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The Benefits and Costs of the Clean Air Act, 1970 to 1990

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In Section 812 of the Clean Air Act Amendments of 1990, Congress required EPA to conduct periodic assessments of the benefits and costs of the Act. The first was to be a retrospective study, followed by a series of prospective assessments. This study represents the retrospective study. The study was designed to answer the following specific question: "How do the overall health, welfare, ecological, and economic benefits of Clean Air Act programs compare to the costs of these programs?" To address the question, EPA developed and contrasted two scenarios: a "control scenario" which reflected actual historical implementation of the 1970 and 1977 Clean Air Acts, and a "no-control scenario" which reflected the assumption that air pollution control programs were frozen at their 1970 levels of scope and stringency. The differences between these two scenarios in terms of economic and environmental outcomes were taken to represent the incremental costs and benefits of the Clean Air Act from 1970 to 1990.

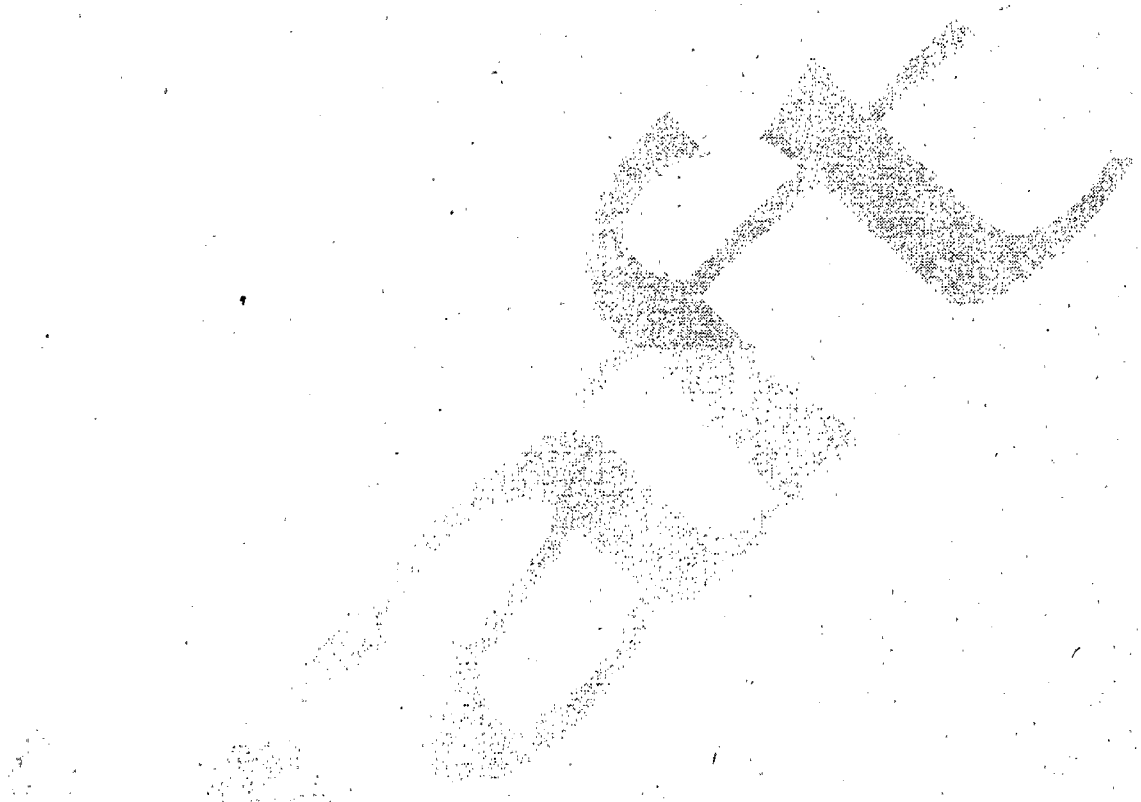
A complex sequence of analytical steps was required to construct and evaluate the control and no-control scenarios and the differences in their respective outcomes. These steps included: (1) direct cost estimation, (2) macroeconomic modeling, (3) emissions modeling, (4) air quality modeling, (5) physical effects estimation, (6) economic valuation, and (7) results aggregation and uncertainty modeling. Each of the first six steps required the adoption of important assumptions and contributed to overall uncertainty in the bottom line estimate of net benefits of the Clean Air Act. The effects of key uncertainties are described, and additional analyses were performed to test the sensitivity of the net benefit results to changes in important assumptions.

Direct expenditures made to comply with 1970 to 1990 Clean Air Act requirements rose steadily in inflation-adjusted terms throughout the study period. Annualizing this stream of expenditures to correct for accrual of costs and benefits beyond 1990 and adjusting the individual year costs for inflation to reflect their equivalent value in 1990 yielded a direct compliance cost estimate for the 1970 to 1990 period equal to 523 billion dollars. This estimate of direct compliance cost is presented as a single point estimate because potentially important uncertainties in the cost estimates could not be quantified. Potential additional costs not reflected in this point estimate include (a) indirect economic effects, such as adverse effects on capital formation, and (b) potential adverse effects on production-related technological innovation, net of potentially beneficial pollution control-related technological innovation.

Direct benefits of the 1970 to 1990 Clean Air Act include reductions in airborne emissions of a variety of pollutants and associated improvements in air quality, human health, human welfare, and ecosystem functioning. For the year 1990, sulfur dioxide (SO_2) emissions were 40 percent lower, primarily due to utilities installing scrubbers and/or switching to lower sulfur fuels. Nitrogen oxides (NO_x) emissions were 30 percent lower, mostly because of the installation of catalytic converters on highway vehicles. Volatile organic compound (VOC) emissions were also lower primarily due to motor vehicle controls, with VOC levels 45 percent lower under the control scenario than under the no-control scenario. Motor vehicle controls were also primarily responsible for a 50 percent reduction in carbon monoxide (CO) emissions under the control scenario. Control scenario lead emissions in 1990 were projected to be about 3,000 tons, reflecting a 99 percent reduction from the projected no-control level of 237,000 tons. These emissions reductions were projected to lead to significant improvements in air quality, including lower rates of acid deposition, lower ambient concentrations of a broad range of pollutants, and improved visibility. Using concentration-response functions for a subset of the potential benefits of these improvements in air quality yielded ranges of quantitative estimates of reduced adverse effects on human health and welfare. Ecosystem changes and a large number of additional human health and welfare could not be quantified in this analysis.

Applying economic valuation functions from the economics literature to a number of these adverse effect changes provided measures of the economic value of air quality-related improvements. Because uncertainties in the physical effects estimation and economic valuation steps were modeled quantitatively, the total direct benefit estimate is presented as a range with associated probabilities. The central case modeling results indicate a range for overall monetizable benefits of 10.5 to 40.6 trillion dollars for the 1970 to 1990 period, a range which is estimated to have a 90 percent probability of including the "true" estimate of total direct monetized benefits. The mean of the distribution of the benefits estimate is 23 trillion dollars (expressed in terms of 1990 value, using a five percent discount rate). This mean estimate of direct monetized benefits is approximately 45 times the estimate of direct costs for the 1970 to 1990 period.

There are numerous important uncertainties associated with these results, particularly on the benefits side. Additional research in critical areas of uncertainty, such as particulate matter-related mortality incidence and economic valuation, might lead to more refined estimates. In addition, further research in the areas of ecosystem effects and effects of air toxics, among others, would allow more comprehensive treatment of benefit categories in future assessments.



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The SAB ACCACA is chaired by Richard Schmalensee of MIT. Members include Morton Lippmann of New York University Medical Center, William Nordhaus of Yale University, Paul Portney of Resources for the Future, Kip Viscusi of Duke University, A. Myrick Freeman of Bowdoin College, Maureen Cropper of the World Bank, Ronald Cummings of Georgia State University, Daniel Dudek of the Environmental Defense Fund, Robert Mendelsohn of Yale University, Wayne Kachel of MELE Associates, William Cooper of Michigan State University, Thomas Tietenberg of Colby College, Paul Lioy

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Executive Summary

Purpose of the Study

Throughout the history of the federal Clean Air Act, questions have been raised as to whether the health and environmental benefits of air pollution control justify the costs incurred by industry, taxpayers, and consumers. For the most part, questions about the costs and benefits of individual regulatory standards continue to be addressed during the regulatory development process through Regulatory Impact Analyses (RIAs) and other analyses which evaluate regulatory costs, benefits, and design variables such as scope, stringency, and timing. There have never been, however, any comprehensive, long-term, scientifically valid and reliable studies which answered the broader question:

"How do the overall health, welfare, ecological, and economic benefits of Clean Air Act programs compare to the costs of these programs?"

To address this void, Congress added to the 1990 Clean Air Act Amendments a requirement under section 812 that EPA conduct periodic, scientifically reviewed studies to assess the benefits and the costs of the Clean Air Act. Congress further required EPA to conduct the assessments to reflect central tendency, or "best estimate," assumptions rather than the conservative assumptions sometimes deemed appropriate for setting protective standards.

This report is the first in this ongoing series of Reports to Congress. By examining the benefits and costs of the 1970 and 1977 Amendments, this report addresses the question of the overall value of America's historical investment in cleaner air. The first Prospective Study, now in progress, will evaluate the benefits and costs of the 1990 Amendments.

Study Design

Designing a study to evaluate effectively the benefits and costs of the entire Clean Air Act over a 20-year period throughout the U.S. was a formidable challenge, particularly given limitations in historical data, scientific tools, and available resources . As a result of these constraints, the levels of geographical, industry-specific and pollutant-specific detail which could be incorporated in the assessment were limited. For example, the emissions data and models available for this study supported development of emissions projections at the state, rather than county, level. While relying on state-level data allows for development of reasonable national-level estimates of the benefits and costs of the Clean Air Act, it is not possible use the results of this study to identify costs and benefits of the Clean Air Act for particular locations. The need to rely on national aggregate compliance expenditure data similarly precludes estimation of costs and benefits by individual company or industry. Finally, atmospheric transport and transformation of pollutants from one species to another (e.g., transformation of gaseous sulfur dioxide to particulate sulfates) make it difficult to estimate benefits and costs by individual pollutant.

Another important feature of the current study is that it assesses the benefits and costs of all air pollution control measures implemented from 1970 to 1990 regardless of the motivation for their adoption. This approach makes it unnecessary to speculate about which actions were taken by states and local governments in response to Clean Air Act mandates and which actions were taken for other reasons.

By allowing for acceptable limitations on the disaggregation of results by government sector, industrial sector, pollutant, and location, the study could be designed and data and models could be selected which allowed EPA to measure the total benefits and costs of historical air pollution controls. In fact, even with these limitations, the present study represents the most extensive evaluation ever completed of the Clean Air Act.

