Least Cost Control Strategies to Reduce Ozone in the Northeastern Urban Corridor

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Least Cost Control Strategies to Reduce Ozone in the Northeastern Urban Corridor

Final Report

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NOTICE

ABSTRACT

This report describes the results of a study to assess least-cost control strategies to reduce ozone in the northeastern urban corridor. Using a variety of statistical methods, photochemical modeling, and economic analyses, the research distilled the relative impacts of reductions in emissions of nitrogen oxides (NOx) and volatile organic compounds (VOC) in different geographical areas on ozone improvement in the Metro-East Corridor that stretches from Washington to Boston. The research used a statistical representation of the predicted concentrations of ozone from a photochemical model to simplify the relationships between input emission variables and output ozone concentrations. The research developed quantitative factors which convey the expected ozone improvement in the Metro-East Corridor from NOx and VOC emission reductions from different source types and source subregions (e.g., transportation sources of NOx and VOC in the Metro-East itself; utility sources of NOx in Ohio, West Virginia, and Virginia; transportation sources of NOx in Pennsylvania, etc.) The research provided a ranking of emission control alternatives in terms of cost-effectiveness. The research demonstrated the need for a mix of emission reductions on both local and regional scales, building on the work of the Ozone Transport Assessment Group (OTAG).

Key Words: air quality, ozone, nitrogen oxides (NOx), volatile organic compounds (VOC), least-cost pollution control

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EXECUTIVE SUMMARY

This study was commissioned by the New York State Energy Research and Development Authority (NYSERDA) for evaluating the cost-effectiveness of different regulatory strategies for reducing ozone concentrations in the northeastern urban corridor extending from Washington, D.C. to Boston, Massachusetts (the receptor region). Since observed concentrations of ozone in the urban corridor often exceed the federal standards during the summer months on many occasions, Baltimore, New York City and Philadelphia have been designated as "severe" nonattainment regions by the U.S. Environmental Protection Agency (EPA). The analysis presented in this report considers a wide variety of measures for reducing the emissions of Nitrogen Oxides (NO_X) and Volatile Organic Compounds (VOC) which are the two primary precursors of ozone. Emission reductions are evaluated in all major sectors of the economy (transportation, electric utility, industrial, commercial and residential). The emission reductions from point sources (primarily elevated NO_X emissions from utility and industrial boilers) cover all northeastern and midwestern states whereas the emission reductions from ground-level sources of NO_X and VOC (primarily from the transportation and commercial sectors) cover the northeastern states only. All other sources of emissions, including biogenic sources, are held constant throughout the analysis at the levels in the base inventory of emissions. The inventory used in this study is the EPA's inventory for 2007, which assumes that all existing mandated federal controls on emissions have been implemented. Hence, the analysis evaluates the effects of additional reductions of emissions beyond the base in 2007; these include the reductions of NO_X emissions in EPA's current State Implementation Plan (SIP) call.

The study was conducted in two phases. In the first phase, a series of 38 different patterns of emission reductions were specified to predict the changes in the maximum ozone concentration (both 1-hr and 8-hr average) in the northeastern urban corridor (the receptor region) for a typical ozone episode. The same meteorological conditions were used for all scenarios and they correspond to the ones that prevailed in July 1995 when observed ozone concentrations in the receptor region greatly exceeded the federal standards. The photochemical model of ozone production UAM-V (Urban Airshed Model - V) was used to predict the maximum levels of ozone concentration in the receptor region for all scenarios. Statistical methods were used to estimate emission weights for different types of emissions (e.g. elevated NO_X) and different subregions (e.g. a specified group of states). Each emission weight represents the effectiveness of reducing a specific type of emissions from a specific subregion on reducing the maximum concentration of ozone in the receptor region.

The second phase of the analysis uses the Regional Economic Model of Air Quality (REMAQ) to determine the cost-effectiveness of reductions in NO_X and VOC emissions. The REMAQ combines the estimated emission weights from the first phase with estimates of the costs per ton of reducing emissions to provide a direct way to rank control options in terms of the cost per unit of ozone reduced in the receptor region. The control options considered in this study represent realistic controls based on known technologies, and they include combustion and post-combustion controls for different types of boilers, low emission vehicles and improved reformulated (federal Phase II) gasoline. The analytical framework of the REMAQ makes it possible to compare the total costs of a conventional command and control strategy (e.g. setting maximum allowed rates of emissions of NO_X for utility boilers), traditional trading of emissions (e.g. setting a maximum total level of emissions and determining the least-cost combination of control options to meet it), and weighted trading of emissions (e.g. setting a target reduction of ozone in the receptor region and determining the least-cost combination of control options to meet it).

The key findings of the study are as follows:

- Large point sources of NO_X in Ohio, West Virginia, and Virginia contribute significantly to ozone levels in the Northeastern urban corridor for both the existing 1-hour standard and the proposed 8-hour standard for ozone. Large scale emission reductions from these sources are cost-effective ways to reduce ozone concentrations in the northeastern urban corridor.
- The effects of ground-level emissions on ozone in the northeastern urban corridor vary significantly between NO_X and VOCs. Ground-level NO_X has the largest effect per ton of emissions on the ozone formation. The transportation sector is an important source of potential emission reductions, and low emission vehicles are more cost-effective than advanced reformulated gasoline beyond current federal standards.
- VOC emissions from rural areas in the northeastern states and upstate New York have little impact on ozone formation in the current 1-hour ozone nonattainment areas in the Metro East because of the prevailing meteorology associated with high ozone events over the northeastern urban corridor. The northeastern states would still be able to meet standards if the EPA granted a waiver for the requirement to provide VOC offsets for emission sources in all source subregions of the OTR (Ozone Transport Region), except the Inner Zone of the OTR.
- The trading of emissions among different sources and subregions can reduce the costs of meeting environmental goals to reduce ozone concentrations in the receptor region. The cost savings are substantially larger if the market for emissions is expanded to include NO_X and VOC emissions from all sectors of the economy and not just NO_X emissions from utility and industrial point sources.

Four important limitations of the study should be clarified. The first limitation is that the emission weights are estimated for a single ozone episode (July 1995). The meteorological conditions in this episode are assumed implicitly to represent a typical Bermuda high when ozone concentrations in the receptor region are high. More recent analyses using other ozone episodes and a larger number of UAM-V runs than this study show that the above findings are consistent with the varying meteorological conditions that lead to high concentrations of ozone in the northeastern urban corridor for the range of emission controls considered. The second limitation is that the statistical model for estimating the emission weights (Multivariate Ozone Response Surface [MORS]) is linear. It should be noted, however, that the fit of the linear model is excellent for the range of emission reductions used in the analysis. The third limitation is that the costs of control options are taken from existing publications. Standard engineering sources were used, such as the Electric Power Research Institute data on NO_X controls on boilers, and the California Air Resources Board data on controls of NO_X and VOC on transportation. The fourth limitation is that the controls considered for transportation focus primarily on modifying engines and fuel, rather than reducing vehicle miles traveled (VMT). The cost-effectiveness of alternative ways to reduce VMT deserves further analysis because the incremental cost of reducing ozone concentrations reaches very high levels for the control options considered in this study.

One final qualification is to note that the benefits from reducing emissions and lowering ozone concentrations are counted in the northeastern urban corridor only. Hence, showing that controls on elevated sources of NO_X in the Midwest, for example, are cost-effective is a stringent test since the additional benefits of improved air quality in the Midwest are not included in the estimated benefits.

Section 1

INTRODUCTION

The urban corridor of the northeastern United States, stretching from Washington D.C. to Boston, is the largest continuous area where ambient ozone (O_3) concentrations in excess of the 1-hour standard of 0.12 ppm[1] continue to prevail. With Baltimore, New York City, and Philadelphia receiving the designation of "severe" nonattainment for concentrations in excess of 180 ppb and Washington D.C., Greater Connecticut, Providence RI, Springfield MA, and Boston receiving designations of "serious" for concentrations in excess of 160 ppb, federal mandates have been implemented to reduce emissions of ozone precursors: volatile organic compounds (VOCs) and nitrogen oxides (NO_x) emissions. With the introduction by the U.S. Environmental Protection Agency (EPA) of an 8-hour ozone standard of 0.08 ppm on July 18, 1998, most of the eastern United States can expect to be classified as ozone nonattainment.[2]

There are five classes or categories of ozone nonattainment within the 1-hour standard: marginal (121 ppb to 138 ppb), moderate (138 ppb to 160 ppb), serious (160 ppb to 180 ppb), severe (180 ppb to 280 ppb), and extreme (280 ppb and above). The control requirements that are imposed on a nonattainment area depend on its classification according to this designation. Attainment dates vary for each category, as do the level of emission reduction requirements. For example, with a "severe" classification for New York City, VOC emissions sources greater than 25 tons per year are considered major and any modification or any new VOC sources must purchase offset allowances of 1.3 tons for every ton of VOC emitted. However, in the case of Washington D.C. which is classified as "serious," major sources of VOC are those that are greater than 50 tons per year and any modification or new industries could purchase offsets at a 1.2 to 1 ratio. In the case of moderate nonattainment areas, there is a mandated 15 percent reduction in VOC emissions by 1996, but no specific emission limit for new sources. Offsets can also be purchased at a 1.15 to 1 ratio. Marginal nonattainment only requires the use of reasonably available control technology, permit programs for new sources, and the continuation of vehicle maintenance programs. Offsets can be purchased at a 1.1 to 1 ratio with marginal nonattainment.

At the present, it is likely that nearly all of the Northeast may need to impose substantial VOC and NO_x emission reductions because of potential nonattainment with the 8-hour standard. This poses a particularly

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1. The five other severe nonattainment areas were Chicago, Houston, Milwaukee, Sacramento, and Ventura County, CA [http:\\www.epa.gov\oar\oaqps\greenbk\onc.html]. Los Angeles has an "extreme" ozone nonattainment designation with exceedences in excess of 280 ppb. The ozone level used in determining attainment is equal to the fourth highest 1-hour concentration of O_3 over a consecutive three year period. The rounding rule used by EPA actually renders this to be a 1-hour concentration threshold of 125 ppb.

2. The new standard is based on the fourth highest 8-hour running average concentration of O_3 in a year. The average over three years of this value cannot exceed 0.08 ppm. The rounding rule used by EPA actually renders this to be an 8-hour concentration threshold of 85 ppb.

difficult economic challenge for these states. Consequently, policy-makers face the possibility of having to implement tighter emissions requirements at higher levels of marginal control cost. The challenging questions that policy-makers now face are:

- What alternative approaches are available to reduce ozone concentrations?
- Among the alternatives available, which are the most cost-efficient control strategies?

This study has been commissioned by the New York State Energy Research Development Authority (NYSERDA) to evaluate the cost-effectiveness of different emission reduction measures (i.e., technologies) and regulatory strategies to reduce ozone concentrations in the Northeast. This report identifies feasible emission control measures to reduce ozone concentrations. For each measure, we estimate daily emissions reductions and the cost of implementing the control measures. We then compare the cost of implementing different emission control measures under regulatory strategies of command and control, traditional emissions trading, and weighted emissions trading. The results and methodology used in this study have been supported in a more recent study performed by Stratus Consulting for U.S. EPA.[3]

BACKGROUND

Major urban parts of the Northeast have been designated as ozone nonattainment areas since the 1-hour ozone standard of 0.12 ppm was implemented in 1973. The areas in the eastern United States that are in nonattainment of the 1-hour standard are shown in Figure 1-1. The black hatched areas indicate monitored exceedences. Area exceedences based on photochemical modeling results are displayed in yellow for the 1995 emissions inventory and red for the 2007 emissions inventory. The overlay of the yellow and red areas of exceedences results in orange areas.[4] The modeled nonattainment areas for the 1-hour standard agree well for the most part with the actual monitored exceedences. For the 8-hour standard that was promulgated recently, EPA has yet to identify and designate the areas of attainment and nonattainment.

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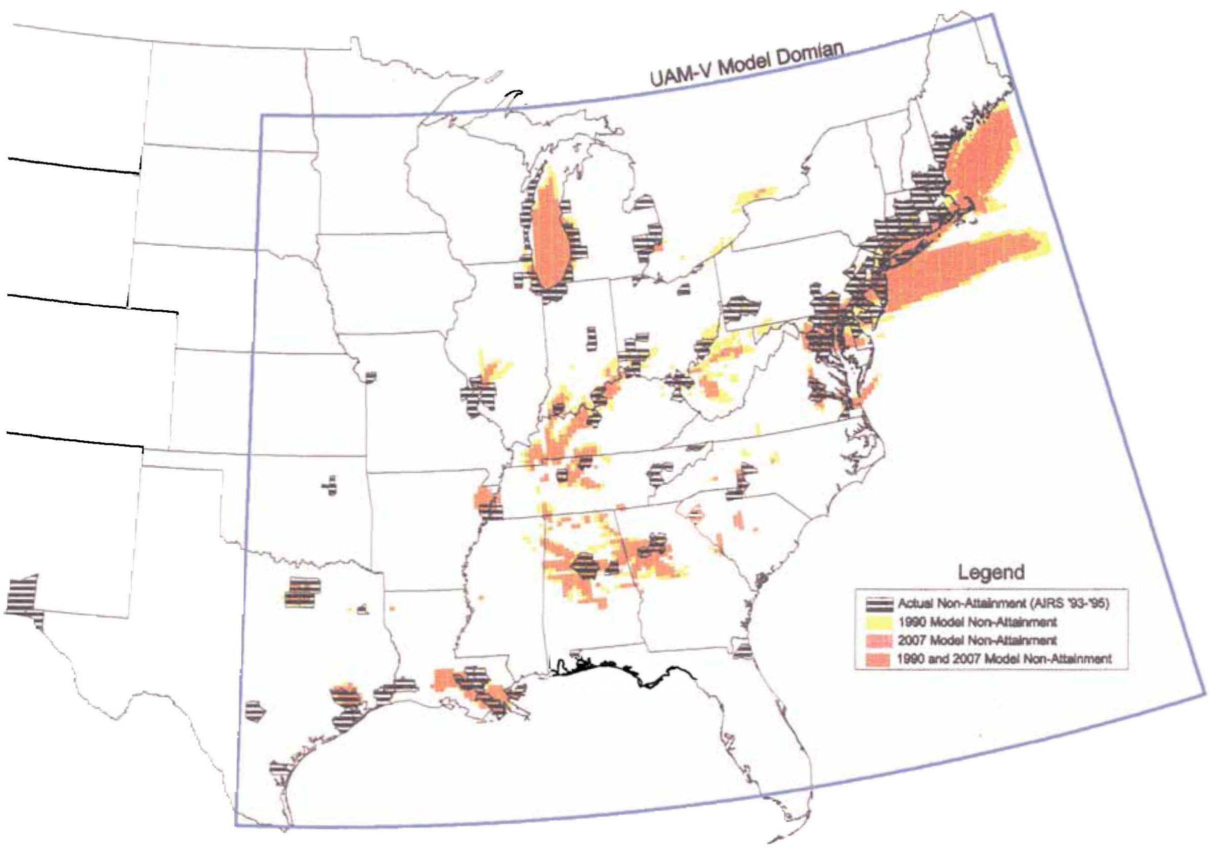
3. This study applied similar techniques and methodology used to evaluate the EPA's SIP Call using data from the 1988, 1991, 1993, and 1995 ozone episodes.

4. The July 7-18, 1995, ozone episode meteorology is used for both the 1995 and the 2007 emissions inventory.

Figure 1-1. Areas of One-Hour Ozone Standard Nonattainment (>120 ppb).



Consequently, we rely predominantly on the photochemical modeling results to predict which counties are likely to exceed the 8-hour threshold (presented in Figure 1-2). The large orange area suggests that a major portion of the New York State might be designated as nonattainment under the 8-hour ozone standard. Recognizing the severity of the ozone problem in the Northeast, Congress established the ozone transport region (OTR) as part of the 1990 Clean Air Act Amendments and required demonstration of attainment of the 1-hour ozone National Ambient Air Quality Standards (NAAQS) through various regulatory steps.

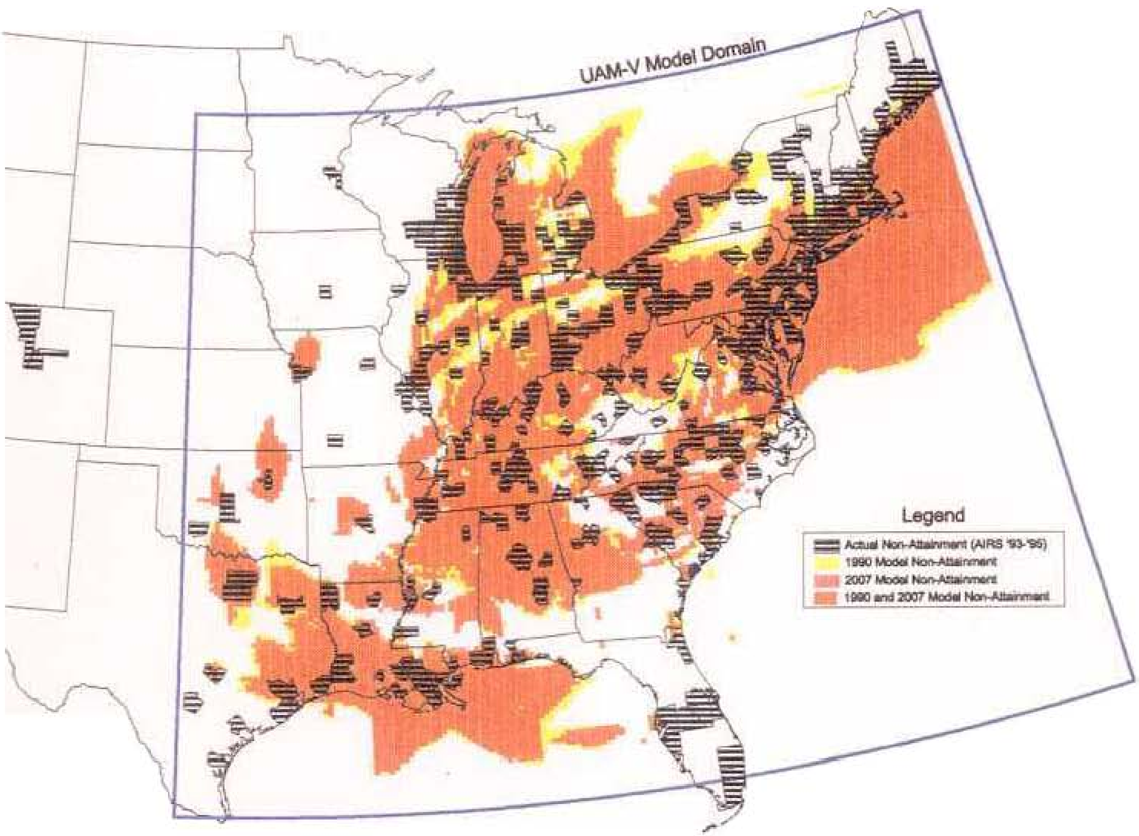
The traditional regulatory methods of emission control fall into three general approaches: (1) command and control regulation, (2) market-based approaches, and (3) reductions on a seasonal basis. Command and control regulation typically prescribes a specific emission control technology or the maximum emission rates allowed (e.g., 0.15 lb/MBtu). Command and control regulation remains the most common form of emission control because of its regulatory simplicity and its ability to meet target emission reduction levels at specific sources. Market-based approaches such as tradable credits or allowances are typically more cost-effective forms of reducing emissions, but are more complicated to implement and track. Seasonal control measures can be either command and control or market based and target reducing emissions during the summer ozone season when they matter most.[5]

Implementation of different emission reduction programs has had little success in dealing with the stubbornness of the ozone problem in the Northeast. This can be attributed in part to the complexity of the chemistry associated with the ozone formation and the downwind distances that precursor emissions and ozone itself can travel. Originally discovered in California and thought to be a local pollutant, the importance of long-range transport of ozone and its precursors, especially in the eastern United States, has emerged in the scientific literature in the past two decades but associated policy responses have been limited and relatively slow. Some of the responses included the creation of the Lake Michigan Air Director's Consortium, the Grand Canyon Visibility Commission, and the Ozone Transport Commission (OTC), the last two established as part of the 1990 Clean Air Act Amendments. The issue of long-range transport came to a head in late 1994 when many states across the nation found it difficult to develop meaningful State Implementation Plans (SIPs) showing how they would attain the 1-hour ozone NAAQS due to pollution being transported into their states from upwind areas. This realization started an intensive and ground-breaking effort led by the Environmental Commissioners of the States (ECOS) "to identify and recommend a strategy to reduce transported ozone and its precursors, which, in combination with other measures, will enable attainment and maintenance of the ozone standard" (Ozone Transport Assessment Group [OTAG], 1997).

The Ozone Transport Assessment Group (OTAG) encompassed the 37 eastern states and the District of Columbia, and conducted a number of very important technical analyses, especially the development

5. EPA defines the summer ozone season as May 1 to September 30.

Figure 1-2. Areas of Eight-Hour Ozone Standard Nonattainment (>80 ppb).



of consistent emissions inventories and widely accepted modeling protocols. The general conclusion of the OTAG was that pollutant transport can be a significant contributor to regional ozone problems. This study builds on the OTAG analysis. It uses emission inventories and modeling protocols developed by EPA and OTAG and is designed to augment the OTAG modeling effort by providing a greater detail on the ozone problem in the Northeast through subregional analysis (one of the areas of further research identified as necessary by OTAG).

OVERVIEW OF ANALYSIS

The results of this report should provide policy-makers with quantitative information of the relative cost-effectiveness of different emission control programs and different regulatory strategies for implementing these control programs to reduce ozone concentrations in the Northeast from Washington D.C. to southern Maine. To accomplish this objective, we applied the Regional Economic Model for Air Quality (REMAQ). REMAQ is an integrated modeling framework that simultaneously assesses improvements in air quality and emission control costs. Through REMAQ, we identify least-cost strategies to reduce ozone concentrations in the Northeast. A brief outline of the model structure of REMAQ is provided here. A more detailed discussion of the components of REMAQ is presented in Section 2.

REMAQ simultaneously evaluates the costs, emission reductions, and ozone reductions from different emission reduction programs (i.e., technologies) and regulatory strategies. REMAQ determines the contribution and estimated uncertainty of emissions transport to ozone accumulation by using the concept of emission weights. Emission weights are discrete values that define the marginal contribution of ozone precursor emissions from source categories (e.g., power plant boilers, various vehicle types, industrial combustion sources, and so forth) in a subregion (i.e., area of influence) on ozone in a receptor area (i.e., area of violation). The economic component of REMAQ estimates the marginal cost of emission reductions for individual power plant boilers, vehicle type, or type of combustion source (to the level of specific boiler type) at increasing levels of emission reductions. These marginal control costs, which are specific to emissions type and source category, are combined with the emission weights to identify cost-effective control strategies to reduce ozone concentrations.

POLICY IMPLICATIONS OF APPLYING REMAQ

The application of REMAQ holds significant advantages for policy-makers over the current methods used to assess the efficiency of ozone control strategies. From an implementation standpoint, probably the most important advantage of the REMAQ modeling approach is that the emission weights developed by this analysis can be directly incorporated into an emissions trading program in which they act like exchange rates between different subregions and species. The common practice for evaluating ozone control strategies is to use dollar-per-ton removal costs for emissions as the basis for assessing the economic efficiency of emission reduction programs. This implies that the contribution of a ton of precursor emissions to ozone is the same everywhere, independent of location, stack elevation, and chemical species. The results of OTAG modeling and previous studies by the authors show this to be false (OTAG, 1997, pp. 30-32; OTAG, 1998; Dorris et al., 1996). This approach of "a ton is a ton" ignores the possibility of controlling ozone with smaller, but better-focused emissions reductions in designing ozone control strategies. Thus, instead of focused emissions reductions where the reductions are both least costly and most effective, a typical result of the "a ton is a ton" approach would be the adoption of emission reduction programs with similar removal costs per ton applied across a broad range of sources in an entire subregion (or even nationally). However, to achieve air quality objectives more cost-effectively, there is a need to accurately determine the effects of different emission source locations and types on ozone levels in receptor subregion(s), and to combine this with emission control cost data.

This study builds on the previous work performed by Dorris et al. (1996). Results of the Dorris study indicate that substantial cost savings can be achieved by a shift from uniform emissions standards to more focused emission control policies. Those results are confirmed in this study. The techniques developed in this study are currently being applied by Dr. Dorris on a national scale in evaluating the current EPA's SIP Call.

OBJECTIVES

The objectives of this report have both a research and a policy analysis component. The research objectives of this report are to extend and validate the results of the previous application of REMAQ to a broader spatial scale and to apply more advanced photochemical modeling techniques building upon OTAG's work. The policy analysis objectives are as follows:

- Determine the extent of emissions transport on ozone in the urban corridor of the Northeast.
- Compare source-receptor relationships for the 1-hour versus the 8-hour ozone thresholds.
- Determine the potential to reduce overall compliance costs through alternative emission reduction strategies that achieve roughly the same level of ozone improvement as that of the proposed strategies under the current EPA Section 110 SIP Call.
- Evaluate the cost-effectiveness of emission reductions from different economic sectors and states.
- Compare costs and ozone improvements resulting from "command and control" regulation to alternative emission trading strategies.

- Use these results to develop specific policy options focused on the reduction of ground-level ozone over the Northeast.

To provide a peer review for the research and policy objectives of this study, NYSERDA assembled a policy advisory group (PAG). The PAG provided critical input to the project scientists by assisting with the direction of this study and the development of policy questions to be addressed.

ORGANIZATION OF REPORT

This report is organized to provide a basic background of the REMAQ model, discuss the specific methods and modeling assumptions used in each segment of the model, and present the results of the analyses conducted along with the implications of the results for policy-making. Thus, Section 2 provides an overview of the REMAQ model, with details about calculating the model parameters, spatial aggregation, weighting, and costs. Section 3 discusses the development of the emissions weights. Section 4 provides the results on emission weights. Section 5 discusses the emission control cost assumptions and calculations used in the model. Section 6 reports the results of applying REMAQ's economic evaluation framework to a command-and-control approach and to conventional and weighted emissions trading to identify how one meet environmental objectives at the lowest cost. Section 7 summarizes the report and provides a discussion of the major policy implications. Several appendices have also been included to provide a greater level of technical information than that included in the main report. Appendix A presents the results from the multivariate ozone response surface estimation. Appendix B provides the NO_x point source emission control cost algorithms. Appendix C discusses the modeling assumptions and sources to determine emission reduction costs. Appendix D discusses the economics of reformulated gasoline as an emission control option for the transportation sector. Appendix E discusses the photochemical modeling process used to examine source-receptor relationships.

Section 2

THE REGIONAL ECONOMIC MODEL FOR AIR QUALITY

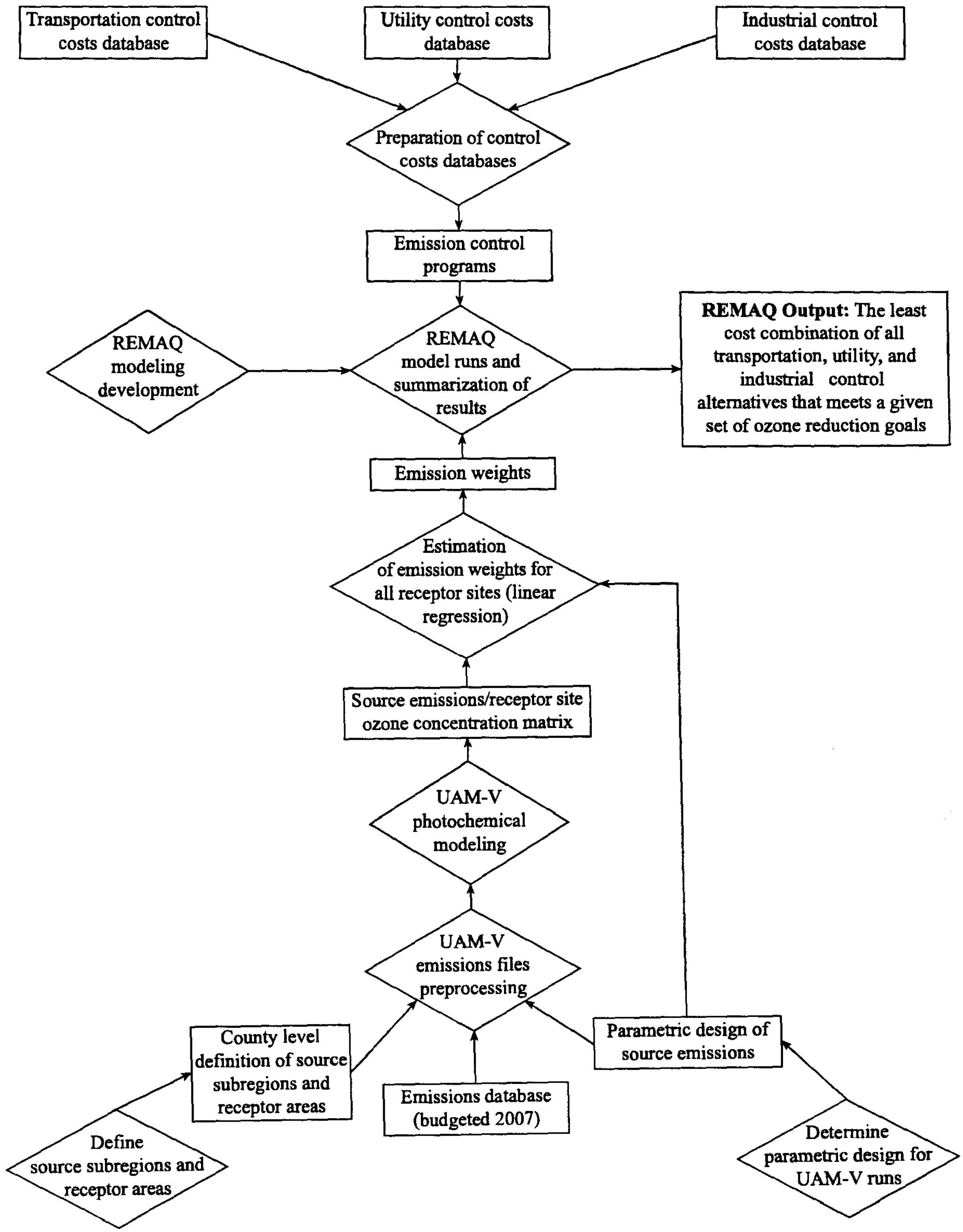
This section discusses the structure of and inputs to the Regional Economic Model for Air Quality (REMAQ). The first part of this section presents a brief overview of REMAQ. After discussing the basic structure of the model, the two main components of REMAQ are described in the next two parts of Section 2. The second part of Section 2 is on the development of emission weights. It examines the photochemical modeling and provides a short overview of the regression analysis necessary to develop emission weights from this modeling. The third part of Section 2 describes how cost curves are developed for each sector and subregion. REMAQ is then examined in more detail in the last part of this section to show how it uses the control costs, in conjunction with the emission weights, to determine the most cost-efficient methods of reducing ozone levels a receptor region.

OVERVIEW OF REMAQ

REMAQ is an economic model that incorporates source-receptor relationships of ozone formation with emission control costs in order to identify least-cost options for reducing ozone concentrations. The emission weights specify the relationships between sources of emissions and the formation of ozone in a receptor region. These weights are combined with an emission control cost database that characterizes the costs of the control programs. The database contains cost estimates and cost savings for each measure. The database includes measures for utility generation, transportation, industry, and households.

REMAQ uses the combination of the emission control programs and the emission weights to determine the least-cost method of achieving a given set of ozone air quality objectives. This is done by identifying the least-cost programs in terms of $/$O_3$-ppb (dollars per ppb of ozone) reduced in the receptor region. For a given set of control costs for the precursor emissions, a direct comparison of different control options can be made for any given source in the two dimensions relevant to policy-makers: the effectiveness of a control measure and the associated costs of reducing ozone levels. This metric also allows a comparison between different regulatory programs (e.g., a command and control regime versus different emissions trading options). REMAQ can also operate in an optimization framework to select the most cost-effective strategies for ozone reductions and calculates the costs and amount of emission reductions by economic sector and subregion and the amount of ozone removed in a receptor area. The model computes this solution by using an optimization program that minimizes the costs of ozone reductions. An overview of the REMAQ model structure is illustrated in Figure 2-1. The top of the figure shows the control cost inputs necessary for REMAQ. The lower half of the figure reviews the steps involved in the determination of the emission weights.

Figure 2-1. REMAQ Methodology Flow Diagram.



OVERVIEW OF EMISSION WEIGHTS

For REMAQ to evaluate the cost-effectiveness of different emission reduction strategies, it is first necessary to determine the spatial and chemical species effects of different emission sources on ozone levels in the receptor area. The contributions of a source subregion and species (e.g., NO_x and VOC) toward ozone formation in a receptor area are expressed through emission weights. The emission weights are estimated through the application of the Urban Airshed Model-V (UAM-V), a photochemical model of ozone production developed by Systems Applications International (SAI), using specified changes in the pattern of emissions in the base inventory of the precursor emissions from all sources and subregions.

Photochemical Modeling

The accumulation of harmful levels of ozone is a nonlinear process involving complex photochemical reactions over a large spatial scale. The effectiveness of reductions of emissions of the ozone precursors on decreasing ozone concentration varies among sources. The type of emission and its location (i.e., the physical characteristics of the problem) must be taken into account to understand the efficacy of emissions controls in reducing ozone concentrations in a receptor area.

The dynamic (i.e., physical and chemical) characteristics of ozone production and accumulation are simulated in photochemical models. Photochemical grid models like the Regional Oxidant Model (ROM) and the UAM-V are designed to simulate these physiochemical processes using a framework of three-dimensional grid cells that extend a pattern of square grids (typically 5 to 50 km on a side) over a specified modeling domain (such as a state or a subregion) and in several (often 3 to 9) vertical layers up to a few thousand meters in altitude. Photochemical grid models calculate the concentrations of ozone and other chemical species in each grid cell at a specific time, typically hourly values extending over one to two week periods (or episodes). These calculations are based on the summation of new emissions into the grid square, transport in and out of grid cells (both horizontally and vertically), photochemical reactions in each time period, and surface deposition. Each of these four processes can be nonlinear, especially the chemistry. The drivers of the model are large data sets that contain the emission inventories of the precursors of ozone and the meteorological conditions for each grid cell. For example, in a typical OTAG simulation 125,650 grid cells and 12 days would be modeled, resulting in 36,187,200 values for each model variable (of which there are over one hundred). The computational requirements of a photochemical grid model are substantial and when combined with the complexity of solving multiple nonlinear equations present a significant barrier to their widespread application in policy analysis. In particular, photochemical grid models have never been used to evaluate the economics of emission control strategies; their primary use has been to assess the ability to meet a specified ozone standard by looking at the predicted ozone concentrations based on a preselected control strategy using historical meteorological conditions (ozone episodes). Similarly, most economic evaluations of ozone control strategies stop short of evaluating the effects on ozone concentrations, addressing only the costs of reducing emissions of the precursors. Thus, this study integrates air quality modeling and economic analyses into a framework that can be used to identify cost-effective emission control strategies to mitigate the ozone problem.

Multivariate Ozone Response Surface (MORS)

REMAQ enables existing photochemical modeling to contribute more directly to the economic evaluation of emission control policies by developing a compact analytic form to represent the predictions from the photochemical model and utilizing this information in an economic model. The compact model is called a Multivariate Ozone Response Surface (MORS), which uses a statistical approach to incorporate the results from a series of photochemical model simulations into a set of emission weights for each source type and subregion. As stated below, these weights represent the average relationship between source emissions and ozone concentrations in the receptor region and are suitable for use in policy analysis.

The UAM-V[1] photochemical model is run for a series of simulations to develop a MORS using a multivariate linear regression technique to relate the predicted changes in ozone concentration in a receptor region reported by UAM-V (the dependent variable) to the key UAM-V inputs of precursor emission species (NO_x and VOC) and the location of these emissions (the explanatory variables). Each MORS is developed from a carefully designed series of UAM-V runs reflecting a range of values for the key explanatory variables for each source subregion. Thus, the estimated slope coefficients of the regression equation from the MORS can be interpreted as emission weights that relate changes in ozone precursor emissions in a given source location to changes in ozone at the receptor location. The measure of ozone is the average for the grid cells in the receptor region (weighted for area) of the predicted maximum concentration of ozone during an episode. (the meteorological conditions for a selected episode are kept the same for all runs used to estimate the MORS). An emission decrease of one ton per day of an ozone precursor is more effective for sources with high emission weights. For example, if emissions from one source have a weight twice that of emissions from another source, a ton of emissions from the first source contributes twice as much to ozone in the receptor location as does a ton of emissions from the second source. To achieve the same average decrease in ozone concentration, the first source would need to make only half the emissions reduction that the second source would need to make.

