by EPA, particularly with respect to electric utilities; and our ozone episodes are different than EPA's, although we both consider Bermuda High-type meteorology. Most important, our analysis is designed to meet not only emissions reduction targets, as EPA's, but also aggregate ozone exposure reduction goals.

In spite of these differences, we feel that our modeling results are qualitatively comparable to EPA's. For instance, our analysis supports the EPA analysis that a utilities-only emissions trading system saves money over the OTAG scenario applied to all sources. We find, for instance, that overall costs are $1.581 billion with emissions trading relative to $1.726 billion with our OTAG regulatory scenario, where all of the savings is experienced by the utilities in the program (whose costs fall from $1.114 billion to $969 million annually).

The overall conclusion from this analysis is that the Bay jurisdictions should not be indifferent to design features of a NOx trading program. Along some design dimensions, one option may dominate. More frequently, there is a tradeoff to the Bay jurisdictions between gaining greater nitrate loading reductions and bearing greater costs, with ozone benefits not being determinative.

In terms of source participation, the Bay jurisdictions benefit more when the trading program includes point and mobile sources (in this paper we are not addressing how such a program could be structured, or even the extent to which it might be possible). This program delivers more nitrate loading reductions, no lower ozone benefits, and lower costs to Bay jurisdiction sources than either EPA's plan, a plan that adds point sources, or the OTAG scenario.

A number of analysts have been urging EPA to modify their proposed trading system to account for locational and source-specific differences in the productivity of NOx emissions reductions in creating ozone and in the transport of ozone to other regions. Account could be taken by altering the trading ratios for emission trades between sources in different regions and of different types. We examined this issue and found that substantial cost savings are possible, over and above those available from emissions trading. If locational differences could be fully taken advantage of, costs would fall to $805 million for an all-source trading ratio program, relative to $987 for an all-source one-for-one trading program.

Interestingly, whether EPA's utility-only trading program should depart from a one-for-one design depends on the type of episode underlying this program. For the average of our "typical" episodes, EPA's one for one trading program performs just as well as a trading ratio program. With NE trading ratios, however, utilities would save $200 million with a trading ratio program.

This finding prompted us to test the sensitivity of our results against the four episode types (an average and a NE, MW, and SE episode type). We find that the episode-type affects costs, emission reductions and nitrate loadings (indirectly); and that the NE episode features the most productive meteorology for reducing ozone through NOx emissions reductions. As a result, using this episode to model least-cost ozone reductions and emissions trading yields the lowest costs to the airshed and the Bay to meet ozone exposure reduction goals. This is a particularly important result because the NE episode is the one usually associated with nonattainment of ozone standards. Therefore, attainment considerations and our results all point towards specifying trading ratios in terms of a NE episode.

Still, because NE conditions occur only during a part of the summer, it is relevant to consider what losses may arise if conditions differ. We find, first, that with SE meteorology, it is not even feasible to reach the given ozone exposure reduction goal. Further, through the domain, the occurrence of other conditions can result in a 15 percent loss in ozone exposure reductions if the allocation of NOx emissions reduction is optimized for a NE episode but some other episode-type occurs. The Bay jurisdictions would experience smaller losses unless a SE episode dominates the summer.

Finally, we envision the Bay jurisdictions deciding that the NOx trading program designed solely for ozone reductions may not meet their needs. A particular problem might be that the seasonal program being advanced by EPA will result in nitrate loading reductions far short of those from a year-round program. Therefore, we examine the consequences to the Bay jurisdictions of creating a trading zone, leaving the other airshed states to create their own trading zone (with no trading of emissions between zones). We find that this autonomy comes at a price, i.e., the cost of the additional nitrate loading reductions to the Bay jurisdiction sources are fairly high. The sources pay $5 million more than they otherwise would to obtain an extra 140,000 pounds of nitrate loading reductions, or $36/pound. However, this is only a small amount below what they would pay in the CAC scenario -- $38/pound.

## D. Limitations and Caveats

This analysis has a number of limitations that temper the usefulness and credibility of its conclusions. Additional research is planned to address some of these concerns (particularly numbers 3,4,6,7, and 8).

1. The source domain is limited to 13 states plus D.C. rather than the 22 or 36 commonly considered in discussion about limiting NOx emissions in the Eastern U.S.