Nevertheless, significant uncertainties usually pervade assessments of environmental program costs and benefits, and the present study is no exception. While there is uncertainty associated with each step of the analysis, resources permitted quantitative assessment of only the physical effects estimation and economic valuation steps. Potential biases and errors in other steps of the analysis are evaluated qualitatively, however, and the Project Team believes that the physical effects and valuation steps are the ones which contribute most to aggregate uncertainty in the estimate of net benefits.

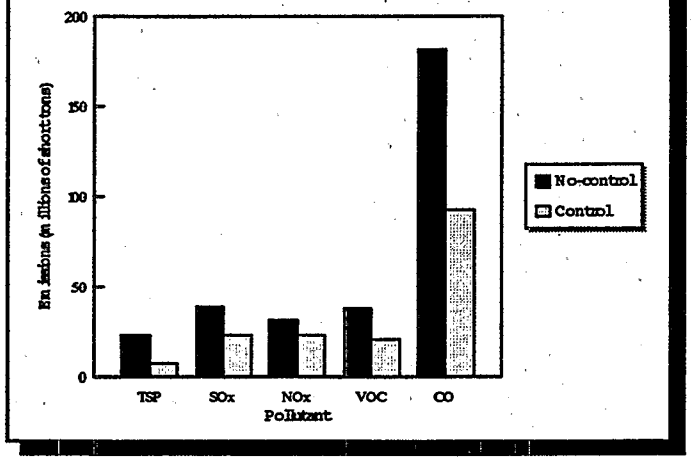
The study derived the benefit and cost estimates by examining the differences in economic, human health, and environmental outcomes under two alternative scenarios: a "control scenario" and a "no-control scenario." The control scenario reflects actual historical implementation of clean air programs and is based largely on historical data. The no-control scenario is a hypothetical scenario which reflects the assumption that no air pollution controls were established beyond those in place prior to enactment of the 1970 Amendments. Each of the two scenarios were then evaluated by a sequence of economic, emissions, air quality, physical effect, economic valuation, and uncertainty models to yield the differences between the scenarios in economic, human health, and environmental outcomes. Details of this analytical sequence are presented in Chapter 1 and are summarized in Figure 4 of that chapter.

Summary of Results

Direct Costs

To comply with the Clean Air Act, businesses, consumers, and government entities all incurred higher costs for many goods and services. The costs of providing goods and services to the economy were higher primarily due to requirements to install, operate, and maintain pollution abatement equipment. In addition, costs were incurred to design and implement regulations, monitor and report regulatory compliance, and invest in research and development. Ultimately, these higher costs

Figure 1. 1990 Control and No-control Scenario Emissions (in millions of short tons).



of production were borne by stockholders, business owners, consumers, and taxpayers.

To tally total compliance expenditures made pursuant to the Clean Air Act over the period from 1970 to 1990, actual dollars spent in a given year were adjusted to reflect their value in 1990. The purpose of this adjustment was to correct for the effects of inflation so that expenditure levels in different years could be compared. In addition, an adjustment was made to the raw expenditure data to take account of the fact that some expenditures in any given year were for control equipment which provided benefits for more than one year. This "annualization" adjustment, which is similar to calculating the value of a year's worth of mortgage payments for a house, allowed for a comparison of costs in a given year with the value of the emissions reductions achieved in that same year. Figure 2 summarizes the historical data on Clean Air Act compliance costs, adjusted both for inflation and for the value of long-term investments in equipment.

These direct cost data show that, even on an inflation-adjusted basis, direct costs to meet requirements of the Clean Air Act increased steadily over the period of the analysis at a rate of increase equal to a little under four percent per year. This compares to a growth rate in Real Gross Domestic Product (GDP) over the same period of about 3¼ percent. The implication of this comparison with GDP is that air pollution control expenditures during the study period increased at a slightly faster rate than total output by the U.S. economy.

Indirect Economic Effects

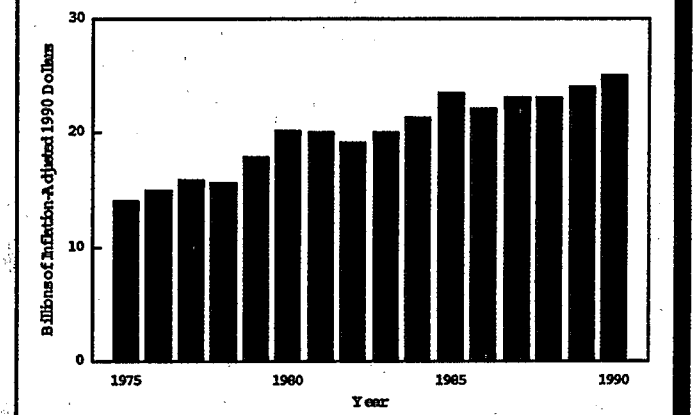
As part of the process of developing input data to model emissions under the control and no-control scenarios, a macroeconomic model was run to estimate changes in key economic variables such as total electricity sales. In addition to providing economic indicators needed for the emissions modeling, the macroeconomic model provided estimates of other, indirect economic effects of the Clean Air Act. For example, the macroeconomic model runs for the control and no-control scenario provided estimates of overall economic production, prices for goods and services from various sectors, levels and patterns of investment, and patterns of employment.

While these indirect effect estimates are reported in Chapter 2 and Appendix A, EPA concluded that these projections do not provide a complete and satisfactory indication of the overall economic effects of the Clean Air Act and are not presented as primary results of this analysis. The principal limitation of the macroeconomic results is that, while they do reflect some of the indirect economic costs of air pollution control expenditures, they do not reflect the indirect economic benefits of the pollution control achieved. For example, the macroeconomic model results suggest there was a decrease in total U.S. production due to displacement of capital investment caused by pollution control expenditures; however, the macroeconomic modeling does not capture the increase in total U.S. production achieved by reductions in air pollution-related worker illness and absenteeism. The macroeconomic modeling also omits the economic expenditure reductions achieved by the Clean Air Act, such as reductions in medical expenses for air pollution-related illness and disease. These savings would offset, at least in part, the capital displacement effect of compliance expenditures. This imbalanced treatment by the macroeconomic model of the effects of the Clean Air Act led EPA to conclude that the most appropriate comparisons in the context of this study are between the direct costs and the direct benefits of the Clean Air Act.

Emissions

Emissions were substantially lower by 1990 under the control scenario than under the no-control scenario, as shown in Figure 2. Sulfur dioxide (SO_2) emissions were 40 percent lower, primarily due to utilities installing scrubbers and/or switching to lower sulfur fuels. Nitrogen oxides (NO_x) emissions were 30 percent lower by 1990, mostly because of the installation of catalytic converters on highway vehicles. Volatile organic compound (VOC) emissions were also lower primarily due to motor vehicle controls, with 1990 VOC levels 45 percent lower under the control scenario than under the no-control scenario. Motor vehicle controls were also primarily responsible for a 50 percent reduction in 1990 carbon monoxide (CO) emissions under the control scenario.

Figure 2. Total Direct Costs of the CAA (in billions of inflation-adjusted 1990 dollars.)



For particulate matter, it is important to recognize the distinction between reductions in directly emitted particulate matter and reductions in ambient concentrations of particulate matter in the atmosphere. As discussed further in the next section, changes in particulate matter air quality depend both on changes in emissions of total suspended particulates (TSP) and on changes in emissions of gaseous pollutants, such as sulfur dioxide and nitrogen oxides, which can be converted to particulate matter through chemical transformation in the atmosphere. Emissions of primary, or directly emitted, total suspended particulates were 75 percent lower under the control scenario by 1990 than under the no-control scenario. This substantial difference is primarily due to vigorous efforts in the 1970s to reduce visible emissions from utility and industrial smokestacks.

Control scenario lead emissions for 1990 were projected to be about 3,000 tons, reflecting a 99 percent reduction from the projected no-control level of 237,000 tons. The vast majority of the difference in lead emissions under the two scenarios is attributable to reductions in the use of leaded gasoline. By 1990, reductions in highway vehicle emissions accounted for 221,000 of the total 234,000 ton reduction in total airborne lead emissions under the control scenario. (Results for lead are not shown in Figure 2 because the absolute levels of lead emissions are measured in thousands, not millions, of tons, and will not be discernible on a graph of this scale.)

As discussed in Chapter 3, there are important uncertainties in the emissions estimates. These include (a) potential errors in the macroeconomic model results used to configure the emissions models and (b) potential overestimation of no-control scenario emissions resulting from the use of 1970 emission factors throughout the 1970 to 1990 period.

Air Quality

The substantial reductions in air pollutant emissions achieved by the Clean Air Act translate into significantly improved air quality throughout the U.S. For sulfur dioxide, nitrogen oxides, and carbon monoxide, the improvements in air quality under the control scenario were treated as proportional to the estimated reduction in emissions. This is because, for these pollutants, changes in ambient concentrations in a particular area are strongly related to changes in emissions in that area. While the differences in control and no-control scenario air quality for each of these pollutants do vary from place to place because of local variability in emissions reductions, the overall national average improvements in air quality for these pollutants achieved by 1990 were: 40 percent improvement for sulfur dioxide, 30 percent improvement for nitrogen oxides, and 50 percent improvement for carbon monoxide.

The differences in ambient concentrations of ground-level ozone and particulate matter under the two scenarios, however, are not so straightforward. Long-range transport through the atmosphere, non-linear formation mechanisms, and other complexities meant that air quality models had to be used to translate changes in emissions of precursors of these pollutants to changes in ambient concentrations of ozone or particulate matter. Estimating differences in acid deposition and visibility in the Eastern U.S. also required the use of air quality models since they are not necessarily directly proportional to changes in emissions of pollutants which cause these effects.

Reductions in ground-level ozone were achieved through reductions in its precursor pollutants, particularly volatile organic compounds and nitrogen oxides.[1] The differences in ambient ozone concentrations estimated under the control scenario varied significantly from one location to another, primarily because of local differences in the relative proportion of VOC and NO_x, in meteorology and in precursor emissions reductions. On a national average basis, however, ozone concentrations in 1990 under the control scenario were about 15 percent lower than projected under the no-control scenario. For several reasons, this overall reduction in ozone is significantly less than the estimated 30 percent reduction in precursor nitrogen oxides and the estimated 45 percent reduction in precursor volatile organic compounds. First, significant natural (i.e., biogenic) sources of VOCs limit the level of ozone reduction achieved by reductions in man-made (i.e., anthropogenic) VOCs. Second, current knowledge of atmospheric photochemistry suggests that ozone reductions will tend to be proportionally smaller than reductions in precursor emissions. Finally, the plume model system used to estimate changes in urban ozone for this study is incapable of handling long-range transport of ozone from upwind areas, complex flow phenomena, and multi-day pollution events in a realistic manner. While regional scale ozone models address these phenomena more effectively, resource limitations precluded using these more sophisticated but costly models for this study.