In general, the more UAM-V runs that are performed, the greater the information available on the spatial effects of precursor emissions on ozone in the receptor region. The estimation of a linear MORS using the UAM-V results to describe the behavior of changes in ozone, an inherently nonlinear process, achieves remarkably good results over the range of emission reduction considered. The usefulness of a MORS to policy-makers also depends on the design of the subregions over which emission control policies need to be developed (all sources of emissions must be assigned to a subregion). In the development of a MORS,

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1. We chose to use the UAM-V model over the ROM model because UAM-V was used by the OTAG.

there is a tradeoff between the number of available photochemical modeling runs and the number of source subregions that are considered. This tradeoff affects the accuracy of the estimated emission weights. However, the adverse effects on accuracy can be reduced, to some extent, through the design of the levels of the input variables (emissions by type and subregion) used in the UAM-V scenarios (see Section 3).

OVERVIEW OF EMISSIONS CONTROL COSTS

REMAQ uses an emissions control cost curve developed for each source type and subregion, expressed as the net present value (NPV) in 1995 dollars of total control costs divided by the NPV of summer tons of emissions reduced. Total costs include capital and variable costs of emissions reduction for the period of June 10 to September 10. This time period is two months shorter than the summer ozone season defined by EPA, but more closely represents the period of high ozone exceedences in the Metro East corridor stretching from Washington, D.C. to Boston. Since this study focuses on the cost-effectiveness of ozone control in Metro East, emissions reductions which occur when ozone is not a problem in this region (e.g., winter) are ignored. The implication of this short summer season is that seasonal control measures may become more cost-effective when compared to fixed control measures such as catalysts in cars, which operate beyond the summer season. Cost curves for both NO_x and VOC reductions are developed for the appropriate sources.

For point sources (i.e., industrial and utility boilers), cost curves for NO_x control are developed for each individual source using algorithms from the Cornell/Carnegie Mellon University (CCMU) model (Czerwinski, Minsker, and Mount, 1994), which are based on 1993 and 1994 data. Each point source can use any of several control technologies in the combustion of the fuel and two controls which chemically reduce NO_x emissions to N_2 in the exhaust stream (i.e., post-combustion). It is important to note that these are *marginal* control costs, so they take into account control technologies which are already installed and, thus, unavailable for further control. The level of installed controls varies significantly across the subregions examined in this study; thus, the marginal costs of emissions control vary as well.

Cost estimates for control programs in other sectors are also developed, including Low Emission Vehicle, Reformulated Gasoline, off-road mobile sources, and heavy-duty diesel vehicle programs. These costs come from a variety of sources depending on where the most recent and most reliable data were available. In addition, control strategies designed to change consumer behavior (and especially designed to reduce miles traveled) are also included.

SUMMARY OF THE REMAQ MODEL

REMAQ determines cost-effective strategies for ozone attainment by minimizing the total cost of emission control programs subject to a constraint that sets the required level of ozone reduction in the receptor region. Each emission control program has a specific emissions reduction (tons per day) and an annual cost per ton associated with it. The equations below illustrate the form of REMAQ used to select the optimum set of control programs:

Minimize Emission Control Costs:

> Total Cost = Sum for all Source Subregions and all Control Programs (Tons of Emissions Removed by a Control Program in a Subregion × Cost per Ton Reduced of a Control Program in a Subregion)

Subject to an Ozone Reduction Constraint

> Sum for all Source Subregions and all Control Programs (Tons of Emissions Removed by a Control Program in a Subregion × Emission Weight for a Control Program in a Subregion) $\geq \Delta$ Ozone.

where Δ Ozone is a specified reduction in the predicted maximum concentration of ozone in the receptor region (in parts per billion). The equations above are written as though separate emission weights are estimated for each control program. In practice, the structure of the MORS is simpler. Emission weights are estimated for three different types of emissions (VOC, ground level NO_x and elevated NO_x) in each source subregion. Hence, a control program uses the appropriate emission weight in the second equation for the type of emission that is reduced (e.g. elevated NO_x from controls on power plants). For some control programs, however, the cost refers to a combination of types of emissions (e.g. VOC and ground level NO_x from a low emission vehicle program), and two emission weights are used to predict the effect of one control program on reducing ozone concentrations.

In REMAQ, the emission control costs are combined with MORS emission weights to rank the cost-effectiveness of different control programs in terms of dollars per ton of ozone reduced in the receptor region. REMAQ's Economic Evaluation Framework (EEF) module selects the control programs with the lowest average cost in units of O_3 reduced per ton of emissions reduced. This is not necessarily the same as selecting the control programs with the lowest average costs per ton of emissions reduced. Different emission control programs are compared using the average cost in \$/ppt of ozone reduced as the common unit for all types in all subregions of programs. The EEF of REMAQ selects the most cost-effective control programs for reducing ozone concentrations in the receptor region and also calculates the annual costs and amount of emission reductions by economic sector and subregion for the selected programs.

Section 3
DETERMINATION OF EMISSION WEIGHTS

OVERALL APPROACH

Section 2 shows how the very complex and computer intensive photochemical modeling can be transformed using REMAQ into a compact analytic form for policy analysis (see Appendix C). In this section, the focus is on the estimation of the emission weights used in REMAQ. The overall approach is shown in Figure 3-1. The essential step for estimating the emission weights is the statistical modeling at the bottom right of the figure. The first part of this section describes how that model is formulated.

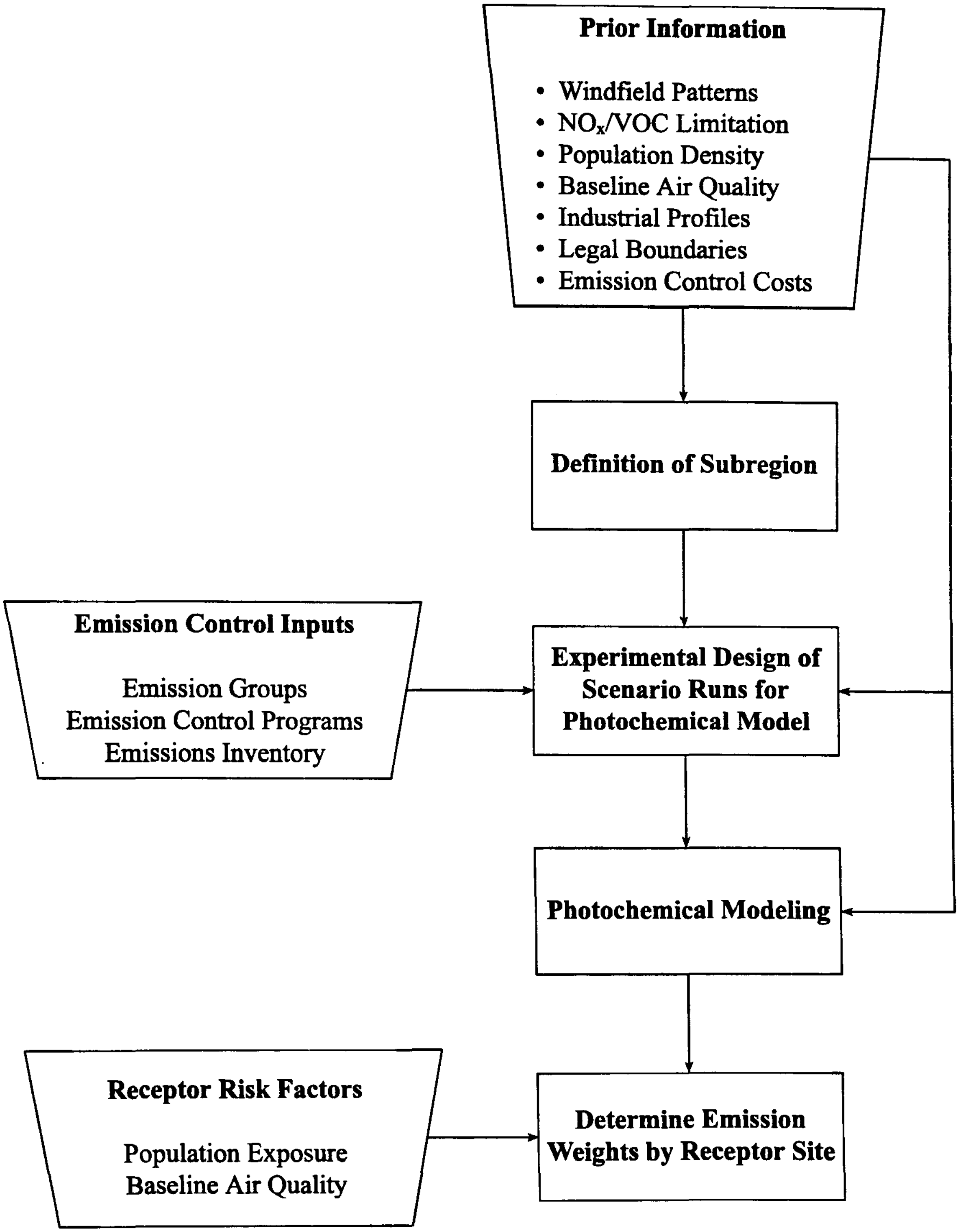
THE STATISTICAL MODEL

A Multivariate Ozone Response Surface, MORS, which is a statistical representation of the predicted concentrations of ozone from a photochemical model, is used to simplify the relationships between input emission variables and output ozone concentrations. A MORS is estimated from a series of photochemical model simulations of ozone concentrations in the receptor region over a set of different spatial patterns of emission reductions from a base inventory of emissions. Regression techniques are then used to fit a response surface (McKay et al., 1979) to predict the output ozone concentrations from the input emission levels.

The variables used in a MORS are a measure of ozone concentration (e.g., the average of the maximum 1-hour or 8-hour ground level concentrations for all grid cells in the receptor region) and emission levels grouped by type and source subregion. Ozone concentration is the dependent variable of a MORS, and the emissions by type and source subregion are the explanatory variables. The general form of Equation 3-1 represents this relationship:

$$O_3 = f[(\Sigma_j\, NOx1_{jk}, \Sigma_j\, NOx2_{jk}, \Sigma_j\, VOC_{jk}) \text{ for } k = 1, 2, \ldots, K, Z] + e \qquad (3\text{-}1)$$

Figure 3-1. Development of Emission Weights.



where:

O_3	= specified measure of ambient ozone concentration in the receptor region (ppb)
$NOx1_{jk}$	= daily tons of ground-level NO_x emitted by economic sector j in subregion k
$NOx2_{jk}$	= daily tons of elevated NO_x emitted by economic sector j in subregion k
VOC_{jk}	= daily tons of volatile organic compounds emitted by economic sector j in subregion k
j	= 1,2,...,J corresponding to economic sectors such as transportation, industrial, commercial, residential, and utility
k	= 1,2,...,K corresponding to source subregions
Z	= meteorological characteristics of the episode
e	= stochastic residual or error term.

If Equation 3-1 is specified as a linear relationship, the coefficients (i.e., the slopes of the regression surface) are emission weights. Each coefficient represents the marginal contribution of one type of emissions from a source subregion to ozone concentrations in the receptor region.

Since each emission weight used in REMAQ is estimated in a MORS, it is important to use a sufficiently large data set to get accurate estimates of these weights. However, each data point is the result of one UAM-V model run, and as a result, there is a tradeoff between accuracy and the cost of doing many runs. The accuracy of the estimated emission weight can be increased without substantially increasing the number of runs required by reducing the number of explanatory variables in the regression. Reducing the number of subregions, for example, is one way to do this, but one also wants sufficient spatial resolution in the analysis to represent the characteristics of the photochemical model effectively.

The number of source subregions for emissions defines the spatial resolution of the REMAQ model, while the number of source types of emissions defines the sectoral resolution. At one extreme, it is possible to treat all the sources within the entire modeling domain as identical. At the other extreme, it would be possible to specify a MORS with emission weights for each individual source. The first approach does not provide a suitably detailed analysis, while the second would require an unrealistically large data set and computational resources because the number of runs of the photochemical model must be larger than the number of emission weights. Thus, an intermediate approach must be taken, and the specific choices of subregions and types of emissions are discussed in the following subsections.

SOURCE SUBREGIONS OF EMISSIONS

The spatial aspects of ozone are simplified for this study by grouping together sources into different subregions, which are specified before running the photochemical model. Source subregions are generally developed along political boundaries (e.g., counties or states) even though air quality problems do not confine themselves to such boundaries. These boundaries are most relevant to policy-makers because they form partitions along which emission control policies can be developed. However, the subregions are also specified to represent the main meteorological characteristics of a typical ozone episode in the northeastern states.

The modeling domain used in this study and the individual source subregions for emissions are shown in Figure 3-2. "Metro East" is the single receptor region of interest in this study, and it includes counties in the Amtrak corridor stretching along the eastern seaboard from Boston, MA to Washington, D.C.[1] Metro East was chosen as the receptor region because it includes the cities in the northeastern states in which most violations of existing standards for ozone occur. In addition, it is consistent with the underlying meteorology of ozone formation and has also been the focus of past regulatory policies.

For the other subregions that represent emission sources only, "Northeast" includes Vermont, parts of New Hampshire, Maine, and upstate New York. "Band I," "Band II," and "Band III" form a series of rings around the Metro East subregion. The definition of these three subregions is based on the southwesterly windfield associated with the Bermuda high that characterizes the ozone episodes of interest. Band I is the closest upwind subregion to Metro East, Band II is the next closest, followed by Band III. The more distant subregions are relatively large because emissions from these subregions are expected to have negligible effects on ozone concentrations in Metro East.

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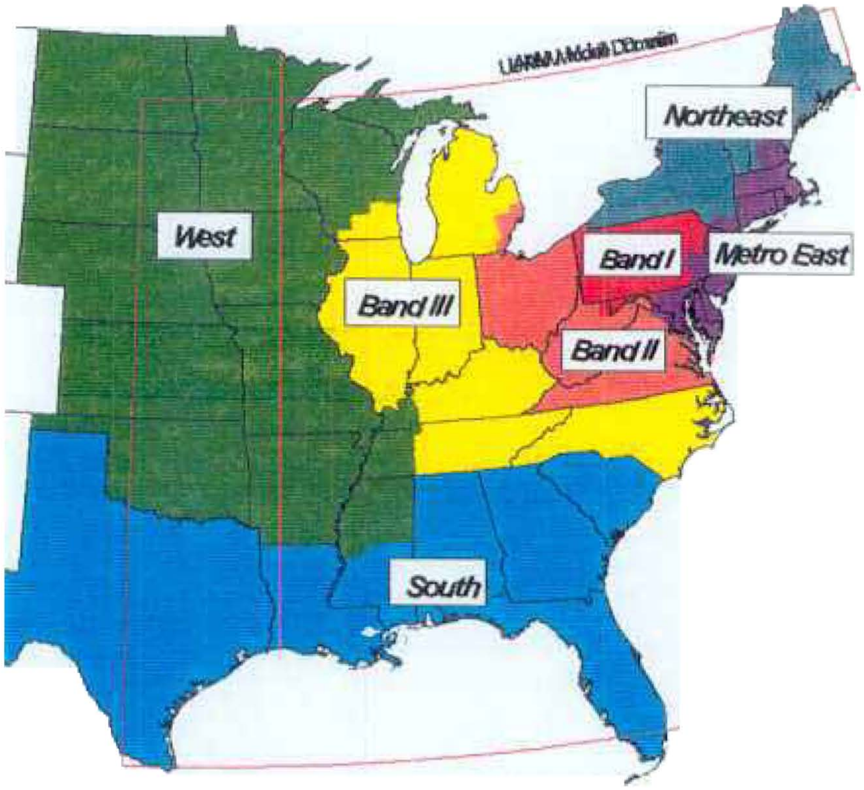
1. The Metro East subregion corresponds to the OTC's (Ozone Transport Commission) definition of "inner zone" as it is defined for the purposes of the OTC's NOx Control Memorandum of Understanding, and Northeast corresponds roughly to the "northern zone," and Band I to the "outer zone" of the OTR (Ozone Transport Region).

Figure 3-2. Source Subregions.



TYPES OF EMISSIONS

To keep the number of emission weights in a MORS manageable, emissions are grouped into three types: (1) ground-level NO_x, (2) elevated NO_x (for stack heights greater than 300 feet), and (3) ground-level VOC. Ground-level NO_x (NOx1) include all area, mobile, and small point sources. This leaves large electric utility, industrial, and institutional boilers as sources of elevated NO_x (NOx2). All VOC emissions are classified as ground-level emissions.[2] The expectation is that the effects of ground level emissions on ozone concentrations in Metro East will be important for subregions close to the receptor region (i.e. Metro East and Band I), but not for distant subregions. In contrast, emissions of elevated NO_x may travel large

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2. Biogenic emissions and trace VOC emissions from large boilers are an integral part of the emissions inventory. However, these sources cannot be readily controlled, and consequently, are held at the base levels in all runs of the photochemical model.

distances, and as a result, emissions from Band II and Band III, as well as from Metro East and Band I, may be important.

Given the classification of emissions into three types, it is straightforward to identify which economic sectors are the main contributor to each type. This in turn identifies which economic sectors are likely to be affected by additional controls on emissions. Table 3-1 provides a summary of the economic sectors with the greatest potential for reducing emissions for each type of emissions. A more detailed discussion of the types of control programs considered is given in the following subsection.

Table 3-1. Emission Types and the Associated Sources.

Emission Type	Major Sources for Emission Control
Ground-Level NO_x (NOx1)	Transportation, construction equipment, and small boilers
Ground-Level VOC	Transportation, small recreational and commercial engines, fugitive emissions
Elevated NO_x (NOx2)	Electric power plants and large industrial boilers

Since reducing the number of variables increases the accuracy of the estimated MORS, it is possible to use prior knowledge to determine which emission sources are likely to be of little or no influence on ozone concentrations in Metro East. It is then possible to design the photochemical runs to provide more information about the important sources of emissions. Based on the results of past research, ground level and elevated emissions from Metro East, Band I and Northeast are considered to be potentially important. However, emissions of elevated NO_x are expected to be important from more distant regions as well. A summary of the sources of emissions that were evaluated in the analysis is given in Table 3.2.

Table 3-2. Emissions Types and Source Subregions Used in the Analysis.

Emission Type	Source Subregion
VOC	Metro East, Northeast, Band I
NOx1 (ground-level; e.g., transportation)	Metro East, Northeast, Band I
NOx2 (elevated; i.e., utility stack heights greater than 300 feet)	Metro East, Northeast, Bands I – III, West, South

EMISSION CONTROLS IN THE BASE INVENTORY

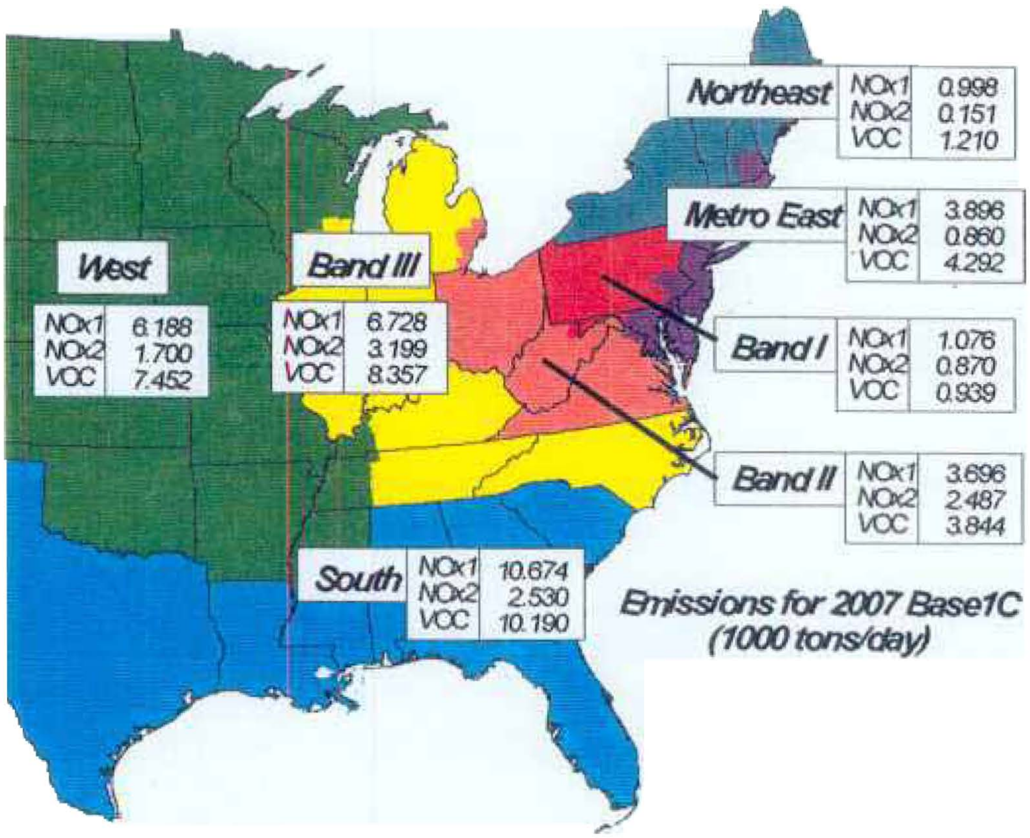
Many federal emission control programs have already been specified by the 1990 Clean Air Act Amendments (CAAA). Consequently, different emissions scenarios in this analysis are specified to represent incremental emission reductions beyond the federal controls that will be in place by the year 2007. The EPA 2007 Base 1C was used as the base emission inventory. It provides projections of emissions in 2007 that incorporate federal controls under Titles I-IV of the 1990 CAAA and the 1988 Amended Resource Conservation Recovery Act (RCRA). Emission sources controlled under these federal initiatives are listed in Table 3-3. The baseline inventory for 2007 includes Reasonably Available Control Technology (RACT) for NO_x on boilers in the Ozone Transport Region (OTR; Metro East + Band I + Northeast in Figure 3-2), Title IV (acid rain related) NO_x controls on boilers outside the OTR, federal (NO_x and VOC) controls for vehicles, and Title III (air toxics related) controls on hazardous emissions, which include a variety of different VOC.

Table 3-3. Emission Controls under the 1990 CAAA and 1988 RCRA.

CAAA-Title I	Point source NO_x emission controls of Reasonable Available Control Technology (RACT) in the OTR
CAAA-Title II	Tier I and II automotive controls (NO_x and VOC emissions) Reformulated gasoline
CAAA-Title III	Solvents, degreasing agents, petroleum and gasoline fugitive and process emissions, hazardous emissions from industry and consumer products, personal products, household products, automotive products, commercial adhesives, architectural coatings, and traffic markings
CAAA-Title IV	NO_x point source emissions beyond the OTR, superseded by Title I regulations for nonattainment areas classified as "moderate" or higher
RCRA	Waste treatment, storage, and disposal facilities (TSDFs)

The most significant reductions in the emissions of ozone precursors from current levels are associated with the Tier I and II vehicle emission controls under Title II, and the NO_x point source controls under Titles I and IV. The base levels of the three types of emissions in the different source subregions are shown in Figure 3-4. Ground level emissions from Metro East are the largest sources for the subregions considered important in Table 3-1. In contrast, elevated NO_x from Band I, outside the OTR, is the largest source of this type of emissions for the subregions in Table 3-1.

Figure 3-4. Emissions for 2007 Base 1C — Metro East.



A wide range of emission controls beyond those specified by existing federal control programs were considered by policymakers in OTAG.[3] These controls form the basis for changing the patterns of emissions in the runs used to estimate the emission weights for the MORS. Emission reductions for NO_x point sources considered in this analysis cover many control options, from inexpensive combustion modifications to selective catalytic reduction (SCR). For the transportation sector, the emission control options correspond to a Low Emission Vehicle (LEV) program, advanced reformulated gasoline, and controls on diesel buses and trucks that have already been adopted in California. Additional engineering controls are included for off-road engines such as small two-stroke and four-stroke engines, and diesel engines used in construction. The final emission controls considered are behavioral programs for the transportation sector, such as making gasoline more expensive. For estimating the emission weights, the relevant information needed for each control option is the level of reduction of emissions by the type of emission and the source subregion.

The selection of a range of economically feasible levels of emission controls used in this analysis is shown in Table 3-4. The 2007 Base refers to the EPA baseline inventory of emissions discussed in the previous section. The Maximum Control is the emission level when all the emission control programs in the analysis are applied. The last row is the percent reduction in emissions from the 2007 Base using maximum controls. Table 3-4 shows that the ability to reduce emissions of NO_x from elevated sources is much greater (i.e. give larger percentage reductions from the 2007 Base) than it is for ground level sources using currently feasible engineering and behavioral controls. However, elevated emissions are only a small part of the total inventory, and as a result, the range of reductions considered in the analysis from all sources is roughly 0-30%. While this is a relatively small amount, it does have an indirect positive effect on the estimated MORS. The linear form of the relationship that is assumed for this analysis fits the data very well, and the anticipated problems of dealing with non-linearities were found to be unimportant.

Table 3-4. Emission Ranges by Source and Subregion (1000 tons/day).

Level of Control	Metro East			Northeast			Band I			Band II	Band III
	NOx1	NOx2	VOC	NOx1	NOx2	VOC	NOx1	NOx2	VOC	NOx2	NOx2
2007 Base	3.896	0.860	4.292	0.998	0.151	1.210	1.076	0.870	0.939	2.487	1.837
Maximum Control	2.762	0.327	3.702	0.704	0.056	1.022	0.756	0.265	0.708	0.722	0.497
Reduction from Base	29%	62%	14%	29%	63%	16%	30%	70%	25%	71%	73%

DEVELOPMENT OF EMISSION WEIGHTS

The next step in the analysis is to develop an experimental design for the runs of the photochemical model. In general, this process draws on the physical profiles used to develop the source subregions and the ranges of emission reductions considered in Table 3-4. The physical profiles direct the focus of the experimental design to the extent and types of emission controls considered. By combining the physical profiles of emissions with the range of emission control programs under consideration by policy-makers, it is possible to define different levels of emission reductions in each subregion. Traditional statistical concerns of collinearity among the regressors to maintain accuracy of the estimated emission weights must also be incorporated into the development of the experimental design. The efficiency of the design of the runs affects the overall fit of the MORS and the size of the statistical confidence interval associated with each

3. Many of these controls have already been adopted in California.

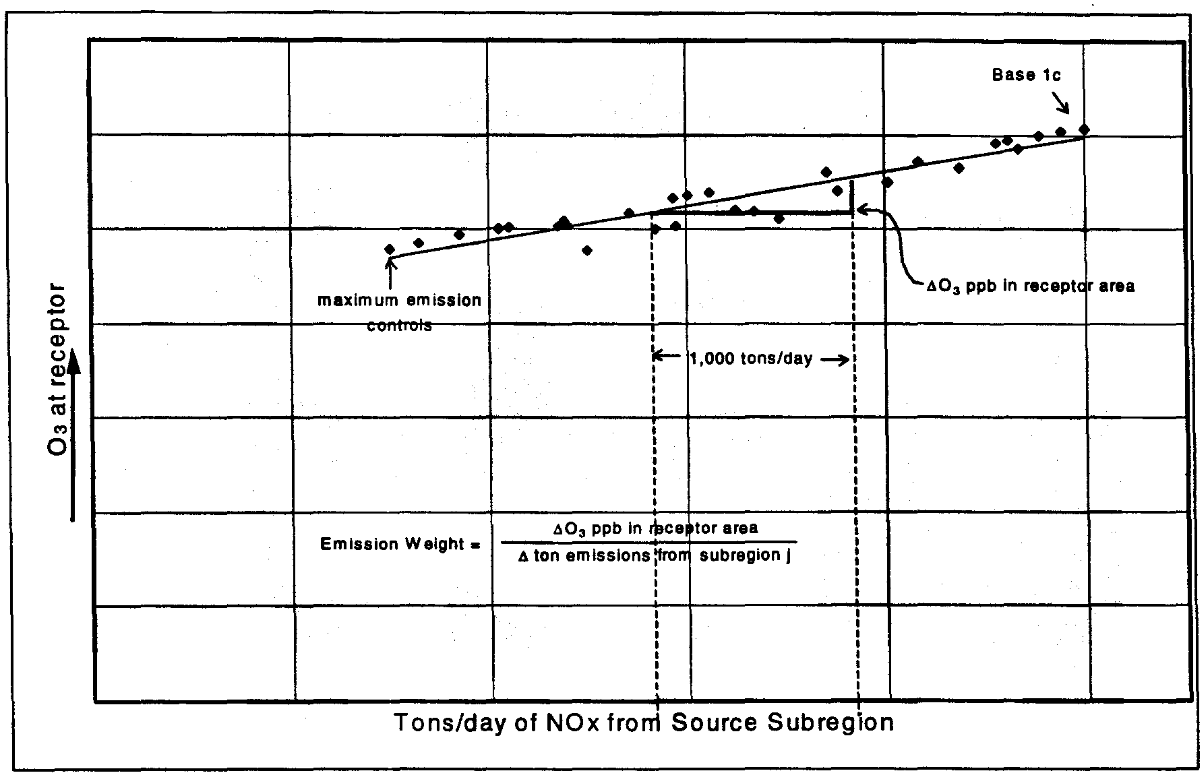
emission weight. The complete experimental design produces a series of 38 different patterns of emissions that are used for the photochemical modeling.

In this analysis, the ozone metric is derived from using the meteorology of the July 1995 ozone and modifications to the 2007 Base inventory of emissions. The July 1995 episode is chosen because it is the most characteristic of high ozone days in Metro East among the episodes considered by OTAG. The photochemical model predicts the 1-hour concentrations for every grid cell in the OTAG states. For Metro East, the receptor region, an average of the 1-hour maximum ozone concentration and an average of the maximum 8-hour ozone concentration for each grid cell in the receptor area (area weighted to account for fine and coarse grid squares) are computed for every scenario. This gives two dependent variables (1-hour and 8-hour ozone) for Metro East.

The photochemical model runs determine the two measures of ozone concentrations corresponding to the specific patterns of emission reductions for each run identified in the experimental design. The final step combines one of the measures of ozone concentration for each photochemical model run with input emission levels by type and source subregion. A regression model is fitted to estimate emission weights that reflect the changes in ozone concentrations in the receptor region due to changes in each type of emissions in each source subregion.

All combinations of emission types and source subregions have individual emission weights that are estimated for a MORS. Each emission weight estimates the reduction in parts per billion of ozone in the

Figure 3-5. Sample MORS Regression (to illustrate the definition of an emission weight).



receptor region for a 1,000 tons per day reduction of the corresponding source of emissions. This concept is illustrated in Figure 3-5 under the assumption that only a single source of NO_x emissions varies (i.e. all other sources are held at the levels in the OTAG 2007 Base 1C). The black diamonds in Figure 3-5 are the values of the ozone metric (vertical axis) and the corresponding emission levels of NO_x (horizontal axis) in the different scenarios. A regression line is fitted through the scatter plot using regression techniques, and the slope of this function is the estimated emission weight. The emission weight measures the predicted reduction in ozone in ppb for a 1,000 tons per day reduction of NO_x in the source subregion. For example, an emission weight of 1.5 corresponds to a 1.5 ppb reduction in ozone associated with a 1,000 tons per day reduction in emissions. The actual estimated emission weights for the 1-hour and 8-hour MORS are presented in the next section.

Section 4
ESTIMATED EMISSIONS WEIGHTS FOR METRO EAST

The emission weights in a Multivariate Ozone Response Surface (MORS) are estimated using 38 different patterns of emissions to predict the corresponding levels of ozone in the receptor region, Metro East (Box et al., 1978). For each of the 38 scenarios, the ozone metric is derived from either the maximum 1-hour concentration of ozone or the maximum 8-hour average concentration of ozone. In both cases, the photochemical model predicts these values for every grid cell in the receptor region, an average of the maximum values for all grid cells in the receptor region (area weighted) is computed for each scenario. This gives two alternative dependent variables (1-hour and 8-hour ozone) for the receptor region.

The 38 scenarios represent 38 possible combinations of reductions of emissions by type and subregion. Each type of emissions in each subregion represents a separate explanatory variable in a regression model, measured by the reduction of emissions from the level in the EPA 2007 Base inventory of emissions. The explanatory variables correspond to the type/subregion combinations shown in Table 3-2, and all other combinations remain unchanged at the levels in the Base and form part of the intercept in the regression model. The estimated intercept is the predicted ozone concentration in Metro East for the base levels of emissions in 2007.

The emission weights estimated using 38 scenarios are consistent with the results of our earlier studies (Dorris et al., 1996). The long-range transport of elevated NO_X emissions, in addition to ground level and elevated emissions from local sources, plays an important role in ozone formation over Metro East. The MORS produced an excellent fit for the 38 scenarios, with over 99% of the variability of both 1-hour and 8-hour ozone levels explained by the regression model. The implication is that the relatively simple linear and additive form of a MORS provides a very good approximation to the predictions from the photochemical model UAM-V for the range of reductions of emissions specified in the 38 scenarios. As a result, there is little room for improving the fit of the MORS by, for example, considering nonlinearities in the relationship between ozone and the different sources of emissions.

The 38 scenarios are designed to give more information about sources of emissions that are within and immediately adjacent to Metro East. For the adjacent subregions, all three types of emissions (elevated NO_x, ground-level NO_x, and VOC) were expected to contribute to ozone in the receptor regions. For subregions that are farther upwind, the expectation was that only elevated NO_x would be important. These expectations were confirmed, and the estimated emission weights for Metro East (the receptor region) are summarized in Table 4-1. Each emission weight estimates the reduction of ozone (area-weighted concentration in ppb) in response to a reduction in each of the three types of emissions (1,000 tons per summer day). For each emission weight, the uncertainty around the point estimates is represented by simultaneous 90% confidence intervals using Bonferroni's procedure.[1] Since the units of every emission weight in a MORS are the same (change of ozone in ppb for a 1,000 ton reduction of emissions), comparing the magnitudes of the emission weights among different emission sources indicates their relative effectiveness in reducing ozone in the receptor region on an absolute scale of a 1,000-ton reduction.

Table 4-1. Estimated Emission Weights* for Metro East Based on Absolute Emissions Reductions.

Subregion of Emission	Emission Type	O_3 ppb in Metro East per 1,000 Tons of Reduction*	
		1-Hour**	8-Hour**
Metro East	NOx1	4.65 ± 0.28	3.95 ± 0.25
	NOx2	2.42 ± 0.20	2.00 ± 0.17
	VOC	0.77 ± 0.45	0.56 ± 0.39
Northeast	NOx1	0.94 ± 1.34†	0.90 ± 1.16†
	NOx2	0.25 ± 1.13†	0.32 ± 1.01†
	VOC	0.02 ± 1.50†	0.14 ± 1.30†
Band I	NOx1	2.22 ± 0.80	2.64 ± 0.70
	NOx2	1.89 ± 0.23	2.03 ± 0.21
	VOC	—	—
Band II	NOx1	—	—
	NOx2	1.07 ± 0.07	1.24 ± 0.07
	VOC	—	—
Band III	NOx1	—	—
	NOx2	0.17 ± 0.08	0.20 ± 0.07
	VOC	—	—

* Reduction in ozone (ppb) for a reduction of emissions (1,000 tons/day).

** 90% simultaneous confidence

† Statistically indeterminate from zero

— No effect is assumed in the model.

1. The Bonferroni procedure increases the size of the confidence interval over the traditional individual confidence interval. Since there are 10 emission weights in a MORS, a 90% simultaneous confidence interval is the same as a 99% individual confidence interval.

The emission weights in Table 4-1 for reductions in the peak 1-hour ozone concentrations emphasize local emission reductions over regional emission reductions. However, emissions from more distant point sources (Bands I, II, III) become more important when examining reductions in the peak 8-hour average ozone concentrations.

The five biggest emission weights corresponding to 1-hour and 8-hour ozone concentrations are shown in order of their magnitude below:

Rank	1-hour	8-hour
1	NOx1 Metro East	NOx1 Metro East
2	NOx2 Metro East	NOx1 Band I
3	NOx1 Band I	NOx2 Band I
4	NOx2 Band I	NOx2 Metro East
5	NOx2 Band II	NOx2 Band II

This ranking indicates that NO_x emissions are the most important precursor to ozone formation. Since most of the Northeast is NO_x-limited, additional molecules of NO_x are needed to form ozone. VOC emissions are less critical to ozone formation as a result. The rankings also indicate that the relative effectiveness of distant NO_x sources increases from the 1-hour to the 8-hour ozone metric due to a reservoir of pollutant build-up above the inversion layer during nocturnal hours. As the sun rises, this reservoir of emissions becomes entrained into the lower mixing layer, resulting in a large source of precursor emissions at ground level (Zhang et al, 1998). In contrast, local sources of elevated NO_x tend to remain several mixing layers above the surface and are primarily entrained to mix with ground-level emissions for a couple of hours around the peak 1-hour ozone concentrations (Zhang et al, 1999).