2. The receptor domain encompasses New England as well as the source regions; however, it is narrower than that considered by EPA and in the OTAG process.
3. The Pechan emissions and cost data are not from the latest Pechan dataset. In addition, the utility data were judged to be less reliable by EPA than the information contained in the Integrated Planning Model (IPM), which EPA uses in its own analyses of NOx trading issues. Our own analysis shows that, at least, the Pechan data contain more abatement options than the IPM dataset.
4. The Pechan mobile source data appear, in light of research conducted at RFF and elsewhere, to seriously underestimate the costs and overestimate the effectiveness of NOx emissions reduction for some mobile source options.
5. The cost data were originally developed in such a way as to leave ambiguous the incremental costs of choosing one abatement option over another. (See Section II.)
6. The S-R coefficients linking NOx emissions to nitrate loadings are not the latest version from RADM and have some limitations. First, the source regions are highly aggregated spatially, often to the state level. Second, there is spotty coverage of some states, or even no coverage, requiring imputations of S-Rs using a nearby state's data. Third, there is spotty information on S-Rs by source-type, with some states provided with separate S-Rs for utility, other major point, and mobile sources, and other states provided with aggregate coefficients. Finally, there are several internal inconsistencies (see Appendix A: Regional Acid Deposition Model Summary Output, in Pechan, 1996) in the S-Rs. In addition, the chemistry of the RADM model has continued to evolve since the underlying model runs were performed, calling into question the credibility of the S-Rs being used to date.
7. The ozone S-R coefficients have not yet been subjected to sensitivity analysis or other tests of their robustness. Such coefficients, even for the same meteorology, may be sensitive to the starting initial concentrations, the magnitude of the emissions change, and whether NOx emissions change at more than one region at a time.
8. Our modeling so far omits consideration of ancillary benefits to NOx emissions reductions to the extent such reductions lower airborne nitrates, which register as lower fine particle concentrations. EPA has recently recognized the health threat posed by such particles in setting a new fine particulate ambient air quality standards.
9. Ancillary benefit estimates are limited to health. Benefits may also be registered for materials, for instance. See Lee et al. (1995) for a full discussion of the benefit pathways for ozone and PM.

# Appendix 1: THE CONVEX HULL AND OPTIMIZATION

This appendix describes the algorithm and mathematical approach used to find the least-cost allocation of NOx emissions reductions to meet the given constraints. This algorithm included determination of the "convex hull," which is basically a marginal cost function containing only technologies not dominated by other technologies, in terms of costs and emission reductions. It also included using the convex hulls for each source in an optimization process, which was written in GAMS (General Algebraic Modeling System).

**Determining convex hull**

Scenarios examining the costs of reduction of pollutants used the 2005 baseline technologies for each source as the starting point in the determination of the convex hulls. Similarly, scenarios examining the costs of pollution reduction, given the fact that CAA technologies were in place, used the 2005 CAA technologies as the starting point in the determination of the convex hulls. The appropriate cost and NOx emissions reduction of every possible technology for each source was computed relative to the appropriate starting point for the analysis. All technologies with NOx emissions reductions less than the NOx emissions reductions of the starting point technology were dropped from the set of feasible options. Technologies having greater NOx 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 NOx 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 NOx emissions reduction, load to emission ratio, technology identifiers, etc.) was retained and this point was used as the new reference point in determining the next point on the convex hull. Technologies having fewer NOx 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 NOx emissions reduction information for the technologies on the convex hull was retained in the form of marginal costs and marginal NOx emissions reductions. Total cost and NOx emissions reduction can be derived from the marginal values since:

Total cost of technology i =

$$\sum_{i} [MC_i * \Delta NR_i]$$

where ΔNR is the marginal NOx emissions reduction.

Total NOx emissions reduction from technology i =

$$\sum_{i} MC_i * \Delta NR_i$$

$$\sum_{i} \Delta NR_i$$

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 MC information) was determined for use in cost minimization scenarios optimizing over gross costs, while the convex hull based on minimum net MC (retaining the gross MC information) was determined for cost minimization scenarios optimizing over net costs.