There are many pollutants which contribute to ambient concentrations of particulate matter. The relative contributions of these individual pollutant species to ambient particulate matter concentrations vary from one region of the country to the next, and from urban areas to rural areas. The most important particle species, from a human health standpoint, are the fine particles which can be respired deep into the lungs. While some fine particles are directly emitted by sources, the most important fine particle species

[1] Ambient NO_2 concentrations are driven by anthropogenic emissions whereas ambient VOCs result from both anthropogenic and biogenic sources (e.g., terpenes emitted by trees).

are actually formed in the atmosphere through chemical conversion of gaseous air pollutants. These species are referred to as secondary particles. The three most important secondary particles are (1) sulfates, which derive primarily from sulfur dioxide emissions; (2) nitrates, which derive primarily from nitrogen oxides emissions; and (3) organic aerosols, which can be directly emitted as well as a result of volatile organic compound emissions. This highlights an important and unique feature of particulate matter as an ambient pollutant: more than any other pollutant, reductions in particulate matter are actually achieved through reductions in a wide variety of air pollutants. In other words, controlling particulate matter means controlling "air pollution" in a very broad sense.

Because of its comprehensive coverage of pollutants contributing to ambient particulate matter, the present study is uniquely configured to illustrate the broader range of benefits achieved by historical reductions in "air pollution." The results of this analysis indicate that the reductions in sulfur dioxide, nitrogen oxides, volatile organic compounds, and directly-emitted particles achieved by the Clean Air Act resulted in an overall, national average reduction in total ambient particulate matter of about 45 percent by 1990.

Reductions in sulfur dioxide and nitrogen oxides also translate into reductions in formation, transport, and deposition of secondarily formed acidic compounds such as sulfate and nitric acid. These are the principal pollutants responsible for acid precipitation, or "acid rain." Regional acid deposition modeling of the control and no-control scenarios indicates that, by 1990, sulfur and nitrogen deposition were significantly higher under the no-control scenario throughout the 31 Eastern States covered by the model. Percentage increases in sulfur deposition under the no-control scenario ranged up to more than 40 percent in the upper Great Lakes and Florida-Southeast Atlantic Coast areas and were due, primarily, to significant projected increases in the use of high-sulfur fuels by utilities in the upper Great Lakes and Gulf Coast states. Nitrogen deposition also was significantly higher under the no-control scenario, with percentage increases reaching levels of 25 percent or higher along the Eastern Seaboard. This higher level of nitrogen deposition can be attributed primarily to higher projected emissions of nitrogen oxides from motor vehicles.

Finally, decreases in ambient concentrations of light-scattering pollutants, such as sulfates and nitrates, under the control scenario were estimated to lead to perceptible improvements in visibility throughout the Eastern states and Southwestern urban areas modeled for this study.

The assumptions required to estimate hypothetical no-control scenario air quality conditions are particularly significant sources of uncertainty in the estimates of air quality change, especially for those pollutants which are not linearly related to changes in associated emissions such as ozone and particulate matter. As discussed in Chapter 4, limitations on geographic coverage and lack of access to state-of-the-art models, among other factors, led to significant uncertainties in the estimates of air quality change between the scenarios. Most of these uncertainties are expected to lead to biases which are either unknown in direction or which tend toward underestimation of air quality improvements due to the Clean Air Act.

Physical Effects

The lower ambient concentrations of sulfur dioxide, nitrogen oxides, particulate matter, carbon monoxide, ozone and lead projected under the control scenario yielded a substantial variety of human health, welfare and ecological benefits. For a number of these benefit categories, quantitative functions

MISPRINT. SEE TABLES 13, 14 FOR CORRECT NUMBERS.

were available from the scientific literature which allowed estimation of the reduction in incidence of adverse effects. Examples of these categories include the human mortality and morbidity effects of a number of pollutants, changes in neurobehavioral effects among children caused by exposure to lead and changes in visibility impairment.

A number of benefit categories, however, could not be quantified and/or monetized for a variety of reasons. In some cases, strong scientific evidence of an effect existed, but data were still too limited to support quantitative estimates of incidence reduction (e.g., changes in lung function associated with long-term exposure to ozone). In other cases, substantial scientific uncertainties prevailed regarding the existence and magnitude of adverse effect (e.g., contribution of ozone to air pollution-related mortality). Finally, there were effects for which there was sufficient information to estimate incidence reduction, but for which there were no available economic value measures; thus reductions in adverse effects could not be expressed in monetary terms. Examples of this latter category include pulmonary function decrements caused by acute exposures to ozone and reduced time to onset of angina pain caused by carbon monoxide exposure.

Table 1. Selected Health Benefits of the CAA, 1970-1990 (in thousands of cases reduced per year, except as noted).

Health Effect		1975	1980	1985	1990
Mortality (PM_{10},O_3,SO_2,Pb) (thousands)	high	38	97	124	140
	mid	20	54	70	79
	low	11	30	40	45
Heart Attacks (Pb) (thousands)	high	1	9	19	24
	mid	1	7	14	18
	low	1	5	10	13
Strokes (Pb) (thousands)	high	1	5	10	13
	mid	1	4	8	10
	low	1	3	6	7
Respiratory symptoms (SO_2) (thousands)		66	187	165	146
Respiratory illness (NO_2) (millions)		1	4	9	15
Hypertension (Pb) (millions)	high	1	6	12	16
	mid	1	5	10	13
	low	1	4	8	10

Table 1 provides a summary of some of the key differences in estimated human health outcomes under the control and no-control scenarios. Results are presented as thousands of cases avoided in 1990 due to control of the pollutants listed in the table and reflect reductions estimated for the entire U.S. population living in the 48 continental states.

Numerous scientific studies were used as sources of the concentration-response functions applied to estimate differences in premature mortality, morbidity, and other outcomes under the two scenarios. These different studies often implied different ranges of estimates for each of these outcomes, contributing to uncertainty in the quantitative estimates of incidence. These uncertainties were captured quantitatively through the use of Monte Carlo computer modeling, which is briefly described later in this Executive Summary and described in more detail in Chapter 7. Table 1 Summarizes the ranges of estimates generated by the Monte Carlo modeling for each of the monetizable benefits of the CAA.

In Table 1, the 5th and 95th percentile estimates for each effect reflect the ends of a range which, based on the available studies, is 90 percent likely to include the true estimated incidence for that effect. The mean estimate for each effect is the average of all of the Monte Carlo-generated estimates for that effect.

Perhaps the most important uncertainty associated with the results presented in Table 1 is in the estimation of particulate matter-related mortality. For this particular endpoint, considerable scientific uncertainty remains concerning the influence of elevated particulate concentrations on prematurity of death. Table 1 presents results based on both long-term and short-term studies. The short-term studies provide a basis for estimating incidences of premature mortality associated with acute episodes of exposure to elevated particulate matter, but the magnitude of life-shortening and the effect of long-term exposures to elevated particulate matter are not discernible from these studies. The long-term studies address these two specific deficiencies in the short-term studies but may be confounded by the effects of pollutant exposures occurring prior to the period of the long-term study. The estimates presented in Table 1 are therefore not additive but should be viewed as an expression of the plausible, overall range of avoided premature mortality associated with particulate matter exposure. The estimates of premature mortalities avoided due to reduced lead exposure *are* additive with the PM-related short-term and long-term exposure estimates.

Adverse human health effects of the Clean Air Act "criteria pollutants" sulfur dioxide, nitrogen oxides, ozone, particulate matter, carbon monoxide, and lead dominate the quantitative estimates presented in this report in part because knowledge of physical relationships is greatest for these pollutants. The Clean Air Act yielded other benefits, however, which are important even though they may have been uncertain and/or difficult to quantify. The other principal benefit categories which, for a variety of reasons, could not be satisfactorily quantified include all benefits accruing from reductions in hazardous air pollutants (also referred to as air toxics); reductions in damage to cultural resources, buildings, and other materials; reductions in adverse effects on wetland, forest, and aquatic ecosystems; and a variety of additional human health and welfare effects of criteria pollutants. A complete list of the major benefits which could not be monetized is provided in Table 2.

In addition to controlling the six criteria pollutants, the 1970 and 1977 Clean Air Act Amendments led to reductions in emissions of a small number of hazardous air pollutants. Control of these pollutants resulted both from regulatory standards set specifically to control hazardous air pollutants and from incidental reductions achieved through programs aimed at controlling criteria pollutants. Due to resource and data constraints, however, reliable quantitative estimates of reductions in adverse effects of hazardous air pollutants could not be developed for this report.

Reductions in both hazardous air pollutants and criteria pollutants likely yielded widespread improvements in the functioning and quality of aquatic and terrestrial ecosystems. In addition to any intrinsic value to be attributed to these ecological systems, human welfare is enhanced through improvements in a variety of ecological services. For example, protection of freshwater ecosystems achieved through reductions in deposition of acidic air pollutants may have improved commercial and recreational fishing. Other potential ecological benefits of reduced acid deposition include improved wildlife viewing, maintenance of biodiversity, and nutrient cycling. Increased growth and productivity of U.S. forests may have resulted from reduced emissions of ozone-forming precursors, especially volatile organic compounds and nitrogen oxides. More vigorous forest ecosystems in turn yield a variety of benefits, including increased timber production; improved forest aesthetics for people enjoying outdoor activities such as hunting, fishing, and camping; and improvements in ecological services such as nutrient cycling and temporary sequestration of global warming gases. Again, due to resource and data limitations these improvements in ecological conditions have not been quantified in this assessment.

Table 2. Major Nonmonetized Benefits of the Clean Air Act.

Pollutant	Nonmonetized Benefits
Particulate Matter	Changes in Pulmonary Function Other Chronic Respiratory Diseases Inflammation of the Lung Chronic Asthma and Bronchitis
Ozone	Changes in Pulmonary Function Increased Airway Responsiveness to Stimuli Centroacinar Fibrosis Inflammation of the Lung Immunological Changes Chronic Respiratory Diseases Extrapulmonary Effects (i.e., other organ systems) Forest and other Ecological Effects Materials Damage
Carbon Monoxide	Decreased Time to Onset of Angina Behavioral Effects Other Cardiovascular Effects Developmental Effects
Sulfur Dioxide	Respiratory Symptoms in Non-Asthmatics Hospital Admissions Agricultural Effects Materials Damage Ecological Effects
Nitrogen Oxides	Increased Airway Responsiveness to Stimuli Decreased Pulmonary Function Inflammation of the Lung Immunological Changes Eye Irritation Materials Damage Eutrophication (e.g., Chesapeake Bay) Acid Deposition
Lead	Cancer Cardiovascular Diseases Reproductive Effects in Women Other Neurological, Metabolic Effects in Children Fetal Effects from Maternal Exposure, inc IQ Loss Ecological Effects
Air Toxics	All Human Health Effects Ecological Effects

In considering the quantitative results of this analysis, it is important to remember the potential significance of the benefits which could not be quantified in physical terms. This caveat is even more significant for consideration of the monetary benefit results presented below. The range of benefits included in the economic assessment is further reduced

because some effects, such as carbon monoxide-related reduction in time to onset of angina attack, could be quantified in physical terms but not valued in economic terms.