While both analysts and decision-makers sometimes talk in terms of total emissions and tons of reductions, it is more common to use percentage reductions of emissions for comparisons. Adopting this latter approach, Table 4-2 presents the rescaled emissions weights calculated on a percentage basis.[2] Since the base emissions are different for each subregion, the relative magnitudes of the results in Tables 4-1 and 4-2 differ from each other. Despite Band II having an emission weight for elevated NO_X less than half the value for Metro East, a 1% reduction in emissions from Band II leads to a greater reduction of ozone in Metro East because the base level of NO_X in Band II is over three times as high as it is in Metro East.

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2. The values in Table 4-2 are found by multiplying the data in Table 4-1 by (0.01*total base emissions in subregion) / 1000 to obtain a rescaled emission weight in parts per trillion / 1% reduction.

Table 4-2. Estimated Ozone Reductions* for Metro East Based on a 1% Emissions Reduction.			
Subregion of Reduction	Emission Type	O_3 Metro East per 1% Reduction	
		1-Hour**	8-Hour**
Metro East	NOx1	181.21 ± 11.10	154.00 ± 9.69
	NOx2	20.84 ± 1.68	17.19 ± 1.50
	VOC	33.23 ± 19.16	24.09 ± 16.71
Northeast	NOx1	9.40 ± 13.38†	8.98 ± 11.61†
	NOx2	0.38 ± 1.70†	0.48 ± 1.52†
	VOC	0.21 ± 18.15†	1.75 ± 15.75†
Band I	NOx1	23.90 ± 8.63	28.39 ± 7.52
	NOx2	16.44 ± 2.01	17.63 ± 1.79
	VOC	—	—
Band II	NOx1	—	—
	NOx2	26.71 ± 1.85	30.90 ± 1.65
	VOC	—	—
Band III	NOx1	—	—
	NOx2	3.03 ± 1.43	3.68 ± 1.27
	VOC	—	—

* Reduction in ozone (ppt) corresponding to a 1% reduction of emissions from the level in the 2007 base.

** 90% simultaneous confidence intervals

† Statistically indeterminate

All emission weights for the Northeast subregion are statistically insignificantly different from zero because the confidence intervals include zero. This results from the northwesterly direction of the windfield, which makes the contribution of these emissions to ozone formation in Metro East relatively small. In addition, less information about changes in emission from Northeast was incorporated into the design of the scenarios. For Band I, both elevated and ground-level NO_x emissions are shown to be significant contributions to ozone accumulation in Metro East. Emissions from elevated NO_x point sources in the Band II (Ohio, West Virginia, and Virginia) are also important contributors to ozone formation in Metro East. A natural extension of this finding suggests that emissions from these distant sources would significantly affect ozone concentrations over Band II, Band I, and Metro East. The combined impact of these emissions over multiple receptor sites makes emission reductions from Band II a potentially effective way to reduce ozone concentrations throughout the northeastern states.

Emissions from elevated NO_x sources in the subregions to the south and west of Band III were found to have an insignificant influence on ozone in Metro East, and these input variables were removed from the final regression analysis. With a limited number of degrees of freedom, this increases the robustness of the estimates for remaining emission weights. As mentioned earlier, reductions of ground-level emissions from subregions that are distant from Metro East were not considered in the scenarios.

By focusing the design of the scenarios on sources that contribute the most to ozone in Metro East, the corresponding emission weights are statistically robust estimates with relatively narrow confidence intervals (i.e., small variance). The narrow confidence intervals are also a result of the good fit of the estimated MORS. The high R^2 further suggest that the MORS serves as an accurate statistical representation of the photochemical model over the relatively small range of emission reductions (about 30% of the base levels in 2007) considered in the scenarios. The emission weights for NO_x2 are generally estimated more accurately than the emission weights for ground-level sources. An explanation for this is that the range of reductions considered in the scenarios were bigger for elevated NO_x2 than for the ground-level sources. This, in turn, reflects the fact that the number and effectiveness of existing control for ground-level sources options are limited compared to controls on utility and industrial boilers.

SUMMARY

In summary, the results show that while local emissions from Metro East, Band I and Northeast contribute significantly to ozone formulation in Metro East, the relatively distant emissions of elevated NO_x from Bands II and III are also important contributors to ozone formation in Metro East. Examining the influence of emissions from these more distant subregions on ozone formation in Metro East is a stringent test because only the effects on ozone in the receptor region is considered in the ozone metric. No account has been made of the additional benefits of reducing ozone within states like Pennsylvania, Ohio, and West Virginia.

The estimated emission weights provide a useful basis for designing a regional emission control strategy, but without consideration of emission control costs, the economic efficiency of different emission control strategies cannot be evaluated. An important next step, presented in Section 5, is to link the emission weights with emission control costs to identify cost-effective strategies in terms of dollars per unit of ozone reduced in the receptor region instead of the usual basis for comparison in terms of dollars per ton of emissions reduced.

Section 5

EMISSION CONTROL COSTS

THE DEFINITION OF CONTROL COST

Since Title I of the 1990 CAAA grants states the flexibility to design emission control initiatives to address ozone compliance, it is important to have a consistent framework to measure the costs over a wide variety of control programs affecting different pollutants. Emission abatement costs in different sectors are determined in terms of decreased emissions during the summer, to reflect the seasonal nature of the ozone problem.

Costs for each control program are expressed as the net present value (NPV) in 1995 dollars of total control costs divided by the corresponding NPV of summer tons of emissions reductions:

$$\text{Average Control Cost} = \frac{\text{NPV (Capital and Variable Cost)}}{\text{NPV (SummerTons Re moved)}} \tag{5-1}$$

Total costs include capital and variable costs, discounted to reflect the timing of expenditures. Summer emissions are defined for the period of June 10 to September 10. This time period is two months shorter than the summer ozone season defined by EPA, but more closely represents the period of observed high ozone concentrations in Metro East. By defining a shorter summer season, seasonal control measures become more cost-effective when compared to fixed control measures such as catalysts in cars, which operate beyond the summer season. The summer tons removed are discounted at a smaller rate than the control costs to reflect the differences in risks and accrued health and environmental benefits over time (Nordhaus, 1990).[1]

CONTROL PROGRAMS

The discussion of emission control programs is divided into two categories: the regulatory base strategy, and additional control programs. The regulatory base strategy focuses on major emission control initiatives that have been endorsed by the OTC and adopted by most of the states in the OTR. The regulatory base strategy serves as a benchmark for the command and control regulation which can be compared with alternative market-based strategies such as traditional emissions trading, interspecies emissions trading, and weighted-emissions trading (Dorris, 1996). The additional control programs include a wide variety of

1. A rate of 9% is used for expenditures and 4% for emissions to discount back to 1994. The 4% for emissions is a social discount rate that accounts for society's time preference to have emission reductions sooner rather than later.

emission controls across sources not addressed in the regulatory base strategy and more stringent emission reductions on sources considered in the regulatory base strategy.

Regulatory Base Strategy

The three components of the regulatory base strategy include (1) NO_x point source controls (at an emission rate limit of 0.15 lb/MMBtu, (2) low emission vehicles (LEV), and (3) advanced reformulated gasoline (RGas). The regulatory base strategy for the analysis applies these three regulations to all states in the OTR (i.e., Metro East, Band I, and Northeast) and the .15 lb/MMBtu standard for NO_x point source controls to the remainder of the OTAG states. In other words, the NO_x point source controls are evaluated over the entire OTAG domain, but the ground-level emission controls affect only sources in the OTR. This is consistent with the sources of emission which have large emission weights (see Table 4.1). It is assumed in all cases that the additional controls add incrementally to the federal controls built into the emissions inventory for 2007.

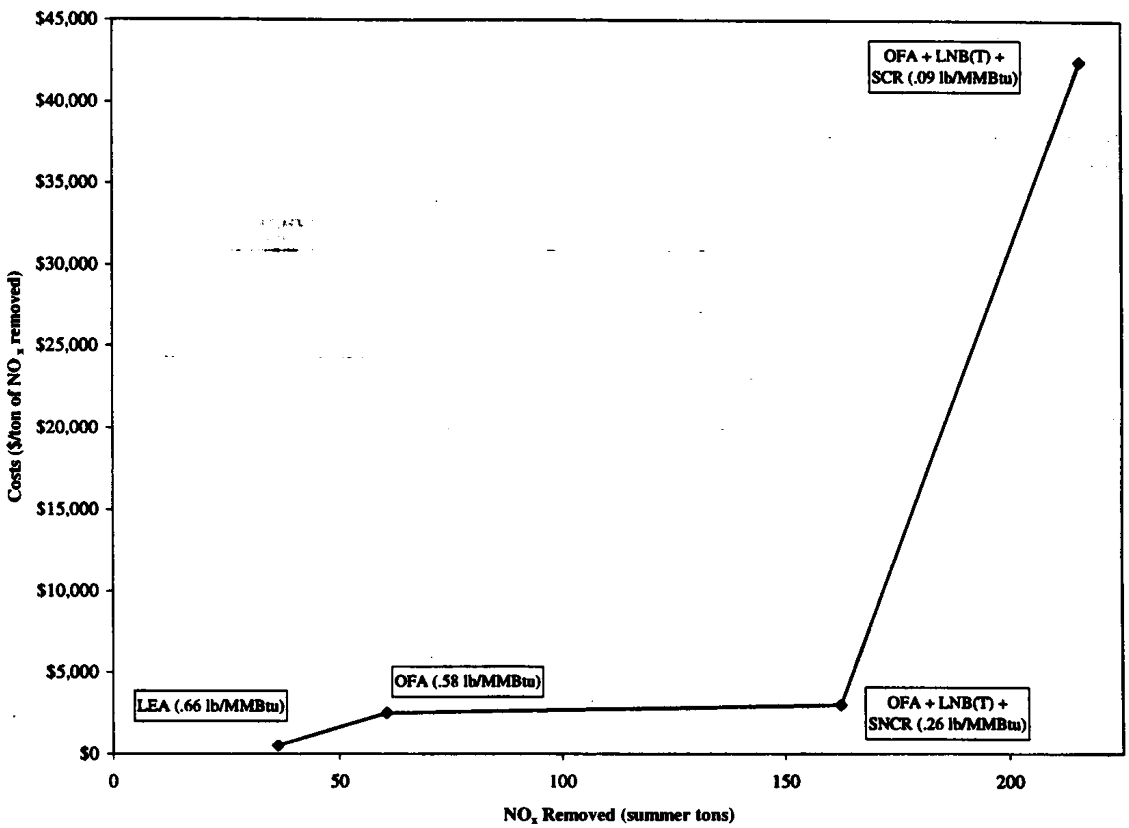
NO_x Point Sources. As major stationary sources of NO_x, electric utility boilers are generally considered central to the control of ozone in the northeastern United States (Czerwinski et al., 1994). This analysis presents summary results on the cost of utility boiler NO_x, control for the "Beyond RACT" period beginning in 2000. The cost algorithms are adapted from the EPA (U.S. EPA 1992a, 1992b, 1993) and the cost of emissions control equipment primarily comes from EPRI proceedings (EPRI, 1993a, 1993b, 1993c). A more detailed discussion of the algorithms used to calculate emission control costs for NO_x point sources is included in Appendix A.

The boiler database has been developed from data collected by the EPA and U. S. Department of Energy (DOE) from Energy Information Administration (EIA) Form 767 for 1993. This database includes boiler identification, location, nameplate capacity, design heat input, SO_2 and NO_x emission rates, firing configuration, and fuel consumption data. Missing data are inputed using published DOE data (EIA, 1996). Boiler capacity factors and heat rates are determined from documented fuel consumption, capacity, and heat input data discussed in Appendix B.

Analysis of NO_x point sources entails the calculation of control costs for emissions reductions beyond those which have already been made in the 2007 base inventory of emissions: the 1995 RACT standards for sources in the OTR and Title IV controls for sources beyond the OTR. By using RACT as a baseline for emissions, it is assumed that the controls installed by utilities will meet the given standard at a minimum total cost. This approach provides a mechanism for predicting the type of controls that will be installed by 2007 at every point source in the base inventory of emissions.

The calculation of summer NO_x reductions assumes that summer NO_x emissions from June 10 to September 10 are 24.9% (91/365) of the annual value. Thus, the daily level of emissions is assumed to be constant throughout the year, with no increase in emissions during the summer peak period.

Figure 5-1. Control Costs for Reductions of NO_x Emissions for Crawford, IL, Coal, Tangentially Fired Boiler (181 MW).



The NO_x control costs are calculated using the cost algorithms developed for the Cornell Carnegie Mellon University Model (CCMU) model (Czerwinski et al., 1994). These algorithms represent the cost of NO_x controls based on 1993 and 1994 data (Czerwinski et al., 1994; EPRI, 1993a; U.S. EPA, 1992b, 1993). NO_x control technologies considered for utility boilers include:

- low excess air (LEA)
- overfire air (OFA)
- low-NO_x burners (LNB)
- low-NO_x burners and overfire air tangential boilers (LNB + OFA-TB)
- selective noncatalytic reduction (SNCR)
- selective catalytic reduction (SCR).

In addition, any combination of applicable combustion modification technologies can be combined with a post-combustion technology to decrease total NO_x emissions. It is assumed that SNCR is used only during the summer season. The structure of the average cost curve of controls for a tangentially fired coal boiler and a wall fired coal boiler is illustrated in Figures 5-1 and 5-2, respectively. Note that the average costs in the smaller boiler (Figure 5.2) are generally much higher than they are for the large boiler (Figure 5.1). Both examples show that the most economically efficient control programs for utility boilers include LEA, OFA, LNB, and SNCR technologies. While the use of SCR gives the largest decreases in NO_x emissions, the average cost per ton of SCR is much higher than with the other technologies (Bechtel, 1996; CTOW, 1996).

Increasing the command and control NO_x emission standard from 0.20 lb/MMBtu to 0.15 lb/MMBtu represents a 25% decrease in NO_x emissions, and increases costs by almost the same amount (24%) from $4,119.9 to $5,119 million (see Tables 5-1 and 5-2). However, the additional costs are not evenly distributed across subregions. For example, the average marginal costs ($/ton) of meeting the more stringent standard in the Metro East, Band I, and Northeast subregions increase by 27%, 37%, and 45%, respectively. Additional reductions are generally less expensive in areas with a higher concentration of dirtier, coal-fired plants. For example, the increase in Band II is less than 20%.

Figure 5-2. Marginal Control Costs for Reductions of NO_x Emissions for Clay Boswell, MN, Coal Wall Fired Boiler (475.9 MW).

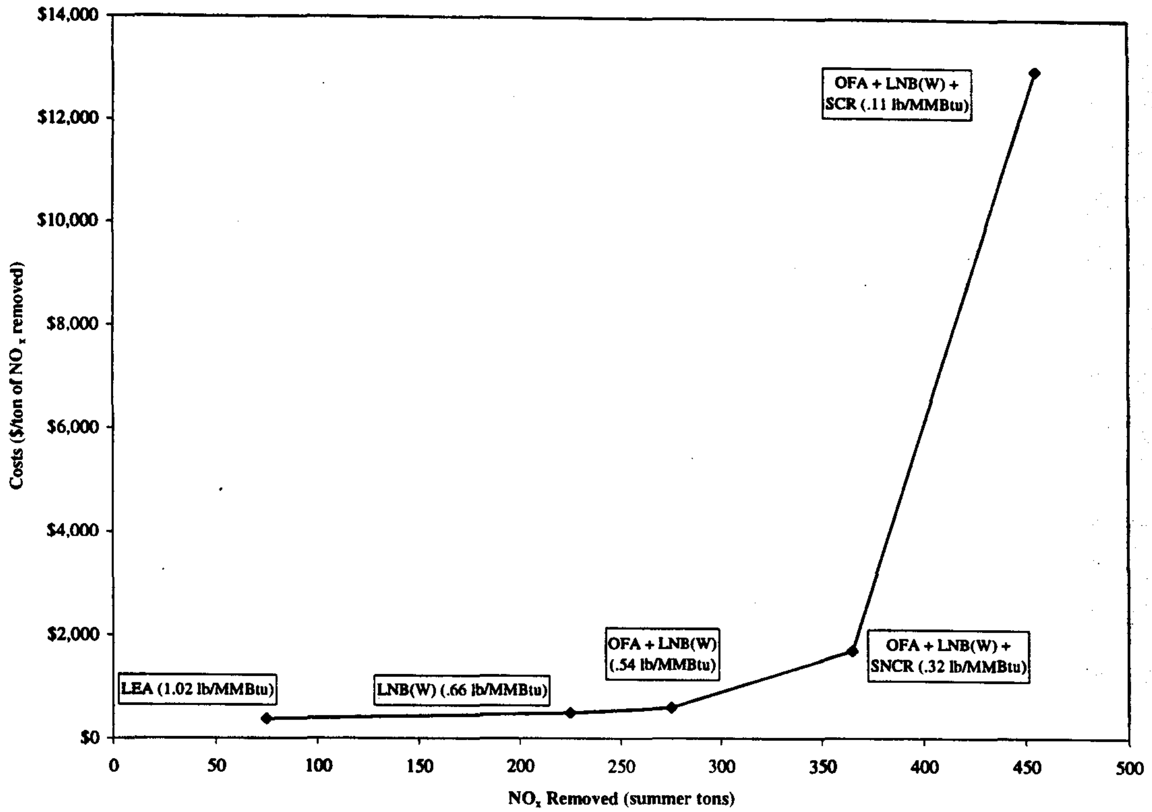


Table 5-1. NO_x Point Source Emission Reduction Costs at 0.15 lb/MMBtu.

Subregion	Summer Emission Reductions (tons)	Average Marginal Cost ($/ton)	Total Annual Cost ($million)
Metro East	47,344	12,615	597
Northeast	10,716	12,798	137
Band I	41,063	12,112	497
Band II	117,450	10,220	1,200
Band III(e)	104,340	9,711	1,013
Illinois	83,514	8,951	748
Central West	97,881	9,464	926
Total	502,308	10,191*	5,119

* This figure is not a total, but an average across subregions, weighted by emissions reductions.

Table 5-2. NO_x Point Source Emission Reduction Costs at 0.20 lb/MMBtu.			
Subregion	Summer Emission Reductions (tons)	Average Marginal Cost ($/ton)	Total Annual Cost ($million)
Metro East	39,685	9,918	394
Northeast	9,043	8,829	80
Band I	33,755	8,852	299
Band II	111,890	9,074	1,015
Band III(e)	96,898	7,936	769
Illinois	81,465	8,415	686
Central West	97,325	9,018	878
Total	470,061	8,765*	4,120
* This figure is not a total, but an average across subregions, weighted by emissions reductions.			

Low Emission Vehicles. The LEV program significantly reduces ground level VOC and NO_x emissions by requiring a set of emission standards for new types of vehicles. Target sales levels are phased in over a decade for transitional LEVs (TLEVs), LEVs, ultra-low emission vehicles (ULEVs), and zero emission vehicles (ZEVs).

To estimate the cost of reducing emissions, the focus is on the difference between the costs of LEV and the current federal programs. Control costs for the LEV program are computed using Equation 5-1. Emission reductions are based on MOBIL 5 model runs and draw on the MARAMA (1992) and NESCAUM (1991) studies evaluating the LEV program (Pechan, 1992). It should be noted that the evaluation of LEV does not account for changes in reactivity of VOC emissions.[2] Determining incremental control costs for larger catalysts, and for engine vehicle design modifications, and health benefits derived from the LEV emission standards is controversial. The major parties concerned, the automobile and petroleum industries and the government, have yet to agree on a consistent methodology for measuring the incremental costs of control technologies for vehicles. For this analysis, the reasoning of the New York State Department of Environmental Conservation is adopted. The automotive industry's methods based on "part-pricing"[3] are assumed to be biased upwards, and the California Air Resources Board (CARB) values for incremental costs are used instead.

The average emission reduction costs estimated for the LEV program in Table 5-3 are just over $2,000/ton for combined tons of NO_x and VOC. The differences in costs among the subregions result principally from

2. This may have a significant effect because the longer chain VOCs, which are more photochemically reactive, have a greater tendency to react with a catalyst, leaving less photochemically reactive VOCs as exhaust.
3. "Part-pricing" is a cost accounting technique of estimating the individual component costs and associated overhead.

the differences in miles traveled per vehicle and the type of driving conditions. The total annual cost of the LEV program for the OTR is approximately $187 million (see Appendix C for more details on transportation sector control programs).

Table 5-3. Emission Reductions and Control Cost of LEV Program.

Subregion	Summer Emission Reductions (thousand tons)	Average Cost ($/ton)	Total Cost ($million)
Metro East	69.2	$1,671	$116
Northeast	19.7	$2,285	$45
Band I	12.5	$2,067	$26
Total	101.4	$1,839	$187

<u>Reformulated Gasoline.</u> Use of reformulated gasoline has been popular as a control strategy because it addresses the control of both the reactivity and the total mass of VOC emissions.[4] The additional cost of RGas is measured from the cost of federal Phase II reformulated gasoline. The lower reactivity of VOC emissions from RGas is determined from the weighted reactivity of total organic gas emissions based on results of chamber experiments (Croes and Holmes, 1992; AQIRP, 1993). The costs of RGas are measured by the loss in consumer surplus; the cost calculation captures the out-of-pocket costs plus the additional cost of consuming less gasoline when it is more expensive. The price effect is greater in Metro East because of the availability of public transportation as a substitute for driving. For this study, an incremental cost of 9 cents/gallon, developed by the Auto/Oil Industry Study (AQIRP, 1993), is used.

RGas increases the variable cost of driving during the summer ozone season, providing an economic incentive consistent with the objective of reducing summer emissions. This differs from the LEV program, which raises the cost of purchasing a new vehicle but does not provide an economic incentive for people to drive less. Accounting for the reduced reactivity of VOC, RGas reduces VOC emissions by about 20% and NO_x emissions by only 10% (AQIRP, 1993). Even though behavioral responses to higher prices are included, RGas has an average cost of over $5,000/ton for NO_x and VOC emissions combined, which is more than twice as high as the cost for the LEV program.

ADDITIONAL CONTROL PROGRAMS

The regulatory base strategy has a relatively narrow focus on only three principal emission control programs (NO_x controls for utilities, LEV, RGas). It is important to evaluate additional control measures and behavioral control programs, especially for sources that currently have only limited controls. Additional engineering controls are evaluated for previously uncontrolled and moderately controlled sources. Additional behavioral control measures focus on reducing emissions from transportation sources in Metro East. These control measures are potentially valuable sources of additional emission reductions as well as potentially more cost-effective alternatives to the three programs in the regulatory base strategy.

Engineering Controls. Additional engineering controls apply combustion and post-combustion controls to sources that are currently relatively uncontrolled. The California Air Resources Board has evaluated and put into law a series of engineering control measures for off-road mobile sources and heavy duty diesel vehicles. Currently, off-road mobile sources do not represent a significant source of emissions compared with gasoline vehicles, but reducing emissions from sources with little or no emission control equipment is relatively inexpensive compared to controlling emissions from on-road construction vehicles that have been subject to controls in the past.

Off-road sources include a large variety of small gasoline engines. These engines range from small two-stroke engines for lawn and garden care to heavy duty diesel engines used extensively for construction. The costs of reducing emissions from small two-stroke off-road engines are less than \$100/ton for VOC emissions, but can exceed \$2,000/ton for NO_x emissions for off-road diesel vehicles.

CARB controls for on-road heavy duty diesel vehicles go beyond federal emissions standards, but these controls are relatively expensive. The cost for NO_x controls is over \$15,000/ton for urban transit buses and over \$10,000/ton for large diesel trucks. (The health benefits of reducing emissions of fine particulate matter is probably the most important environmental issue for policy makers.)

Behavioral Controls. Behavioral programs differ from command and control regulations by introducing economic incentives to induce a change in behavior or a switch to a less polluting activity. Control costs were estimated for a series of behavioral programs designed to reduce vehicle emissions from automobiles in Metro East. Behavioral programs use both the carrot and the stick to induce changes. The carrot is offered through reduced usage costs or tax credits such as lower public transit fares or a tax credit toward the conversion or purchase of a natural gas vehicle or a zero-emission vehicle. The stick has several different forms, including usage fees, consumption taxes, permits, and outright bans. Since feasible alternatives to driving exist in Metro East, additional costs can be charged to drivers to reflect the costs they impose on society for polluting and congestion in urban areas, including increased tolls, emissions based registration fees, metered parking, and a consumption tax on gasoline. However, in most cases, the costs per

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4. Gasoline and the exhaust from conventionally fueled vehicles are highly reactive in the atmosphere because they are rich in aromatics and alkenes.

ton of reducing emissions using behavioral controls, as shown in Table 5-4, are substantially higher than they are for engineering controls (Dorris, 1996).

Table 5-4. Emission Reductions and Costs for Behavioral Control Programs.

Programs	NO_x Reduction	VOC Reduction	Combined Cost, NO_x and VOC ($/ton)
Emissions based tolls	1%	2%	30,000
Lower summer public transit fares	1%	2%	30,000
Emissions based registration fee (annual)	1%	2%	30,000
Increased public transport infrastructure	3%	3%	30,000
Increased turnpike tolls	1%	1%	30,000
Redesigned LEV program	2%	3%	30,000
Summer driving permit ($20-$40)	3%	4%	30,000
HOV lanes	1%	1%	30,000
Natural gas incentive program	0.2%	4.0%	12,000
Clean taxi program	0.2%	1.0%	4,000

SUMMARY

Control costs were developed for a wide range of emission control strategies. These control costs form a database of emission controls for both elevated and ground level sources of emissions. In developing this database, a variety of standard publications were used to give estimates of costs that generally fall between the estimates of industry groups and those of environmental advocacy groups.

The next step in the analysis is to create an integrated framework for analysis of different emission control strategies by combining the emission control costs database with the emission weights presented in Section 4. This integrated framework for evaluating these strategies can deal with command and control regulations, traditional emissions trading, and weighted emissions trading. The results of this integrated analysis determine the amount of emissions reductions and the associated costs by emissions category for each subregion. In addition, the corresponding incremental reductions of ozone concentrations in Metro East are calculated. The results of the economic analysis are summarized in the next section.

Section 6
LEAST COST STRATEGIES FOR REDUCING OZONE CONCENTRATIONS IN METRO EAST

INTRODUCTION

This section reports the results of an economic analysis that combines both the emissions weights estimated in Section 4 and the control cost data presented in Section 5 to determine least-cost strategies for reducing ozone concentrations in the Metro East receptor region. The analysis identifies the economically efficient balance of controls on emissions among electric utilities, transportation, and other sectors. In addition, a comparison is made between setting fixed emission standards for sources of emissions and different types of trading schemes for emissions.

A set of command and control regulations is described in the first part of this section. These controls are relatively expensive to enforce but they provide a basis for comparison with other more economically efficient control strategies. The second part of this section describes the effects of traditional trading of emissions of NO_x among point sources. The objective is to minimize the total cost of reducing emissions of NO_x by a specified amount. The third part of this section describes the effects of weighted trading of emissions among point sources of NO_x, where the objective is to reduce the concentration of ozone in Metro East by a specified amount. The use of weighted trading is extended in the last part of this section to include all sources of emissions from all economic sectors. This provides the framework for identifying the least cost strategy for reducing the concentration of ozone in Metro East by the same amount as the cost and control regulation. The use of intersectoral weighted trading among all sources of emissions reduces the total cost of controls by 60% when compared with the command and control regulation.

COMMAND AND CONTROL REGULATION

For the purpose of this analysis, "command and control" regulation refers to a set of emissions control policies which apply fixed requirements to similar sources of emissions. Examples include standards on the rate of emissions for point sources (such as utility power plants) and technical control requirements for automobiles (such as advanced reformulated gasoline).

Description of the Analysis

The command and control regulation adopted for this analysis is based on the strategies presented in the Ozone Transport Commission's (OTC) Memorandum of Understanding (MOU) in 1995 for reducing ozone in the northeastern Ozone Transport Region (OTR). For the command and control analysis, the emissions trading component of the OTC MOU is not included. The programs recommended by the OTC include 1) an emissions rate limit for utilities and large industrial sources, 2) the use of advanced reformulated gasoline, and 3) a package of low emission vehicles similar to the one adopted in California. The OTC recommended that these programs be applied within the northeastern states (corresponding to Metro East, Northeast, and Band I).[1] In addition, the programs for elevated NO_x are applied to all subregions within the OTAG modeling domain that contribute to ozone in Metro East (Band II and Band III). The latter extension corresponds approximately to the EPA program for controlling point sources of NO_x that was proposed in November 1997, but once again the trading component of the program is not included. A summary of the programs used in the analysis is given in Table 6-1 (note that Band III corresponds to Band III(e) plus Illinois and Chicago[2]). It is assumed that all emission control programs will be implemented regardless of their cost and without consideration of the fact that many sources are forced to "over control" in order to meet the minimum standard.[3]

Overall Results

Under the command and control scenario described above, total emissions of NO_x and VOC combined would be reduced by just over 900,000 tons per summer. Using the MORS (Multivariate Ozone Response Surface) emissions weights, this level of emissions reduction (given the pattern of reductions across sectors and source subregions) translates into a 12.5 ppb reduction in the predicted ozone concentration in the Metro East receptor area (using the 8-hour ozone standard). The total cost of the regulation was estimated by the REMAQ (Regional Economic Model of Air Quality) model at just under $2 billion per year. The results of the command and control regulatory strategy are summarized in Table 6-2.

An important feature of the results for the command and control scenario is the extremely high values of marginal control costs in most subregions, with the highest of nearly $1 million/summer ton in Metro East. These very high marginal costs are symbolic of the main problem with command and control regulations — by imposing the same emission rate requirements on all sources of emissions regardless of cost, some sources face unacceptably high costs of controls.

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1. In addition to the proposed control measures of the OTC, each state in the OTR must submit a state implementation plan (SIP) to EPA. Each state's SIP must include control programs used to demonstrate progress toward meeting the standard for ozone.

2. This breakdown was made for an analysis using Chicago as the receptor region. It is retained here to illustrate how high marginal costs can go with inefficient control strategies.

3. The discrete nature of the NO_x control technologies applied to combustion sources can cause sources to "over-comply" with the NO_x standard of 0.15 lb/MMBtu. For example, when selective noncatalytic reduction (SNCR) is applied to a wall-fired boiler, that boiler typically reaches a NO_x rate between 0.15 and 0.25 lb/MMBtu. The application of selective catalytic reduction (SCR) will in most cases reduce the NO_x rate to less than 0.10 lb/MMBtu. Many sources must install SCR technology to meet this standard, but in doing so, they over-comply with the standard.

Table 6-1. Command and Control Emission Controls Programs.			
Sector	**Description and/or Examples**	**Subregions to which controls were applied**[2]	**Potential Emissions Reductions**
Industrial[1]	Industrial boilers with heat rates greater than 250 MMBtu/hr must meet a NO_x emission standard of.15 lb/MMBtu	Northeast Band I Band II Band III(e) Illinois & Chicago	Elevated NO_x
Transportation	Reformulated Gasoline Low Emissions Vehicles (LEV)	Metro East Northeast Band I	Ground-level NO_x VOC
Utility	All utility boilers are required to meet a minimum NO_x emission standard of.15 lb/MMBtu	Metro East Northeast Band I Band II Band III(e) Illinois & Chicago	Elevated NO_x

1. The industrial controls would be applied to the Metro East subregion; however, no industrial boilers of this size exist in Metro East.
2. In developing the emission weights, Band III included Illinois and Chicago. This subregion has been disaggregated in the section in order to provide economic results at a higher level of detail.

This is economically inefficient and leads to political resistance to implementing environmental programs. Note that while the marginal costs are very high, the average costs in all sectors are much lower (over two orders of magnitude lower for utilities in some cases), indicating that there are many control technologies with relatively low control costs per ton in each subregion and source category.

Although marginal costs are important for implementing an economically efficient strategy, it is sensible to use average costs to compare costs across sectors and subregions with a command and control strategy because they are more representative of the typical costs of the selected controls. The marginal costs represent the most expensive single control within a sector and subregion, and they are not representative of the other selected controls. For example, the marginal cost for utilities in Band I is the lowest by far for utilities, but the average costs for utilities in Band II and Band III are lower than the average cost in Band I.

Table 6-2. Results of Command and Control Regulatory Analysis.[1]

Subregion and Sector Type	Emissions Reduced (1,000 tons/summer) NO$_x$	VOC	Total	Average Cost ($/summer Ton)	Marginal Cost ($/summer ton)	8-Hour Ozone Reduction (ppb)	Average Cost per ppb O$_3$ Reduced ($million)	Annualized Total Cost ($million)
Metro East	**161**	**76**	**237**	**$4,277**	**$972,278**	**6.26**	**$162**	**$1,012**
Industrial[2]	0	0	0	$0	$0	0.00	$0	$0
Transportation	71	76	147	$5,461	$22,759	3.56	$225	$803
Utility	90	0	90	$2,333	$972,278	2.69	$78	$209
Northeast	**37**	**16**	**53**	**$3,254**	**$426,430**	**0.29**	**$598**	**$173**
Industrial	2	0	2	$1,500	$4,714	0.01	$430	$3
Transportation	15	16	31	$4,241	$18,308	0.18	$754	$133
Utility	20	0	20	$1,861	$426,430	0.11	$350	$37
Band I	**100**	**19**	**119**	**$1,697**	**$54,472**	**2.37**	**$85**	**$202**
Industrial	1	0	1	$1,693	$21,715	0.02	$76	$2
Transportation	16	19	35	$2,808	$16,624	0.46	$215	$99
Utility	83	0	83	$1,223	$54,472	1.89	$54	$101
Band II	**219**	**0**	**219**	**$1,030**	**$881,675**	**2.99**	**$75**	**$226**
Industrial	1	0	1	$2,051	$25,000	0.02	$150	$3
Utility	218	0	218	$1,024	$881,675	2.97	$75	$223
Band IIIe	**174**	**0**	**174**	**$1,157**	**$837,740**	**0.38**	**$526**	**$201**
Industrial	4	0	4	$2,649	$25,000	0.01	$1,204	$10
Utility	170	0	170	$1,123	$837,740	0.37	$511	$191
Illinois and Chicago[3]	**112**	**0**	**112**	**$1,387**	**$779,695**	**0.25**	**$630**	**$156**
Industrial	1	0	1	$9,970	$25,000	0.00	$4,530	$12
Utility	111	0	111	$1,291	$779,695	0.24	$587	$144
Total	**803**	**111**	**914**	**$2,155**	**$972,278**	**12.54**	**$157**	**$1,969**
Industrial	**9**	**0**	**9**	**$3,249**	**$25,000**	**0.06**	**$520**	**$29**
Transportation	**103**	**111**	**214**	**$4,844**	**$22,759**	**4.20**	**$246**	**$1,035**
Utility	**691**	**0**	**691**	**$1,309**	**$972,278**	**8.28**	**$109**	**$905**

1. This table summarizes the costs of implementing the command and control programs by subregion and economic sector. Tables 6-3, 6-6, 6-9, and 6-10 present the results of the traditional and weighted trading strategies respectively, and follow the same format as that of command and control.
2. No industrial controls could be applied in Metro East because all boilers are smaller than the minimum size for this regulation.

Emissions reduced are listed for NO$_x$ and VOC emissions. The NO$_x$ total combines both ground-level and elevated NO$_x$ emissions to facilitate the presentation of the results. Similarly, because NO$_x$ and VOC emissions react differently in the formation of ozone, the total column for emissions is presented only as a simple means of representing the effectiveness of the sector's controls in reducing overall emissions.

Average cost (all control cost estimates are in 1995 U.S. dollars) is calculated by dividing the total costs of implementing a particular subregion and sector's control programs by the total summer tons reduced under those programs.

Marginal cost represents the cost per summer ton of implementing the most expensive control program selected for the given subregion and sector. These costs can be very high for utility boilers with capacity factors of less than 10%.

Ozone reduced in Metro East (using the 8-hour standard) is calculated by multiplying the emission weight for each subregion and sector by the tons reduced under the applicable control programs. The total ozone reduced is determined by summing across regions and sectors.

Average cost per O$_3$ reduced per year is calculated by dividing the total costs for each subregion and sector by the total ozone reduction in Metro East (for the associated emission reductions from each subregion and sector).

Total cost represents the total cost of all control programs from each subregion and sector per year.

The total rows at the bottom of the table represent the overall summation and individual sectoral summation for each column except the average and marginal cost columns. For these columns, weighted means (using tons reduced as the weight) were used to summarize the overall and sectoral means.

Utility Sector Results

Under the command and control framework, the utility sector accounts for roughly three quarters of the total tons reduced, and two thirds of the total ozone reduced. Considering the stringency of the standard applied to utility boilers, this result is not unexpected. At the same time, it is relatively efficient for utilities to make a significant portion of the emission reductions. Despite being responsible for two-thirds of the reduction in ozone, the cost of the utility controls represents less than half of the total cost of the regulation. These results reflect the underlying cost characteristics for controlling emissions of NO_x described in Section 5. The costs of combustion modifications and SNCR (selective noncatalytic reduction) are relatively inexpensive compared to the incremental cost of SCR (selective catalytic reduction). There are also important economies of scale in controls so that costs are relatively low for large point sources. High marginal costs occur when SCR is required to meet the mandated emission rate standard for a small utility boiler with a low capacity factor.