<u>Optimization Process</u>:

All optimization scenarios minimized the total cost of NOx emissions reduction:

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

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
x=marginal NOx emissions reduction of technology j given technology j-1 for source k

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

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

Where:
d= marginal NOx emissions reduction as determined by convex hull

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

$$\sum_k a_k \left( \sum_j x_{kj} \right) \geq L$$

Where:

L= nitrate load reduction or ozone reduction from command and control scenarios
a= coefficient converting NOx emissions into nitrate loading or ozone

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

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

Where:
E=NOx emissions reduction constraint

Using the optimal $x_{kj}$ derived from the constrained cost minimization problem and the coefficients $a_k$ , the corresponding nitrate loading or ozone changes were calculated. If the calculated load or ozone reduction was less that L, E was increased and the problem was resolved. This process was repeated until the minimum value of E resulting in a nitrate loading or ozone reduction as least as great as L. was found.

# Appendix 2: UTILITY COST FUNCTIONS

The Pechan database models a wide variety of NOx control strategies for each point source boiler type. This is illustrated below for dry-bottom wall-fired coal boilers. Most of the capital cost equations are variations of the standard form $aX^b$, where X is the maximum nameplate capacity of the boiler. The operating and maintenance (O&M) cost equations can have fixed and/or variable cost components. The variable cost component is related to the maximum nameplate capacity of the boiler, and may or may not be related to the boiler capacity factor.

**Dry-Bottom Wall-Fired Coal Boilers**

| NOx Controls | Average Reduction (%) | Average Cost per Ton ($/ton) | 300 MW: Capital Cost ($1000/ year) | 450 MW: Capital Cost ($1000/ year) | 300 MW: O&M Cost ($1000/ year) | 450 MW: O&M Cost ($1000/ year) |
|---|---|---|---|---|---|---|
| LNB (low-NOx burner) | 46 | 180 | 4,140 | 4,698 | 127 | 167 |
| SNCR (selective non-catalytic reduction) | 48 | 975 | 2,889 | 3,684 | 1,159 | 1,739 |
| NGR (natural gas reburn) | 50 | 521 | 6,355 | 8,106 | 1,390 | 2,085 |
| LNB + OFA (overfire air) | 53 | 300 | 5,477 | 6,215 | 147 | 190 |
| LNB + SNCR | 65 | 493 | 10,870 | 13,982 | 1,255 | 1,772 |
| SCR (selective catalytic reduction) | 75 | 1,815 | 19,066 | 24,318 | 2,178 | 3,267 |
| SNCR+SCR | 88 | 1,315 | 13,404 | 17,096 | 2,804 | 4,206 |
| LNB+OFA+SCR | 90 | 1,590 | 41,066 | 53,928 | 5,546 | 8,183 |

Note: A capacity factor of 0.6148 was assumed.

## Appendix 3: MOBILE SOURCE DATA

Pechan and Associates, Inc. provided the Mobile Source data for the analysis in this paper. This dataset included emission reductions and costs for the following control options (described in more detail in Pechan's *Emission Reduction and Cost Analysis Model for NOx (ERCAM-NOx, 1997)*:

- One of three possible I/M programs
  - Basic I/M
  - Low Enhanced I/M
  - High Enhanced I/M
- Reformulated gasoline
- Reformulated diesel
- LEV program – the emission reduction from LEVs depends on what type of I/M program is in place (LEVs get substantially more reductions if they are initiated in combination with Enhanced I/M; see below for more detail).

There are 7 categories of vehicles listed in Table A3-1 below. The table shows which control options can be applied to which vehicle types in our analysis.

**Table A3-1. Vehicle Type and Control Option Combinations**

| | Heavy Duty Diesel Vehicles (HDDV) | Heavy Duty Gas Vehicles (HDGV) | Light Duty Diesel Vehicles (LDDV) | Light Duty Diesel Trucks (LDDT) | Light Duty Gas Trucks 1 (LDGT1) | Light Duty Gas Trucks 2 (LDGT2) | Light Duty Gas Vehicles (LDGV) |
|---|---|---|---|---|---|---|---|
| **I&M** | | | | | Yes | Yes | Yes |
| **Reformulated Gasoline** | | Yes | | | Yes | Yes | Yes |
| **Reformulated Diesel** | Yes | | Yes | Yes | | | |
| **LEV** | | | | | Yes | | Yes |

### Scenarios and Spatial Aggregation:

Each county in the airshed has some baseline option/s in place as of 1990 (baseline levels include only basic or low enhanced I/M in some counties). There are then two additional possible scenarios for further control:

We use two different **regulatory scenarios** in the paper:

1) **Clean Air Act –** This scenario represents the controls that regions have undertaken or intend to undertake under the Clean Air mandates. States were asked what programs they would implement to conform with CAA.