Economic Valuation

Estimated reductions in the incidence of adverse effects represent an important, and to some people a sufficient, basis for characterizing the benefits of air pollution control. However, to aggregate the full range of benefits and compare them to costs, a common unit of measure must be applied. The metric most commonly used for cost-benefit analysis is dollars. Assigning monetary values to all effects which are amenable to economic valuation permits a summation of monetized direct benefits and subsequent comparison to the direct compliance costs of the Clean Air Act.

Before proceeding through this step, it is important to recognize the substantial controversies and uncertainties which pervade attempts to characterize adverse human health and ecological effects of pollution in dollar terms. To many, dollar-based estimates of the value of avoiding outcomes such as loss of human life, pain and suffering, or ecological degradation do not capture the full and true value to society as a whole of avoiding or reducing these effects. Adherents to this view tend to favor assessment procedures which (a) adopt the most technically defensible dollar-based valuation estimates for analytical purposes but (b) leave the moral dimensions of policy evaluation to those who must decide whether, and how, to use cost-benefit results in making public policy decisions. This is the paradigm adopted in the present study. Given the Congressional mandate to perform a cost-benefit study of the Clean Air Act, the Project Team has endeavored to apply widely-recognized, customary techniques of Applied Economics to perform this cost-benefit analysis. However, EPA believes there are social and personal values furthered by the Clean Air Act which have not been effectively captured by the dollar-based measures used in this study. Therefore, EPA strongly encourages readers to look beyond the dollar-based comparison of costs and benefits of the Clean Air Act and consider the broad social value of the reductions in adverse health and environmental effects which have been achieved as well as any additional adverse social consequences of regulation which may not be reflected in the cost estimates reported herein.

For this report, unit valuation estimates were derived from the economic literature and reported in dollars per case reduced for health effects, and dollars per unit of avoided damage for welfare effects. Similar to estimates of physical effects provided by health studies, the monetary values of benefits for this study are reported both in terms of mean value and as a range of estimates. These value ranges, and approaches used for each of the effects monetized in this study, are described in Chapter 6.

As with other components of the assessment, there may be significant uncertainties concerning estimates of economic values. Potential errors in the estimates arise for a variety of reasons. One important example is the uncertainty caused by application of monetary values derived from one circumstance (e.g., wage compensation demanded for accepting riskier jobs) to a different circumstance (e.g., the value of reductions in risk from exposure to air pollution). Substantial uncertainties also apply in cases where survey-based approaches, such as "contingent valuation," are used to estimate the monetary value of reductions in risk of an adverse effect.

These uncertainties are discussed in detail in Chapter 6 and in Appendix I. As for physical effect estimates, the ranges of unit values were utilized in the Monte Carlo modeling to evaluate the effect of valuation uncertainty on the overall monetary benefit estimates.

Monetized Benefits and Costs

The total monetized economic benefit attributable to the Clean Air Act was derived by applying the unit values (or ranges of values) to the stream of monetizable physical effects estimated for the 1970 to 1990 period. In developing these estimates, steps were taken to avoid double-counting of benefits. In particular, it was assumed that studies providing different measures of a given effect were providing alternative estimates of the same effect. For example, the Project Team had access to several studies estimating air pollution-related "restricted activity days" of various degrees of severity, as well as studies which estimated "lost work days." To avoid double-counting, the various "restricted activity" health effects were considered to be alternative measures for a single, combined endpoint.

Table 3. Central Estimates of Economic Value per Unit of Avoided Effect (in 1990 dollars).

Endpoint	Valuation (mid-estimate)
Mortality	$4,800,000 per case
Heart Attacks	$587,000 per case
Strokes	$587,000 per case
Hospital Admissions	
Respiratory	$7,500 per case
Ischemic Heart Disease	$10,000 per case
Congestive Heart Failure	$8,000 per case
Respiratory Illness and Symptoms	
Upper Respiratory Illness	$18 per case
Lower Respiratory Illness	$10 per case
Acute Bronchitis	$45 per case
Acute Respiratory Symptoms	$17 per case
Work Loss Days	$83 per day
Restricted Activity Days	$38 per day
Asthma Attacks	$32 per case
IQ Changes	
Lost IQ Points	$5,550 per IQ pt.
Incidence of IQ < 70	$52,700 per case
Hypertension	$680 per year per case
Decreased Worker Productivity	Direct Economic Valuation
Visibility	Direct Economic Valuation
Household Soiling	Direct Economic Valuation
Agriculture (Net Surplus)	Est. Change in Econ. Surplus

To evaluate the total, aggregate effect of uncertainties in the analysis, a Monte Carlo computer model was used to calculate the estimates of total monetary benefit.

First, the model selected an estimate of the incidence of a given physical effect from the range of possible incidence estimates calculated during the physical effects step. Second, the model selected an estimate of the economic value of reducing that effect from the range of values derived from the economics literature. Third, the model multiplied the reduction in incidence by the unit value to derive one estimate of the monetary benefit of reductions in that effect. The model then repeated this procedure over and over again sampling new values from the range of estimated incidence and the range of economic value. After running through this procedure thousands of times, a range of possible outcomes was compiled, with each outcome carrying with it a specific probability of occurrence.

The idea behind this approach is to allow presentation of the bottom-line results in a way which captures the combined effect of uncertainties in key variables, such as the concentration-response functions and the unit economic values. It must be emphasized that the results presented herein reflect only the uncertainties in those two particular variables because physical effects estimation and economic valuation

were the only steps for which uncertainty could be captured quantitatively. Quantifying the uncertainties associated with earlier steps in the analytical sequence would have required multiple runs of the macroeconomic, emissions, and/or air quality models. This was far beyond the scope of the resources available for this project. However, prior experience in regulatory analysis has demonstrated repeatedly that the physical effects estimation and economic valuation steps contribute much more to overall uncertainty than these earlier steps.

It is also important to emphasize that the central case results presented below reflect the specific analytical assumptions and methodologies adopted by the Project Team. The potential effects of adopting alternative assumptions for two important variables, the discount rate and the valuation method for particulate-related mortality, were assessed separately by performing additional sets of Monte Carlo model runs. Varying one of these factors at a time, the sensitivity of the net benefit estimates to changes in each of these factors was evaluated quantitatively. These sensitivity analyses, summarized in Chapter 7, indicate that (1) adopting discount rates of three or seven percent in lieu of the central case five percent makes little difference in the mean or range of the net benefit estimates and (2) adjusting the value of particulate-related mortality downward to reflect the relatively advanced age and/or poor health status of many victims may have a potentially significant influence on the benefit estimates but would not be expected to change the basic finding of substantial positive net benefits.

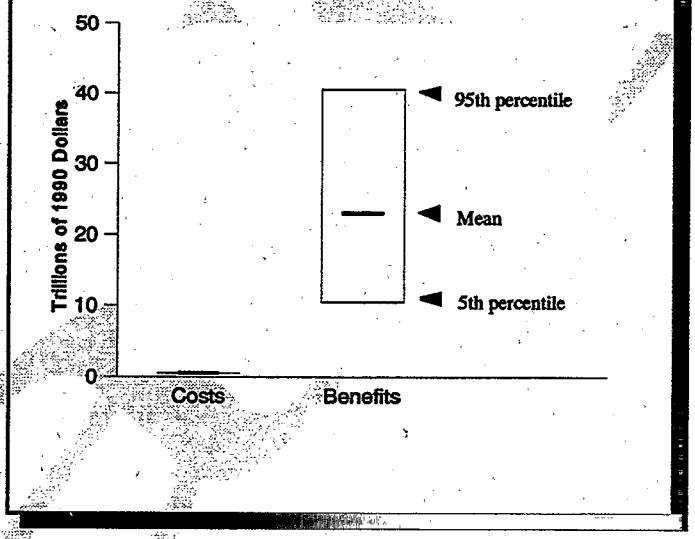
By adding up the ranges of values for each effect obtained by the procedure described above, a range of possible economic values for the estimated differences in physical outcomes under the control and no-control scenarios could be derived. Again, each of the possible outcomes in this range carries with it a certain probability of occurring. Based on this probability information, and adding up the results obtained for the entire 20 year period of the study, the following statement can be made regarding the central case results:

The Project Team found that 90 percent of the credible estimates of the total monetized benefits of the Clean Air Act realized during the period from 1970 to 1990 were within the range of 10.5 to 40.6 trillion dollars, with a central estimate of 23.0 trillion dollars (in 1990-value dollars, discounted at 5 percent). By comparison, annualized direct costs of compliance equal approximately 523 billion dollars over the same period. Subtracting total costs from total benefits, net direct benefits of 1970 to 1990 controls attributed in this analysis to the Clean Air Act were within a 90 percent probability internval of 10.0 to 40.1 trillion dollars, with a central estimate of 22.5 trillion dollars. However, even this broad range of estimated net benefits would be expected to expand further if analytical uncertainties associated with compliance cots, macroeconomic effects, emissions projections, and air quality modeling could be quantified and incorporated in the uncertainty analysis. Finally, the central estimate of 23.0 trillion dollars in benefits may be significantly underestimated due to the exclusion of large numbers of benefits from the monetized benefit estimate (e.g., all air toxics effects, ecosystem effects, numerous human health effects).

Figure 3 provides a graphical representation of the estimated range of total direct monetarized benefits and compares this range to estimated direct compliance costs. Clearly, even the lower bound estimate of monetized benefits substantially exceeds the costs of the historical Clean Air Act. As shown in the more detailed, year-to-year results presented in Chapter 7, monetized benefits consistently and substantially

exceeded costs throughout the 1970 to 1990 period.

Figure 3. Total Direct Costs and Monetized Direct Benefits of the Clean Air Act, 1970 to 1990 (in trillions of 1990 dollars).



Conclusions and Future Directions

First and foremost, the conclusion to be drawn from these results is that the benefits of the Clean Air Act and associated control programs substantially exceeded costs. Even considering the large number of important uncertainties permeating each step of the analysis, it is extremely unlikely that the converse could be true.