Regional Utility Sector Results

Since command and control regulations affect utilities uniformly, the regional results for the utility sector reflect the distribution of utility sources and the levels of emission controls in the 2007 base inventory of emissions. Emission reductions within the Metro East subregion are smaller than some of the upwind subregions (90 thousand tons versus 218 thousand tons in Band II and 281 thousand tons in Band III) because utilities within Metro East tend to be oil and gas fired generation meeting RACT as opposed to coal fired generation in the more rural subregions of Bands I, II and III. Therefore, to meet the 0.15 lb/MMBtu standard in Metro East, a few relatively expensive controls are required for the relatively low emitting plants. In contrast, plants in Bands I, II, and III must reduce emissions significantly to meet the 0.15 lb/MMBtu standard, but many of the controls are relatively inexpensive. Consequently, the average cost of controlling NO_x is higher for Metro East than other subregions ($2,300/ton compared to an average of $1,300 for all subregions).

Under the command and control strategy, reductions of NO_x from the utility sector in the Northeast subregion are the smallest because this subregion generates only a limited amount of electric power. Average NO_x control costs for Northeast utilities ($1,900/ton) are also high compared to similar control costs in Bands I, II, and III because, like the utilities in Metro East, Northeast utilities meet RACT.

However, the most important column in Table 6-2 is the second to last one, giving the average cost of ozone reductions in Metro East per ppb (the actual environmental goal). These values integrate the effectiveness of emission reductions (i.e., the emission weights from the MORS) in each subregion for decreasing ozone concentrations in the Metro East receptor region *and* the corresponding costs per ton of reducing emissions. The analysis produces values for utilities which range over almost two orders of magnitude from $54 million/ppb (Band 1 utility boilers) to $4,530 million/ppb (Illinois and Chicago industrial boilers). Sources in subregions which are relatively far away (Band III) or mostly downwind (Northeast) have higher average costs of reducing ozone than other subregions (Metro East, Band I, and Band II). This suggests that controls in the Northeast and Band III will be relatively inefficient in controlling ozone in the Metro East receptor region because the corresponding emission weights are small.

Importantly, the best opportunities in the utility sector for reducing ozone in Metro East (from an economic efficiency standpoint) come from controls on the high concentration of coal-fired generation units in Band I (within the OTR) and Band II (outside the OTR). The reason for this is that these subregions combine low average costs of controlling emissions with relatively large emission weights. In addition, the large emission weight for utilities in Metro East compensates for the relatively high average cost of controls in this subregion so that the average cost of reducing ozone is low. The overall conclusion for the utility sector is that controls on utilities in Metro East, Band I, and Band II are the most cost-effective for reducing ozone in Metro East, with average costs of less than $100 million per ppb.

Transportation Sector Results

In this analysis of the command and control strategy, controls on emissions in the transportation sector are responsible for almost a quarter of the total emissions reduction and a third of the total ozone reduction. However, these programs cost over $1 billion per year and represent half of the total cost of the command and control strategy. In general, controls on transportation are relatively inefficient compared to controls on utilities. Across all subregions in the OTR, the average cost of controls in transportation programs, measured in dollars per ppb of ozone reduced, is twice as high as the corresponding cost in the utility sector. An underlying reason is that the emission weights for VOC are relatively small or zero in all three subregions of the OTR. Consequently, the cost-effectiveness of using advanced reformulated gasoline is low compared to controls that reduce emissions of NO_x such as low emission vehicles (LEV).

Regional Transportation Sector Results

Over two-thirds of the reductions in transportation sector emissions occur within the Metro East subregion. Since the contribution of those emissions per ton to ozone formation is also the highest (i.e., the emission weights for Metro East are relatively large), nearly 80% of the total reduction of ozone from controls in the transportation sector is attributed to controls in Metro East. The average cost of emission reductions per ton is highest in Metro East because the vehicles in Metro East are driven less than vehicles in other subregions. Therefore, the annualized fixed cost of switching to natural gas for a LEV program, for example, is averaged over fewer miles per vehicle and less emissions per vehicle.

Transportation programs in the Northeast subregion are almost 25% less expensive on average than the programs in Metro East, but this is a relatively expensive means of reducing ozone in Metro East because the emission weights are low for both ground-level NO_x and VOC. Transportation controls in the Northeast are over three times as expensive per ppb for reducing ozone concentration in Metro East compared to the controls in Metro East and Band I.

Transportation programs in Band I are less expensive than transportation controls in the other two subregions because vehicles are driven more in miles. This spreads the fixed costs of installing emission controls over more tons of emissions per vehicle. The low cost per ton of controls is offset by the zero emission weight for VOC to make the cost-effectiveness of transportation controls per ppb in Band I similar to the corresponding controls in Metro East.

Industrial Sector Results

The emission reductions for industrial boilers represent only 1% of total emission reductions from all sources in Table 6-2. Most large industrial boilers can be controlled using similar technologies, and at similar costs, to utility boilers. Since utility boilers are generally larger than industrial boilers, the industrial sector cannot benefit as much from economies of scale. Consequently, the average costs of controls per ton of emissions are higher in the industrial sector than they are in the utility sector. Overall, the industrial sector was responsible for less than 0.5% of the total ozone reduced under the command and control strategy and about 1.5% of the total cost. It was the smallest and least cost-effective program among sectors, and the overall average cost per ppb of ozone reduced is over twice as high as the cost in the transportation sector and five times as high as the cost in the utility sector.

Regional Industrial Sector Results

No industrial programs were applied in Metro East because the subregion does not contain any industrial boilers larger than the minimum size of 250 MMBtu/hr (see Table 6-1). In the Northeast and Band I subregions, industrial controls are relatively inexpensive (just over $1,500/ton). However, because the Northeast subregion has such a low emission weight for elevated NO_x, these controls were a very expensive means of reducing ozone in Metro East. On the other hand, the high emission weights for NO_x from elevated sources (i.e., utility plants and industrial boilers) in Band I make industrial controls in this subregion a relatively efficient means of ozone reduction, with a cost well below $100 million per ppb of ozone in Metro East. Industrial controls in Band II and Band III have higher average costs of removing emissions, and the corresponding emission weights are relatively low, particularly for Band III. The cost of $150 million per ppb of ozone for industrial controls in Band II is twice as high as the cost in Band I, and the cost in Band IIIe is well over $1 billion per ppb of ozone. For Illinois and Chicago, the cost is over $4 billion per ppb of ozone, the highest average cost by far among all sources of emissions.

Summary of Command and Control Results

The command and control regulation affects mainly the transportation and utility sectors. The emission reductions and the associated costs for the industrial sector are relatively small. The total cost of all command and control programs is nearly $2 billion per year, and it is divided fairly equally between the transportation and utility sectors. In contrast, three-quarters of the total emissions reduction and two-thirds of the total ozone reduction in Metro East are attributed to the utility sector. Consequently, the cost-effectiveness of controls in the utility sector is higher than it is for transportation for both the average cost per ton of reducing emissions and the average cost per ppb of reducing ozone. However, the marginal costs of controls per ton of emissions are much higher for the utility sector than they are for the transportation sector. This is an indication that the standards set for the utility sector may be too stringent to be economically efficient relative to emission control programs for the transportation sector. Although combustion controls and SNCR are relatively inexpensive ways of reducing emissions of NO_x in the utility sector compared to transportation controls, requiring the next step to SCR is expensive and an economically inefficient way to reduce ozone in Metro East. These economic problems can be addressed by allowing for trading of emissions among point sources and determining a cost-effective balance between controls in the utility and transportation sectors. The potential for meeting the same environmental goal of reducing ozone in Metro East at a much lower cost is explained at length in the following subsections.

TRADITIONAL TRADING FOR POINT SOURCES

The "traditional trading" of emissions is defined here to mean the exchange of emission allowances on a ton-for-ton basis within one or more sectors across subregions. An example is the acid rain trading program for sulfur dioxide under Title IV of the 1990 Clean Air Act Amendments. In this subsection, traditional trading is applied to NO_x from utility and industrial boilers.

Description of the Analysis

The command and control system described above required the implementation of specific emission control programs regardless of their cost or their effectiveness in reducing ozone concentrations in Metro East. Traditional emissions trading takes a more flexible, goal-oriented approach by making reductions at the sources that can be controlled most cost-effectively. However, the goal of a traditional emissions trading system is to reduce total emissions of NO_x rather than ozone concentrations in Metro East. For this analysis, the traditional trading system applies only to utility and industrial sources, and the goal is to meet the same reductions of NO_x from utility and industrial sources under the command and control regulations in Table 6-2 (700,000 tons). However, there is no size limit applied to boilers, and as a result, a few reductions from industrial boilers in Metro East are found to be cost-effective even though none of these boilers qualified for controls under the command and control regulation.

Overall Results

Under the traditional trading system, the control programs for utility and industrial sources of NO_x were ranked in terms of the cost per ton of reducing NO_x. The least expensive controls were selected until 700,000 tons of NO_x emissions were reduced from utility and industrial sources. The results are summarized in Table 6-3. The total cost of these emission reductions was \$830 million per year at an average cost of nearly \$1,200 per ton. The same level of emission reductions under the command and control framework (from the utility and industrial sectors) cost \$930 million at an average cost of over \$1,300 per ton. Although the savings in cost from traditional trading (11%) are modest, the marginal cost per ton is much lower ($10,000 per ton for traditional trading compared to $25,000 per ton under command and control).

Table 6-3. Results of Traditional Trading for Point Sources.								
Region and Sector Type	Emissions Reduced (1,000 tons/summer) NO$_x$	VOC	Total	Average Cost ($/summer Ton)	Marginal Cost ($/summer Ton)	8-Hour Ozone Reduction (ppb)	Average Cost per ppb O_3 Reduced ($million)	Annualized Total Cost ($million)
Metro East	**87**	**0**	**87**	**$1,626**	**$10,776**	**2.62**	**$54**	**$141**
Industrial	0	0	0	$2,236	$ 3,400	0.00	$102	$0
Utility	87	0	87	$1,626	$10,776	2.62	$54	$141
Northeast	**22**	**0**	**22**	**$1,534**	**$10,710**	**0.11**	**$295**	**$34**
Industrial	2	0	2	$1,590	$ 5,767	0.01	$456	$4
Utility	20	0	20	$1,527	$10,710	0.11	$283	$30
Band I	**85**	**0**	**85**	**$1,318**	**$10,000**	**1.95**	**$57**	**$112**
Industrial	1	0	1	$1,491	$ 5,682	0.02	$67	$2
Utility	84	0	84	$1,315	$10,000	1.93	$57	$111
Band II	**219**	**0**	**219**	**$982**	**$10,686**	**2.99**	**$72**	**$215**
Industrial	1	0	1	$2,159	$ 4,389	0.02	$158	$3
Utility	217	0	217	$974	$10,686	2.97	$71	$212
Band IIe	**175**	**0**	**175**	**$1,131**	**$10,447**	**0.39**	**$514**	**$199**
Industrial	6	0	6	$3,070	$ 5,767	0.01	$1,395	$20
Utility	169	0	169	$1,058	$10,447	0.37	$481	$179
Illinois and Chicago	**112**	**0**	**112**	**$1,171**	**$10,747**	**0.25**	**$532**	**$131**
Industrial	1	0	1	$1,883	$ 5,682	0.00	$856	$2
Utility	111	0	111	$1,165	$10,747	0.24	$529	$129
Total	**699**	**0**	**699**	**$1,187**	**$10,776**	**8.30**	**$100**	**$831**
Industrial	**11**	**0**	**11**	**$2,450**	**$ 5,767**	**0.06**	**$448**	**$30**
Utility	**688**	**0**	**688**	**$1,165**	**$10,776**	**8.23**	**$97**	**$801**

The predicted reduction of ozone concentrations in Metro East under traditional trading in Table 6-3 is an unexpected result. The reduction of ozone is 8.3 ppb, which is almost identical to the corresponding reduction from utility and industrial controls under command and control in Table 6-2. While in general there is no guarantee that a traditional trading scheme will meet the same environmental goal as a command and control program (e.g., a reduction of ozone in Metro East, or a lower rate of acid deposition in an environmentally sensitive subregion under the traditional trading program for sulfur dioxide), in this simulation it did.

Comparing Command and Control with Traditional Trading

In comparison to the command and control system, the traditional trading regime results in slightly fewer NO_x emission reductions in the utility sector and, consequently, more in the industrial sector. Also, traditional trading results in less variation in the average cost of reducing NO_x emissions compared to the command and control system. Under the command and control system, the average cost of utility and industrial controls is between $1,000 and $10,000 per ton, but under the traditional trading system, the range is $1,000 to $3,000 per ton because the most expensive controls were replaced by less expensive alternatives under traditional trading. A summary of the regional differences in emission reductions and the corresponding ozone reductions between the command and control and the traditional trading systems is presented in Table 6-4. The results show that the regional differences are generally quite small.

Table 6-4. Comparison of Command and Control and Traditional Trading for Point Sources.

	Emission Reductions (1,000 tons/summer)			8-Hour Ozone Reduction (ppb)		
Subregion & Sector	Command And Control	Traditional Trading	Net Change	Command And Control	Traditional Trading	Net Change
Metro East	**90**	**87**	**-3**	**2.7**	**2.6**	**-0.1**
Industrial	0	0	0	0.0	0.0	0.0
Utility	90	87	-3	2.7	2.6	-0.1
Northeast	**22**	**22**	**0**	**0.1**	**0.1**	**0.0**
Industrial	2	2	0	0.0	0.0	0.0
Utility	20	20	0	0.1	0.1	0.0
Band I	**84**	**85**	**1**	**1.9**	**2.0**	**0.1**
Industrial	1	1	0	0.0	0.0	0.0
Utility	83	84	1	1.9	1.9	0.0
Band II	**219**	**218**	**-1**	**3.0**	**3.0**	**0.0**
Industrial	1	1	0	0.0	0.0	0.0
Utility	218	217	-1	3.0	3.0	0.0
Band IIIe	**174**	**175**	**1**	**0.4**	**0.4**	**0.0**
Industrial	4	6	2	0.0	0.0	0.0
Utility	170	169	-1	0.4	0.4	0.0
Illinois and Chicago	**112**	**112**	**0**	**0.3**	**0.3**	**0.0**
Industrial	1	1	0	0.0	0.0	0.0
Utility	111	111	0	0.3	0.3	0.0
Total	**701**	**699**	**-2**	**8.3**	**8.3**	**0.0**
Industrial	9	11	2	0.1	0.1	0.0
Utility	692	688	-4	8.2	8.2	0.0

For the utility sector, the biggest regional difference is the lower level of reductions under traditional trading in Metro East. As a result, the average cost of reductions in Metro East decreased substantially from

$2,300 per ton under the command and control system to $1,600 per ton under traditional trading, and the total cost in Metro East fell by almost $70 million to $140 million per year. Small additional decreases in emission reductions from utilities occur in the Northeast, Band II, and Band III subregions under traditional trading. The only increase of emission reductions in the utility sector occurs in Band I, but it is small (less than 2%).

For the industrial sector under traditional trading, the biggest difference is the increase of emission reductions in Band III(e). Small increases are also found in all the other subregions except Illinois and Chicago. In the latter subregion under command and control, 1,200 tons of emission reductions cost $12 million per year to give an average cost of $10,000 per ton. Under traditional trading, 200 tons more are emitted but the cost saving is over $10 million per year, which implies that the additional 200 tons reduced under command and control cost over $50,000 per ton. This illustrates the type of inefficiency that motivates political resistance to command and control regulation, and supports the use of emission trading programs.

Overall, the changes implied by the traditional trading system are relatively minor. Even an 11% savings in total costs using traditional trading is small. The reason that there is not a greater contrast between command and control and traditional trading is that the command and control standard for the rate of emissions is very stringent. The emission rate of 0.15 lb/MMBtu requires that most of the controls are needed to meet the total reduction. In other words, most of the control programs *required* under the command and control system must also be *chosen* under the trading system in order to meet a reduction of 700,000 tons of NO_x per summer. One would expect that the percentage savings in total cost would be larger using traditional trading if a higher rate of emissions had been set as the standard.

WEIGHTED TRADING FOR POINT SOURCES

The "weighted trading" of emissions is defined here to mean the exchange of emissions allowances among sources in different regions and in different sectors based on of the effectiveness of those emissions in meeting a specific environmental goal. Weighted trading in this analysis minimizes the cost of reducing ozone concentrations in Metro East. Weighted trading provides a basis for determining the potential cost saving through incorporating the spatial and species effects of ozone formation. Weighted trading is more sophisticated than traditional trading programs and requires a significant scientific input to establish appropriate trading weights. For this analysis, the emission weights estimated for the 8-hour ozone standard by modeling the July 1995 ozone episode with UAM-V are used (see Table 4-1).

Description of the Analysis

The principal difference between the weighted trading system and the traditional trading system described above is that the goal of traditional trading is to reduce emissions by a specified amount, and the goal of weighted trading is to reduce ozone concentrations by a specified amount in a particular receptor region. Specifically, the goal is to find the least expensive combination of controls on point sources in the utility and industrial sectors to reduce 8-hour ozone concentrations in Metro East by 8.3 ppb. Weighted trading accomplishes this goal by incorporating the emission weights into the analysis so that control programs are ranked in terms of the cost per ppb of reducing ozone concentrations in Metro East rather than the cost per ton of reducing of NO_x. (The product of the tons removed, for a given type of emission using a particular control option, and the corresponding emission weight, for the appropriate subregion, is the measure of the effectiveness of the control.) Weighted trading minimizes the cost of ozone reduction by selecting the control programs that simultaneously are inexpensive and effective (i.e., have a large emission weight), whereas traditional trading ignores effectiveness. Compared to traditional trading, one would expect larger emission reductions in subregions with high emission weights for elevated NO_x (Metro East and Band I), and smaller emission reductions in subregions with low emission weights (Northeast and Band III).

Overall Results

Selecting the least expensive control programs, in terms of the cost per ppb of 8-hour ozone reduced in Metro East, to meet the ozone reduction target of 8.3 ppb for the 8-hour ozone concentration, weighted trading reduced emissions by nearly 650,000 tons at a cost of less than $650 million per year in the five subregions (see Table 6-5). Traditional trading met the target reduction of ozone (8.3 ppb) by reducing emissions of NO_x by 700,000 tons at a total cost of $830 million per year. The implication of the result is that weighted trading among utility and industrial sources results in an 8% decrease of emission reductions and a much bigger 22% decrease in the total costs of the controls compared to traditional trading. Comparing weighted trading with command and control in Table 6-5, the total cost for the utility and industrial sectors is almost one-third lower under weighted trading and still meets the same environmental goal of reducing 8-hour ozone concentrations in Metro East by 8.3 ppb.

Comparing Weighted Trading with Traditional Trading for Point Sources

The use of weighted trading for point sources ensures that the most economically efficient controls will be invoked to meet the given target level of ozone reduction in Metro East. While many of the control programs available for utility and industrial sources in Metro East are more expensive than controls in Band II and Band III, for example, the large emission weights for NO_x in Metro East increase the effectiveness of controls in Metro East. As a result, even though the total number of tons removed decreases under weighted trading compared to traditional trading, the tons removed in Metro East increase by approximately 4%.

Table 6-5. Results of Weighted Trading for Point Sources.								
Region and Sector Type	Emissions Reduced (1,000 tons/summer) NO_x	VOC	Total	Average Cost ($/summer Ton)	Marginal Cost ($/summer ton)	8-Hour Ozone Reduction (ppb)	Average Cost per ppb O_3 Reduced ($million)	Annualized Total Cost ($million)
Metro East	**90**	**0**	**90**	**$2,181**	**$37,527**	**2.75**	**$73**	**$202**
Industrial	0	0	0	$2,321	$4,000	0.01	$375	$5
Utility	90	0	90	$2,178	$37,527	2.73	$72	$197
Northeast	**19**	**0**	**19**	**$ 847**	**$8,506**	**0.10**	**$156**	**$16**
Industrial	2	0	2	$1,065	$2,748	0.01	$306	$2
Utility	17	0	17	$ 822	$8,506	0.09	$146	$14
Band I	**85**	**0**	**85**	**$1,324**	**$12,398**	**1.95**	**$58**	**$113**
Industrial	1	0	1	$1,491	$5,682	0.02	$67	$2
Utility	84	0	84	$1,322	$12,398	1.93	$58	$111
Band II	**219**	**0**	**219**	**$ 987**	**$11,556**	**2.99**	**$72**	**$216**
Industrial	1	0	1	$2,159	$4,389	0.02	$158	$3
Utility	218	0	218	$ 979	$11,556	2.97	$72	$213
Band IIIe	**141**	**0**	**141**	**$ 369**	**$1,907**	**0.31**	**$168**	**$52**
Industrial	3	0	3	$ 627	$1,453	0.01	$285	$2
Utility	138	0	138	$ 364	$1,907	0.30	$165	$50
Illinois and Chicago	**93**	**0**	**93**	**$ 528**	**$1,887**	**0.20**	**$240**	**$49**
Industrial	1	0	1	$ 624	$1,453	0.00	$283	$0
Utility	92	0	92	$ 527	$1,887	0.20	$240	$49
Total	**647**	**0**	**647**	**$ 997**	**$37,527**	**8.30**	**$78**	**$647**
Industrial	**8**	**0**	**8**	**$1,395**	**$5,682**	**0.07**	**$197**	**$14**
Utility	**639**	**0**	**639**	**$ 991**	**$37,527**	**8.23**	**$77**	**$633**

In Bands I and II, emission reductions are nearly unchanged between the traditional and weighted trading systems. The expected increased reductions in Band I (because the emission weights for NO_x are large) did not occur under weighted trading because the marginal costs of additional controls (e.g., installing SCR) were too high.

In moving from traditional trading to weighted trading, emission controls are relaxed in the more distant subregions such as Band III, and in the Northeast (because the emission weights are small). The changes of costs under weighted trading amplify the changes of emission reductions. Costs are over 40% higher in Metro East and over 50% and 70% lower in the Northeast and Band III, respectively.

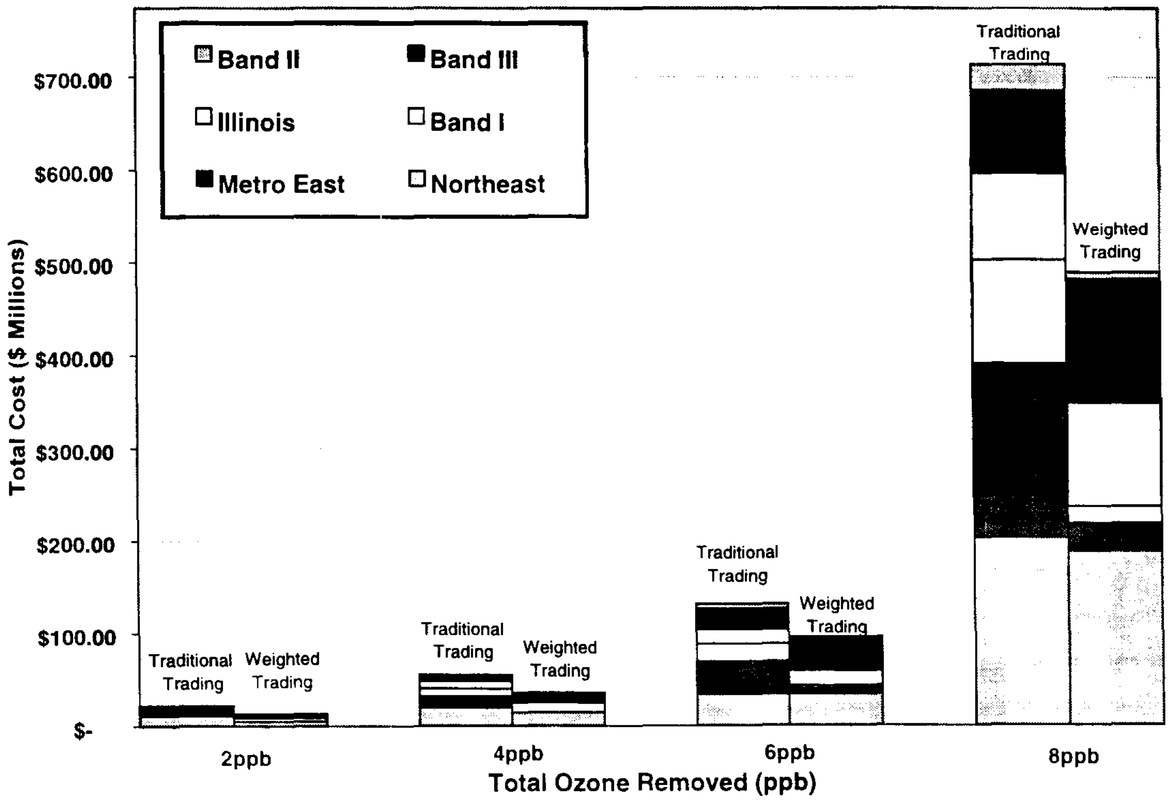
In comparing the weighted and traditional trading systems for different levels of reduction of ozone, it is more evident that certain subregions are more efficient than others at removing ozone (see Table 6-6). For example, for a reduction of only 2 ppb in ozone in Metro East under weighted trading, no additional controls on emissions beyond those in the base inventory are put in the Northeast, Band III(e), and Illinois and Chicago subregions. The most cost-effective programs in the utility and industrial sectors for reducing ozone in Metro East are located in Metro East, in Band I within the OTR, and in Band II outside the OTR. A 4 ppb reduction in ozone under weighted trading requires additional emissions controls in these three subregions even though the total cost of controls more than doubles. For a 6 ppb reduction in ozone, utility controls in Northeast and Band III and industrial controls in Band II are introduced, and the total cost more than doubles again. Finally, for an 8 ppb reduction, controls are introduced in Metro East, Band I and the industrial sector in Band III; and the total cost is over five times as high as it is for a 6 ppb reduction of ozone.

Table 6-6. Comparison of the Costs of Weighted Trading and Traditional Trading for Different Levels of 8-Hour Ozone Reduction.

Region And Sector	2 ppb Traditional Trading	2 ppb Weighted Trading	4 ppb Traditional Trading	4 ppb Weighted Trading	6 ppb Traditional Trading	6 ppb Weighted Trading	8 ppb Traditional Trading	8 ppb Weighted Trading
	Costs in $million							
Metro East	**$1**	**$5**	**$6**	**$10**	**$22**	**$35**	**$90**	**$133**
Industrial	$0	$0	$0	$0	$0	$0	$5	$1
Utility	$1	$5	$6	$10	$22	$35	$85	$132
Northeast	**$0**	**$0**	**$1**	**$0**	**$5**	**$2**	**$28**	**$8**
Industrial	$0	$0	$0	$0	$0	$0	$6	$0
Utility	$0	$0	$1	$0	$5	$2	$22	$7
Band I	**$3**	**$4**	**$8**	**$10**	**$16**	**$16**	**$94**	**$112**
Industrial	$0	$0	$0	$0	$0	$0	$2	$2
Utility	$3	$4	$8	$10	$15	$16	$91	$110
Band II	**$11**	**$5**	**$19**	**$15**	**$34**	**$34**	**$203**	**$188**
Industrial	$0	$0	$0	$0	$1	$1	$3	$3
Utility	$11	$5	$19	$15	$33	$33	$200	$185
Band IIIe	**$5**	**$0**	**$12**	**$0**	**$35**	**$6**	**$187**	**$31**
Industrial	$0	$0	$0	$0	$2	$0	$20	$2
Utility	$5	$0	$12	$0	$33	$6	$168	$29
Illinois and Chicago	**$3**	**$0**	**$8**	**$0**	**$19**	**$3**	**$113**	**$18**
Industrial	$0	$0	$0	$0	$0	$0	$2	$0
Utility	$3	$0	$8	$0	$19	$3	$111	$18
Total	**$22**	**$13**	**$55**	**$35**	**$131**	**$96**	**$714**	**$489**
Industrial	**$0**	**$0**	**$0**	**$0**	**$4**	**$1**	**$38**	**$7**
Utility	**$22**	**$13**	**$54**	**$35**	**$128**	**$95**	**$676**	**$482**

The costs of achieving each additional 2 ppb reduction of ozone increase rapidly under weighted trading and traditional trading (see Figure 6-1). The incremental costs under weighted trading (in million dollars per year) are $13, $22, $61, and $393, respectively. Although the cost of reducing ozone by 8 ppb instead of 6 ppb is very large ($393 million per year), the marginal cost of going from 8 ppb to 8.3 ppb (see Table 6-5) is even larger (over $150 million per year, which corresponds to $1 billion/year if extrapolated to a 2 ppb reduction). The main conclusion is that the marginal costs of additional controls in the utility and industrial sectors increase dramatically if the corresponding reduction of ozone increases above 6 ppb. In fact, achieving an ozone reduction of approximately 8.5 ppb requires the use of all available controls in the utility and industrial sectors for all subregions. This means that at this high level of ozone reduction, weighted and traditional trading would be identical because there is no flexibility to make changes in controls. The next step in the analysis is to compare the cost-effectiveness of controls in other sectors with controls in the utility and industrial sectors.

Figure 6-1. Comparison of Regional Differences between Weighted Trading and Traditional Trading for Different Levels of Ozone Reduction.



INTERSECTORAL WEIGHTED TRADING

"Intersectoral weighted trading," trading of emissions is defined here to allow trading among all species of emissions and among all economic sectors and subregions, thereby allowing a much more flexible system than weighted trading NO_x among point sources only. This additional flexibility makes it possible to make substantial reductions in the cost of meeting environmental objectives. The sectors included in the analysis and examples of their emission sources are listed in Table 6-7.

Table 6-7. Emission Sources Included for Intersectoral Weighted Trading.

Emission Sector	Types of Processes/Activities That Are Controlled	Types of Emissions
Industrial	▸ Combustion boilers ▸ Production processes	▸ Elevated NO_x ▸ Ground-Level NO_x ▸ VOCs
Institutional/Commercial	▸ Production processes	▸ Ground-Level NO_x ▸ VOCs
Residential	▸ Small combustion engines	▸ Ground-Level NO_x ▸ VOCs
Transportation	▸ Combustion engines	▸ Ground-Level NO_x ▸ VOCs
Utility	▸ Electricity generation	▸ Elevated NO_x ▸ Ground-Level NO_x

The goal set for intersectoral weighted trading is to minimize the cost of reducing the concentration of 8-hour ozone by 8.3 ppb in Metro East by drawing upon control programs for ground-level and elevated NO_x and VOC. This is the same environmental objective set for weighted trading (and traditional trading) among point sources in the previous subsection of the report. It provides the opportunity to determine whether reductions of emissions in sectors other than the utility and industrial sectors are more cost effective.

Overall Results

The results for intersectoral weighted trading are summarized in Table 6-8. The total cost in meeting an 8.3 ppb reduction of ozone concentration in Metro East is only \$180 million per year, compared to \$650 million and \$830 million, respectively, for weighted and traditional trading for point sources of NO_x only. Many expensive controls in the utility and industrial sectors are replaced under intersectoral weighted trading by controls on ground-level emissions, particularly in the transportation sector. In fact, over half of the total cost of intersectoral weighted trading is attributed to controls on transportation in Metro East (mainly LEV programs). The traditional and weighted trading systems for point sources required emission reductions of 700,000 and 650,000 tons of NO_x per summer, respectively. The intersectoral weighted trading system, on the other hand, met the same reduction in ozone by reducing total emissions by less than 400,000 tons per summer, but most of these reductions (over 300,000 tons) still occur in the utility sector.

Table 6-8. Results for Intersectoral Weighted Trading to Reduce Ozone by 8.3 ppb.

Region and Sector Type	Emissions Reduced (1,000 tons/summer) NO$_x$	VOC	Total	Average Cost ($/summer ton)	Marginal Cost ($/summer ton)	8-Hour Ozone Reduction (ppb)	Average Cost per ppb O$_3$ Reduced ($million)	Annualized Total Cost ($million)
Metro East	**122**	**11**	**133**	**$964**	**$2,067**	**4.42**	**28**	**124**
Industrial	0	1	1	$236	$713	0.00	38	0
Inst/Comm	8	0	8	$873	$1,800	0.33	20	7
Residential	6	9	15	$364	$713	0.25	19	5
Transportation	44	1	45	$1,907	$2,067	1.90	45	84
Utility	64	0	64	$449	$2,067	1.94	15	29
Northeast	**6**	**0**	**6**	**$239**	**$446**	**0.04**	**36**	**1**
Industrial	0	0	0	$161	$161	0.00	46	0
Utility	6	0	6	$239	$446	0.04	36	1
Band I	**68**	**0**	**68**	**$266**	**$1,300**	**1.57**	**12**	**18**
Industrial	1	0	1	$428	$800	0.01	19	0
Inst/Comm	2	0	2	$713	$713	0.05	25	1
Residential	0	0	0	$598	$713	0.00	21	0
Transportation	2	0	2	$513	$1,200	0.07	18	1
Utility	63	0	63	$244	$1,300	1.44	11	15
Band II	**163**	**0**	**163**	**$190**	**$644**	**2.23**	**14**	**31**
Industrial	0	0	0	$345	$600	0.00	25	0
Utility	163	0	163	$190	$644	2.23	14	31
Band III	**20**	**0**	**20**	**$89**	**$100**	**0.04**	**41**	**2**
Utility	20	0	20	$89	$100	0.04	41	2
Total	**379**	**11**	**390**	**$457**	**$2,067**	**8.30**	**21**	**177**
Industrial	**1**	**1**	**2**	**$321**	**$800**	**0.02**	**24**	**0**
Inst/Comm	**10**	**0**	**10**	**$845**	**$1,800**	**0.37**	**21**	**8**
Residential	**6**	**9**	**15**	**$364**	**$713**	**0.25**	**19**	**5**
Transport	**46**	**1**	**47**	**$1,839**	**$2,067**	**1.96**	**44**	**86**
Utility	**316**	**0**	**316**	**$247**	**$2,067**	**5.70**	**14**	**78**

The overall conclusion is that most of the emission reductions and most of the ozone reduction in Metro East are still attributed to controls in the utility sector but the total cost is much lower when controls in all sectors are considered.

The average cost of controls ($/ton) and the average cost of reducing ozone ($/ppb) in the utility sector are among the lowest in Table 6-8. Nevertheless, the analysis in the subsection on weighted trading demonstrates that additional controls on utilities have very high marginal costs (i.e., going from 6 ppb to 8 ppb of ozone in Figure 6-1). Note that the ozone reduction attributed to utility controls is only 5.7 ppb in Table 6-8. Consequently, controls in other sectors are more cost-effective, including controls in the residential and the institutional and commercial sectors which are not considered at all under command and control regulation. Over 90% of the non-utility reduction of emissions occurs in Metro East. In contrast, the biggest reduction of emissions in the utility sector occurs in Band II outside the OTR.

Comparing Intersectoral Weighted Trading and Weighted Trading for Point Sources

The results in Table 6-5 for weighted trading identify the most cost-effective controls on point sources of NO_x in the utility and industrial sectors to reduce 8-hour ozone in Metro East by 8.3 ppb. The results in Table 6-8 identify the most cost-effective controls for all sources of emissions to meet the same reduction of ozone. The corresponding emission reductions and ozone reductions are compared in Table 6-9.

Total emission reductions (and the associated ozone reductions) under intersectoral weighted trading are higher in Metro East and lower in all other subregions. In particular, emission reductions in Band III fall by over 200,000 tons to less than 10% of the level under weighted trading for point sources only. Emission reductions for utilities in all subregions, including Metro East, are lower under intersectoral weighted trading. All increases of emission reductions under intersectoral weighted trading are associated, as expected, with ground-level sources of NO_x and VOC. All reductions of VOC occur in Metro East and most of these are in the residential sector (e.g., controls on two-stroke engines). Reductions of NO_x in transportation from LEV programs in Metro East are the biggest new source of reductions, and these reductions account for almost half of the total cost of all controls in all regions.

The four most important types of controls under intersectoral weighted trading, in terms of reducing ozone in Metro East, are putting controls on NO_x from utilities in Metro East, Band I, and Band II and controls on transportation in Metro East. This illustrates the major difference between ground-level emissions, which have a relatively local effect on ozone, and elevated emissions of NO_x from utilities, which can affect ozone in Metro East from sources in Band II outside the OTR as well as from sources in Metro East and Band I.