2) **OTAG –** This includes LEV in regions that indicated an intention to implement LEV.

Then there are all the possible **least cost scenario** results which depend on what is being optimized. Under the least cost scenarios, regions are free to adopt whatever controls will be lowest cost and still meet the regional nutrient or emissions reduction objectives. To do the optimization, the counties are aggregated into about 74 groups. The groups are separated by state, and within states by ozone attainment vs. non-attainment status, and MSA status (MSAs are occasionally separate if they are large enough to be required to do I/M under the CAA-100,000+ population, but are not part of a non-attainment area). In the Pechan dataset, LEV vehicles could be introduced only by region: the possible regions are Massachusetts and New York, the rest of the Ozone Transport Region (OTR) and states outside the OTR. In our least cost analysis, LEV vehicles can be introduced in any of the 84 regions.

Table A3-2 shows the resulting controls under different scenarios.

**Problems with the Mobile Source data:**

There were several problems we found in the Pechan dataset on mobile sources used in this analysis. We list them below and indicate what correction we took, if any, in each case.

1. The Pechan dataset assumed that I/M programs could only be applied to LDGVs but in many regions light duty trucks are inspected as well (LDGT1). We extended the possible set of options counties could implement to include I/M for LDGT1.
2. Motorcycles were an $8^{th}$ type of vehicle in the original dataset but were dropped for this analysis because there were some inconsistencies in the data.
3. Emissions of VOC for LDGVs are higher under low enhanced I/M than they are under basic I/M. This appears to be because of different assumptions in the MOBILE Model which is used to forecast emission reductions under different I/M scenarios. For example, basic I/M runs assumed 100% compliance whereas low enhanced assumed 96% compliance. We have not changed these forecasts in this version of the paper. However, it means that Low Enhanced I/M is unlikely to be chosen in the least cost scenarios (higher cost and less emission reduction than basic I/M).
4. Pechan revised its original cost estimate for high enhanced I&M from $5.70/vehicle as listed in ERCAM-NOx, to $15.70/vehicle. We use the revised estimate in this paper.

**Table A3-2. Mobile Source NOx Abatement Technologies For CAA, OTAG and Least Cost (OTAG) Scenarios**

| Mobile Source Technologies for Mobile Source Regions (n=84) | | | | | | |
|---|---|---|---|---|---|---|
| Abatement Options | | | | Distribution of Technologies (%) | | |
| I&M | Reform Gas | ReformDiesel | LEV | CAA | OTAG | Least Cost Scenario for OTAG |
| 0 | 0 | 0 | 0 | 38.1 | 0.0 | 27.4 |
| 0 | 0 | 0 | 6 | 0.0 | 26.2 | 0.0 |
| 0 | 0 | 0 | 8 | 0.0 | 11.9 | 0.0 |
| 0 | 1 | 0 | 0 | 3.6 | 0.0 | 0.0 |
| 0 | 1 | 0 | 6 | 0.0 | 1.2 | 0.0 |
| 0 | 1 | 0 | 8 | 0.0 | 2.4 | 0.0 |
| 2 | 0 | 0 | 0 | 3.6 | 0.0 | 0.0 |
| 2 | 0 | 0 | 8 | 0.0 | 3.6 | 0.0 |
| 2 | 1 | 0 | 0 | 4.8 | 0.0 | 0.0 |
| 2 | 1 | 0 | 6 | 0.0 | 1.2 | 0.0 |
| 2 | 1 | 0 | 8 | 0.0 | 3.6 | 0.0 |
| 4 | 0 | 0 | 0 | 20.2 | 0.0 | 0.0 |
| 4 | 0 | 0 | 4 | 0.0 | 4.8 | 0.0 |
| 4 | 0 | 0 | 6 | 0.0 | 15.5 | 0.0 |
| 4 | 1 | 0 | 0 | 4.8 | 0.0 | 0.0 |
| 4 | 1 | 0 | 4 | 0.0 | 2.4 | 0.0 |
| 4 | 1 | 0 | 6 | 0.0 | 2.4 | 0.0 |
| 6 | 0 | 0 | 0 | 7.1 | 0.0 | 3.6 |
| 6 | 0 | 0 | 5 | 0.0 | 0.0 | 9.5 |
| 6 | 0 | 0 | 7 | 0.0 | 1.2 | 56.0 |
| 6 | 0 | 0 | 9 | 0.0 | 6.0 | 3.6 |
| 6 | 1 | 0 | 0 | 17.9 | 0.0 | 0.0 |
| 6 | 1 | 0 | 5 | 0.0 | 2.4 | 0.0 |
| 6 | 1 | 0 | 7 | 0.0 | 15.5 | 0.0 |