A second important implication of this study is that a large proportion of the monetized benefits of the historical Clean Air Act derive from reducing two pollutants: lead and particulate matter[2] (see Table 1). Some readers may argue that, while programs to control these two pollutants may have been worthwhile, many other historical Clean Air Act programs would not pass a benefit-cost test when considered in isolation. While this may or may not be true, this analysis provides no evidence to support or reject such conjectures. On the cost side, the historical expenditure data used in this analysis are not structured in ways which allow attribution of control costs to specific programs or standards. On the benefit side, most control programs yielded a variety of benefits, many of which included reductions in other pollutants such as ambient particulate matter. For example, new source performance standards for sulfur dioxide emissions from coal-fired utility plants yielded benefits beyond those associated with reducing exposures to gaseous sulfur dioxide. The reductions in sulfur dioxide emissions also led to reductions in ambient fine particle sulfates, yielding human health, ecological, and visibility benefits. Even so, it is likely that some specific historical programs or standards may not have yielded monetized benefits in excess of costs.

Although this study indicates that the 1970 to 1990 cumulative direct benefits of the Clean Air Act clearly exceeded the cumulative direct cots of those controls, these results do not inform the question of whether additional future controls would be cost-beneficial. Examining this issue is the central purpose of the first Section 812 Prospective Study now in progress.[3] As the first in an ongoing series of benefit-cost

[2]Ambient particulate matter results from emissions of a wide array of precursor pollutants, including sulfur dioxide, nitrogen oxides, and organic compounds.

[3]Nevertheless, a recent peer-reviewed study examining sulfate-related health benefits alone of the 1990 Amendments pertaining to acid rain control found benefits well in excess of the total estimated costs of the 1900 Amendments. See hagler Bailly Consulting, *Human Health Benefits from Sulfate Reductions Under Title IV of the 1990 Clean Air Act Amendments,* submitted to U.S. Environment Protection Agency/Office of Air and Radiation/Office of Atmospheric Programs/Acid Rain Division, November 1995.

studies of future Clean Air Act programs, this study will evaluate the specific, incremental costs and benefits of new control programs initiated subsequent to the 1990 Amendments.

As for the Retrospective Study, the first Prospective Study is designed to complement, not substitute for, the regulatory analyses developed for individual programs or standards. Nevertheless, the first Prospective Study will differ from the Retrospective Study in that it will be structured to allow comparisons of the costs and benefits of major program areas, rather than just the totality of the 1990 Amendments. This will be feasible largely because, unlike the Retrospective Study, substantial high-quality, program-specific benefit and cost data have already been developed which are consistent with the specific control and baseline scenarios to be analyzed.

Another important element of the first Prospective Study is that the Project Team will seek, resources permitting, to expand the range and depth of the benefits analysis. In particular, efforts will be made to develop valid and reliable estimates of the benefits of future reductions in hazardous air pollutants, one of the key program areas addressed more fully by the 1990 Amendments than by the 1970 and 1977 Amendments. In addition, efforts will be made to provide a more thorough and informative assessment of the ecological effects of reductions in both criteria pollutants and hazardous air pollutants. EPA's view is that ecological benefits of air pollution controls were substantial, even if they could not be quantified for this study. In addition, ecological effects have social and political significance far in excess of any apparent dollar-based, economic measures of value. As such, the Project Team places a high priority on improving and expanding assessment of ecological effects of air pollution controls.

This retrospective study highlights important areas of uncertainty associated with many of the monetized benefits included in the quantitative analysis (see Appendix J). The sensitivity analyses presented in Chapter 7 demonstrate that additional research in the areas of particulate matter-related mortality incidence, valuation of premature mortality, and valuation of particulate-related chronic bronchitis, among others, might reduce critical uncertainties in future assessments.

Finally, the results of this retrospective study provide useful lessons with respect to the value and the limitations of cost-benefit analysis as a tool for evaluating environmental programs. Cost-benefit analysis can provide a valuable framework for organizing and evaluating information on the effects of environmental programs. When used properly, cost-benefit analysis can help illuminate important effects of changes in policy and can help set priorities for closing information gaps and reducing uncertainty. Such proper use, however, requires that sufficient levels of time and resources be provided to permit careful, thorough, and technically and scientifically sound data-gathering and analysis. When cost-benefit analyses are presented without effective characterization of the uncertainties associated with the results, cost-benefit studies can be used in highly misleading and damaging ways. Given the substantial uncertainties which permeate cost-benefit assessment of environmental programs, as demonstrated by the broad range of estimated benefits presented in this study, cost-benefit analysis is best used to inform, but not dictate, decisions related to environmental protection policies, programs, and research.

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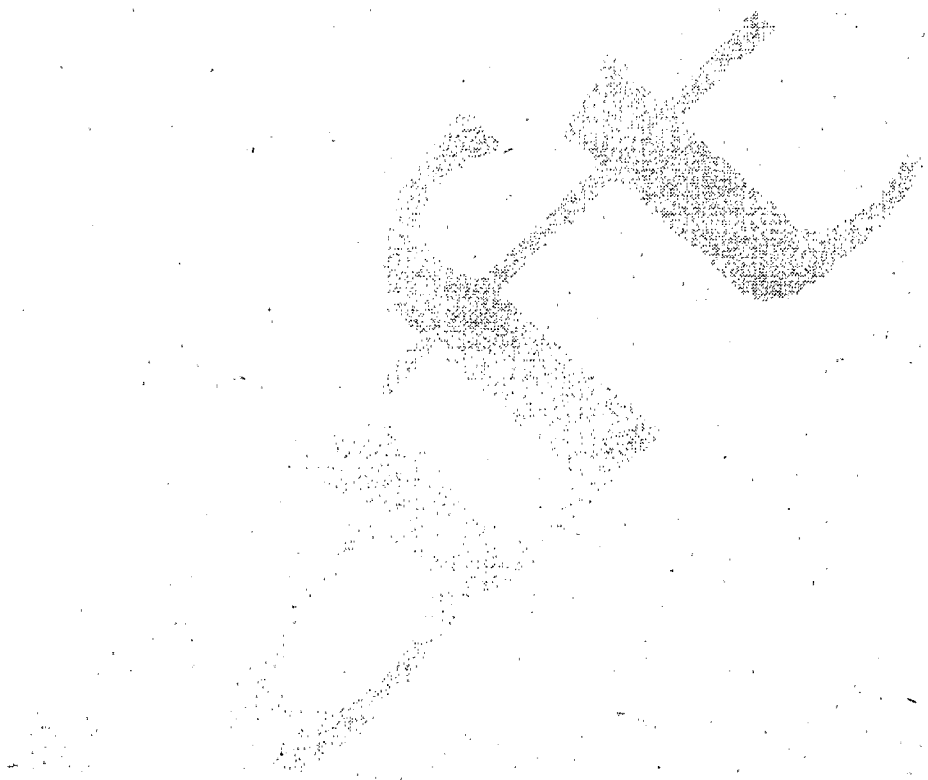

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Acronyms and Abbreviations

μeq/L	microequivalents per liter
$\mu g/m^3$	micrograms per cubic meter
μg	micrograms
μm	micrometers, also referred to as microns
ACCACA	SAB Advisory Council on Clean Air Compliance Analysis
ACCACAPERS	SAB ACCACA Physical Effects Subcommittee
AGSIM	AGricultural SImulation Model
AIRS	EPA Aerometric Information Retrieval System
Al^{3+}	aluminum
ANC	acid neutralizing capacity
ANL	Argonne National Laboratories
APPI	Argonne Power Plant Inventory
AQCR	Air Quality Control Region
ARGUS	Argonne Utility Simulation Model
ASI	Acid Stress Index
ATERIS	Air Toxic Exposure and Risk Information System
ATLAS	Aggregate Timberland Assessment System
AUSM	Advanced Utility Simulation Model
BEA	Bureau of Economic Analysis
b_{ext}	total light extinction
BG/ED	Block Group / Enumeration District
BI	atherothrombotic brain infarction
BID	Background Information Document
BP	blood pressure
BTU	British Thermal Unit
c.i.	confidence interval
CA	cerebrovascular accident
CAA	Clean Air Act
CAAA90	Clean Air Act Amendments of 1990
CAPMS	EPA's Criteria Air Pollutant Modeling System
CARB	California Air Resources Board
CASAC	SAB Clean Air Scientific Advisory Committee
CDC	Centers for Disease Control (now CDCP, Centers for Disease Control and Prevention)
CERL	EPA/ORD Corvallis Environmental Research Laboratory (old name; see NERL)
CEUM	ICF Coal and Electric Utility Model
CHD	coronary heart disease
CIPP	changes in production processes

CO	carbon monoxide
CO_2	carbon dioxide
COH	coefficient of haze
COHb	blood level of carboxyhemoglobin
COPD	chronic obstructive pulmonary disease
CPUE	catch per unit effort
CR	concentration-response
CRESS	Commercial and Residential Simulation System model
CSTM	Coal Supply and Transportation Model
CTG	Control Techniques Guidelines
CV	contingent valuation
CVM	contingent valuation method
D.C.	District of Columbia
DBP	diastolic blood pressure
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
$DFEV_1$	decrement of forced expiratory volume (in one second)
dL	deciliter
DOC	Department of Commerce
DOE	Department of Energy
DOI	Department of Interior
DRI	Data Resources Incorporated
dV	DeciView Haze Index
DVSAM	Disaggregate Vehicle Stock Allocation Model
EC	extinction coefficient
EDB	ethylene dibromide
EDC	ethylene dichloride
EFI	Electronic Fuel Injection
EI	Electronic Ignition
EIA	Energy Information Administration
EKMA	Empirical Kinetics Modeling Approach
ELI	Environmental Law Institute
EOL	end-of-line
EPA	Environmental Protection Agency
EPRI	Electric Power Research Institute
ESEERCO	Empire State Electric Energy Research Corporation
ESP	electrostatic precipitator
FERC	Federal Energy Regulatory Commission
FEV_1	forced expiratory volume (in one second)
FGD	flue gas desulfurization

FHWA	Federal Highway Administration
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
FIP	Federal Information Processing System
FR	Federal Register
FRP	Forest Response Program
GDP	gross domestic product
GEMS	Graphical Exposure Modeling System
GM	geometric mean
GNP	Gross National Product
GSD	geometric standard deviation
H_2SO_4	sulfuric acid
ha	hectares
HAP	Hazardous Air Pollutant
HAPEM-MS	Hazardous Air Pollutant Exposure Model - Mobile Source
HNO_3	nitric acid
hp	horsepower
HTCM	Hedonic Travel-Cost Model
ICARUS	Investigation of Costs and Reliability in Utility Systems
ICD-9	International Classification of Diseases, Ninth Version (1975 Revision)
ICE	Industrial Combustion Emissions model
IEc	Industrial Economics, Incorporated
IEUBK	EPA's Integrated Exposure Uptake Biokinetic model
IMS	Integrated Model Set
IPF	iterative proportional fitting
IQ	intelligence quotient
ISCLT	Industrial Source Complex Long Term air quality model
J/W	Jorgenson / Wilcoxen
kg	kilograms
km	kilometers
lbs	pounds
LRI	lower respiratory illness
m/s	meters per second
m	meters
m^3	cubic meters
Mm	megameters
MMBTU	million BTU
MOBILE5a	EPA's mobile source emission factor model
mpg	miles per gallon
Mpls	Minneapolis
MRAD	minor restricted activity day