Comparing Intersectoral Weighted Trading and Command and Control

The results in the previous subsection show that it is relatively inexpensive to meet an 8-hour ozone reduction of 8.3 ppb using controls on all sources of emissions under intersectoral weighted trading. The next step in the analysis is to determine the effectiveness of intersectoral weighted trading in meeting a

Table 6-9. Comparison of Weighted Trading for Point Sources and Intersectoral Weighted Trading to Reduce Ozone by 8.3 ppb.						
	Emission Reductions (1,000 tons/summer)			8-Hour Ozone Reductions (ppb)		
Subregion and Sector	Weighted Trading for Point Sources	Intersectoral Weighted Trading	Net Change	Weighted Trading for Point Sources	Intersectoral Weighted Trading	Net Change
Metro East	**90**	**133**	**43**	**2.7**	**4.4**	**-0.1**
Industrial	0	1	1	0.0	0.0	0.0
Inst/Comm	0	8	8	0.0	0.3	0.3
Residential	0	15	15	0.0	0.3	0.3
Transportation	0	45	45	0.0	1.9	0.0
Utility	90	64	-26	2.7	1.9	-0.7
Northeast	**19**	**6**	**-13**	**0.1**	**0.1**	**-0.1**
Industrial	2	0	-2	0.0	0.0	-0.0
Utility	17	6	-11	0.1	0.1	-0.1
Band I	**85**	**68**	**-17**	**2.0**	**1.6**	**-0.4**
Industrial	1	1	-0	0.0	0.0	-0.0
Inst/Comm	0	2	2	0.0	0.1	0.1
Residential	0	0	0	0.0	0.0	0.0
Transportation	0	2	2	0.0	0.1	0.1
Utility	84	63	-21	1.9	1.4	-0.5
Band II	**219**	**163**	**-56**	**3.0**	**2.2**	**-0.8**
Industrial	1	0	-1	0.0	0.0	-0.0
Utility	218	163	-54	3.0	2.2	-0.7
Band IIIe	**141**	**20**	**-121**	**0.3**	**0.0**	**-0.3**
Industrial	3	0	-3	0.0	0.0	-0.0
Utility	138	20	-118	0.3	0.0	-0.3
Illinois and Chicago	**92**	**0**	**-92**	**0.2**	**0.0**	**-0.2**
Industrial	1	0	-0	0.0	0.0	0.0
Utility	92	0	-92	0.2	0.0	-0.2
Total	**646**	**390**	**-256**	**8.3**	**8.3**	**0.0**
Industrial	7	2	-5	0.1	0.0	-0.1
Inst/Comm	0	10	10	0.0	0.4	0.4
Residential	0	15	15	0.0	0.3	0.3
Transportation	0	47	47	0.0	2.0	2.0
Utility	639	316	-323	8.2	5.7	-2.5

larger reduction of 12.5 ppb of 8-hour ozone, which corresponds to the reduction in Table 6-2 using the command and control regulation. The results for intersectoral weighted trading are summarized in Table 6-10. The most important result is that the total cost of $780 million per year in the five subregions is 60% less than the cost under command and control, saving well over $1 billion per year in cost.

Nevertheless, comparing the results in Tables 6-10 and 6-8 for different reductions of ozone, the total cost for a 12.5 ppb reduction in ozone is over four times as high as the total cost for an 8.3 ppb reduction

Table 6-10. Results of Intersectoral Weighted Trading to Reduce Ozone by 12.5 ppb.

Region and Sector Type	Emissions Reduced (1,000 tons/summer) NO_x	VOC	Total	Average Cost ($/summer ton)	Marginal Cost ($/summer ton)	8-Hour Ozone Reduction (ppb)	Average Cost per ppb O_3 Reduced ($million)	Annualized Total Cost ($million)
Metro East	**169**	**56**	**225**	**$1,800**	**$13,078**	**6.61**	**$61**	**$406**
Industrial	0	1	1	$543	$3,400	0.01	$87	$1
Inst/Comm	17	0	17	$2,341	$5,300	0.76	$54	$41
Residential	10	22	32	$1,286	$3,400	0.59	$72	$42
Transportation	57	33	90	$2,191	$13,078	2.68	$74	$197
Utility	85	0	85	$1,489	$12,815	2.59	$49	$126
Northeast	**29**	**0**	**29**	**$1,441**	**$2,882**	**0.21**	**$196**	**$41**
Industrial	1	0	1	$366	$911	0.00	$105	$0
Inst/Comm	1	0	1	$761	$2,330	0.00	$77	$0
Residential	1	0	1	$713	$713	0.01	$72	$1
Transportation	12	0	12	$2,790	$2,882	0.12	$286	$34
Utility	14	0	14	$415	$2,684	0.08	$77	$6
Band I	**103**	**0**	**103**	**$1,288**	**$8,620**	**2.47**	**$54**	**$132**
Industrial	1	0	1	$1,491	$5,682	0.02	$67	$2
Inst/Comm	4	0	4	$2,258	$5,300	0.11	$78	$9
Residential	0	0	0	$2,008	$3,400	0.00	$69	$0
Transportation	16	0	16	$1,847	$4,000	0.46	$64	$29
Utility	82	0	82	$1,132	$8,620	1.88	$50	$93
Band II	**207**	**0**	**207**	**$751**	**$4,103**	**2.83**	**$55**	**$156**
Industrial	1	0	1	$2,158	$3,925	0.02	$158	$3
Utility	206	0	206	$742	$4,103	2.81	$54	$153
Band IIIe	**121**	**0**	**121**	**$237**	**$658**	**0.27**	**$108**	**$29**
Industrial	1	0	1	$362	$600	0.00	$164	$0
Utility	120	0	120	$236	$658	0.26	$107	$28
Illinois and Chicago	**68**	**0**	**68**	**$253**	**$643**	**0.15**	**$115**	**$17**
Industrial	0	0	0	$477	$600	0.00	$217	$0
Utility	68	0	68	$252	$643	0.15	$115	$17
Total	**697**	**56**	**753**	**$1,038**	**$13,078**	**12.54**	**$62**	**$782**
Industrial	**4**	**1**	**5**	**$1,061**	**$5,682**	**0.05**	**$109**	**$6**
Inst/Comm	**22**	**0**	**22**	**$2,293**	**$5,300**	**0.87**	**$57**	**$50**
Residential	**11**	**22**	**33**	**$1,275**	**$3,400**	**0.59**	**$72**	**$43**
Transportation	**85**	**33**	**118**	**$2,207**	**$13,078**	**3.25**	**$80**	**$260**
Utility	**575**	**0**	**575**	**$736**	**$12,815**	**7.77**	**$55**	**$423**

in ozone (the reduction of ozone is only one and a half times bigger in comparison). In other words, the marginal costs of reducing ozone are increasing rapidly.

At the higher level of ozone reduction in Table 6-10, utilities account for over half of the total cost and transportation for about one-third of the total cost. About 75% of the total reduction of emissions come from controls in utilities and only 15% from controls in transportation, but on a per ton basis the controls on transportation are more effective for reducing ozone.

Comparing the results from different subregions in Table 6-10, over half of the total cost is concentrated in Metro East. The next highest cost is in Band II (20%), followed by Band I (15%) and Northeast (5%). The costs in Metro East are mainly for controls in transportation (50%) and in utilities (30%). Although the cost of the controls in other sectors in Metro East is small (20% of the total), these reductions are relatively important compared to the same sectors in other subregions. The corresponding cost shares in Band I are 20% and 70% for transportation and utilities, respectively. In Bands II and III, controls on utilities account for most costs, and in Northeast, the transportation sector accounts for over 80% of the costs and the utility sector for only 15%.

Looking at emission reductions by sector, over 55% of the total reductions for utilities occur in Bands II and III, outside the OTR. In contrast, over three-quarters of the reductions in transportation, residential, and institutional/commercial sectors occur in Metro East, which is the receptor region for ozone. This illustrates the important differences in the effects on ozone formation of elevated emissions from utilities and ground-level emissions from transportation.

The levels of emission reductions and the associated ozone reductions under intersectoral weighted trading and command and control are compared in Table 6-11. Total emission reductions are over 15% lower under intersectoral weighted trading. The percentage difference for the utility sector is similar in magnitude. In contrast, the percentage difference in the transportation sector is much larger (45%), but this smaller reduction of emissions is partly offset by larger emission reductions in other sectors (residential and institutional/commercial). The total emission reduction in each subregion is lower under intersectoral weighted trading, particularly in Band III, the subregion farthest away from Metro East, and in Northeast, the subregion largely downwind from Metro East. The differences in total emission reductions in Metro East, Band I and Band II are relatively small in comparison.

Even though the levels of total emission reductions under intersectoral weighted trading are lower in Metro East and Band I, the corresponding levels of ozone reduction are larger. This shows that controls are adjusted under intersectoral weighted trading to be more effective in physical terms as well as more efficient in economic terms. Effectiveness improves by putting more emphasis on controls that reduce NO_x instead of VOC, because the emission weight for VOC is small compared to the weights for NO_x.

Table 6-11. Comparison of Command and Control and Intersectoral Weighted Trading to Reduce Ozone by 12.5 ppb.						
	Emission Reductions (1,000 tons/summer)			8-Hour Ozone Reductions (ppb)		
Subregion and Sector	Command and Control	Intersectoral Weighted Trading	Net Change	Command and Control	Intersectoral Weighted Trading	Net Change
Metro East	**237**	**225**	**-12**	**6.3**	**6.6**	**0.4**
Industrial	0	1	1	0.0	0.0	0.0
Inst/Comm	0	17	17	0.0	0.7	0.8
Residential	0	32	33	0.0	0.6	0.6
Transportation	147	90	-57	3.6	2.7	-0.9
Utility	90	85	-5	2.7	2.6	-0.1
Northeast	**53**	**29**	**-24**	**0.3**	**0.2**	**-0.1**
Industrial	2	1	-1	0.0	0.0	-0.0
Inst/Comm	0	1	1	0.0	0.0	0.0
Residential	0	1	1	0.0	0.0	0.0
Transportation	31	12	-19	0.2	0.1	-0.1
Utility	20	14	-6	0.1	0.1	-0.0
Band I	**119**	**103**	**-16**	**2.3**	**2.5**	**0.1**
Industrial	1	1	0	0.0	0.0	0.0
Inst/Comm	0	4	4	0.0	0.1	0.1
Residential	0	0	0	0.0	0.0	0.0
Transportation	35	16	-20	0.4	0.5	-0.0
Utility	83	82	-1	1.9	1.9	-0.0
Band II	**219**	**207**	**-12**	**3.0**	**2.8**	**-0.2**
Industrial	1	1	0	0.0	0.0	-0.0
Utility	218	206	-12	3.0	2.8	-0.2
Band IIIe	**174**	**121**	**-53**	**0.4**	**0.3**	**-0.1**
Industrial	4	1	-3	0.0	0.0	-0.0
Utility	170	120	-50	0.4	0.3	-0.1
Illinois and Chicago	**112**	**68**	**-45**	**0.2**	**0.1**	**-0.1**
Industrial	1	0	-1	0.0	0.0	0.0
Utility	111	68	-44	0.2	0.1	**-0.1**
Total	**914**	**753**	**-162**	**12.5**	**12.5**	**0.0**
Industrial	9	6	-4	0.1	0.1	**-0.0**
Inst/Comm	0	22	22	0.0	0.9	0.9
Residential	0	33	33	0.0	0.6	0.6
Transportation	214	118	-96	4.2	3.3	-1.0
Utility	691	575	-116	8.3	7.8	-0.5

LEAST COST STRATEGIES FOR REDUCING OZONE

The use of intersectoral weighted trading determines the least cost pattern of controls for all sectors and subregions for any given target level of ozone reduction (up to the maximum level when all control options are used). The costs of the least cost strategies for different levels of ozone reduction are summarized in

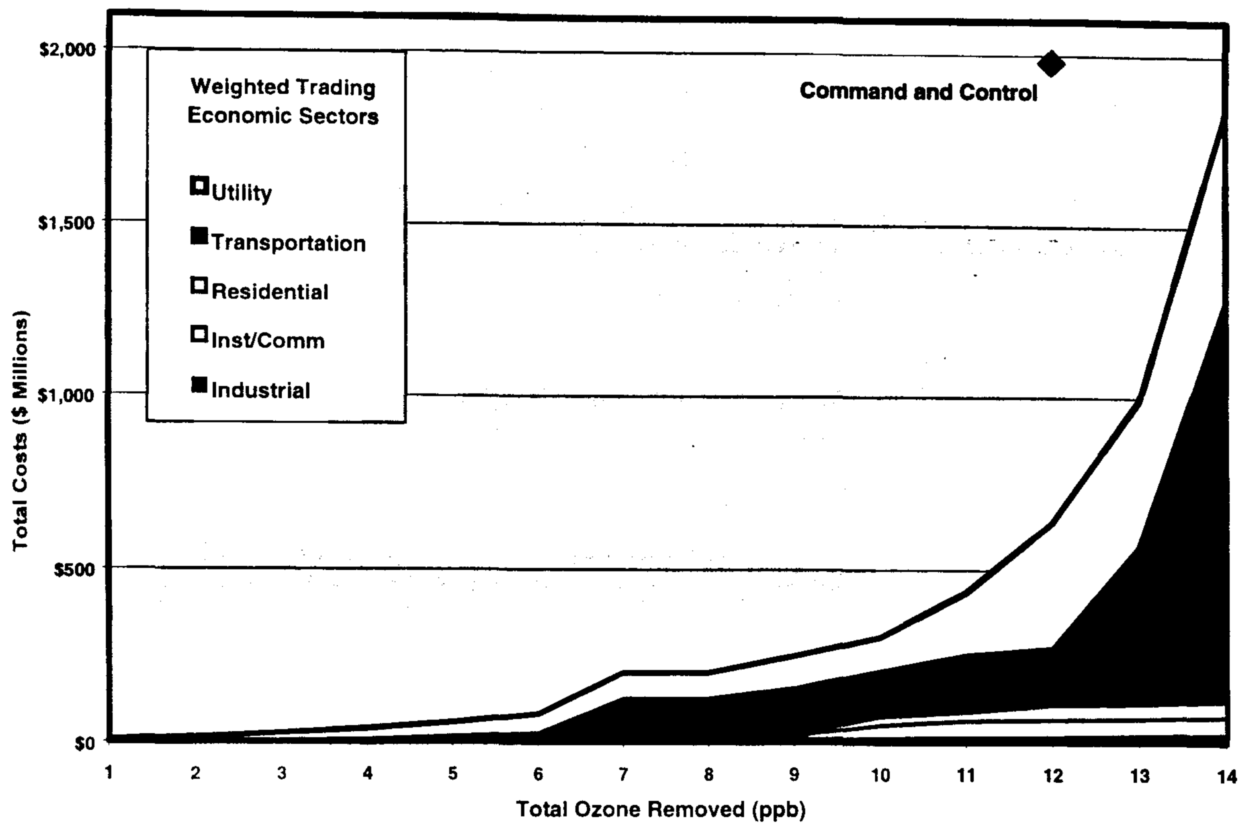
Table 6-12 and Figure 6-2. Comparing the costs by sector and subregion as the levels of ozone reduction increase shows in which sectors and subregions the most cost-effective controls are found.

Table 6-12. The Cost of Intersectoral Weighted Trading for Different Levels of 8-Hour Ozone Reduction (million dollars per year).

Subregion and Sector	2 ppb	4 ppb	6 ppb	8 ppb	10 ppb	12 ppb	14 ppb	16 ppb	18 ppb
Metro East	**$5**	**$11**	**$33**	**$112**	**$199**	**$337**	**$1,320**	**$2,058**	**$5,502**
Industrial	$0	$0	$0	$0	$0	$1	$1	$5	$19
Inst/Comm	$0	$0	$5	$7	$37	$41	$41	$41	$41
Residential	$0	$0	$5	$5	$25	$42	$43	$43	$69
Transportation	$0	$1	$6	$72	$101	$134	$1,070	$1,767	$5,151
Utility	$5	$10	$18	$28	$36	$120	$166	$202	$222
Northeast	**$0**	**$0**	**$0**	**$1**	**$3**	**$6**	**$53**	**$59**	**$1,227**
Industrial	$0	$0	$0	$0	$0	$0	$2	$2	$8
Inst/Comm	$0	$0	$0	$0	$0	$0	$2	$2	$2
Residential	$0	$0	$0	$0	$0	$0	$4	$4	$4
Transportation	$0	$0	$0	$0	$0	$0	$35	$35	$1,174
Utility	$0	$0	$0	$1	$2	$5	$11	$16	$39
Band I	**$4**	**$10**	**$17**	**$18**	**$46**	**$132**	**$178**	**$219**	**$1,157**
Industrial	$0	$0	$0	$0	$0	$2	$2	$2	$15
Inst/Comm	$0	$0	$1	$1	$1	$9	$9	$9	$9
Residential	$0	$0	$0	$0	$0	$0	$0	$0	$17
Transportation	$0	$0	$1	$1	$29	$29	$57	$98	$1,000
Utility	$4	$10	$15	$15	$16	$92	$111	$111	$116
Band II	**$5**	**$14**	**$25**	**$31**	**$34**	**$106**	**$210**	**$216**	**$226**
Industrial	$0	$0	$0	$0	$1	$1	$3	$3	$3
Utility	$5	$14	$25	$31	$33	$105	$207	$213	$223
Band IIe	**$0**	**$0**	**$0**	**$2**	**$7**	**$28**	**$38**	**$59**	**$211**
Industrial	$0	$0	$0	$0	$0	$0	$2	$2	$20
Utility	$0	$0	$0	$2	$7	$27	$36	$57	$191
Illinois and Chicago	**$0**	**$0**	**$0**	**$0**	**$4**	**$16**	**$24**	**$52**	**$154**
Industrial	$0	$0	$0	$0	$0	$0	$0	$0	$10
Utility	$0	$0	$0	$0	$4	$16	$23	$52	$144
Total	**$13**	**$35**	**$74**	**$164**	**$293**	**$625**	**$1,823**	**$2,663**	**$8,476**
Industrial	$0	$0	$0	$0	$1	$4	$9	$14	$76
Inst/Comm	$0	$0	$6	$8	$38	$50	$52	$52	$52
Residential	$0	$0	$5	$5	$25	$43	$46	$46	$90
Transportation	$0	$1	$7	$73	$130	$163	$1,162	$1,900	$7,325
Utility	$13	$34	$57	$78	$98	$366	$555	$651	$934

At the lowest ozone reduction of 2 ppb, most controls are on utilities in Metro East, Band I, and Band II, with no controls on transportation at all. (Minor programs in the residential sector are also introduced.) At 4 ppb, the total cost increases by roughly $20 million per year, and controls on utilities in Metro East, Band I, and Band II still dominate this total. The first minor controls on transportation in Metro East are

Figure 6-2. The Cumulative Costs of Efficient Strategies by Sector for Different Levels of 8-Hour Ozone Reduction.



also made. At 6 ppb, the total cost increases by $40 million per year and the relative importance of utilities begins to decline, with controls on transportation and other sources of ground-level emissions making up a quarter of the total cost. At 8 ppb, the total cost increases by $90 million per year and the importance of transportation in Metro East increases substantially to over 40 % of the total cost. In addition, the first controls on utilities in Band IIIe occur. At 10 ppb, controls are expanded in the residential and the commercial and institutional sectors along with more controls on transportation in Metro East and Band I. Relatively few additional controls are put on utilities, and the total cost increases by $130 million per year. To reach the next step of 12 ppb, most additional controls are put on utilities in Metro East, Band I, and Band II. Additional controls on transportation dominate the final three steps for 14 ppb, 16 ppb, and 18 ppb. The total cost rises by $330 million, $1.2 billion, and over $5 billion for these three steps. For the largest reduction of 18 ppb of ozone, most of the available controls in all sectors and regions are used.

An interesting feature of the results in Table 6-12 is that the total cost approximately doubles for each additional 2 ppb of ozone reduction until a level of 16 ppb is reached ($20, $40, . . . , $2560 million). The final step at 18 ppb is over three times the total cost of a reduction of 16 ppb. The overall implication is that the marginal cost of reducing ozone by more than 12 ppb (i.e., to levels higher than the 12.5 ppb reached under command and control in Table 6-2) is prohibitively high.

SUMMARY

The integration of the emission weights and control costs can lead to much more economically efficient strategies for meeting specific environmental objectives compared to the command and control approach to controlling emissions [Section 6.2]. Traditional trading of NO_x among point sources gave modest cost reductions of about 11% [Section 6.3] over the command and control scenario. The benefits of trading were limited by the stringent standards for rates of emissions that were required under the command and control strategy. With less stringent standards, there would be more opportunities to adjust the combinations of controls among point sources, and as a result, the economic benefits of trading would be greater.

The total cost of controls can be reduced more substantially from command and control (30%) if weighted trading of NO_x among point sources is used [Section 6.4]. Under this system, the same environmental goal for reducing ozone in Metro East was met using a smaller total reduction of emissions. This was accomplished by adjusting the spatial pattern of emissions to be more cost-effective by cutting back on controls in regions with low emission weights for NO_x.

If the concept of weighted trading is extended to cover emissions from all economic sectors, additional savings in costs can be achieved [Section 6.5]. Under intersectoral weighted trading, the total cost of controls to meet the same environmental goal as command and control was reduced by 60% to $780 million per year for the five subregions, saving over $1 billion per year. Cost-effective controls on utilities were located primarily in Metro East and Band I, inside the OTR, and in Band II, outside the OTR. The most cost-effective controls on ground level emissions were LEV programs for transportation and controls in the residential and commercial/institutional sectors in Metro East. Hence, controls on ground level emissions in the least cost strategy were concentrated in the selected receptor region for ozone. In contrast, the controls on utility and industrial sources of elevated NO_x extended over a much larger region, and the largest single source of emission reductions was made in the utility sector in Band II, outside the OTR. The conclusion is that controls on relatively distant sources of elevated NO_x in the Midwest are cost-effective in reducing ozone concentrations along the northeastern seaboard.

A simple rule-of-thumb for costs is that the minimum total cost of controls for reducing 8-hour ozone in Metro East doubles for every additional 2 ppb reduction from the initial $20 million per year to attain a 2 ppb reduction of ozone. Reducing ozone by 2 ppb beyond the 12.5 ppb reached under command and control (Table 6-11) would cost well over $1 billion more per year, and the total cost of this least cost strategy would be similar in magnitude to the total cost of the command and control strategy in Table 6-2. In other words, the economically efficient strategy that costs the same as the command and control strategy would reduce 8-hour ozone in Metro East by only another 2 ppb. Most of the additional costs of reducing 8-hour ozone by more than 12.5 ppb occur in the transportation sector, but the extremely high marginal cost of achieving these reductions in ozone implies that the additional controls are not very cost-effective. It would be sensible to evaluate other types of policies for transportation that are not included in this analysis. For example, an investment in improving the infrastructure of public transportation could be a better approach. The objective would be to reduce ground-level emissions, particularly in Metro East, by cutting vehicle miles traveled for commuting and transferring some freight from roads to rail. The controls considered in the analysis, which depend mainly on engine modification and the use of improved fuels, have relatively little effect on miles traveled.

Section 7
POLICY IMPLICATIONS

The analysis conducted for this study compares the cost-effectiveness of different emission control technologies and regulatory systems to achieve emission reductions and lower concentrations of ozone in Metro East. The results of the analysis are discussed here to elucidate the implications and to make recommendations for consideration by policy makers. To that end, this chapter reviews several issues: spatial and species effects of ozone formation, cost-effectiveness of emission control technologies, implications of VOC offsets for economic growth, and regulatory structures to achieve environmental objectives.

SPATIAL AND SPECIES EFFECTS

Through application of the UAM-V photochemical model, a series of emission weights were estimated that identify the marginal contribution of a ton of emissions from each source subregion on ozone formation in Metro East and the statistical uncertainty associated with this relationship. The results provide an important extension to OTAG photochemical modeling analysis through the development of emission weights as a measure of the contribution to ozone formation and the evaluation of species effects. The policy implications of these results are that ozone reductions in Metro East will require a mix of emission reductions that will have both regional and local components.

Large NO_x point sources from Band II (Ohio, West Virginia, and Virginia) have emission weights that are almost half as big as local NO_x point sources for the 1-hour standard and almost two-thirds of local NO_x point sources for the 8-hour standard. Extending the receptor region farther west to include nonattainment areas such as Pittsburgh would further increase the relative importance of Band II emissions for addressing the regional ozone nonattainment problem in the northeast. Since the ozone problem increases in spatial scale when the 8-hour standard is considered, the need for large-scale regional emission reductions increases as well.

The effects of ground-level emissions on ozone formation in Metro East vary significantly between NO_x and VOC (species effect) and by source subregion. Ground-level NO_x from Metro East has the largest emission weight for both the 1-hour and 8-hour ozone standard compared to all emission sources. However, the relative magnitude of this effect compared to NO_x point sources from Band II declines from a factor of 4 to 1 for the 1-hour standard to a factor of 3 to 1 for the 8-hour standard. Of greater importance for ozone policy are the relatively small emission weights for VOC emissions. VOC emissions in Metro East have emission weights four and eight times smaller for the 1-hour and 8-hour standards, respectively, compared to the weights for ground-level NO_x from Metro East. In fact, the weights for elevated NO_x emissions in

Band II for the 8-hour standard is over twice as large as the weight for VOC in Metro East. In addition, VOC emission weights for all other source subregions are either very small or zero. This suggests that NO_x emissions are generally the limiting factor in ozone formation over Metro East under the typical meteorological conditions for an ozone episode

COST-EFFECTIVENESS OF EMISSION CONTROL PROGRAMS

The multisector analysis of emission control technologies considers a broad range of emission reduction programs across the residential, industrial, utility, and transportation sectors. The emission programs include emission controls under current consideration by policy makers in the OTR and emission reduction programs adopted or proposed in California.

Since an objective of this report is to provide an analysis for policy-makers on the least-cost steps for reducing ozone concentrations in Metro East, a ranking of emission control programs in terms of cost-effectiveness is provided in Table 7-1. These results are a consolidation of the hundreds of individual emission control programs with different marginal control costs, especially for NO_x point sources. The rank ordering of emission control programs based on average removal costs provides a basis for comparing different emission control strategies. The results for the utility and industrial sectors correspond to the combination of controls adopted under intersectoral weighted trading in Table 6-10. (These costs are lower than they are under command and control in Table 6-2 because SCR is not installed.)

Transportation Controls

The transportation control programs consist of traditional emission reduction measures such as an LEV program and enhanced reformulated gasoline as well as unconventional measures such as behavioral control programs to reduce vehicle use and encourage public transit, and controls on off-road emissions from construction equipment. Since the transportation sector comprises approximately 35% of the VOC and NO_x emissions inventory, it is an important source of potential emission reductions. Transportation control measures that reduce NO_x emissions tend to be more cost-effective than programs that focus on reducing VOC emissions. The results presented in Section 6 also provide a basis for evaluating the cost-effectiveness of specific transportation controls measures, such as hybrid electric vehicles.

Table 7-1. Rank Ordering of the Cost-Effectiveness of Emission Control Programs.					
Source Subregion	Economic Sector	Program Description	$Million/ ppb of O_3 Reduced	$Thousand/ ton of emissions	Ozone Removed (ppb)
Band I	Transportation	Mobile Construction Equipment NO_x	20	0.6	0.07
Metro East	Transportation	Mobile Construction Equipment NO_x	22	1.0	0.24
Metro East	Inst/Comm/Res	Non_Utility NO_x	48	2.0	1.20
Metro East	Utility	Utility NO_x	49	1.5	2.59
Band I	Utility	Utility NO_x	50	1.1	1.88
Band II	Utility	Utility NO_x	54	0.7	2.81
Band I	Industrial	Industrial NO_x	67	1.5	.02
Band I	Inst/Comm/Res	Non_Utility NO_x	72	2.1	0.11
Northeast	Utility	Utility NO_x	77	0.4	.08
Metro East	Industrial	Industrial NO_x	87	0.5	.01
Metro East	Transportation	LEV	107	1.5	2.18
Band III	Utility	Utility NO_x	107	0.2	0.26
Band II	Industrial	Industrial NO_x	158	2.2	.02
Metro East	Transportation	Natural Gas Program Trucks	220	8.6	0.20
Metro East	Transportation	HDDV Truck/Bus Program	301	13.1	0.77
Metro East	Residential	Small Engine VOCs	393	2.4	0.15
Band I	Transportation	HDDV Truck/Bus Program	419	12.2	0.05
Metro East	Transportation	RGas	446	12.5	1.38
Northeast	Transportation	LEV	633	2.2	0.13
Northeast	Transportation	HDDV Truck/Bus Program	1,135	11.2	0.08
Northeast	Transportation	RGas	1,887	10.7	0.05
Metro East	Transportation	Lower summer public transit fairs	2,372	27.5	0.05
Metro East	Transportation	HOV lanes	2,453	28.0	0.03

<u>LEV.</u> The LEV program reduces both VOC and NO_x emissions from vehicles in nearly equal portions. The cost-effectiveness of this program is reduced in source subregions other than Metro East, because the VOC emissions have a small impact on ozone formation in Metro East. Nonetheless, the magnitude of the emission weights for ground-level NO_x emissions and VOC in the Northeast subregion make LEV a cost-effective control program. The costs in $/ppb of ozone removed for the LEV in Metro East and Northeast is $100 million/ppb and $600 million/ppb, respectively.

Application of the LEV program to areas that are not immediately upwind of the nonattainment areas or have a zero VOC emission weight may be of marginal benefit. However, the regional scope of projected 8-

hour nonattainment areas suggests that application of the LEV program on a regional basis is a cost-effective means of improving air quality.

Reformulated Gasoline. Although RGas is a summer emission control program, the cost-effectiveness of this program is less than that of the LEV program because most reductions for RGas are heavily weighted toward reducing VOC emissions. Nonetheless, RGas is a cost-effective source of emission reductions from Metro East at $450 million/ppb of ozone removed.

Natural Gas Incentive Program for Trucks. The natural gas incentive program for trucks was first piloted in Connecticut to provide incentives for the conversion of delivery trucks and other diesel trucks to natural gas. This program reduces NO_x emissions and provides an ancillary benefit of reduced particulate emissions. In the densely populated Metro East, this program is estimated to be cost-effective at $220 million/ppb of ozone removed.

Diesel Trucks. Diesel truck emission programs as proposed by CARB reduce both NO_x and particulate emissions. Although the removal costs of NO_x emissions are about $12,000 per ton, this program is relatively cost-effective to source subregions with large ground-level NO_x emissions weights. Consequently, this program is cost-effective for Metro East and Band I, at removal costs of $301 million/ppb and $420 million/ppb, respectively. The program is only marginally cost-effective when applied to the Northeast, at a cost of over $1 billion/ppb.

Hybrid Electric Vehicles. Recently, attention has been focused on the use of hybrid vehicles to reduce emissions in metropolitan areas. Hybrid vehicles significantly reduce VOC and NO_x emissions by using an electric vehicle drive system with a gasoline motor to charge the batteries. For hybrid vehicles to be cost-effective relative to other transportation and nontransportation control programs, the cost of emissions reductions in Metro East times the emission weights for Metro East should be less than $650 million/ppb. This roughly corresponds to a marginal control of $20,000 per ton for the three month summer ozone season.

Public Transportation. Public transit has the potential to significantly reduce vehicle miles traveled in the densely populated Metro East subregion. This is distinct from elsewhere in the country where distances traveled are greater and the public transit infrastructure is relatively poor. With investment in upgrading the public transit infrastructure, the attractiveness of substituting driving for a comfortable and efficient mass transit system increases. In addition, the recent shift in the last few years to large sport utility vehicles with relatively high emission rates compared to automobiles (about 1.5 to 2 times larger) increases the air quality benefits of reducing vehicle miles traveled.

NO_x Point Source Controls

Control Level. Control of NO_x point sources principally affects the utility sector, with about 10% of NO_x emissions coming from the industrial, commercial, and residential sectors. Utility controls provide a significant amount of ozone benefit because of the amount of emission reductions available and the relative cost-effectiveness of these controls. Utility emission controls in Metro East, Band I and Band II are among the most cost effective ways to reduce ozone concentrations in Metro East at a cost of roughly $50 million/ppb. These three sources of controls all have relatively large effects on reducing ozone compared to other control programs (LEV in Metro East is the only other individual program which has a major effect on ozone). It should also be noted that the average cost of controls gets smaller as the distance from the receptor region increases (e.g. the cost is $1500/ton in Metro East compared to $700/ton in Band III). The cost effectiveness and the ozone reduced are much lower for controls on utilities in Northeast and Band III. Controls in the industrial sector on point sources have a very small impact on ozone in Metro East, and are also relatively expensive in terms of $/ppb of ozone reduced.

Impact of Deregulation. Deregulation of the electric generation industry may shift the spatial geographic patterns of electric generation emissions. Recognizing that the lower cost of electric generation resides in Band I and II and from sources with relatively high emission rates, this could have an adverse effect on air quality in Metro East. Assuming that 2,000 MW can be transmitted from Band II sources to Metro East during the summer ozone season, this will increase emissions by approximately 36 tons/day, or an increase in ozone formation of about 0.045 ppb/day.[1] At the same time, however, plants in Metro East will generate less electricity and thereby produce fewer emissions. A reduction of 36 tons of emission in Metro East would translate into a 0.072 ppb/day reduction in ozone formation. The net effect of the shift in generation would be roughly 0.027 ppb/day reduction in ozone concentrations in Metro East. If the emission rates from Band II are higher than .15 lb./MBtu, then the effects on ozone concentrations in Metro East would be even smaller.

IMPLICATIONS OF VOC OFFSETS

The results of the emission weights analysis described in Section 4 indicate that VOC emissions from rural areas have a negligible impact on ozone formation in any of the receptor regions analyzed in this study. The photochemical modeling reveals that the weights for VOCs are either not statistically different from zero or are relatively small. This results suggests that the OTR could benefit by petitioning the EPA to receive a

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1. The calculation for this estimate is as follows: (2000 MW/hr)*(24 hr/day)*(10 MBtu/MW)*(0.15 lb/MBtu) = 72,000 lb/day or 36 tons/day. (36 tons/day)*(1.24 ppb/thousand tons of emissions) = 0.0447 ppb/day.

waiver for the requirement to provide VOC offsets for emission sources in all source subregions with perhaps the exception of the Inner Zone (Metro East). Removal of the VOC offset requirement for new industrial sources would create a more business-friendly environment since these offsets can represent a significant cost to new businesses. The current market price of VOC offsets is over $2,500 per ton (Cantor/Fitzgerald, 1998).

REGULATORY STRUCTURES

Command and Control

Command and control regulation holds the advantage of providing a fixed emission standard across every emission source and is the most common form of regulation. Command and control regulation can be cost-effective across a narrowly defined source category with relatively homogenous costs, such as small two-stroke engines. However, command and control regulation can be relatively costly compared to trading strategies when there is variability in control costs among emission sources, such as utility boilers. Thus, command and control regulation is appropriate for controls on small engines, RGas, and LEV. Including these sources in a weighted trading program, however, determines the economically efficient pattern of controls among sectors, and determines which control programs should be implemented to meet a specific reduction of ozone in Metro East.

Traditional Trading

Traditional trading, as developed by EPA to reduce SO_2 emissions, minimizes the cost of reducing emissions by a given amount. It allows sources with high control costs to purchase emission reductions from sources with low control costs. The results in Section 6 indicate that traditional emissions trading can achieve the same level of emission reductions from point sources of NO_X as command and control regulation for about 12% less cost using a 0.15 lb/MBtu NO_x standard on boilers. This saving more than doubles at a 0.20 lb/Mbtu standard. The cost savings are relatively small from trading at the more stringent standard because most control options have to be implemented. The increase in savings at the less stringent standard comes from avoiding the high marginal cost of emission reductions from SNCR to SCR. Using the stringent standard, there are more cost-effective controls available in the transportation sector than the emission reductions from installing SCR.

The disadvantage of traditional trading is the inability to incorporate the spatial and species effects of ozone formation. Traditional trading works best when the spatial and species effects of all sources are relatively constant in a region. For large NO_x point sources, where there is a significant degree of variability in control costs among sources, traditional trading can provide significant cost advantages over command and control regulation. However, it is difficult to implement traditional emissions trading across large regions with heterogeneous effects on air quality and ensure that air quality objectives will be maintained. For example, emission reductions in Northeast provide considerably less ozone reduction in Metro East than the same level of reduction from Band I or Metro East.

Weighted Emissions Trading

Weighted trading is an extension of traditional emissions trading that combines the cost of controls and the effectiveness of controls on ozone formation. The emission weights act as exchange rates for emissions between the different source subregions and enable emissions to be traded in common units of ozone removed in the receptor region. This provides a substantial cost advantage over traditional trading for reaching an air quality objective when the effectiveness of controls varies among sources, as it does with ozone formation. It also provides a basis for increasing the spatial scale of controls to large regions, and to allow for emissions trading among species (e.g. ground-level and elevated NO_x sources and VOC). This increased spatial and species scale is important to address the lack of liquidity that exists in typical offset programs, for example. The results suggest that weighted emissions trading can reach the same reduction of ozone in Metro East as the command and control strategy at about 40 percent of the total cost ($800 million/year compared to $2 billion/year under command and control).