**Codes:**
I&M: 0 = none; 2 = Basic; 4 = Low Enhanced; 6 = High Enhanced
Reformulated Gasoline: 0 = none; 1 = in place
Reformulated Diesel: 0 = none; 1 = in place
Low Emission Vehicles: 0 = none; 4 = Minimum National LEV (Mass. and NY); 5 = Maximum National LEV (MA and NY); 6 = Minimum National LEV (other OTR states); 7 = Maximum National LEV (other OTR states); 8 = Minimum National LEV (outside the OTR); 9 = Maximum National LEV (outside the OTR)

5. The emission reductions assumed by LEVs in combination with High Enhanced I/M are huge, and they come at a relatively low cost. Table A3-3 below shows the emissions reductions and costs for various mobile source options in Allegheny County. The emissions reductions are relative to the base case which is no mobile source controls. It is clear from the table why either LEV with High Enhanced I/M or High Enhanced I/M alone is chosen in the least cost scenario. We believe these estimates are due to some very optimistic assumptions in the MOBILE inventory Model (Harrington, McConnell and Cannon, 1998). We plan to examine the sensitivity of our results to these assumptions, and to reassess some of these estimates in future work.

**Table A3-3. LDGV Emissions Reductions and Costs for Mobile Source Controls Allegany County, Maryland**

| | NOx Emissions Reductions (Tons/Ozone Season) | Total Cost ($1990/Ozone Season) | Cost Effectiveness ($1990/Ton NOx Reduced) |
|---|---|---|---|
| No Controls | 0 | 0 | N/A |
| Reformulated Gasoline | 27 | 261,189 | 9,811 |
| Basic I&M | 7 | 113,349 | 15,434 |
| Low Enhanced I&M | 8 | 113,349 | 14,817 |
| High Enhanced I&M | 110 | 312,208 | 2,846 |
| LEV | 21 | 92,866 | 4,398 |
| LEV + Low Enhanced I&M | 28 | 206,215 | 7,285 |
| LEV + High Enhanced I&M | 199 | 405,074 | 2,038 |

## REFERENCES

Bloyd, Cary, et al. 1996. *Tracking and Analysis Framework (TAF) Model Documentation and User's Guide*, ANL/DIS/TM-36, Argonne National Laboratory, December.

Chesapeake Executive Council. 1987. Chesapeake Bay Agreement, Annapolis, MD.

Clinton, President William J. 1997. Presidential Directive on the Implementation of New National Ambient Air Quality Standards for Ozone and Fine Particulates.

Dennis, R. L. 1997. "Using the Regional Acid Deposition Model to Determine the Nitrogen Deposition Airshed of the Chesapeake Bay Watershed," in Joel Baker, ed., *Atmospheric Deposition to the Great Lakes and Coastal Waters* (Pensacola, FL: Society of Environmental Toxicology and Chemistry), pp. 393-413.

Guthrie, Paul D., and Alan J. Krupnick. 1998. *Air Quality Benefits Associated with Reductions in Atmospheric Nitrogen Deposition to the Chesapeake Bay, Modeling Methodology: Ozone and Fine Particles*.

Guthrie, Paul D., Gerard E. Mansell, and Dongfen Gao. 1998. *Assessment of Potential Air Quality Benefits Associated with NOx Emissions reductions in the Chesapeake Bay Airshed*.