MSCET	Month and State Current Emission Trends
MTD	metric tons per day
MVATS	EPA's Motor Vehicle-Related Air Toxics Study
MVMA	Motor Vehicle Manufacturers Association
Mwe	megawatt equivalent
N	nitrogen
NA	not available
NAAQS	National Ambient Air Quality Standard
NAPAP	National Acid Precipitation Assessment Program
NARSTO	North American Research Strategy for Tropospheric Ozone
NATICH	National Air Toxics Information Clearinghouse
NCLAN	National Crop Loss Assessment Network
NEA	National Energy Accounts
NERA	National Economic Research Associates
NERC	North American Electric Reliability Council
NERL	EPA/ORD National Exposure Research Laboratory (new name for CERL)
NESHAP	National Emission Standard for Hazardous Air Pollutants
NHANES	First National Health and Nutrition Examination Survey
NHANES II	Second National Health and Nutrition Examination Survey
NIPA	National Income and Product Accounts
NMOCs	nonmethane organic compounds
NO	nitric oxide
NO_2	nitrogen dioxide
NO_3^-	nitrate ion
NO_x	nitrogen oxides
NPTS	Nationwide Personal Transportation Survey
NSPS	New Source Performance Standards
NSWS	National Surface Water Survey
O&M	operating and maintenance
O_3	ozone
OAQPS	EPA/OAR Office of Air Quality Planning and Standards
OAR	EPA Office of Air and Radiation
OMS	EPA/OAR Office of Mobile Sources
OPAR	EPA/OAR Office of Policy Analysis and Review
OPPE	EPA Office of Policy Planning and Evaluation
ORD	EPA Office of Research and Development
OZIPM4	Ozone Isopleth Plotting with Optional Mechanism-IV
PACE	Pollution Abatement Costs and Expenditures survey
PAN	peroxyacetyl nitrate
PAPE	Pollution Abatement Plant and Equipment survey

Pb	lead
PbB	blood lead level
PCB	polychlorinated biphenyl
PES	Pacific Environmental Services
pH	the logarithm of the reciprocal of hydrogen ion concentration, a measure of acidity
PIC	product of incomplete combustion
PM_{10}	particulates less than or equal to 10 microns in aerometric diameter
$PM_{2.5}$	particulates less than or equal to 2.5 microns in aerometric diameter
POP	population
Pop_{mild}	exposed population of exercising mild asthmatics
Pop_{mod}	exposed population of exercising moderate asthmatics
ppb	parts per billion
PPH	people per household
pphm	parts per hundred million
ppm	parts per million
PPRG	Pooling Project Research Group
PRYL	percentage relative yield loss
PURHAPS	PURchased Heat And Power
PVC	polyvinyl chloride
r^2	statistical correlation coefficient, squared
RAD	restricted activity day
RADM	Regional Acid Deposition Model
RADM/EM	RADM Engineering Model
RAMC	Resource Allocation and Mine Costing model
RfD	reference dose
RIA	Regulatory Impact Analysis
ROM	Regional Oxidant Model
RRAD	respiratory restricted activity day
RUM	Random Utility Model
s.e.	standard error
SAB	Science Advisory Board
SAI	Systems Applications International
SAQM	SARMAP Air Quality Model
SARA	Superfund Amendment Reauthorization Act
SARMAP	SJVAQS/AUSPEX Regional Modeling Adaptation Project
SCC	Source Classification Code
SEDS	State Energy Data System
SIC	Standard Industrial Classification
SIP	State Implementation Plan
SJVAQS	San Joaquin Valley Air Quality Study

SMSA	Standard Metropolitan Statistical Area
SO_2	sulfur dioxide
SO_4^{2-}	sulfate ion
SOS/T	State of Science and Technology (refers to a series of NAPAP reports)
SRaw	Specific Airway Resistance
STAR	Stability Array weather database
TAMM90	Timber Assessment Market Model (revised version)
TEEMS	Transportation Energy and Emissions Modeling System
TIUS	Truck Inventory and Use Surveys
TRI	Toxic Release Inventory
TSP	total suspended particulate
U.S.	United States
UAM	Urban Airshed Model
URI	upper respiratory illness
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
VC	vinyl chloride
VMT	vehicle miles traveled
VOC	volatile organic compounds
VOP	Vehicle Ownership Projection
VR	visual range
W126	index of peak weighted average of cumulative ozone concentrations
WLD	Work Loss Day
WTP	willingness to pay

1

Introduction

Background and Purpose

As part of the Clean Air Act Amendments of 1990, Congress established a requirement under Section 812 that EPA develop periodic Reports to Congress estimating the benefits and costs of the Clean Air Act itself. The first such report was to be a retrospective analysis, with a series of prospective analyses to follow every two years thereafter. This report represents the retrospective study, covering the period beginning with passage of the Clean Air Act Amendments of 1970, until 1990 when Congress enacted the most recent comprehensive amendments to the Act.

Since the legislative history associated with section 812 is sparse, there is considerable uncertainty regarding Congressional intent behind the requirement for periodic cost-benefit evaluations of the Clean Air Act (CAA). However, EPA believes the principal goal of these amendments was that EPA should develop, and periodically exercise, the ability to provide Congress and the public with up-to-date, comprehensive information about the economic costs, economic benefits, and health, welfare, and ecological effects of CAA programs. The results of such analyses might then provide useful information for refinement of CAA programs during future reauthorizations of the Act.

The retrospective analysis presented in this Report to Congress has been designed to provide an unprecedented examination of the overall costs and benefits of the historical Clean Air Act. Many other analyses have attempted to identify the isolated effects of individual standards or programs, but no analysis with the present degree of validity, breadth and integration has ever been successfully developed. Despite data limitations, considerable scientific uncertainties, and severe resource constraints; the EPA Project Team was able to develop a broad assessment of the costs and benefits associated with the major CAA programs of the 1970 to 1990 period. Beyond the statutory goals of Section 812, EPA intends to use the results of this study to help support decisions on future investments in air pollution research. Finally, many of the methodologies and modeling systems developed for the Retrospective Study may be applied in the future to the ongoing series of section 812 prospective studies.

Clean Air Act Requirements, 1970 to 1990

The Clean Air Act establishes a framework for the attainment and maintenance of clean and healthful air quality levels. The Clean Air Act was enacted in 1970 and amended twice -- in 1977 and most recently in 1990.

The 1970 Clean Air Act contained a number of key provisions. First, EPA was directed to establish national ambient air quality standards for the major criteria air pollutants. The states were required to develop implementation plans describing how they would control emission limits from individual sources to meet and maintain the national standards. Second, the 1970 CAA contained deadlines and strengthened enforcement of emission limitations and state plans with measures involving both the states and the federal government. Third, the 1970 Act forced new sources to meet standards based on the best available technology. Finally, the Clean Air Act of 1970 addressed hazardous pollutants and automobile exhausts.

The 1977 Clean Air Act Amendments also set new requirements on clean areas already in attainment with the national ambient air quality standards. In addition, the 1977 Amendments set out provisions to help areas that failed to comply with deadlines for achievement of the national ambient air quality standards. For example, permits for new major sources and modifications were required.

The 1990 Clean Air Act Amendments considerably strengthened the earlier versions of the Act. With respect to nonattainment, the Act set forth a detailed and graduated program, reflecting the fact that problems in some areas are more difficult and complex than others. The 1990 Act also established a list of 189 regulated hazardous air pollutants and a multi-step program for controlling emissions of these toxic air pollutants. Significant control programs were also established for emissions of acid rain precursors and stratospheric ozone-depleting chemicals. The biggest regulatory procedural change in the Act is the new permit program where all major sources are now required to obtain an operating permit. Finally, the amendments considerably expanded the enforcement provisions of the Clean Air Act, adding administrative penalties and increasing potential civil penalties.

Section 812 of the Clean Air Act Amendments of 1990

Section 812 of the Clean Air Act Amendments of 1990 requires the EPA to perform a "retrospective" analysis which assesses the costs and benefits to the public health, economy and the environment of clean air legislation enacted prior to the 1990 amendments. Section 812 directs that EPA shall measure the effects on "employment, productivity, cost of living, economic growth, and the overall economy of the United States" of the Clean Air Act. Section 812 also requires that EPA consider all of the economic, public health, and environmental benefits of efforts to comply with air pollution standards. Finally, section 812 requires EPA to evaluate the prospective costs and benefits of the Clean Air Act every two years.

Analytical Design and Review

Target Variable

The retrospective analysis was designed to answer the following question:

> ***"How do the overall health, welfare, ecological, and economic benefits of Clean Air Act programs compare to the costs of these programs?"***

By examining the overall effects of the Clean Air Act, this analysis complements the Regulatory Impact Analyses (RIAs) developed by EPA over the years to evaluate individual regulations. Resources

were used more efficiently by recognizing that these RIAs, and other EPA analyses, provide complete information about the costs and benefits of specific rules.

Focusing on the broader target variables of "overall costs" and "overall benefits" of the Clean Air Act, the EPA Project Team adopted an approach based on construction and comparison of two distinct scenarios: a "no-control scenario" and a "control scenario." The no-control scenario essentially freezes federal, state, and local air pollution controls at the levels of stringency and effectiveness which prevailed in 1970. The control scenario assumes that all federal, state, and local rules promulgated pursuant to, or in support of, the CAA during 1970 to 1990 were enacted. This analysis then estimates the differences between the economic and environmental outcomes associated with these two scenarios.

Key Assumptions

Two key assumptions were made during the scenario design process to avoid miring the analytical process in endless speculation. First, the "no-control" scenario was defined to reflect the assumption that no additional air pollution controls were imposed by any level of government or voluntarily initiated by private entities after 1970. Second, it is assumed that the geographic distribution of population and economic activity remains the same between the two scenarios.

The first assumption is an obvious simplification. In the absence of the CAA, one would expect to see some air pollution abatement activity, either voluntary or due to state or local regulation. It is conceivable that state and local regulation would have required air pollution abatement equal to –or even greater than– that required by the CAA; particularly since some states, most notably California, have done so. If one were to assume that state and local regulations would have been equivalent to CAA standards, then a cost-benefit analysis of the CAA would be a meaningless exercise since both costs and benefits would equal zero. Any attempt to predict how states' and localities' regulations would have differed from the CAA would be too speculative to support the credibility of the ensuing analysis. Instead, the no-control scenario has been structured to reflect the assumption that states and localities would not have invested further in air pollution control programs after 1970 in the absence of the federal CAA. That is, this analysis accounts for all costs and benefits of air pollution control from 1970 to 1990. Speculation about the fraction of costs and benefits attributable exclusively to the federal CAA is left to others.