Appendix A
DESCRIPTION OF REGRESSION TECHNIQUE

The first 15 scenarios (including the EPA Base 1c) in the series of photochemical model UAM-V runs focus exclusively on emissions reductions from elevated NOx2 from point sources in different subregions. This initial sample was extended by a series of 9 scenarios involving changes of point and mobile sources, and a second series of 14 scenarios involving changes of point, area, and mobile sources. The total size of the final sample for estimating a MORS is 38, which is over 50% larger than the planned size in the original research proposal. These additional modeling simulations improve the accuracy of the emission weights.

The main statistical problem with the larger sample was that the relationships between ozone concentrations and ground-level emissions were not nearly as well-determined as the relationships to elevated NOx2 (the unexplained variability in the MORS is larger with the full sample of 38 observations than it is with the initial sample of 15 observations). Since this situation required special attention, we adopted a weighted regression technique to deal with it.

In a standard regression specification, the unexplained variability (residual variance) is assumed to be the same for each observation, and, consequently, each observation is given the same weight in the estimation. However, this assumption does not hold for our data, and application of standard regression analysis [i.e., ordinary least squares (OLS)] will result in a biased estimate of variance for each parameter implying the t-test and F-tests and standard error estimates are invalid. Recognizing that the residuals are not uniformly distributed, we apply a variation of generalized least squares (VGLS) estimation to produce unbiased estimators with a smaller variance-covariance matrix (i.e., a more efficient estimator).

Instead of minimizing the sum of squared residuals as in OLS estimation, we minimize an appropriately weighted sum of squared residuals. Observations that are expected to have large residuals because the variance of their associated disturbances is known to be large are given smaller weight. Observations whose residuals are expected to be large because other residuals are large are given smaller weight. The VGLS procedure thus produces a more efficient estimator by minimizing a weighted sum of squared residuals. The weighted sum of squared residuals is determined by emission category in VGLS instead of by the variance-covariance matrix as in standard GLS.

It is appropriate to weight emissions by category given the way the data are generated in the full sample of 38 observations for estimating a MORS. The weighting is applied to specify that the unexplained variability is different in the three different blocks of observations (NOx2 only, ground level NO_x and VOC, and all sources). The first step is to estimate a least squares fit to the data and compute the unexplained residuals. The three appropriate subsets of the computed residuals are then used to estimate three different residual variances. In the second step, the inverse of the square root of the corresponding residual variance is used to weight each observation in a weighted regression. In other words, an observation associated with a large residual variance is given a small weight in the regression. This is equivalent to giving the weighted observations the same amount of unexplained variability, and giving more importance in the estimation process to observations that can be predicted accurately. The residuals for the first 15 data points (elevated NO_x) are given their full weight and the residuals for the next 14 data points (ground-level VOC and NO_x) are weighted at a factor of 0.8. The last nine data points that combine reductions of ground-level and elevated sources are given a weight of 0.6.

Appendix B

NO_x POINT SOURCE EMISSION CONTROL COST ALGORITHMS

FEDERAL EMISSION CONTROLS

The Clean Air Act Amendments (CAAA) of 1990 have served as a guide to the latest revisions of the control algorithms for nitrogen oxides (NO_x) used as part of the Regional Economic Model for Ozone Compliance (REMOC) emission control. In particular, the provisions of Title IV pertaining to acid rain and of Title I pertaining to ambient air quality standards will affect electric utility emissions of NO_x. The emission cost algorithms in REMOC were originally developed for the New York Power Pool for research supported by the New York State Department of Public Service.

Title IV of the CAAA regarding the control of sulfur dioxide (SO_2) and NO_x emissions responsible for acid rain apply specifically to electric utility boilers. The major focus of Title IV is a market-based trading system of SO_2 allowances that will be used to reduce SO_2 emissions by 10 million tons from 1980 emission levels. Title IV also includes provisions regarding utility emissions of NO_x; however, these provisions are not as extensive as those for SO_2. Title IV divides the utility boiler population into two groups: Phase I boilers are a specified list of large boilers with high SO_2 emissions that are affected by the regulations in 1995; Phase II boilers are the remaining boiler population and are subject to regulation in 2000. Only coal-fired boilers are affected by the Title IV NO_x regulations. For Phase I boilers, the NO_x limits are 0.50 lb/MMBtu for dry-bottom wall-fired boilers and 0.45 lb/MMBtu for dry-bottom tangentially fired boilers. For Phase II boilers, the NO_x limits for dry-bottom wall-fired and tangentially fired boilers are reduced to 0.45 lb/MMBtu and 0.38 lb/MMBtu, respectively. The NO_x limits for other types of coal-fired boilers included in this analysis are 0.94 lb/MMBtu for cyclone boilers, 0.86 lb/MMBtu for wet-bottom boilers, and 0.29 lb/MMBtu for fluidized-bed boilers.

Title I of the Clean Air Act Amendments of 1990, pertaining to ambient air quality standards, does not directly address electric utility emissions. Although the Title I provisions do not provide specific limits for NO_x emissions, utilities are generally required to install control technologies equivalent to Reasonably Available Control Technology (RACT).

The definition of RACT varies by state, with most states requiring utility boilers to meet emissions limits similar to or slightly more stringent than the Phase II limits of Title IV in 1995. In Pennsylvania, RACT is defined as a technology, specifically low-NO_x burners coupled with overfire air. For states with limited non-attainment areas, RACT provisions may be imposed only on specific utility boilers. In the OTR, more stringent standards are to be imposed in 1999 and 2003. These standards are 55% to 65% reductions or

0.20 lb/MMBtu for 1999 and up to 75% reduction or 0.15 lb/MMBtu for 2003, depending on the location of the boiler.

NO_x CONTROLS

The costs of NO_x controls are calculated using algorithms and cost data adapted from the U.S. Environmental Protection Agency's (U.S. EPA) Integrated Air Pollution Control System (IAPCS), data from other U.S. EPA publications, reports published by the Electric Power Research Institute (EPRI), conference proceedings, and published magazine articles (see references at the end of this appendix). The NO_x control technologies considered are low excess air (LEA), overfire air (OFA), low-NO_x burners (LNB), low-NO_x burners with overfire air/tangential (LNB/OFA-T), selective noncatalytic reduction (SNCR), and selective catalytic reduction (SCR). Most of the combustion modification technologies (OFA, LNB, LNB/OFA-T) can be combined with the post-combustion technologies (SNCR, SCR) to provide greater NO_x reductions at lower overall cost. In addition, OFA and LNB (LNB + OFA) can be combined as well. Capital and O&M costs for each control technology are calculated separately and are added together for the combined technologies.

LOW EXCESS AIR

Low excess air controls NO_x emissions by reducing the amount of oxygen available to combine with nitrogen in the combustion air or fuel to form NO_x. LEA is assumed to provide a 15% reduction in NO_x emissions. For coal-fired boilers, the capital costs of LEA are calculated using the following equation:

$$\text{Capital Cost} = [0.170 * (\text{Heat Rate}) * \text{MW}] + 18{,}847,$$

and O&M costs are determined using:

$$\text{O\&M Cost} = [0.0031 * (\text{Heat Rate}) * \text{MW}] + 349.4.$$

For oil- and gas-fired boilers, the capital cost for LEA is:

$$\text{Capital Cost} = [0.1057 * (\text{Heat Rate}) * \text{MW}] + 15{,}698,$$

and the O&M cost is:

$$\text{O\&M Cost} = [0.00196 * (\text{Heat Rate}) * \text{MW}] + 291.0,$$

where:

MW	=	Plant capacity in MW
Heat Rate	=	Plant heat rate in Btu/kWh.

Low excess air provides the added benefit of improving the efficiency of the boiler. This benefit is accounted for by lowering the plant heat rate. LEA is assumed to improve plant heat rate by 1.5%.

OVERFIRE AIR

Overfire air reduces NO_x emissions by staging the combustion process in the boiler. Combustion air is restricted in the lower portion of the boiler where the burners are located. This restriction lowers NO_x emissions by limiting the amount of oxygen available to form NO_x in a manner similar to LEA. Overfire air is injected above the burners to complete the combustion process. The excess level of oxygen in the overfire air region lowers the peak flame temperature of the combustion process, the primary variable influencing thermal (non-fuel) NO_x formation. OFA is assumed to provide a 20% reduction in NO_x emissions. For coal-fired boilers, OFA is assumed to result in a 1.0% reduction in boiler efficiency. Capital and O&M costs for OFA are:

$$\text{Capital Cost} = 5{,}835.6 * \text{MW}$$

$$\text{O\&M Cost} = 108.18 * \text{MW}.$$

LOW-NO_x BURNERS

Low-NO_x burners reduce NO_x emissions in a manner similar to OFA; however, the combustion process is staged within the burner flame rather than the entire furnace. LNB is applicable to wall and opposed boilers with either dry or wet bottoms. For dry-bottom boilers, capital costs are calculated using the following equation:

$$\text{Capital Cost} = 83{,}223 * (\text{MW})^{0.648}.$$

For retrofit installations on dry-bottom boilers, there are no additional O&M costs; however, for new installations on dry-bottom boilers, O&M costs are determined using the following equation:

$$\text{O\&M Cost} = 1{,}142.8 * (\text{MW})^{0.648}.$$

Low-NO_x burners are assumed to provide a 45% reduction in NO_x emissions on dry-bottom boilers. For wet-bottom boilers, LNB are assumed to provide NO_x reductions of 25%. Capital costs for these installations are calculated as follows:

$$\text{Capital Cost} = 98{,}828 * (\text{MW})^{0.648}.$$

For retrofit installations on wet-bottom boilers, there are no additional O&M costs; however, for new installations on wet-bottom boilers, O&M costs are determined using the following equation:

$$\text{O\&M Cost} = 1{,}357.1 * (\text{MW})^{0.648}.$$

LOW-NO_x BURNERS WITH OVERFIRE AIR/TANGENTIAL (LNB/OFA-T)

Low-NO_x burners with overfire air/tangential were designed as a low-cost retrofit control technology for tangentially fired boilers. There are three types of LNB/OFA-T, known as Levels 1, 2, and 3. LNB/OFA-T Level 1 uses close-coupled overfire air immediately above the burner assembly to reduce NO_x emissions by staging the combustion process in the entire furnace. LNB/OFA-T Level 2 uses offset air within the burner assembly and separated overfire air above the burner assembly to stage the combustion process in a manner similar to LNB used with OFA. LNB/OFA-T Level 3 combines the close-coupled overfire air of LNB/OFA-T Level 1 with the offset and separated overfire air of LNB/OFA-T Level 2. LNB/OFA-T Level 3 is the only LNB/OFA-T implemented in the CCMU model. LNB/OFA-T Level 3 is assumed to provide a NO_x reduction of 45%. Capital costs for the LNB/OFA-T Level 3 system are calculated using the following equation for retrofit installations:

$$\text{Capital Cost} = 417.43 * \text{MW}^{-0.49} * (\text{MW} * 1{,}000).$$

For new LNB/OFA-T Level 3 installations, the capital costs are calculated using the following equation:

$$\text{Capital Cost} = 321.10 * \text{MW}^{-0.49} * (\text{MW} * 1{,}000).$$

No O&M costs are assumed for retrofit installations of LNB/OFA-T Level 3. For new LNB/OFA-T Level 3 installations, O&M costs are determined using the following equation:

$$\text{O\&M Cost} = 4.409 * \text{MW}^{-0.49} * (\text{MW} * 1{,}000).$$

SELECTIVE NONCATALYTIC REDUCTION

In the SNCR process, chemically enhanced urea is injected into the boiler within a specified temperature zone. The urea reacts with NO_x in the flue gas to form nitrogen and water. SNCR is assumed to be capable of 50% to 70% NO_x removal for boilers less than 300 MW and 50% to 60% removal for boilers 300 MW and larger. The difference in removal rates is due to the difficulty of providing adequate chemical coverage and shorter residence times in the proper temperature range in the larger boilers. Capital costs for SNCR are calculated using the following equation:

$$\text{Capital Cost} = 159{,}912.0 * \text{MW}^{0.514}.$$

O&M costs for SNCR are determined for both nonchemical O&M costs and chemical usage costs before being combined to determine the total O&M costs. Nonchemical O&M costs for SNCR are calculated using the following equation:

$$\text{Nonchemical O\&M} = 24{,}162.7 * MW^{0.514}.$$

Costs for the urea reagent depend on the NO_x concentration after any combustion modifications and the degree of NO_x removal required. These parameters are used to determine the normalized stoichiometric ratio (NSR) for SNCR. The NSR is calculated using the following equation:

$$NSR = 2.562 * (\%\ NO_x\ \text{removal}),$$

where:

% Removal	=	Percent NO_x removal required.

Once the NSR has been calculated, the cost of reagent for SNCR is determined using the following equation:

$$\text{Reagent Cost} = 0.002857 * NSR * (NO_x\ \text{conc.}) * (\text{heat rate}) * MW * CF * (\text{urea cost}),$$

where:

NSR	=	Normalized stoichiometric ratio
NO_x Conc.	=	Inlet NO_x concentration in lb/MMBtu
Heat Rate	=	Plant heat rate in Btu/kWh
MW	=	Plant capacity in MW
CF	=	Plant capacity factor
Urea Cost	=	Cost of urea in $/ton.

SELECTIVE CATALYTIC REDUCTION

Selective catalytic reduction is similar to SNCR in that a reagent is used to reduce NO_x in the flue gas to nitrogen and water; however, in the SCR process, the reagent is ammonia and a catalyst bed is used to allow the reaction to occur in a lower temperature range than the SNCR process. SCR units can be installed before the air heater (hot-side) or after the air heater and any SO_2 or particulate matter controls (cold-side). Hot-side SCR installations have problems associated with catalyst poisoning, masking, and erosion due to the particulate matter in the flue gas. In addition, SO_2 in the flue gas can react to form SO_3 and combine with the ammonia to form ammonium bisulfate, a sticky solid that can plug the air heater. Cold-side SCR installations generally avoid these problems because of their location after any control technologies. This allows cold-side SCR installations to have smaller catalyst beds than hot-side SCR units. However, cold-side SCR installations require the flue gas to be reheated to the proper temperature range, generally resulting in higher leveled costs than hot-side SCR installations. A hot-side SCR installation is assumed in the CCMU model.

First, various parameters necessary to calculate the costs of the SCR system are calculated. The ratio of ammonia to NO_x varies by the fuel type based on a critical NO_x inlet concentration. For NO_x emissions below the critical NO_x inlet concentration, the NH_3/NO_x molar ratio is equal to the required NO_x removal rate plus 0.020. For NO_x emissions greater than or equal to the critical NO_x inlet concentration, the NH_3/NO_x molar ratio is equal to the required NO_x removal rate plus 0.013. The critical NO_x inlet concentration varies by fuel type as follows:

Fuel	Critical NO_x Inlet Concentration
Coal	1.091 lb/MMBtu
Oil	1.008 lb/MMBtu
Gas	0.955 lb/MMBtu.

The ammonia injection rate can then be determined using the following equation:

$$NH_3 \text{ Injection} = 0.0003702 * (NH_3/NO_x) * MW * (\text{heat rate}) * (NO_x \text{ conc.}),$$

where:

NH_3/NO_x	=	NH_3/NO_x molar ratio
MW	=	Plant capacity in MW
Heat Rate	=	Plant heat rate in Btu/kWh
NO_x Conc.	=	Inlet NO_x concentration into the SCR unit in lb/MMBtu.

Next, the catalyst volume is calculated. Required catalyst volume varies depending on the plant capacity factor, catalyst life, number of catalyst layers, required NO_x removal rate, and flue gas flow rate. Plant capacity factor and catalyst life are used to determine an adjustment factor for overall catalyst life using the following equation:

$$f(CF) = (0.15 * CF * 8{,}760) / [(\text{cat. life}) * 8{,}000],$$

where:

CF	=	Plant capacity factor
Cat. Life	=	Catalyst life in years.

For coal-fired units, catalyst life is assumed to be four years. Catalyst life for oil-fired units is assumed to be six years, and catalyst life of eight years is assumed for gas-fired units. The adjustment for the number of capacity layers uses the results of this function as follows:

$$k / k_0 = \left(\frac{1}{\text{Layers}}\right) \sum_{n=1}^{\text{Layers}} \left\{[1 - f(CF)]^{[(\text{Cat. Life})(n) - 1]}\right\},$$

where:

Layers	=	Number of catalyst layers
Cat. Life	=	Catalyst life in years.

The space velocity for the SCR unit, a measure of the volumetric flow rate of flue gas through the SCR unit per cubic foot of catalyst, is determined using the following equation:

$$SV = 3{,}000 * (k/ko) * \{\ln(1 - 0.80) / \ln[1 - (\%\ \text{removal})]\},$$

where:

k/ko	=	Adjustment for number of catalyst layers
% Removal	=	Percent NO_x removal required.

The catalyst volume required by the SCR unit can then be calculated using the following equation:

$$CV = [60 * (\text{flue gas})] / SV,$$

where:

Flue Gas	=	Flue gas flow rate in scfm
SV	=	Space velocity in 1/hr.

Material costs for the SCR unit are calculated for each equipment section and summed to provide the total direct capital cost. For the ammonia handling equipment, material costs are determined using the following equation:

$$NH_3\ \text{Handling} = 26{,}743.0 * (\text{retrofit factor}) * (NH_3\ \text{inj.})^{0.639},$$

where:

Retrofit Factor	=	1.00 for new installations
	=	1.35 for retrofit installations
NH_3 Inj.	=	Ammonia injection rate in lb/hr.

For the reactor and initial catalyst loading, the material costs are calculated using the following equation:

$$\text{Reactor \& Cat.} = \{1.472 * (\text{retrofit factor}) * [145{,}822 * (\#\ \text{reactors}) + 316.9 * CV]\} + [CV * (\text{cat. cost})],$$

where:

Retrofit Factor	=	1.00 for new installations
	=	1.35 for retrofit installations
# Reactors	=	Number of reactors
	=	2 if capacity < 750 MW
	=	4 if capacity ≥ 750 MW
CV	=	Catalyst volume in ft^3
Cat. Cost	=	Catalyst cost in $\$/ft^3$.

The material cost for the flue gas handling equipment is calculated using the following equation:

$$\text{Flue Gas Handling} = 301.48 * (\text{retrofit factor}) * (\text{flue gas})^{0.694},$$

where:

Retrofit Factor	=	1.00 for new installations
	=	1.35 for retrofit installations
Flue Gas	=	Flue gas flow rate in acfm.

For the process control system, the material costs are determined using the following equation:

$$\text{Controls} = 113{,}893 * (\#\ \text{reactors}),$$

where:

# Reactors	=	Number of reactors
	=	2 if capacity < 750 MW
	=	4 if capacity ≥ 750 MW.

For the air preheater modifications, the material costs are given by the following equation:

$$\text{Air Preheater Mod.} = 23.51 * (\text{retrofit factor}) * (\text{flue gas})^{0.772},$$

where:

Retrofit Factor	=	1.00 for new installations
	=	1.35 for retrofit installations
Flue Gas	=	Flue gas flow rate in acfm.

The total direct capital costs are the sum of the material costs for ammonia handling, the reactor and initial catalyst loading, flue gas handling, the control system, and the air preheater modifications. Indirect capital costs are then added to the direct capital costs. General facilities costs are assumed to be 10% of the direct capital costs, as are the engineering fees, process contingency factor, and project contingency factor. These four costs are added to the total direct capital costs to calculate the total plant cost. Royalties are assumed to be 0.5% of the total plant cost, and preproduction costs are assumed to be 2.0% of the total plant cost plus one month of O&M costs. These two items are then added to the total plant cost to give the total capital cost.

O&M costs for the SCR unit are divided into fixed and variable components. The first fixed O&M cost item calculated is operating labor, which is determined using the following equation:

$$\text{Op. Labor} = (CF / 0.628) * [1{,}341 + (5.363 * MW)] * (\text{labor } \$),$$

where:

CF	=	Plant capacity factor
MW	=	Plant capacity in MW
Labor $	=	Labor cost in $/hr.

The next fixed O&M cost is analysis and landfill labor, which is calculated using the following equation:

$$\text{A\&LF Labor} = (CF / 0.628) * [1{,}671 + (1.185 * MW)] * (\text{labor } \$),$$

where:

CF	=	Plant capacity factor
MW	=	Plant capacity in MW
Labor $	=	Labor cost in $/hr.

Maintenance labor and materials are assumed to be 4% of the total plant cost calculated in the capital costs. These costs are divided as 40% labor and 60% materials. Overhead and administration costs are assumed to be 30% of all labor requirements (operating, analysis and landfill, and maintenance). Total fixed O&M costs are the sum of operating labor costs, analysis and landfill labor costs, maintenance labor and materials costs, and overhead and administration costs.

Variable O&M costs include the cost of all consumables used by the SCR unit as well as the cost of catalyst replacement and disposal. Reagent costs are calculated using the following equation:

$$\text{Reagent Cost} = (8{,}760 / 2{,}000) * (NH_3 \text{ inj.}) * CF * (NH_3 \text{ cost}),$$

where:

NH_3 Inj.	=	Ammonia injection rate in lb/hr
CF	=	Plant capacity factor
NH_3 Cost	=	Ammonia cost in $/ton.

Catalyst replacement costs are calculated using the following equation:

$$\text{Cat. Repl.} = \left(\frac{CV}{\text{Layers}}\right)(\text{Cat. Cost})\left\{\frac{i(1+i)^{(\text{Layers})(\text{Chg Rate})}}{(1+i)^{(\text{Layers})(\text{Chg Rate})}-1}\right\}\sum_{n=\text{Chg Rate}}^{(\text{Layers})(\text{Chg Rate})}\frac{1}{(1+i)^n},$$

where:

CV	=	Catalyst volume in ft^3
Layers	=	Number of catalyst layers
Cat. Cost	=	Catalyst cost in $\$/ft^3$
Disc. Rate	=	Discount rate (i)
Chg Rate	=	Rate of catalyst layer change in years.

The cost of catalyst disposal is determined using the following equation:

$$\text{Cat. Disp.} = [0.024 * (\text{cat. repl.}) * (\text{cat. disposal } \$)] / (\text{cat. cost}),$$

where:

Cat. Repl.	=	Cost of catalyst replacement from above
Cat. Disposal $	=	Cost of catalyst disposal in $/ton
Cat. Cost	=	Catalyst cost in $\$/ft^3$.

The cost of electricity consumption is calculated using the following equation:

$$\text{Elec.} = \{[5.801 * (\text{flue gas})] - 545{,}133\} * (\text{CF} / 0.628) * [(\text{elec. \$}) / 1{,}000],$$

where:

Flue gas	=	Flue gas flow rate in acfm
CF	=	Plant capacity factor
Elec. $	=	Electricity cost in mills/kWh.

The cost of steam consumption by the SCR unit is determined using the following equation:

$$\text{Steam} = 841.8926 * \{[33.29 * (NH_3 \text{ inj.}) * \text{CF}] - 14.91\} * [(\text{steam \$}) / 1{,}000],$$

where:

NH_3 Inj.	=	Ammonia injection rate in lb/hr
CF	=	Plant capacity factor
Steam $	=	Steam cost in $/1000 lb.

Total variable O&M costs are the sum of the reagent costs, catalyst replacement and disposal costs, electricity cost, and steam cost. Total O&M costs for the SCR unit are the sum of the total fixed O&M costs and the total variable O&M costs.

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Appendix C

MODELING ASSUMPTIONS AND SOURCES TO DETERMINE EMISSION REDUCTION COSTS

INTRODUCTION

The diversity of emission control programs evaluated makes it important to have a consistent framework to measure emission abatement costs. Emission abatement costs in different sectors are determined in terms of dollars per ton of emissions removed during the summer. This provides a logical basis for evaluating controls in different economic sectors on a basis consistent with the seasonal nature of tropospheric ozone. The cost figures will invariably be larger than calculations that measure the marginal emission reductions on an annual basis.

$$\text{Control Cost} = \frac{\text{NPV (Capital + Variable)}}{\text{NPV (Summer Tons Removed)}} \cdot \tag{C-1}$$

This method of measuring costs balances the seasonal and temporal benefits of a wide range of control programs and provides a consistent basis to evaluate their costs.

Table C-1. Economic Determinants Used in Emission Reduction Cost Calculations.

	Capital Cost Expenditure Present Value Factor	Emission Present Value Factor
Transportation	9%	4%
Recreational Vehicles	9%	4%
Off-Road Utility Engines	9%	4%
NO_x Point Sources	9%	4%

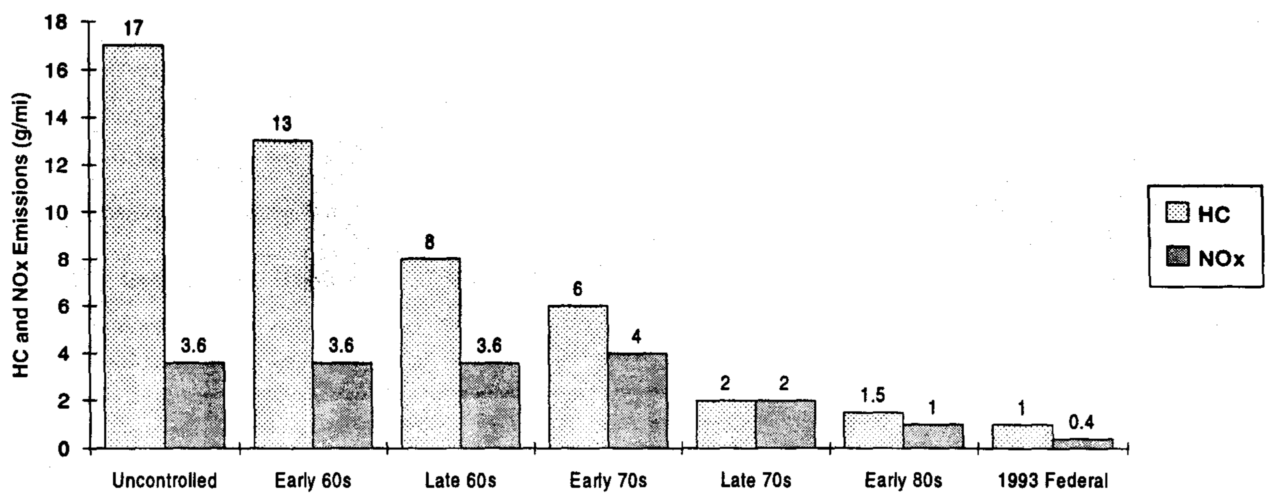
Total costs include capital and variable costs and, when applicable, the loss in consumer welfare from an increase in retail prices. All costs are discounted to account for the time value of expenditures.

The following sections discuss the calculation of costs of emission reductions used as inputs in the Regional Economic Model for Ozone Compliance (REMOC). The discussion focuses on the following emission control programs: (1) California low emissions vehicle (Cal LEV) program, (2) increasing vehicle fuel economy, (3) lawn and garden and utility engines, and (4) heavy-duty off-road and on-road diesel engines. The emission controls discussed provide insight into the full range of controls to be used in the economic analysis of emission control strategies.

BACKGROUND OF VEHICLE EMISSION CONTROLS

Standards for new vehicle exhaust emission rates have decreased dramatically over the last 30 years, as shown in Figure C-1. Uncontrolled hydrocarbon (HC) emissions from 1950s passenger cars consisted of exhaust emissions of about 8.7 grams per mile (g/mi), crankcase emissions of about 4 g/mi, evaporative emissions of about 4 g/mi, and refueling emissions of about 0.3 g/mi (for vehicle average fuel economy of 20 mpg) (Sierra Research, 1994). Emission controls began in the early 1960s with positive crankcase ventilation (PCV) systems, which reduced "blow by gases" at the intake manifold. The PCV system reduced HC emissions from 17 g/mi to 13 g/mi. HC emissions were further reduced to 8 g/mi by exhaust emission standards established in the late 1960s. Evaporative emissions were cut by 50% in the late 1970s, reducing total HC emissions to 6 g/mi. Exhaust gas recirculation (EGR) introduced in the late 1970s further reduced NO_x emissions to about 2 g/mi. Over the same period, HC emissions were reduced to 2 g/mi with stricter evaporative emissions controls. The introduction of catalysts in 1980 cut NO_x emissions to 1.5 g/mi and VOC to 1.0 g/mi (Sierra Research, 1994; Seinfeld, 1986).

Figure C-1. Evolution of Vehicle Emission Standards for Ozone Precursors.



Source: Sierra Research (1994).

Emissions from gasoline engines can be reduced through changes in engine design, combustion chamber conditions, catalytic after-treatment, and fuel composition. The advances in emission reduction technology for vehicles over the last 30 years have resulted in extremely efficient and reliable controls. An ordinary vehicle catalyst, once properly warmed up, reduces HC and CO emissions to virtually zero, and NO_x is reduced 90% or more (Weaver and Turner, 1993). So the preponderance of HC and CO emissions, typically 80%, are produced under cold-start conditions where the catalyst has not achieved its optimum operating temperature (Hellman et al., 1989).[1] Compliance with more stringent emission standards therefore focuses on reducing cold start emissions with technologies such as electrically preheated catalysts.

CALIFORNIA LOW EMISSIONS VEHICLE PROGRAM

Under Section 177 of the Clean Air Act, states retain the authority to enforce vehicle emission standards that differ from federal standards. However, these standards must be identical to the California standards and adopted at least 2 years before an affected model year. This legal provision established conditions for the Ozone Transport Commission (OTC) October 29, 1991, memorandum of understanding to adopt the Cal LEV program (OTC, 1994). The reasons the OTC adopted the LEV program include the following:

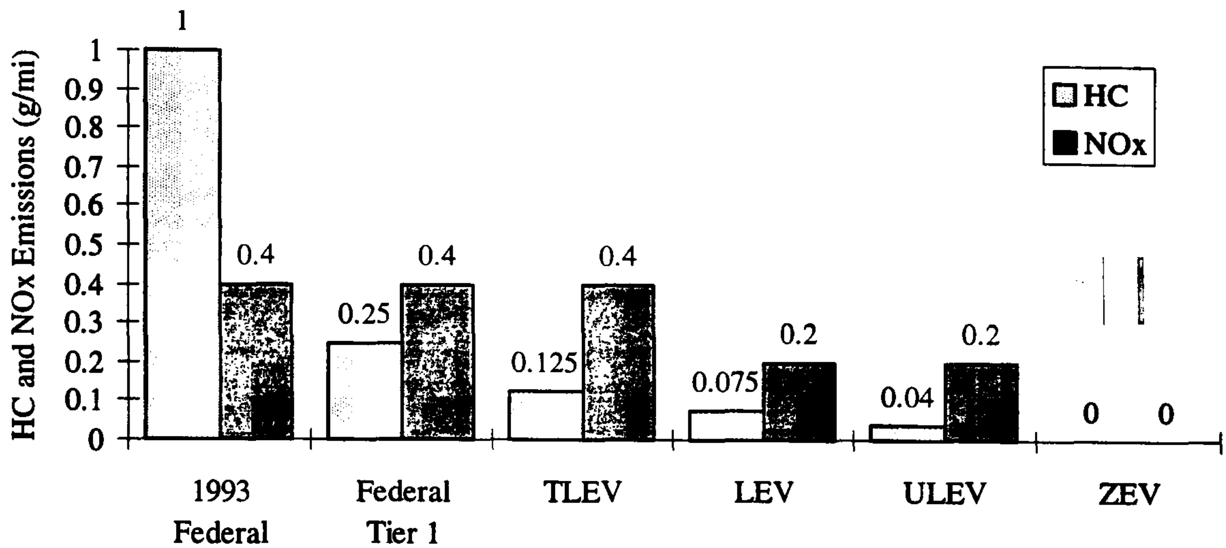
- A severe and widespread long-term ozone nonattainment problem still exists in the subregion.
- Motor vehicles continue to be the largest source of ozone precursor, CO, and some air toxic compounds.
- U.S. EPA supports the adoption of the California LEV program as an element of an attainment and maintenance strategy for the states in the ozone transport subregion.

As northeastern states take steps to implement the California LEV program, it is important to evaluate the cost-effectiveness of adopting the program.

Federal U.S. EPA and California Air Resources Board (CARB) emission standards are shown in Figure C-2 for passenger car and light duty truck (LDT1). The California emission standards are distinguished from the federal U.S. EPA standards not only in stringency but also in program requirements. CARB's program has been characterized by NESCAUM (1991) as "technology forcing approach . . . and is more flexible and responsive than the federal program." NESCAUM's characterizations of the California LEV program can be attributed to emissions averaging between vehicle emission categories and an emission banking and trading system. The averaging of emissions between the categories enables manufacturers to meet each year's NMOG standard, as shown in Table C-2.

1. Hellman's results show 80% of the HC and OC emissions produced with the federal test procedure by modern emission-controlled vehicles are produced during the first two minutes of the cold start, before the catalyst warms up.

Figure C-2. Comparison of Emission Standards of Federal Vehicles and Cal LEV Vehicles.



Source: Weaver & Turner.

Table C-2. Implementation Schedule for Cal LEV Program for Passenger Cars and Light Duty Trucks under 3,750 lb Test Weight.

Model Year	Fleet Avg. Standard NMOG (g/mi)	Fed Tier 0.25 g/mi % of Fleet	TLEV 0.125 g/mi % of Fleet	LEV 0.075 g/mi % of Fleet	ULEV 0.04 g/mi % of Fleet	ZEV 0.00 g/mi % of Fleet
1994	0.250	90%	10%			
1995	0.231	85%	15%			
1996	0.225	80%	20%			
1997	0.202	73%		25%	2%	
1998	0.157	48%		48%	2%	2%
1999	0.113	23%		73%	2%	2%
2000	0.073			96%	2%	2%
2001	0.070			90%	5%	5%
2002	0.068			85%	10%	5%
2003	0.062			75%	15%	10%
2004	0.062			75%	15%	10%
2005	0.062			75%	15%	10%
2006	0.062			75%	15%	10%
2007	0.062			75%	15%	10%

Note: The standard by vehicle type is for NMOG emissions in units of grams/mile (g/mi).
Source: Weaver and Turner (1993).

The average emission standard for a fleet is determined by multiplying the emission certification rate (shown in the header row of Table C-2) for each vehicle type (i.e., Fed. Tier 1, TLEV, LEV, ULEV, ZEV) by the percent of vehicles in the fleet. In practice, the regulations offer manufacturers the flexibility to shift vehicles between emission categories so long as the fleet average standard is met. In addition, the banking and trading program permits manufacturers to earn marketable credits for sale or purchase with other manufacturers as well as for use between years. Manufacturers must meet the ZEV percentage requirements, but can trade ZEV credits to achieve the requirement.[2]

EMISSION CONTROL TECHNOLOGY

Vehicle exhaust emissions are primarily controlled with an exhaust gas catalytic converter (Calvert J. et al., 1993). The catalytic converter achieves additional emission reductions beyond the design controls of the combustion chamber. The three-way catalyst increases the rate of reaction of unburned hydrocarbons and CO with the existing oxygen in the exhaust, while simultaneously promoting the reduction of NO to nitrogen and oxygen. Catalysts are more effective in oxidizing the highly reactive hydrocarbon species such as olefins and formaldehyde. The less reactive short-chain hydrocarbon species such as methane, ethane, and propane are difficult to oxidize and are more likely to constitute hydrocarbon exhaust emissions. Carter (1990) showed that the incremental ozone reactivities for short-chain hydrocarbon species are about 10 times less reactive than the long-chain hydrocarbon species for typical VOC/NO_x ratios in the Northeast. Thus, catalytic converters working at design conditions typically achieve substantial reductions in ozone forming potential.

Combustion chamber stoichiometry varies with vehicle operating conditions, with acceleration typically being fuel rich and steady operation nearly stiochiometric (i.e., complete combustion). The three-way catalyst can simultaneously oxidize HC and CO while reducing NO_x if combustion conditions are within a narrow operating range. This air-fuel ratio is maintained with an oxygen sensor and a computer controlled system continuously maintaining the desired air-fuel ratio.

The efficacy of the catalyst also depends on the type of fuels burned. Fuels high in sulfur, lead or phosphorus from engine oil mixing will form deposits in the catalytic converter, blocking the pores and reducing the reactive surface of the catalyst. Catalyst efficiency is also reduced by excessively high temperatures, which cause the metal crystals to sinter together, thereby reducing the reactive surface area.

EMISSION TEST PROCEDURES

Exhaust emission testing was designed in the early 1970s to reflect the highly transient nature of actual vehicle operations in an urban environment. The Federal Test Procedure (FTP), used by both U.S. EPA and

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2. Weaver and Turner (1993) present a detailed listing of both California and Federal emission standards.

CARB, measures exhaust emissions produced over a prescribed driving cycle on a chassis dynamometer. In the early 1970s, when the cycle was first developed, the limited capabilities of the chassis dynamometers at that time made it necessary to limit speeds and acceleration rates. With a top speed of 57 mph and an equally modest acceleration rate, dynamometer tests fall far short of the top speed and acceleration rates most vehicles experience. The test program does follow the "stop-and-go" pattern of urban driving, but with unrealistically slow acceleration rates.