Harrington, Winston, V. McConnell, M. Cannon. 1998. *A Behavioral Analysis of EPA's MOBILE Emission Factor Model*, Discussion Paper, Resources for the Future, Washington, D.C., Summer.

Jones-Lee, M. W., M. Hammerton and P. R. Phillips. 1985. "The Value of Safety: Results of a National Sample Survey," *The Economic Journal*, vol. 95, pp. 49-72.

Krupnick, Alan J., and Dallas Burtraw. 1996. "The Social Costs of Electricity: Do the Numbers Add Up?" *Resource and Energy Economics*, vol. 18, pp. 423-466.

Krupnick, Alan J., Wallace E. Oates, and Eric Van den Berg. 1983. "On Marketable Air-Pollution Permits: The Case for a System of Pollution Offsets," *Journal of Environmental Economics and Management*, vol. 10, no. 3 (September), pp. 233-247.

Lee, Russell., Alan Krupnick, Dallas Burtraw, and others. 1995. *The External Costs and Benefits of Fuel Cycles*, A Study by the U.S. Department of Energy and the Commission of the European Communities, prepared by Oak Ridge National Laboratory and Resources for the Future (McGraw-Hill, Utility Data Institute, New Jersey), December.

Linker, L. C., G. E. Stigall, C. H. Chang, and A. S. Donigian. 1996. "Aquatic Accounting: Chesapeake Bay Watershed Model Quantifies Nutrient Loads," *Water Environment and Technology*, vol. 8, no. 1, pp. 48-52.

Miller, Paul, Praveen Amar, Marika Tatsutani. 1997. *The Long-range Transport of Ozone and its Precursors in the Eastern United States*, NESCAUM, March.

Montgomery, David. 1972. "Markets in Licenses and Efficient Pollution Control Programs," *Journal of Economic Theory*, vol. 5, no. 3, December, pp. 395-418.

Pechan, E. H., Associates, Inc. 1996a. *Analysis of Costs and Benefits of a National Low Emission Vehicle Program*, EPA-68-D3-0035, Work Assignment No, II-67, prepared for Vehicle Programs and Compliance Division, U.S. Environmental Protection Agency, Washington, D.C., November.

Pechan, E. H., Associates, Inc. 1996b *Atmospheric Nitrogen Deposition Loadings to the Chesapeake Bay: An Initial Analysis of Cost Effectiveness of Control Options*, EPA-68-D3-0035, Work Assignment No. II-76, prepared for Chesapeake Bay Program Office, U.S. Environmental Protection Agency, Annapolis, MD, November.

Pechan, E. H., Associates, Inc. 1997. *The Emission Reduction and Cost Analysis Model for NOx (ERCAM-NOx)*, EPA-68-D3-0035, Work Assignment No. II-63, prepared for Ozone Policy and Strategies Group, U.S. Environmental Protection Agency, Research Triangle Park, NC, September.

Rao, S. T., Tim Mount, and Gary Dorris. 1997. *Least Cost Options for Ozone Improvement in the Eastern United States: Estimating Emission Weights*, prepared for the New York State Energy Research and Development Authority.

Sisler, James F. 1996. *Spatial and Seasonal Patterns and Long Term Variability of the Composition of the Haze in the United States--An Analysis of Data from the IMPROVE Network*, Cooperative Institute for Research in the Atmosphere, Colorado State University, July.

Tietenberg, Thomas H. 1985. *Emissions Trading, an Exercise in Reforming Pollution Policy* (Washington, D.C.: Resources for the Future, Inc.).

U.S. Environmental Protection Agency (EPA). 1993. Regional Interim Emission Inventories (1987-1991), Volume I: Development Methodologies. EPA-450/R-93-021a, U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, Research Triangle Park, NC, May.

U.S. Environmental Protection Agency (EPA). 1997. Finding of Significant Contribution and Rulemaking for Certain States in the Ozone Transport Assessment Group Region for Purposes of Reducing Regional Transport of Ozone. 62 FR 60318, U.S. Environmental Protection Agency, November.

U.S. Environmental Protection Agency (EPA). 1998. Supplemental Notice for the Finding of Significant Contribution and Rulemaking for Certain States in the Ozone Transport Assessment Group Region for Purposes of Reducing Regional Transport of Ozone. 40 CFR Part 52, U.S. Environmental Protection Agency, March.