The second assumption concerns changing demographic patterns in response to air pollution. In the hypothetical no-control world, air quality is worse than that in the historical "control" world particularly in urban industrial areas. It is possible that in the no-control case more people, relative to the control case, would move away from the most heavily polluted areas. Rather than speculate on the scale of population movement, the analysis assumes no differences in demographic patterns between the two scenarios. Similarly, the analysis assumes no changes in the spatial pattern of economic activity. For example: if, in the no-control case, an industry is expected to produce greater output than it did in the control case, that increased output is produced by actual historical plants, avoiding the need to speculate about the location or other characteristics of new plants providing additional productive capacity.

Analytical Sequence

The analysis was designed and implemented in a sequential manner following seven basic steps which are summarized below and described in detail later in this report. The seven major steps were:

- direct cost estimation
- macroeconomic modeling
- emissions modeling
- air quality modeling
- health and environmental effects estimation
- economic valuation
- results aggregation and uncertainty characterization

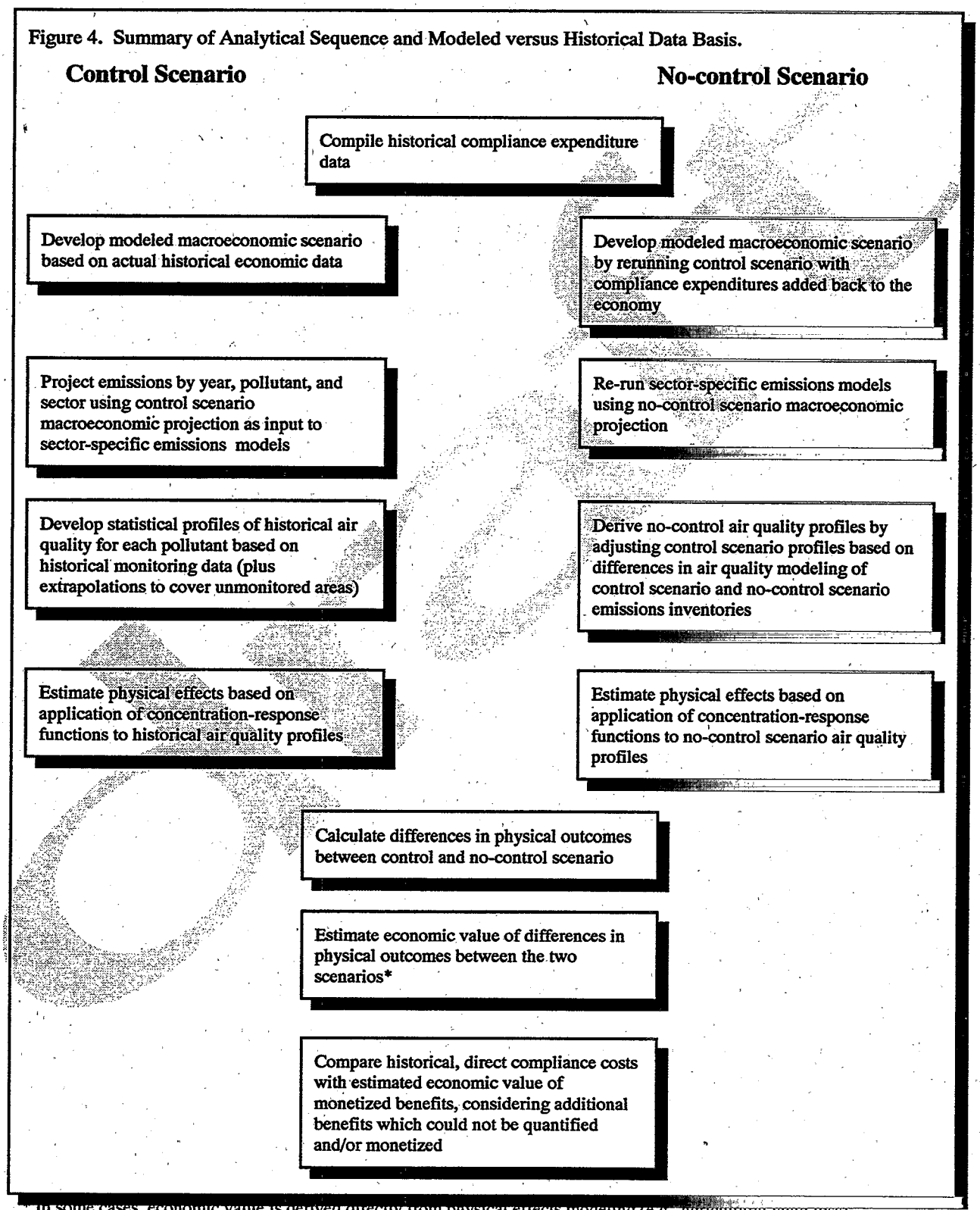
By necessity, each of these components had to be completed in a sequential manner. The emissions modeling effort had to be completed entirely before the air quality models could be configured and run; the air quality modeling results had to be completed before the health and environmental consequences of air quality changes could be derived; and so on. The analytical sequence, and the modeled versus actual data basis for each analytical component, are summarized in Figure 4 and described in the remainder of this section.

The first step of the analysis was to estimate the total direct costs incurred by public and private entities to comply with post-1970 CAA requirements. These data were obtained directly from Census Bureau and Bureau of Economic Analysis (BEA) data on compliance expenditures reported by sources, and from EPA analyses. These direct cost data were then adopted as inputs to the macroeconomic model used to project economic conditions–such as production levels, prices, employment patterns, and other economic indicators–under the two scenarios. To ensure a consistent basis for scenario comparison, the analysis applied the same macroeconomic modeling system to estimate control and no-control scenario economic conditions.[4] First, a control scenario was constructed by running the macroeconomic model using actual historical data for input factors such as economic growth rates during the 1970 to 1990 period. The model was then re-run for the no-control scenario by, in essence, returning all post-1970 CAA compliance expenditures to the economy. With these additional resources available for capital formation, personal consumption, and other purposes, overall economic conditions under the no-control scenario differed from those of the control scenario. In addition to providing estimates of the difference in overall economic growth and other outcomes under the two scenarios, these first two analytical steps were used to define specific economic conditions used as inputs to the emissions modeling effort, the first step in the estimation of CAA benefits.[5]

[4] Using modeled economic conditions for both scenarios has both advantages and disadvantages. The principal disadvantage is that historical economic conditions "predicted" by a macroeconomic model will not precisely duplicate actual historical events and conditions. However, this disadvantage is outweighed by the avoidance of distortions and biases which would result from comparing a modeled no-control scenario with actual historical conditions. By using the same macroeconomic model for both scenarios, model errors and biases essentially cancel out, yielding more robust estimates of scenario differences, which are what this analysis seeks to evaluate.

[5] For example, the macroeconomic model projected different electricity sales levels under the two scenarios, and these sales levels were used as key input assumptions by the utility sector emissions model.

Figure 4. Summary of Analytical Sequence and Modeled versus Historical Data Basis.



* In some cases, economic value is derived directly from physical effects modeling (e.g., agricultural yield loss).

Using appropriate economic indicators from the macroeconomic model results as inputs, a variety of emissions models were run to estimate emissions levels under the two scenarios. These emissions models provided estimates of emissions of six major pollutants[6] from each of six key emitting sectors: utilities, industrial processes, industrial combustion, on-highway vehicles, off-highway vehicles, and commercial/residential sources. The resulting emissions profiles reflect state-wide total emissions from each pollutant-sector combination for the years 1975, 1980, 1985, and 1990.[7]

The next step toward estimation of benefits involved translating these emissions inventories into estimates of air quality conditions under each scenario. Given the complexity, data requirements, and operating costs of state-of-the-art air quality models–and the afore-mentioned resource constraints–the EPA Project Team adopted simplified, linear scaling approaches for a number of pollutants. However, for ozone and other pollutants or air quality conditions which involve substantial non-linear formation effects and/or long-range atmospheric transport and transformation, the EPA Project Team invested the time and resources needed to use more sophisticated modeling systems. For example, urban area-specific ozone modeling was conducted for 147 urban areas throughout the 48 contiguous states.

Up to this point of the analysis, both the control and no-control scenario were based on modeled conditions and outcomes. However, at the air quality modeling step, the analysis returns to a foundation based on actual historical conditions and data. Specifically, actual historical air quality monitoring data from 1970 to 1990 are used to define the control scenario. Air quality conditions under the no-control scenario are then derived by scaling the historical data adopted for the control scenario by the ratio of the modeled control and no-control scenario air quality. This approach takes advantage of the richness of the historical data on air quality, provides a realistic grounding for the benefit measures, and yet retains the analytical consistency conferred by using the same modeling approach for both scenarios. The outputs of this step of the analysis are statistical profiles for each pollutant characterizing air quality conditions at each monitoring site in the lower 48 states.[8]

The control and no-control scenario air quality profiles were then used as inputs to a modeling system which translates air quality to physical outcomes –such as mortality, emergency room visits, or crop yield losses– through the use of concentration-response functions. These concentration-response functions are in turn derived from studies found in the scientific literature on the health and ecological effects of air pollutants. At this point, estimates were derived of the differences between the two scenarios in terms of incidence rates for a broad range of human health and other effects of air pollution by year, by pollutant, and by monitor.[9]

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[6] These six pollutants are total suspended particulates (TSP), sulfur dioxide (SO_2), nitrogen oxides (NO_x), carbon monoxide (CO), volatile organic compounds (VOCs), and lead (Pb). The other CAA criteria pollutant, ozone (O_3), is formed in the atmosphere through the interaction of sunlight and ozone precursor pollutants such as NO_x and VOCs.

[7] By definition, 1970 emissions under the two scenarios are identical.

[8] The one exception is particulate matter (PM). For PM, air quality profiles for both Total Suspended Particulates (TSP) and particulates less than or equal to 10 microns in diameter (PM_{10}) were constructed at the county level rather than the individual monitor level.

[9] Or, for PM, by county.

In the next step, economic valuation models or coefficients were used to estimate the economic value of the reduction in incidence of those adverse effects which were amenable to such monetization. For example, a distribution of unit values derived from the economic literature was used to estimate the value of reductions in mortality risk associated with exposure to particulate matter. In addition, benefits which could not be expressed in economic terms were compiled and are presented herein. In some cases, quantitative estimates of scenario differences in the incidence of a nonmonetized effect were calculated.[10] In many other cases, available data and techniques were insufficient to support anything more than a qualitative characterization of the change in effects.