The limited range of the FTP test program provides an opportunity for manufacturers to take advantage of the "off-cycle" conditions that are effectively uncontrolled. For example, the fuel rich mixture of near full throttle acceleration results in the shut-off of exhaust gas recirculation. This results in bypassing an integral part of the exhaust gas emission control system, which results in a substantial increase in NO_x and HC emissions. CARB performed a series of tests under different acceleration patterns in an attempt to characterize the potential impacts of by-passing the emission control system. Table C-3 summarizes CARB's test results of several passenger vehicles from 1988 through 1990 model years over 11 different driving cycles. The high level of hydrocarbon emissions during acceleration represents only a small fraction of a typical driving cycle, and the increase in average emissions will be smaller than shown in Table C-3. The CARB 1991 test result indicated that the existing test procedures used to assess vehicle emissions are likely to underestimate HC and CO emissions. Although provisions of the 1990 CAAA require reevaluation of the FTP cycle, it remains the basis for all light-duty vehicle emission standards listed in Table C-2.

Table C-3. Emissions Produced in Different Standard and High-Acceleration Test Driving Cycles.

Cycle	HC (g/mi)	CO (g/mi)	NO_x (g/mi)	Ozone (g/mi)
M1	0.73	3.90	0.61	2.71
M2	4.62	145.23	0.77	23.63
M3	1.87	62.31	1.34	9.77
M4	3.73	130.00	0.72	19.79
M5	3.56	140.31	2.25	19.76
M6	3.64	113.64	0.85	18.60
M7	1.87	73.80	0.75	10.39
M8	1.54	67.13	1.10	8.89
M9	4.40	164.40	1.76	23.91
FTP	0.33	2.69	0.44	1.27
NYCC	0.47	5.42	0.79	1.89
Source: CARB (1991).				

Evaporative emissions test procedures measure the evaporative emissions due to (simulated) diurnal heating and cooling, and evaporation from the carburetor under hot-shutdown conditions. The evaporative emissions test shares the same design deficiency as the exhaust emissions test of failing to exercise the evaporative control system under conditions as severe as actual use (CARB, 1990b). CARB (1990b) provides a comprehensive review of the deficiencies of the evaporative test procedure:

1. The present test procedure does not measure emissions venting from the tank during operation.

2. The present test procedure does not measure or account for the "puffs" of vapor emitted from pressurized tanks when the cap is removed for refueling.

3. The present procedure fails to measure resting losses due to permeation of hoses and plastic tanks, fuel weathering, and back-purge during cool-down.

DETERIORATION SCHEDULES

Deterioration schedules are annual incremental reductions in the effectiveness of an emission control device. The durability requirements of CARB and U.S. EPA require the emission control system to demonstrate durability through 50,000 miles of operation in an accelerated test cycle. To guard against emission increases due to defective emission controls, U.S. EPA and CARB randomly check vehicles to ensure compliance. This random check program forces manufacturers to design and build far more durable and effective emission control systems in order to avoid recalling vehicles. The calculation of the emission savings achieved through the Cal LEV program over the federal program assumes vehicles on average meet the 50,000 mile emission standard. However, most manufacturers feel that a design margin of at least 30%, and preferably more, is needed between the certified emission levels and the standard. The margin is necessary to provide a reasonable allowance for in-use deterioration.

CARB and U.S. EPA use different assumptions to determine vehicle emissions deterioration rates. The methodology used by CARB in its evaluation of the LEV program (CARB, 1990b) assumes that the emissions deterioration rate will occur as a constant percent of the zero mile emission rate up to 50,000 miles and then the deterioration rate is readjusted at a different but constant percent of the 50,000 mile emission rate. On the other hand, U.S. EPA believes all LDV have the same deterioration rates regardless of the initial level of emissions.[3] For example, CARB would determine that the deterioration factors for the TLEV will be 50% lower than for future federal vehicles since the emission standard drops from 0.25 g/mi to 0.125 g/mi (NESCAUM, 1991). The effect of the two different deterioration rates on vehicle certification standards is presented in Figures C-3 and C-4.

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3. U.S. EPA's vehicle emissions deterioration rates also follow the pattern of 0-50,000 miles and 50,000 miles and beyond of CARB, but with only one rate for all vehicles.

Figure C-3. Cal LEV Standards under U.S. EPA Deterioration Rates.

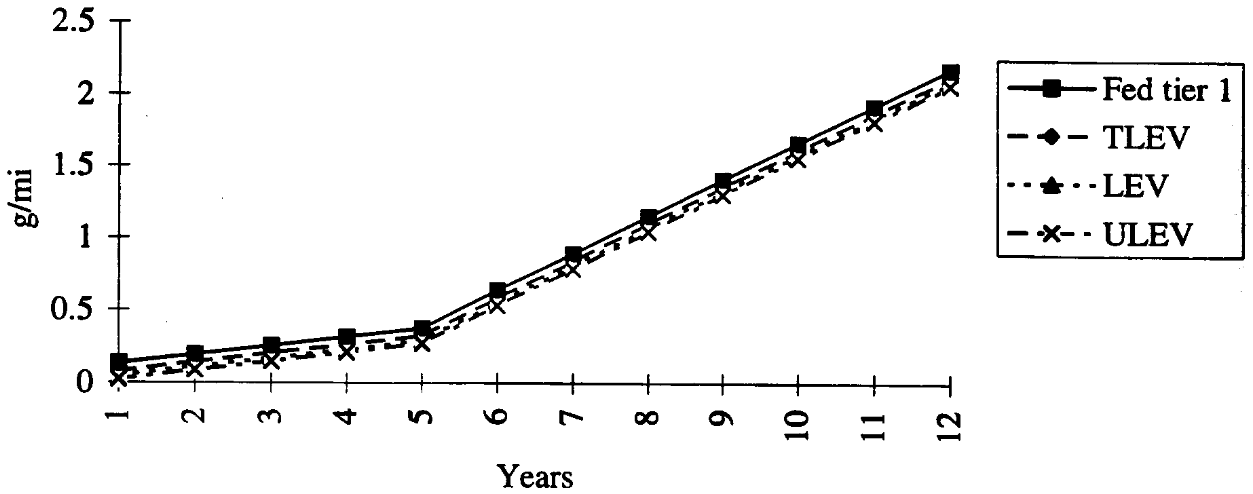
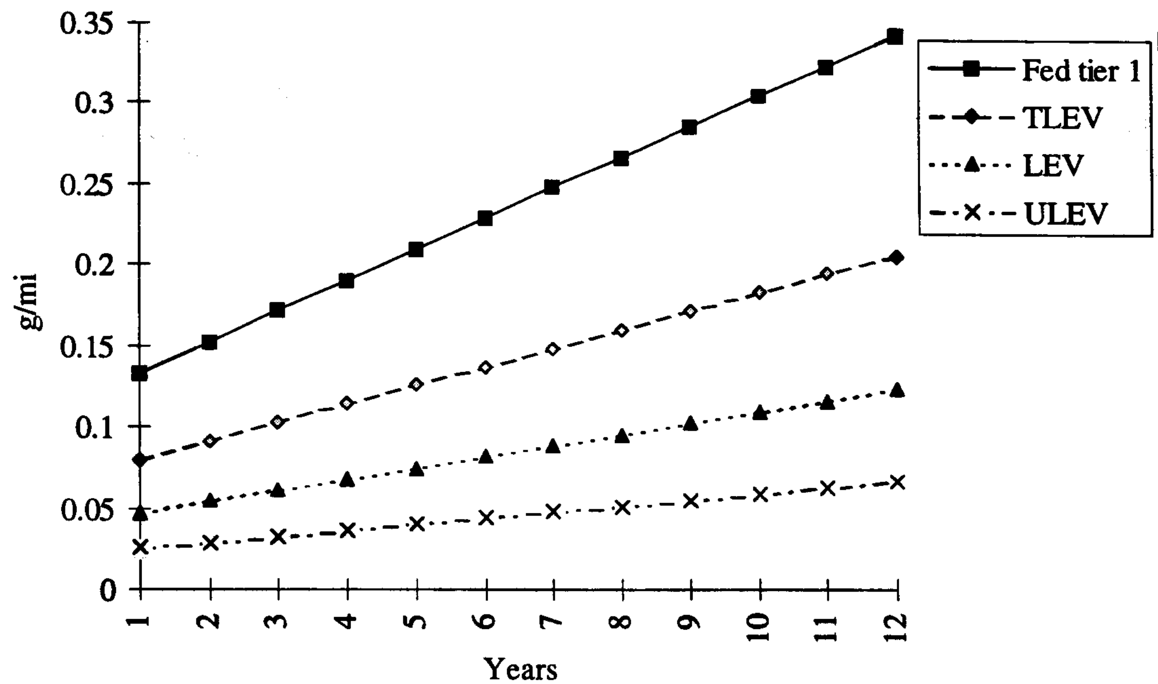


Figure C-4. Cal LEV Standards under CARB Deterioration Rates.



The differences between CARB's and U.S. EPA's deterioration factors relate to claims of the current underestimation of VOC emissions by the FTPs. Several studies have measured roadside emissions of VOC and CO and have found emissions far in excess of values reported on their last inspection maintenance (I/M) check (Lawson et al., 1990; Pierson et al., 1990). This again raises concern over the quality of vehicle emissions data as well as the amount of credits given to I/M programs in determining VOC emissions. In addition, the Van Nuys tunnel study, performed in 1992, further suggests that the ambient measurements of VOC concentrations differ from concentrations produced by exhaust emission models such as Mobil 5 (Pierson et al., 1990; U.S. EPA, 1993). If Mobil 5 underestimates exhaust emissions, adopting CARB deterioration rates will further exacerbate the model's shortcomings. On the other hand, the deficiencies and inherent biases of Mobil 5 should remain independent of determining automotive exhaust deterioration rates (Furey and Monroe, 1981). Furthermore, CARB supports their slow deterioration rates with dynamometer (tread mill) tests that show little deterioration between 50,000 and 100,000 mile emission levels. Consequently, we develop cost estimates based on the CARB deterioration rates.

VEHICLE FLEET CHARACTERISTICS

Estimation of the present value of emission savings attained through the California LEV program rests upon a variety of factors other than emission rates: (1) summer vehicle miles traveled (VMT), (2) VMT decay rate by vehicle age, (3) vehicle attrition rate, (4) growth rate of vehicle fleet, and (5) growth rate of VMT. These values are inputs into MOBIL 5 and are integral to determining removal costs.

Summer VMT for each of the sub-regions uses data from each state's 1993 SIP filing (i.e., NY, NJ, CT, and PA). The aggregate summer VMT for each subregion is divided by the vehicle fleet size to determine average vehicle miles traveled. The annual VMT values are then adjusted for declining VMT with vehicle age and vehicle attrition from Sierra Research (1994). The total VMT for each sub-subregion includes a 0.40% increase in vehicle population and a total VMT growth rate of 1.1%. The VMT values serve as the basis of the emission reduction calculations.

The aggregate emission reductions achieved by the LEV program for the model year 2007 draw upon NESCAUM (1992) results and several supplemental MOBIL 5 runs performed over urban and rural counties of New York State. These results also determine the percentage reduction in vehicle emissions of the LEV program for 2007 for each of the four regions shown in Table C-4. NESCAUM (1992) uses MOBIL 5 to evaluate the LEV program for New Jersey, Pennsylvania, Maryland, Delaware, and Virginia. The MOBIL 5 modeling results also serve as a basis to determine the first year g/mi emissions factor for each category of vehicle.

Table C-4. NO_x and VOC Percent Emission Reduction of the Cal LEV Program over Fed Tier 1 Vehicles.

Subregion	Percent Reduction LDV + LDT1	
	NO_x	VOC
Northeast	28	25
Metro East	38	38
Band I	32	27
Average	34	32

CONTROL COSTS

The control costs for the California LEV program are subject to significant controversy and suffer from large disparities among estimates. The wide range of costs estimates shown in Table C-5 illustrates this problem.

Table C-5. Control Cost Estimates per Vehicle.

Control Measure	CARB	Manufacturers	Sierra Research
TLEV	$105	$666	$344
LEV	$146	$1,291	$775
ULEV (small cars)	$214	$1,668	$610
ULEV (mid-size)	$214	$2,229	$1,347
ZEV	$1,436	$34,145	$12,588

The CARB cost estimates are produced by an independent state agency and are less likely to suffer from a bias to exaggerate costs than estimates by the Manufacturers or Sierra Research, as a subcontractor the for American Automobile Manufacturers Association.

U.S. EPA has developed a standard retail price equivalent (RPE) technique that was developed with the intent of bringing some reproducibility to cost estimates for emission control systems. The principal equation for determining the RPE is as follows:

$$RPE = ((SP + AL + AO) \times MM + RD + TE + WC) \times DM , \qquad (C\text{-}2)$$

where:

RPE is the retail price equivalent

SP is the supplier price charged to the auto assembler for the components and subassemblies involved

AL is the direct cost of assembly labor for installing the components

AO is the manufacturer's assembly overhead cost per unit

MM is the manufacturer's markup percentage, to account for corporate overhead and profit

RD is the manufacturer's research and development cost per unit

TE	is the manufacturer's tooling cost per unit
WC	is the manufacturer's added warranty cost, per unit
DM	is the dealer's markup percentage.

The RPE equation provides a consistent estimate of control costs but is likely to overstate these costs by using component pricing. As Commissioner Jorling[4] noted, a change in vehicle cosmetics or a simple design change such as a tilt steering wheel calculated under the RPE formula would result in equally excessive costs (NY State Senate, 1991). Common markup values are a 5.7% dealer markup and a 19.2% manufacturers' markup with assembly overhead value of 40% of direct labor (Weaver and Turner, 1993).

For example, modifications necessary to move from the federal emissions standard to the LEV standard have been achieved through a slight increase in catalyst size and moving the catalyst closer to the engine exhaust manifold. The tooling modifications for manufacturing are minimal; the difference in assembly time is minimal. The economies of scale achieved through integrated component assembly and simultaneous system assembly is not accurately captured by U.S. EPA's RPE. The RPE falls short of reflecting the actual consumer cost for improving the emissions control system. A marginal cost analysis may be more reflective of the true cost. The marginal cost difference before and after regulations are promulgated will likely be a constant plus the new catalyst cost.

The automobile industry numbers are subject to a great deal of skepticism because of past claims in 1970 and 1977 that proposed regulation will "cripple the industry" (NY State Senate, 1991). The automobile industry's pricing of emission controls component by component can only be fairly compared to a vehicle priced component by component (e.g., the cost of wheel covers, electric windows, fancy stereos). The automobile industry claims that the CARB numbers do not adequately account for capital costs. Commissioner Jorling in his 1991 testimony countered the automobile industry's claims of excessive cost and excessively onerous costs by stating that vehicle manufactures "should identify what the capital cost to plant is that they charge off for cosmetic changes to the vehicle annually" (NY State Senate, 1991).

In support of the LEV technology and CARB's control cost estimates are emission certification tests performed by CARB on a 1993 Ford Escort TLEV and a 1992 Oldsmobile Achieva. At the 50,000 mile certification level, the Escort's emissions were still 23% below the LEV standard, while emissions from the Achieva were even lower. These emission rates were surprising considering little or none of the advanced technologies available to reduce emissions were implemented. In light of these recent test results, we will only consider the CARB control cost estimates and will dismiss the other values as over inflated and unreliable estimates.

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4. New York State Department of Environmental Control Commissioner in 1991.

COST-EFFECTIVENESS

Table C-6 shows VOC and NO_x incremental reduction costs using CARB deterioration rates. The TLEV program has VOC control costs ranging from $882/ton to $1,191/ton in Metro East and NO_x removal costs between $822/ton and $1,042/ton. VOC removal costs are slightly more than twice as large for LEV certified vehicles and about one and a half time larger for NO_x reductions. ULEV vehicles removal costs are about 1.5 times larger for both NO_x and VOC emissions, except for Northeast where the penalty for having a less developed I/M program causes the removal costs to double.[5] Zero emission vehicles (ZEV) are the most controversial part of the LEV emissions program from a cost-effectiveness as well as a technical perspective.[6] For ZEVs, the removal costs for both VOC and NO_x emissions range from approximately $13,000 to $35,000.

The weighted average removal cost for the LEV program under CARB deterioration factors is shown in the bottom row of Table C-6. These costs are determined by a weighted average of the incremental control costs for the individual LEV categories from the program implementation year of 1996 through 2007. Despite the enormous costs of ZEV, the LEV program costs are less than $3,000 per ton for all subregions.

INCREASING THE CAFE STANDARD

Improvements in the car average fuel efficiency (CAFE) standard, from an estimated baseline level of 28 mpg to 35 mpg, offer the opportunity to simultaneously increase transport fuel efficiency, reduce exhaust emissions, and preserve the environment. These improvements are technologically achievable with existing production automobiles. Fuel economy has been commonly synthesized as a function of three characteristics: (1) vehicle weight, (2) tire pressure, and (3) frontal area times velocity cubed. Improved fuel economy will inevitably rest on vehicle design changes that will involve technological advances to maintain rider comfort, but will also involve "nonpecuniary" costs as vehicles become less powerful, smaller, and have diminished impact resistance.

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5. It should be noted that the assumptions used for credits associated with enhanced I/M programs have not been field tested and lack supporting studies.

6. Much of the recent and intense research activity can be attributed to this mandate. In response to the California mandate, domestic automobile manufacturers in cooperation with the U.S. Department of Energy (DOE) have assembled a consortium focused on developing a more durable and longer range battery (Pechan, 1992).

Table C-6. Emission Control Cost of the Cal LEV Program in $/Ton.			
Vehicle Type	**Northeast**	**Metro East**	**Band I**
VOC TLEV	1,191	882	985
VOC LEV	1,454	1,077	1,202
VOC ULEV	2,041	1,511	1,687
VOC ZEV	17,663	13,078	14,602
Total VOC	**2,686**	**1,989**	**2,220**
NO_x TLEV	1,042	822	901
NO_x LEV	1,347	1,063	1,165
NO_x ULEV	2,613	1,875	2,055
NO_x ZEV	20,609	14,786	16,209
Total NO_x	**2,947**	**2,181**	**2,390**

The measurement of cost-effectiveness draws upon results of a similar study performed by Krupnick (1990) which attempts to account for behavioral responses to increased fuel efficiency (Krupnick et al., 1991). The effect of lower variable operating costs causing a "rebound effect" (see Dahl and Sterner, 1991) is incorporated. Cost of VOC emission changes are not calculated for CNG or methanol fueled vehicles.

It is difficult to determine the effect of VOC reductions from improved fuel economy. Competing arguments exist. A mass balance approach would lead to the conclusion that less VOC emissions would be produced since less gasoline is consumed. On the other hand, the market forces of a competitive automobile industry require vehicles to meet a gram per mile emission standard regardless of fuel economy, and in the absences of other market incentives to reduce vehicle emissions[7] manufacturers would meet the minimum standard as a least cost production strategy. Thus, even if one accepts the mass balance approach to VOC emissions, competition would encourage manufacturers to reduce expenditures on emission controls (for example, reducing catalyst size).

Krupnick relies on a pilot study by Rusin (1989) to sort out these competing hypotheses. As Krupnick states, "Although the Rusin results do not establish a clear cause and effect relationship between fuel efficiency and the HC emissions, they are the best evidence currently available on this subject." Rusin analyzed data obtained from U.S. EPA on hydrocarbons (HC) tailpipe emission of 845 new 1989 non-California passenger cars. Rusin finds a 0.00345 g/mi reduction in HC emission for each 1 mpg increases in fuel efficiency. However, the vehicles Rusin tested are subject to a less stringent emissions standard than the model year vehicle in 2007 and the mean fuel economy was 29.3 mpg versus our predicted fuel

7. For example, a fee based on the emission level of a vehicle could be either developed from U.S. EPA test characteristics and made part of the vehicle registration fee. This would endogenise vehicle emissions into the cost to consumers.

economy of 35 mpg. Furthermore, Rusin's fitted relationship has an R^2 of 0.16 that lacks statistical robustness in predicting HC emissions, particularly if a higher fuel and significantly cleaner exhaust controls are used. Thus, with some misgivings, we adjust Rusin's relationship for the differences in fuel economy and determine emissions reductions as a percentage reduction.[8]

NO_x formation in internal combustion engines is primarily produced through oxidation of N_2 at the high temperatures of combustion, but can also be formed through oxidation of nitrogen-containing compounds in the fuel (Seinfeld, 1986). NO_x formed by the first pathway is called thermal NO_x and the second pathway is called fuel NO_x. Thermal NO_x formation has been modeled as an exponential function of temperature that begins significant and rapid formation at temperatures above 2100°F.[9] Temperature control in the combustion chamber has been maintained by adjusting combustion stoichiometry, gas dynamics in the combustion chamber, point of ignition, and the specific heat of fuel (Seinfeld, 1986). The relationship for NO_x control and vehicle fuel economy follow a similar pattern as HC controls. The levels of NO_x emissions contemporaneously shift downward with HC emission levels as vehicle fuel economy improved in the 1970s through the mid-1980s.

Furthermore, the Federal Tier 1 emission standards (shown in Table C-7) for both NO_x and HC are based on vehicle weight. The relationship between HC emission and vehicle weight follows a pattern shared by NO_x emissions that can be characterized as the percentage reduction for NO_x corresponds to the percentage reduction for HC. Adopting this method of characterizing NO_x reductions diverges from Krupnick's derivation, which entirely ignores reductions in NO_x emissions due to an erroneous scientific explanation.

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8. Rusin gets the following result:

$$HC = 31.4 - \underset{(0.117)}{0.638}\, mpg + \underset{(0.002)}{0.005}\, mpg^2; \qquad R^2 = 0.16;$$

Where HC is hydrocarbon emissions in g/mi, multiplied by 100. Standard errors are in parentheses. The equation is adjusted by solving for a new intercept value with HC emissions of .125 g/mi and a fuel economy of 35 mpg. The HC emission level corresponds to the Cal LEV emissions reduction program TLEV vehicle at 50,000 miles, which after adjusting for vehicle deterioration has approximately the same emission characteristics as a federal vehicle for Rusin's test 1989 test vehicles. The percent reduction in HC emissions for the fuel economy improvements are applied to the LEV and ULEV vehicles. As vehicles become cleaner, the amount of emissions reduction decreases.

9. However, the Zeldovich mechanism predicts an equilibrium amount of NO can be formed and rapidly quenched. The short ignition time occurring from the spark at top dead center to flame propagation starts the NO formation process. As the piston recedes to bottom dead center, the gases rapidly cool with the expansion, "freezing" the NO (i.e., the chemical reactions which would remove the NO become much slower) at the levels formed during combustion.

Table C-7. Federal Emission Standards for Passenger Cars and LDTs.			
	5 Yr/50,000 Mi		
Vehicle Type (nondiesel)	NMHC	NO_x	CO
LDVs and LDTs of 3,750 lbs or less	0.25	0.4	3.4
LDVs and LDTs of 3,750-5,750 lbs	0.32	0.7	4.4
LDT over 5,750 lbs	0.39	1.1	5.0

ESTIMATES OF REBOUND EFFECT

Since improved fuel economy lowers the variable cost of driving, consumers are expected to increase VMT with more fuel efficient cars. This "rebound-effect" has been recently characterized by Dahl (1992) and Jones (1993). Jones' more recent analysis confirms Dahl's results for short-run price elasticities of -.13 and introduces a long-run price elasticity of -.27.[10] We use the later figure to capture the effect of increased fuel efficiency on cars.

CONTROL COSTS

Since the focus is on costs associated with improving environmental quality, cost estimates for increases in fuel efficiency involve direct costs paid by customers and not the nonpecuniary costs associated with driving a more fuel efficient vehicle. In addition, the effect of increased vehicle prices on the demand for new vehicles is ignored. To the extent that new vehicle demand is reduced by the new CAFE standards,[11] the cost-effectiveness is reduced as the average age of the vehicle fleet increases. A comprehensive study would also credit environmental benefits through option valuation of future land use for reductions in refinery waste and capacity.

Following the lead of Krupnick, an econometric study by Walls (1992) is used to determine the cost of improved fuel efficiency. Walls find that a 1 mpg increase in fuel efficiency raises an average car's price by $135, and assuming linearity would raise an average car's price by $945. Walls' estimates include "nonpecuniary" costs to consumers for driving vehicles that are likely to be smaller and slower than they prefer. Ignoring the more abstract "nonpecuniary" costs, data from MVMA 1992 are used to develop an

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10. Jones price elasticity results are developed from an AR(1) log linear model. Although Jones concludes that the long-run effects are statistically significant, he also states ". . . we are not completely sure about the long-run rebound effect, but if it does exist it is probably not very large."

11. Additional costs may be incurred as new technologies are used to increase customer satisfaction. These technologies could include light weight composite material and ceramics to reduce frame and engine weight, respectively, or smart vehicles which adjust tire pressure to road smoothness (i.e., higher tire pressure on smooth highways and lower pressure on rough municipal roads).

alternative cost to increased fuel economy. The additional costs for improved fuel economy are determined from data that combine fuel economy improvement and emission control costs from 1972 to 1980 and are assumed to be proportioned equally. Again, assuming linearity of costs with improvements in fuel economy, a value of $206 is used for the 7 mpg improvement in fuel economy.

Omission of the nonpecuniary costs of smaller vehicles may be more consistent with a procedure that also omits the effects of "shared" environmental and financial benefits of improved fuel economy. However, societal benefits (some of which are intergenerational) such as reduced hazardous and nonhazardous refinery waste, reduced refinery size and storage, reduced greenhouse gas emissions, improved balance of trade and national security are also omitted. Although these benefits are considered exogenous to a consumers choice set and should not be accrued to the purchasing consumer, they nonetheless do contain a measurable value (Mitchel and Carson, 1989). Recognizing that the "nonmarket" value of externalities of gasoline consumption is comparable to the "nonpecuniary" costs, the base case will only consider the direct costs determined from MVMA 1992.

COST-EFFECTIVENESS

The cost-effectiveness of control programs was calculated for the summer ozone season based on Mobil 5 emission rates and driving patterns for each subregion.[12] The percentage of emission reductions for each subregion is shown in Table C-8. Although the increase in CAFE standards from 28 mpg to 35 mpg would likely be implemented over a five year period, the incremental reductions in emissions for the entire 7 mpg increase are measured for a twelve year vehicle life beginning in 2000. The net present value of emission savings and added vehicle costs are calculated for the base year of 1994. Costs are equally apportioned between NO_x and HC emissions.

Table C-8. Percent Emission Reduction and Reduction Cost of Increasing the CAFE Standard for Cal LEV and Fed Tier 1 Light Duty Vehicles.

	Percent Reduction		Removal Cost Cal LEV		Removal Cost Fed Tier 1	
Subregion	NO_x	VOC	NO_x $/ton	VOC $/ton	NO_x $/ton	VOC $/ton
Northeast	28%	25%	121,264	106,841	52,596	14,703
Metro East	38%	38%	103,974	85,777	45,097	12,004
Band I	32%	27%	102,356	89,344	44,395	12,964

The removal costs for increasing the CAFE standard for Cal LEV are in excess of $35,000/ton for both VOC and NO_x emissions, as shown in Table C-7.[13] The incremental cost-effectiveness does improve by

12. Mobil 5 runs were supported by NYDEC and also drew heavily upon Pechan (1992).

13. Cost estimates are based on the CARB methodology for the vehicle emission deterioration rate.

almost a factor of ten for vehicles meeting the less stringent federal emission standard, but these costs still remain significantly above other control measures. Increasing CAFE standards as an emissions reduction strategy, remains relatively expensive.

This static analysis fails to account for a large number of other environmental and economic factors that may make improvements in fuel economy an economically tenable strategy. For example, the value added or multiplier effect for expenditures on gasoline consumption in New York State that has negligible production and refining capacity remains relatively low compared to almost any other purchase such as electronic equipment and food items.[14] The net societal gains of other attributes, such as reducing dependency on imported oil, may have sufficient countervailing weight to justify implementation of higher CAFE standard, but inclusion of these factors attributes more appropriately resides with a state or federal energy plan.

LAWN, GARDEN, AND UTILITY ENGINES

Sources and Emission Characteristics

The CAA limits the authority of states to promulgate regulations of off-road mobile sources. Section 209 (e) (1) establishes a federal pre-emption prohibiting states from regulating new engines under 175 horsepower used in farm and construction vehicles and equipment. However, small engines used for lawn and garden and utility applications have been subject to regulations in California beginning in 1995.[15]

CARB defines the utility equipment category to include a variety of equipment that use engines 25 horsepower or less. This category includes classification of equipment as handheld and nonhandheld. Table C-9 lists some the equipment included in the utility category.

Table C-9. Utility and Lawn and Garden Equipment.

Nonhandheld	Handheld
Walk Behind Mowers	Chain Saws
Riding Mowers	String Trimmers
Compressors	Edge Trimmer
Portable Refrigeration Units	Blowers
Pumps	
Generators	

14. This statement is made on a value added basis from NYSIP (1993) and NYSEO (1994) data.

Control of utility engine emissions offers great promise for reducing emissions when compared on a unit of work basis to control for on-road heavy-duty truck engines. Table C-10 shows these disparities in emissions rates.

Table C-10. Comparison of Exhaust Emission Rates for Utility and On-Road Heavy-Duty Truck Engines.

Application	Emissions in g/bhp-hr		
	HC	CO	NO_x
Current 2-Stroke Utility Engines	150-300	200-700	1-5
Current 4-Stroke Utility Engines	4-50	200-700	1-5
On-Road Heavy-Duty Truck Engines	1.1	14.4	5

Small engine emissions hold great promise for emission reductions, but the chemical dynamics of internal combustion engines may limit the ability to reduce these emissions to equal those of larger engine sizes.

The relatively high formation of hydrocarbon emissions in small utility engines results from a narrower cylinder chamber. The narrower chamber has a higher combustion chamber surface area compared to the larger diameter of more powerful engines. The combustion process begins with spark ignition of the compressed air-fuel mixture in the center of the cylinder and a flame that propagates across the cylinder. As the flame approaches the walls, which are relatively cooler, the flame is quenched, leaving a very thin layer of unburned or absorbed fuel on the walls and in the crevice between the piston and the cylinder wall above the piston ring (Seinfeld, 1986). The unburned hydrocarbons are then discharged from the engine. As the surface to volume ratio increases, total hydrocarbon emissions increase as well.

CARB has found it appropriate to divide the utility category by engine class with handheld engines greater than 50 cc and less than 50 cc and nonhandheld engines greater than 225 cc and less than 225 cc.[16] The reason for this fine delineation with different emission standards is the increasing level of hydrocarbon emissions as the surface to volume ratio of the combustion chamber increases. The 50 cc engine category is for handheld equipment that represents primarily two stroke engines. As shown in Table C-11, two-stroke engines have HC and CO emission levels that are much higher than their 4-stroke counterparts. Two-stroke engines are uniquely fitted for use in hand-held equipment because of their high power to weight ratio and

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15. These regulations have subsequently been approved by the U.S. EPA.

16. Handheld engines are defined by CARB "as a handheld piece of equipment, performance of the equipment's requisite function must require that the operator support the equipment's full weight. The engine and equipment must require the capability of operating in any position to properly perform its design function."

multipositional operation. Recognizing the special function of 2-stroke engines, CARB designs a separate emission standard shown in Table C-11.[17]

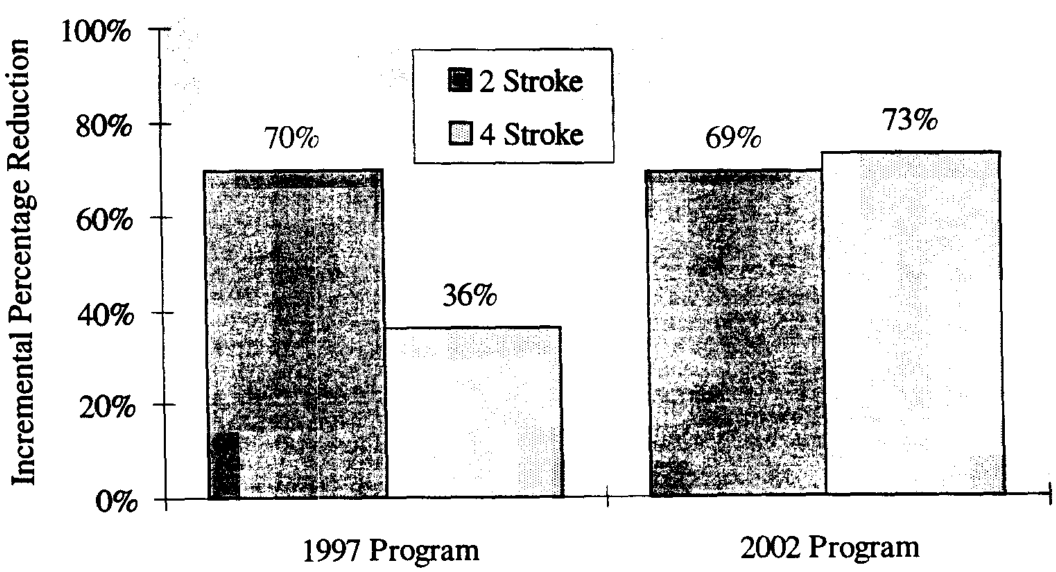
Table C-11. Exhaust Emission Standards for Utility Engines in g/bhp-hr.

Calendar Year	Engine Class	HC + NO_x	HC	NO_x
1997-2002	<225 cc (nonhandheld)	12	—	—
	•225 cc (nonhandheld)	10	—	—
	<50 cc (handheld)	—	180	4
	•50 cc (handheld)	—	120	4
2002	all 225 cc (nonhandheld)	3.2	—	—
2002	all 50 cc (handheld)	—	50	4

EMISSION REDUCTIONS AND CONTROL COSTS

The percentage of emission reductions produced by the CARB emission standards are shown in Figure C-5. Unlike emission reduction measurements for the transportation sector, it is assumed that utility engines follow similar operating conditions and consequently achieve the same level of reduction in all regions. Emission reductions for the 2002 program are measured as incremental reductions in emissions from the 1997 program.

Figure C-5. Percentage VOC Emissions Reductions for Utility Engines.



17. CARB standards contain an additional category for engines with less than 20 cc displacement. CARB's implementation dates are several years ahead of those listed in Table C-11.

The 1997 regulations for both 2-stroke and 4-stroke engines can likely be met with carburetor modifications to comply with the proposed handheld equipment standards. CARB claims "this standard will promote the use of the cleanest current technology in 2-stroke engines, and maintain handheld product line availability" (CARB, 1990a). The use of 2-stroke engines in mowers and generators would cease with the 1997 standards because it is not feasible for 2-stroke engines to comply with the nonhandheld equipment standards. For these applications, it is likely 4-stroke engines or electric motors will replace the dirtier 2-stroke engine.

The 2002 standards are emission reductions of 60 to 70% above the 1997 standard. Achievement of the standards may be met through the use of a catalyst technology that has proven capable of reductions of 90% and greater. According to CARB (1990), catalysts have achieved 80% reduction in HC emissions in Europe since 1989. Table C-12 shows the projected increases in equipment costs for meeting the 2002 standards are only moderately above the cost of compliance with the 1997 standards.

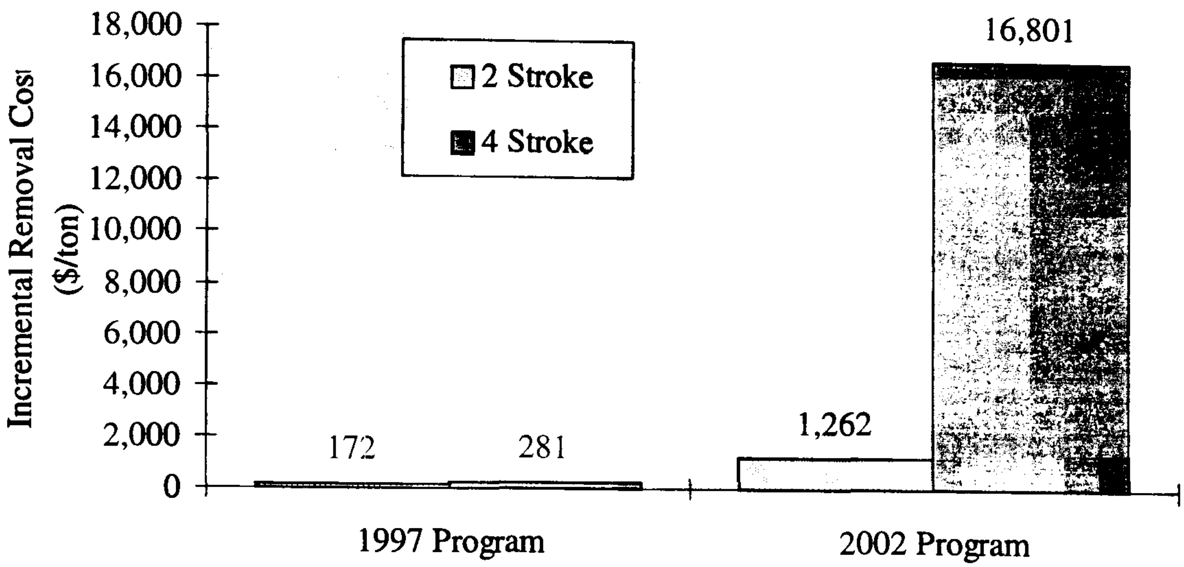
Table C-12. Control Costs for Utility Engines in 1994 Dollars.

Year	2-Stroke	4-Stroke
1997	30	40
2002	35	60

The incremental removal costs for HC plus NO_x emissions presented in Figure C-6 are under $300 per ton for the 1997, but increase to approximately $1,262 and $16,801 per ton for 2-stroke and 4-stroke engines, respectively, in 2002. The 4-stroke engine emissions control costs for 2002 rise precipitously because of the high capital cost and relatively small mass of emissions reductions.[18] The mass of annual emissions was assumed to be equivalent to CARB's, but only 75% of these emissions were assumed to occur during the ozone season. In addition, cost estimates ignored the reduced health risks to utility engine operators from using the cleaner engines.

18. The capital costs for emissions compliance may be over stated because CARB relied on manufacturer cost estimates. In addition, these cost estimates only assumed the modifications would be required for California. The proliferation of these cleaner utility engines across the Northeast would likely further reduce costs.

Figure C-6. Emissions Reduction Costs for 2-Stroke and 4-Stroke Engines.



HEAVY-DUTY OFF-ROAD AND ON-ROAD ENGINES

Heavy-duty engines are used in both diesel trucks, urban transit busses and mobile off-road diesel equipment. The diesel engine is used extensively in heavy-duty vehicles due to its high efficiency, reliability and durability. However, the diesel engine also creates large amounts of NO_x and particulate matter (PM) emissions compared to the Otto cycle gasoline engine. NO_x and PM emissions from heavy-duty engines represent a significant portion of New York City's air quality problem. Heavy-duty diesel engines are responsible for over 50% of controllable and inhalable PM and a significant portion of NO_x emissions.

The ability to promulgate extensive regulation for mobile off-road diesel engines is preempted by Section 209 (e) (1) of the CAA,[19] but engines greater than 175 hp are open to regulation. This section will focus on the development of emission control costs for heavy-duty vehicles, transit buses, and heavy-duty off-road engines between 175 hp and 750 hp.

Heavy-duty engine classes are distinguished by gross vehicle weight for on-road vehicles and by engine size for off-road nonmobile engines. Table C-13 defines the class characteristics for the heavy-duty vehicle fleet. With the preponderate share of VMT in this category produced by and the inextricable link of VMTs to emissions, heavy-duty diesel engine vehicles are of primary concern. Off-road engines greater than

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19. Approximately 30% of the applications are farm and construction equipment which uses engines of 175 hp or less.

175 hp are almost exclusively diesel. As a result, the cost analysis of this section will focus on diesel engine emission controls.

Table C-13. Heavy-Duty Class Characteristics.

Characteristics	Heavy-Duty Class		
	Light	Medium	Heavy
GVW Range (lbs)	8,500-14,000	14,001-33,000	>33,000
General Power Range (hp)	70-170	170-250	>250
Percentage Diesel	30	56	100
Percentage Gasoline	70	44	0

The heavy-duty off-road equipment category includes construction and farm equipment, as well as mining, forestry and industrial implements and equipment (CARB, 1992). The reliability and compactness of diesel engines as a power source enables the same engine model to be used in multiple applications. Table C-14 list some the equipment included in the heavy-duty off-road category.

Table C-14. Heavy-Duty Off-Road Equipment.

Rollers/Compactors	Trenchers/Ditchers
Excavators	Motor Graders
Backhoe Loaders	Off-Highway Trucks
Scrapers	Track Loaders
Tractors	Crawler Tractors
Rough Terrain Forklifts	Wheel Loaders
Pavers	Wheel Dozers
Air Compressors	Mobile Cranes

EMISSION STANDARD

Table C-15 shows the historic and projected NO_x and PM_{10} emission standards for heavy-duty diesel engines. The 1993 and 1998 emission standards for on-road heavy-duty diesel vehicles correspond to the existing federally mandated standard, but more stringent regulations are adopted from CARB (1992) and CARB (1993). NO_x emission standards for trucks and buses are consistently one third of the standards for 150 to 750 hp off-road engines. The NO_x emissions standards for trucks and buses are significantly higher than they are for passenger automobiles with 10.1 g/mi in 1998 and 5.1 g/mi in 2002.

Table C-15. Historic and Projected Heavy-Duty Diesel Engine Emissions.

	NO_x g/mi	PM_{10} g/mi
1988		
Trucks	6	0.25
Buses	6	0.25
1993		
Trucks	5	0.25
Buses	5	0.1
Diesel 150-750 hp	14.3	0.4
1998		
Trucks	4	0.1
Buses	4	0.05
Diesel 150-750 hp	6.9	0.4
2002		
Trucks	2	0.05
Buses	2	0.05
Diesel 150-750 hp	5.8	0.4
Diesel >750 hp	6.9	0.4

Off-road diesels greater than 750 hp are currently unregulated and in the year 2002 would be subject to the 1998 emission standards for 150 to 750 hp class engines. The disparity in emission standards between on-road and off-road engines has existed despite the compatibility between on-road and off-road engine technology, and will likely continue to persist. Consequently, the analysis considers less stringent standards for off-road engines than for heavy-duty trucks and buses.

Although PM emissions are not ozone precursors, they are included in Table C-15 because of their relative toxicity compared with other emissions. In addition, the carbonaceous nature of PM emissions from diesel engines makes these particles especially important in reducing visibility because of their small size and ability to absorb light. PM emission standards are also of concern because of the trade-off between NO_x and PM.

Table C-16 shows the incremental NO_x reductions for off-road and on-road diesel engines from the 1998 federal emission standards for diesel trucks. The incremental reductions in emissions are measured in terms of reductions from the 1998 standard and also in terms of reduction in the 2007 emission inventory for on-road and off-road heavy-duty diesel engines. The inventory reductions in 2007 are 60% of the absolute reductions for the 2002 model year engine and 100% for the 1998 model year. These reductions result from an expected 10 year engine life that results in a 60% penetration level for both on-road and off-road diesels.

Table C-16. Heavy-Duty Diesel Engines: Percent Reduction and Cost.		
1998	**NO_x Percent Reduction**	**Cost $/ton**
175-750 hp	52%	$713
2002	**NO_x Percent Reduction**	**Cost $/ton**
175-750 hp	16%	$2,330
>750 hp	52%	$75
Urban Transit Bus	50%	$15,386
Heavy-Duty Truck	50%	$10,770

A short-coming in using a 60% penetration level is the failure to account for the impact of the increased cost of cleaner diesel engines on new engine purchases. Although this short-coming can not be entirely dismissed, the cost increases are relatively small compared to the large base price for most heavy-duty diesel applications. The emission reduction levels also assume a constant rate of operation throughout the entire life.

DIESEL CONTROL TECHNOLOGY

Simultaneously controlling both PM and NO_x emissions and still maintaining fuel economy poses a challenge to manufacturers and adds complexity to efficient regulation of air pollution. Countervailing factors in the combustion process make simultaneous reductions in NO_x and PM emissions difficult. Methods that reduce NO_x emissions, such as cooler peak cylinder temperatures, retarded injection timing and longer combustion duration, tend to increase PM emissions. The trade-off also exists between shorter combustion duration, which tends to increase peak temperatures and pressures, thereby increasing NO_x production, and limiting PM.

The formation of NO_x emissions follows the same process as for Otto cycle engines discussed previously. PM emissions are formed from incomplete combustion of heavy hydrocarbon chains as they break down during the combustion process. In addition, lubricating oil that is entrained into the exhaust or partially burned during combustion further contributes to PM emissions (CARB, 1993a). HC and CO emissions are less important factors since controlling for complete combustion to reduce particulates also controls HC and CO emissions.

Control of emissions to meet the 1993 diesel engine standard has been primarily accomplished through engine design changes: air induction systems, combustion chamber modifications, oil control, and fuel injection systems, and adjustments in injection timing.[20] Since many of the more fundamental engine

20. Recognizing NO_x formation as a function of temperature and residence time at high temperatures, control of NO_x and PM emissions within a diesel engine is linked to the rate at which fuel is burned in the combustion chamber. Combustion residence time has been reduced by retarding the injection timing, thus

design modifications were used to meet the 1991 standards, the more stringent emission standards in future years will likely require exhaust after-treatment, fuel reformulation and engine design improvements. Table C-17 lists some of these perspective technologies (CARB, 1993a).

Table C-17. Diesel Engine Emission Control Technologies.

1993	1998
High Pressure Fuel Injection	Injection Rate Shaping
Air-to-Air After-Cooling	Advanced Turbochargers
Fuel Injection Electronic Control	Advanced Electronic Control
Responsive Turbocharging	Low Inlet Manifold Temperature
Efficient Combustion Chambers	Exhaust Gas Recirculation
Oil Control	Low Aromatic, High Cetane Fuel
	Oxygenates, Additives
	Advanced Traps
	Oxidation Catalysts with low sulfur fuel
	DE- NO_x Catalysts with low sulfur fuel
	Hybrid – Electric Drive

The use of gaseous fuels can nearly eliminate PM emissions and facilitate meeting NO_x emission standards, although the impact on production of ultrafine particles is uncertain. In addition, gaseous fuels have high hydrogen to carbon ratios and consequently produce less CO_2 on a workload basis. The two principal gaseous fuels for heavy-duty engine use are natural gas (NG)[21] and liquid petroleum gas (LPG). Natural gas is primarily methane and is distributed through pipelines that can limit its availability in rural areas. LPG is a by-product of the refining process of crude oil and may be constrained from broad application because of production constraints. None-the-less, these two fuels hold enormous promise for urban transit buses, delivery vehicles, and nonroad engines. For stringent large emission standards of urban transit buses and heavy-duty delivery trucks, gas fired engines are more cost effective than diesel engines.

CONTROL COSTS

CARB (1993), using a study by ACUREX, provides an extremely detailed analysis of engine modifications for diesel trucks and urban transit buses. The analysis calculate control costs on a cents/mile basis from the

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reducing the peak combustion temperatures which affects the amount of NO_x formed. Potential increases in PM emissions result from incomplete combustion associated with injection retarding. The increase in PM emissions can be off-set by changing engine design characteristics to impart organized swirl, causing the injected fuel to burn more completely within the engine cylinder. High pressure injection increases vaporization and mixing which simultaneously reduces PM emissions and NO_x formation.

21. Natural gas is compressed for transportation purposes and is referred to as compressed natural gas (CNG).

1998 baseline vehicle. These costs are derived from an estimated 40,000 miles per year for both diesel trucks and diesel buses. The low speed and successive stop and start driving cycle of urban transit buses results in average lifetime operating costs three times greater than heavy-duty diesel trucks. The increase in costs for achieving the 2002 emission standards is 3.5 cents/mi for trucks and 5 cents/mi for buses. Control costs for off-road engines relied on results from CARB 1992.

Table C-16 shows the marginal removal costs for heavy-duty engines. The costs for off-road engine reductions for 175-750 hp engines are $713/ton for 1998 emission standards and $2,330/ton for the 2002 emission standards and $75/ton for engines greater than 750 hp. The disparity in costs between the two engine classes results from the difference in the stringency of the emission standards. On-road diesels have considerably higher removal costs with urban transit buses at $15,386/ton and trucks at $10,770/ton for the 2002 NO_x standards. However, the cost of both of these emission reduction programs would be reduced if we were to apply a credit to account for reductions in PM emissions.

The regulation of heavy-duty diesel engines holds significant promise to reduce NO_x and PM emissions. Regulation of off-road diesel engines has a cost-effectiveness advantage over regulation of heavy-duty diesel vehicles. The disparity in control cost-effectiveness shows the large potential that can be achieved from extending emission regulations to new sources.

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Appendix D
REFORMULATED GASOLINE

INTRODUCTION

The Clean Air Act Amendments of 1990 have several key provisions that have made the introduction of reformulated gasoline an important emission control policy. However, assessing the cost-effectiveness of reformulated gasoline is complicated by the air chemistry of pollution formation and the behavioral response of consumers. This appendix accounts for these factors in determining the marginal cost of California reformulated gasoline (RFG) over federal reformulated gasoline for the subregions of the Ozone Transport Subregion (OTR).[1] The principal findings indicate that the reduced reactivity of hydrocarbon emissions and the reduced mass of hydrocarbon emissions are of equal importance. In addition, the behavioral response of consumers also has a discernible impact on fuel consumption and cost effectiveness. These findings imply that the introduction of California reformulated gasoline can be a cost-effective policy as part of a more focused and carefully targeted emissions control strategy.

BACKGROUND

Throughout most of the OTR the control of volatile organic compounds (VOCs) has been identified as critical to reducing ozone levels.[2] The attractiveness of RFG as a potential control strategy results from three principal factors: (1) it controls both the reactivity and total mass of VOC emissions, (2) it is consistent with the seasonal nature of ozone formation, and (3) it reduces emissions from existing vehicles without requiring vehicle modifications. The merits of alternative fuels as a credible strategy to control vehicle exhaust emissions relate to several key provisions of the 1990 Clean Air Act Amendments (CAAA):

- all gasoline sold in the nine worst nonattainment areas must use reformulated gasoline producing a 15% decrease in mass emission of VOCs and air toxics from aggregated 1990 levels and a 25% decrease by 2000

- requirements to control fugitive emissions during transport and refueling

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1. The Ozone Transport Region includes the 11 East Coast states, from Maryland to Maine, several counties in Northern Virginia, and Washington, D.C.

2. NRC (1991) and Carter (1994). The effectiveness of reformulated gasoline in reducing ozone concentrations depends upon local conditions.

- a requirement for a study of air toxics emission of motor vehicles and fuels that addresses the need for and feasibility of controlling these emissions

- nonattainment areas may choose to adopt the reformulated fuels program.

In addition, states can seek further emission reductions to improve air quality by requiring motor vehicles and/or fuel sold in their state to meet the more stringent standards set by California. It has become an issue of debate whether or not states that adopt the California vehicle emission standard must also adopt the California gasoline standards. The State of New York recently adopted the California low emission vehicle (LEV) standards, but not the California gasoline standards. This requires vehicle manufacturers to meet the California LEV standards without the added benefit of the "cleaner" California gasoline. Vehicle manufacturers filed a law suit in 1994 contending that the State of New York must adopt either the California standards for both vehicles and gasoline or neither. However, New York State and most of the OTR did adopt the California LEV program while reserving the latitude to use California gasoline as an additional emissions control measure.

Adding to this debate is the issue of whether California reformulated gasoline is cost-effective in reducing summer ozone concentrations. The magnitude of costs for California gasoline, an estimated cost of $100 million per year for New York alone, has broad economic consequences beyond the transportation sector. To determine the cost of introducing California reformulated gasoline, we consider fuel composition, reduced reactivity of VOC emissions, price effect on demand, fuel efficiency, and fugitive emissions from fuel transport and storage.

The rest of this appendix is organized as follows. The first subsection provides a description of the composition of reformulated fuels, followed by a discussion in the second subsection on the factors affecting emission reductions. The third subsection addresses fugitive emissions from refueling, storage and transport. The fourth through sixth subsections address the response and effect of the increase in fuel costs to consumers. The seventh subsection presents the final results and the eighth subsection provides references.

REFORMULATED GASOLINE

Gasoline and the exhaust from conventionally fueled vehicles are highly reactive in the atmosphere because they are rich in complex hydrocarbons (aromatics and alkenes). A large variety of different compositions for reformulated gasoline contain different proportions of methyl- or ethyl-tributyl ethers (MBTE and ETBE, respectively) and lower levels of other components of conventional gasoline (such as aromatics and olefin fractions). Federal RFG Phase 2, as defined by U.S. EPA, becomes the baseline fuel across the entire

OTR in 2000.[3] Drawing on results of several studies,[4] the following analysis focuses on the incremental cost-effectiveness of California reformulated gasoline (CARB 2) compared to federal reformulated gasoline (U.S. EPA Phase 2). The composition of these fuels is summarized in Table D-1.

Table D-1. Conventional and Reformulated Gasoline.

	Baseline	Reformulated	
	Conventional	U.S. EPA Phase 2	CARB 2
Aromatics, vol. %	31.9	27.6	22.0*
Oxygen, wt. %	.4	2.1	2.0**
Olefins, vol. %	15.8	11.6	4.0*
Benzene, vol. %	1.69	0.95*	0.80*
Sulfur, wppm	449	124	30*
RVP, psi	8.2	6.7	6.6*
Distillation, °F	0	3.0	15.0
50%	210	203	199*
90%	351	344	290*

*At maximum specification.
**At minimum specification.
Source: NYSERDA (1994).

FACTORS AFFECTING EMISSION REDUCTIONS

Reformulated gasoline is a refined petroleum product whose composition is similar to conventional gasoline. The composition of the gasoline is altered to make exhaust products less photochemically reactive, to reduce toxicity, and to lower emission levels.[5] The composition is adjusted by increasing octane enhancing oxygenate organic compounds such as MTBE and ETBE, which reduce levels of aromatics and olefin hydrocarbons. MTBE is the favored additive.[6] The marginal benefits of California reformulated gasoline can be calculated in three ways: (1) using emission reductions based on a mass removal basis and ignoring the differences in reactivity between VOC emissions, (2) using weighted reactivity of total organic gases (TOG) emissions based on results of empirical chamber experiments, and (3) performing air quality

3. Federal RFG Phase 2 has slightly more stringent reed vapor pressure (RVP) standards for states south of Pennsylvania than northern states in the OTR. The more stringent RVP standards are to control the higher rate of evaporative emissions from the higher summer temperatures. Federal RFG Phase 2 Class B has an RVP of 6.8, and Class C has an RVP of 7.6.

4. AQIRP (1993), NRC (1991), and Pechan (1992).

5. A shift from conventional gasoline would presumably reduce atmospheric levels of some toxic compounds such as benzene, toluene, and xylenes, but may also introduce higher levels of formaldehyde, acetaldehyde, dimethyl sulfate, and peroxacetyl nitrate. CARB (1991 and 1992).

6. Pechan (1992).

simulations, which account for the change in emissions species as well as spatial and temporal variation of emission concentrations. For this analysis, the first and second methods are used to assess the benefits of CARB 2 compared with those of U.S. EPA Phase 2 gasoline in 2007, assuming the adoption of the Cal LEV program. The third method is impractical for the study because results from a photochemical model of air quality were not available.

Mass Reductions

Estimates of the mass reduction in VOC emissions draw on a Pechan 1992 study. Mass reductions are determined using MOBIL 5.0. This model requires a significant amount of technical input on vehicle fleet characteristics and driving patterns. A shortcoming of the model is the lack of technical detail used to assess the benefits of reformulated gasoline. The fuel input parameters describe only a few physical characteristics and omit compositional parameters. Consequently, the output of MOBIL 5.0 produces nonspecied VOC emissions based on estimates of evaporative and tail pipe emissions for each vehicle type under specified driving conditions.

Reactivity Estimates

Different VOCs vary considerably in the rate at which they react to increasing ozone levels. Species such as formaldehyde, olefins, and some aromatic hydrocarbons exhibit very high photochemical reactivity, whereas paraffins and alcohols are much less reactive. Reformulated gasoline reduces reactivity by lowering highly reactive aromatics and olefins. Changes in incremental reactivity estimates are based on AQIRP (1993). Failing to adjust the different removal costs for different species and for the composition of emission reductions may underestimate the cost-effectiveness of California reformulated gasoline. For example, Carter (1990) shows that changes in incremental reactivity are equally as effective in reducing ozone as reductions in the mass of emissions.

We use the maximum incremental reactivity (MIR) scale developed by Carter (1990) to account for reactivity.[7] Carter's reactivity factors are designed for low VOC-to-NO_x ratios, and the ambient conditions of the Northeast support the application of these factors as a measure of incremental impacts on ozone formation.[8] The application of Carter's reactivity measure draws on results of AQIRP (1993), which uses a

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7. The fact that incremental reactivities depend on environmental conditions means that no single scale can predict incremental reactivities, or even ratios of incremental reactivities, under all conditions. Reactivity scales oversimplify the complexities of ozone formation, but are a practical choice to ignoring reactivities altogether.

8. Carter (1994).

speciated reactivity weighted estimate of TOG emissions from light duty vehicles.[9] Carter's reactivity weighted scales have been developed over a limited range of atmospheric conditions, but these estimates serve as a reasonable basis for making a comparison of the relative contribution of different gasoline compositions toward ozone formation.[10]

The uncertainty in the MIR reactivity factors results from a series of complex and highly uncertain inputs. For example, the use of Carter's reactivity scales introduces simplifications in the chemistry, composition, and dynamics of the emissions inventory as well as atmospheric transport. The MIR reactivity factors have an average uncertainty of 35% for a 95% level of confidence.[11]

The reactivity estimates are based on measurements from the current vehicle fleet and operating patterns for 2007. The actual vehicle fleet in 2007 will achieve significant reductions in VOC emissions over the current vehicle fleet by adopting existing federal programs. Ignoring these reductions will bias the results toward favoring the use of reformulated fuels. However, the overestimation of emission reductions may be counterbalanced in AQIRP by the use of MOBIL 4.0 to project vehicle emissions. AQIRP acknowledges the likely underestimation of VOC emissions with MOBIL 4.0 and states, "If the models also underestimate vehicle emissions in future years, then the light-duty vehicle contribution to ozone and Reactivity Weighted Emissions will likely be underestimated as well."[12]

AQIRP examined a compositional matrix of four fuel variables: aromatics, MTBE, olefins, and the 90% distillation temperature (T_{90}). AQIRP also evaluated the effects of reducing RVP, adding ETBE (17 vol.%), ethanol (10 vol.%), and 85% methanol, but found little or no statistically significant reductions in the reactivity weighted emissions. The specific reactivities presented in Table D-2 account for exhaust, evaporative and running losses, and refueling/bulk storage.[13] The MIR reactivity factors are developed from an initial fuel matrix with significantly higher aromatics, olefins, and T_{90}.

Table D-2. Reactivity Weighted TOG Emissions from Light-Duty Vehicles

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9. TOG is defined as the sum of all hydrocarbon, aldehyde, ketone, ether and alcohol emissions, and includes methane.

10. Carter's reactivity measures account for the effects of changes in NO_x emissions that may be associated with changes in TOG emissions. The diurnal effects of ozone transport and the spatial variation of emissions are not considered in the weighted reactivity factors.

11. AQIRP (1993).

12. AQIRP (1993).

13. According to AQIRP (1993), the relative contribution of TOG emission for Los Angeles was 79% evaporative and 21% running loss in 1995, and will be 86% evaporative and 14% running loss in 2010.

(effect of change in fuel variables from U.S. EPA phase 2 to CARB 2 on MIR reactivity factors — percent decrease in potential tons ozone per day).			
Aromatic	**MTBE**	**Olefins**	**T_{90}**
2.3%	2.8%	9.3%	4.9%

The assumption of linearity for the incremental reactivities is dampened by two adjustment factors. The first factor accounts for declining incremental reactivities with changes in the composition of the emissions inventory and is set at 0.90. The second factor adjusts incremental reactivities by 0.80 to account for the decrease in emissions reactivity under the California LEV program versus the test fleet.[14] The first and second adjustment factor are combined to produce a composite adjustment factor of 0.72.

Reactivity weighted emissions for TOG account for changes in both the mass and the composition of TOG emissions, and result in a 15% reduction compared with a 4% reduction in TOG mass. The incremental reactivity estimates shown in Table D-2 indicate that reducing olefin content and T_{90} have the largest effects, providing nearly three quarters of the reactivity weighted reductions.[15] MTBE displaces some of the more reactive hydrocarbons commonly associated with evaporative losses from refueling and storage, which account for 16% of the reactivity adjusted reductions.[16] Reducing the aromatic content provides the remaining 13% of the reactivity based reductions.

The reactivity weighted TOG values of AQIRP (1993) tend to predict larger fuel effects than the Urban Airshed Model (UAM) results for the Northeast analyzed by AQIRP. For example, the reactivity scales match the direction of UAM results for olefins, but they predict a greater magnitude of ozone reduction. One explanation for the disparities between the reactivity scales and UAM may rest with the underestimation of VOC emissions from the transportation sector in the emissions inventory used with UAM. A more plausible explanation may be the confounding effects of transport of ozone and its precursors from other regions.[17]

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14. The adjustment factors do not account for reductions in CO emissions or NO_x emissions and their potential benefits for reducing ozone production. Estimates made using Carter's reactivity factors for CO indicate that CO contributes 5% to 16% of the combined TOG and CO reactivity weighted emissions.

15. The high incremental reactivities for olefin reductions coincide with Carter's (1994) results, which show olefins have high incremental reactivities relative to other hydrocarbon species.

16. AQIRP (1993).

17. Aromatic VOCs react with NO_x. A reduction in aromatics enables NO_x to act as a sink for ozone in the near field, but may increase ozone concentrations in far field rural sites with higher VOC/ NO_x ratios.

Another important component of the fuel composition matrix shown in Table D-1 is the sulfur content. Early results by Furey and Monroe (1981) found that reduced sulfur content increases catalyst life and the effectiveness of exhaust systems. In contrast to these early results, Calvert et al. (1993) find that sulfur is an inhibitor of catalyst efficiency.[18] AQIRP (1993) showed that reducing fuel sulfur content can contribute directly to reductions in mass emissions (HC, CO, and NO_x); toxic emissions (benzene, 1,3 butadiene, formaldehyde, and acetaldehyde); and potential ozone formation. The 75% reduction in sulfur concentrations in the fuel accounts for 9% of the reactivity based reductions in the AQIRP (1993) study.[19]

DECREASE IN DEMAND

The projected increase in gasoline prices of 8.5¢/gal reduces the demand for gasoline. The magnitude of this response varies by subregion because of the difference in the elasticity of demand and initial level of consumption. Long-run elasticities of demand are taken from a recent comprehensive survey by Dahl and Sterner (1991). Their survey concludes, "We find a fair degree of agreement concerning average short-run and even long-run income and price elasticities." Recognizing Metro East's extensive public transportation system, Dahl's results are used in a model developed by Drollas (1984):

$$G = f(P_{gas}, P_{trans}, Y, P_{veh}, P_{gas_{t-1}}, Y_{t-1}, G_{t-1}). \qquad \text{(D-1)}$$

The explanatory variable P_{trans} has been introduced to account for the substitution effect of public transportation and yields slightly higher price and income elasticities compared to the other subregions. The lack of a broad-based and significant public transportation system in the remainder of the OTR makes the application of the partial adjustment model shown in Equation D-2 more appropriate:

$$G = f(P_{gas}, Y, G_{t-1}). \qquad \text{(D-2)}$$

Metro East has a well developed public transportation system. Consequently, the elasticity for Metro East was determined as a weighted average of the elasticities produced by Equations D-1 and D-2.[20] The long-run price and income elasticities for the four subregions are presented in Table D-3.

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18. Sulfur inhibits catalyst efficiency by occupying active sites of the catalyst. This effect is reversible by using low sulfur fuel.

19. The reactivity per gram of NMOG increases slightly with the reduced sulfur content, but this effect is offset by a decrease in the total mass of NMOG emissions. Unlike the other components of reformulated gasoline, a decrease in sulfur concentrations produces a slightly more reactive gasoline. However, this increase in reactivity is offset by an even larger decrease in the mass of hydrocarbon emissions.

20. A weight of two-thirds was applied to elasticities of Equation D-2 and one-third to the elasticities of Equation D-3.

Table D-3. Long-Run Price and Income Elasticities.		
Subregion	**LR Price Elasticity**	**Income Elasticity**
Northeast	-0.6	0.8
Metro East	-0.71	1.01
Band I	-0.6	0.8

The long-run price elasticities are negative and range between -0.6 and -0.77, an expected range for a normal good. The income elasticities are perhaps a little more illuminating for explaining differences among the subregions. Elasticities greater than 1.0 are considered luxury goods, and income elasticities less than 1.0 are considered staple commodities. The availability of public transportation as a lower cost substitute for private automobiles in Metro East makes gasoline a luxury good with income elasticities greater than 1.0. For Northeast and Band I, gasoline does not have an effective substitute and appropriately has income elasticities less than 1.0.

The reformulated fuels program increases the variable cost of driving during the summer ozone season, providing an economic incentive consistent with the objective of reducing summer emissions. This differs from historical automotive emissions control programs, which raise the cost of purchasing a new vehicle but do not provide any economic incentive for consumers to curb the polluting activity. The costs of California Phase 2 reformulated gasoline are measured as incremental costs above U.S. EPA Phase 2 reformulated gasoline.[21] The uncertainty of these cost estimates build on the uncertainty in projecting costs for U.S. EPA Phase 2 gasoline. The cost of changing gasoline in the Northeast was analyzed for both NYSERDA (1994) and AQIRP (1993), but with strikingly different results. While the NYSERDA study reports an incremental increase of 12.9 cents per gallon, AQRIP estimates a 7.2 cent per gallon increase, which corresponds more closely to Pechan's and CARB's estimates. Balancing these conflicting results, an increase of 8.5 cents per gallon has been chosen.[22]

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21. DRI/McGraw Hill (1991) and Pechan (1992) developed identical cost premiums of 8.6 cents per gallon for federal Phase 2 reform in the eastern states. Although the agreement between these two reports is comforting, these cost values are based on very limited information about the formulation of federal Phase 2 and are consequently subject to large uncertainty.

22. A small bias may be introduced in the measurement of cost-effectiveness because of differences in the definition of the duration of the summer ozone season. The calculations are based on a summer ozone season starting June 15 and ending September 15. U.S. EPA defines the summer ozone season as starting in May 1 and ending September 30. The planned federal reformulated fuels program will sell "summer" gasoline over the U.S. EPA ozone season. However, the month and a half difference will undoubtedly increase the total program costs, but will have little if any impact on the incremental cost-effectiveness ratio of expenditures divided by tons removed.

The increase in price reduces demand and further contributes to emission reductions by the amounts shown in Table D-4.

Table D-4. Percent of Total Reduction in Demand and Percent of Emissions Reductions from the Price Increase of CARB 2 Gasoline over U.S. EPA Phase 2 Gasoline.

Subregion	Percent Change in Demand	Percent of Emissions Reductions for (NO_x and VOC) Attributable to Price Changes	Increase in Price (cents/gallon)
Northeast	-3.7%	6.9%	8.5
Metro East	-4.5%	9.0%	8.5
Band I	-3.9%	8.1%	8.5

The more densely populated Metro East achieves 9.0% of the emissions reductions from the price effect. This corresponds to a decrease in demand over original consumption levels of 4.5 for Metro East. The relatively more rural subregions of Northeast and Band I reduce demand by slightly less than 4%, and attribute 6.9 and 8.1%, respectively, of their emission reductions to the price effect. The differences in percent reduction between subregions are primarily a function of each subregion's price and income elasticity.

MILES PER GALLON

Increases in gasoline prices cause consumers to shift to more fuel efficient vehicles.[23] The effect on total emissions as consumers purchase more fuel efficient cars is confounded by several factors: (1) increase in VMT, (2) lower emissions rate with increased fuel economy, (3) higher cost of more fuel efficient cars, and (4) the ability to predict these effects based on past behavior. The effects of these competing factors are considered below.

The federal CAFE standards implemented in the early 1970s as a result of the first Middle East oil embargo are rightfully credited with raising fuel economy standards. This assertion can be inferred from the data. The price for gasoline in 1994 dollars has remained unchanged at $1.12 per gallon for 1972 and 1992, but over the same 20 year period the average passenger car fuel economy has risen from 13.9 mpg to 27.8 mpg. On the other hand, the data also include the effect of increasing gas prices on consumer choice. From 1979 to 1980, gasoline prices increased by nearly 50 cents per gallon (1994 dollars), and the average new vehicle fuel economy rose by 4 mpg.

Equation D-3 is estimated by OLS regression with data from MVMA (1993) and NYSERDA (1994):

$$mpg = 15.2 + 0.0569 \text{ fuel price} + 0.486 \, mpg_{t-1} - 1.796 \text{ cents/mi}, R^2 = .99. \quad \text{(D-3)}$$

$$(0.0147) \qquad (0.1164) \qquad (0.3526)$$

The lagged structure of Equation D-3 is used to capture the fact that adaptation takes time and is dynamic. This widely used representation of dynamic behavior is referred to as a geometric lag and has the properties of a partial adjustment model.[24] The anticipated 8.5 cent rise in fuel prices associated with CARB 2 give rise to approximately 0.5 mpg increase in fuel economy.[25] This gain in fuel economy is too small to have a significant effect on emissions.

LOSS OF CONSUMER SURPLUS

The loss of utility due to higher prices at the pump from introducing reformulated gasoline is measured by the loss in consumer surplus. Using the change in consumer surplus as a measure of cost, "out-of-pocket" costs plus the additional value consumers receive from consuming the original level of gasoline are combined. Cost estimates performed by environmental and industry groups uniformly underestimate the costs of switching to reformulated gasoline by omitting the dead weight loss. However, the dead weight represents less than 1% of the total cost of reformulated gasoline, as shown in Table D-5.

Table D-5. Dead Weight Loss of California Reformulated Gasoline.

Subregion	Percentage of Total Program Costs from the Loss in Consumer Surplus
Northeast	-0.075%
Metro East	-0.083%
Band I	-0.078%

23. Although CARB 2 gasoline reduced fuel economy by a few percent, this effect is probably too small to change consumer behavior.

24. The partial adjustment model captures changes in consumer behavior that require adaptive response time. For example, if fuel prices change in one year, but consumers have an established purchase pattern of large and less fuel efficient vehicles, then today's consumption becomes a function of today's price structure as well as earlier purchase patterns.

25. The regression was performed in Excel using Analysis Tools. Plots of the residuals are relatively homogeneously dispersed, showing little sign of autocorrelation.

RESULTS

Emissions savings with CARB 2 are based on estimated fleet emissions from 2000 through 2007. Vehicle fleet average emissions are based on implementation of the California LEV program in 1998 for LDV, LDT1, and LDT2. The effects of electric vehicles and fleet vehicles powered by natural gas are removed before calculating the fleet average emissions. Additionally, the effects of enhanced inspection and maintenance for subregions are integrated into the average fleet emissions and are a factor in explaining the regional differences in costs among subregions shown in Table D-6.

Table D-6. Reactivity Adjusted Percent Removal and Emissions Reduction Cost of CARB 2 Gasoline

Subregion	Percent Reduction		Removal Cost $/Ton	
	NO_x	VOC	NO_x	VOC
Northeast	7%	17%	20,542	2,802
Metro East	6%	21%	27,241	2,388
Band I	6%	18%	18,742	2,408

The differences in control costs between subregions can be explained by the different driving patterns and vehicle inspection and maintenance programs.[26] Adjusting VOC emissions for ozone reactivity improves the cost-effectiveness of VOC emission reductions by approximately 25%. Emission reduction costs are allocated equally between NO_x and VOC emissions. Thus, the smaller reduction in NO_x emissions results in higher removal costs compared to VOCs.

CONCLUSION

Adoption of CARB 2 gasoline by the OTR provides additional reductions in the mass and reactivity of precursor emissions (NO_x and VOC) to tropospheric ozone. The cost-effectiveness of CARB 2 gasoline considers the additional cost over U.S. EPA Phase 2 gasoline. Common methods used to assess the cost-effectiveness of reformulated gasoline measure the mass of emission reductions and the additional costs of production. This approach has been shown to overlook some important factors concerning the benefits of introducing reformulated gasoline.

The reactivity, or ozone forming potential, varies considerably among VOC species. Accounting for this effect results in a 15% reduction in the potential formation of ozone from VOCs, compared with a 4% reduction in the mass of TOG. The estimated increase in price of 8.5 cents per gallon further reduces

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26. The low penetration of the enhanced inspection and maintenance program in Northeast increases the effectiveness of California reformulated gasoline and consequently produces lower control costs.

demand. The price effect accounts for approximately 5% and 3.5% reduction in demand for urban and rural areas, respectively. This decrease in demand results in a relatively insignificant loss in consumer surplus equal to less than 1% of the total program costs. The price increase also causes a negligible increase in the fuel efficiency of motor vehicles. Evaporative emission reductions from storage and transport are approximately 7% for Metro East and 2% for Band I and Northeast.

The overall cost of reformulated gasoline is relatively high compared with other emission control options. However, for the urban corridor of Metro East, reformulated gasoline provides an additional potentially cost-effective source of precursor emission reductions. In addition, unlike most emission control programs, the use of reformulated gasoline can be limited to the summer ozone season.

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Least Cost Control Strategies to Reduce Ozone in the Northeastern Urban Corridor

Report 00-01

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