Finally, the costs and monetized benefits were combined to provide a range of estimates for the partial, net economic benefit of the CAA with the range reflecting quantified uncertainties associated with the physical effects and economic valuation steps.[11] The term "partial" is emphasized because only a subset of the total potential benefits of the CAA could be represented in economic terms due to limitations in analytical resources, available data and models, and the state of the science.[12] Of paramount concern to the EPA Project Team was the paucity of concentration-response functions needed to translate air quality changes into measures of ecological effect. In addition, significant scientific evidence exists linking air pollution to a number of adverse human health effects which could not be effectively quantified and/or monetized.[13]

Review Process

The CAA requires EPA to consult with an outside panel of experts –referred to statutorily as the Advisory Council on Clean Air Act Compliance Analysis (ACCACA)– in developing the section 812 analyses. In addition, EPA is required to consult with the Department of Labor and the Department of Commerce.

The ACCACA was organized in 1991 under the auspices and procedures of EPA's Science Advisory Board (SAB). Organizing the review committee under the SAB ensured that review of the section 812 studies would be conducted by highly qualified experts in an objective, rigorous, and publicly open

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[10] For example, changes in forced expiratory volume in one second (FEV_1) as a result of exposure to ozone were quantified but could not be expressed in terms of economic value.

[11] Although considerable uncertainties surround the direct cost, macroeconomic modeling, emissions modeling,, and air quality modeling steps, the ranges of aggregate costs and benefits presented in this analysis do not reflect these uncertainties. While the uncertainties in these components were assessed qualitatively, and in some cases quantitatively, resource limitations precluded the multiple macroeconomic model, emissions model, and air quality model runs which would have been required to propagate these uncertainties through the entire analytical sequence. As a result, complete quantitative measures of the aggregate uncertainty in the cost and benefit estimates could not be derived. However, the ranges presented do reflect quantitative measures of the uncertainties in the two most uncertain analytical steps: physical effects estimation and economic valuation.

[12] It should be noted that there is some uncertainty associated with the estimates of economic costs as well and that some omitted components of adverse economic consequences of pollution control programs may be significant. For example, some economists argue that the economic costs of the CAA reported herein may be significantly underestimated to the extent potential adverse effects of regulation on technological innovation are not captured. Nevertheless, it is clear that the geographic, population, and categorical coverage of monetary cost effects is significantly greater than coverage of monetized benefits in this analysis..

[13] For example, while there is strong evidence of a link between exposure to carbon monoxide and reduced time of onset of angina attack, there are no valuation functions available to estimate the economic loss associated with this effect.

manner. The SAB ACCACA has met many times during the development of the retrospective study to review methodologies and interim results. While the full ACCACA retains overall review responsibility for the section 812 studies, some specific issues concerning physical effects and air quality modeling have been referred to subcommittees comprised of both ACCACA members and members of other SAB committees. The ACCACA Physical Effects Subcommittee has met several times and has provided its own review findings to the full ACCACA. Similarly, the ACCACA Air Quality Subcommittee, comprised of members and consultants of the SAB Clean Air Scientific Advisory Committee (CASAC), has held several teleconference meetings to review methodology proposals and modeling results.

With respect to the interagency review process, the EPA has expanded the list of consulted agencies to include the Department of Energy, the Council on Environmental Quality, the National Acid Precipitation Assessment Program, The Office of Management and Budget, and the Council of Economic Advisors. While several meetings have been held with the Interagency Review Group during the course of the retrospective study, the EPA Project Team now provides opportunity for integrated interagency, expert panel, and public consultation through the public review meetings conducted by the SAB ACCACA.

Report Organization

The remainder of the main text of this report summarizes the key methodologies and findings of retrospective study. The direct cost estimation and macroeconomic modeling steps are presented in Chapter 2. The emissions modeling is summarized in Chapter 3. Chapter 4 presents the air quality modeling methodology and sample results. Chapter 5 describes the approaches used and principal results obtained through the physical effects estimation process. Economic valuation methodologies are described in Chapter 6. Chapter 7 presents the aggregated results of the cost and benefit estimates and describes and evaluates important uncertainties in the results.

Additional details regarding the methodologies and results are presented in the appendices and in the referenced supporting documents. Appendix A covers the direct cost and macroeconomic modeling. Appendix B provides additional detail on the sector-specific emissions modeling effort. Details of the air quality models used and results obtained are presented or referenced in Appendix C. The effects of the CAA on human health and visibility; aquatic, wetland, and forest ecosystems; and agriculture are presented in Appendices D, E, and F, respectively. Appendix G presents details of the lead (Pb) benefits analysis. Air toxics reduction benefits are discussed in Appendix H. The methods and assumptions used to value quantified effects of the CAA in economic terms are described in Appendix I. Appendix J describes some areas of research which may increase comprehensiveness and reduce uncertainties in effect estimates for future assessments, and describes plans for future Section 812 analyses.

2

Cost and Macroeconomic Effects

The costs of complying with Clean Air Act (CAA) requirements through the 1970 to 1990 period affected patterns of industrial production, capital investment, productivity, consumption, employment, and overall economic growth. The purpose of the analyses summarized in this chapter was to estimate those direct costs and the magnitude and significance of resulting changes to the overall economy. This was accomplished by comparing economic indicators under two alternative scenarios: a control scenario serving as the historical benchmark, including the historical CAA as implemented; and a no-control scenario which assumes historical CAA programs did not exist. The estimated economic consequences of the historical CAA were taken as the difference between these two scenarios.

Data used as inputs to the cost analysis can be classified into two somewhat overlapping categories based on the information source: survey-based information (generally gathered by the Census Bureau) and information derived from various EPA analyses. For the most part, cost estimates for stationary air pollution sources (e.g., factory smokestacks) are based on surveys of private businesses that attempt to elicit information on annual pollution control outlays by those businesses. Estimates of pollution control costs for mobile sources (e.g., automobiles) are largely based on EPA analyses, rather than on direct observation and measurement of compliance expenditures. For example, to determine one component of the cost of reducing lead emissions from mobile sources, the Project Team used an oil refinery production cost model to calculate the incremental cost required to produce unleaded (or less-leaded, as appropriate) rather than leaded gasoline, while maintaining the octane level produced by leaded gasoline.

As is the case with many policy analyses, a significant uncertainty arises in the cost analysis as a consequence of constructing a hypothetical scenario. With this retrospective analysis covering almost twenty years, difficulties arise in projecting alternative technological development paths. In some cases, the analytical assumptions used to project the alternative scenario are not immediately apparent. For example, the surveys covering stationary source compliance expenditures require respondents to report pollution abatement expenditures -- implicitly asking them to determine by how much the company's costs would decline if there were no CAA compliance requirements. While a response might be relatively straightforward in the few years following passage of the CAA, a meaningful response becomes more difficult after many years of technical change and investment in less-polluting plant and equipment make it difficult to determine the degree to which total costs would differ under a "no CAA" scenario. In cases such as this, assumptions concerning the alternative hypothetical scenario are made by thousands of individual survey respondents. Where cost data are derived from EPA analyses, the hypothetical scenario assumptions are, at least in theory, more apparent. For example, when determining the incremental cost caused by pollution-control requirements, one needs to make assumptions (at least implicitly) about what an auto would look like absent pollution control requirements. In either case, the need to project

hypothetical technology change for two decades introduces uncertainty into the assessment results, and this uncertainty may be difficult to quantify..

The remainder of this chapter summarizes the basic methods and results of the direct compliance cost and macroeconomic analyses. Further detail regarding the modeling methods and assumptions employed, as well as additional analytical results, are presented in Appendix A.

Direct Compliance Costs

Compliance with the CAA imposed direct costs on businesses, consumers, and governmental units; and triggered other expenditures such as governmental regulation and monitoring costs and expenditures for research and development by both government and industry. Although expenditures unadjusted for inflation—that is, expenditures denominated in "current dollars"—increased steadily from $7 billion to $19 billion per year over the 1973 to 1990 period,[14] annual CAA compliance expenditures adjusted for inflation were relatively stable, averaging near $25 billion (in 1990 dollars) during the 1970s and close to $20 billion during most of the 1980s (see Table 4). Aggregate compliance expenditures were somewhat less than ½ of one percent of total domestic output during that period, with the percentage falling from 2/3 of one percent of total output in 1975 to ⅓ of one percent in 1990.

Table 4. Estimated Annual CAA Compliance Costs ($billions).

	Expenditures		Annualized Costs $1990 at:		
Year	$current	$1990	3%	5%	7%
1973	7.2	19.6	11.0	11.0	11.1
1974	8.5	21.4	13.2	13.4	13.7
1975	10.6	24.4	13.3	13.6	14.0
1976	11.2	24.1	14.1	14.6	15.1
1977	11.9	24.1	15.3	15.9	16.6
1978	12.0	22.6	15.0	15.8	16.7
1979	14.4	24.8	17.3	18.3	19.3
1980	16.3	25.7	19.7	20.8	22.0
1981	17.0	24.4	19.6	20.9	22.3
1982	16.0	21.6	18.6	20.1	21.7
1983	15.5	20.1	19.1	20.7	22.5
1984	17.3	21.6	20.1	21.9	23.8
1985	19.1	22.9	22.5	24.4	26.5
1986	17.8	20.8	21.1	23.2	25.4
1987	18.2	20.6	22.1	24.2	26.6
1988	18.2	19.8	22.0	24.3	26.7
1989	19.0	19.8	22.9	25.3	27.8
1990	19.0	19.0	23.6	26.1	28.7

Although useful for many purposes, a summary of direct annual expenditures may not the best cost measure to use when comparing costs to benefits. Capital expenditures are investments, generating a stream of benefits and opportunity cost[15] over the life of the investment. The appropriate accounting technique to use for capital expenditures in a cost/benefit analysis is to annualize the expenditure. This technique, analogous to calculating the monthly payment associated with a home mortgage, involves spreading the cost of the capital equipment over the useful life of the equipment using a discount rate to account for the time value of money.

[14] Due to data limitations, the cost analysis for this CAA retrospective starts in 1973, missing costs incurred in 1970-72.. This limitation is not likely to be significant, however, because relatively little in the way of compliance with the "new" provisions of the 1970 CAA was required in the first two years following passage.

[15] In this context, "opportunity cost" is defined as the value of alternative investments or other uses of funds foregone as a result of the investment.

For this cost/benefit analysis, "annualized" costs reported for any given year are equal to O&M expenditures — including R&D and other similarly recurring expenditures — plus amortized capital costs (i.e., depreciation plus interest costs associated with the existing capital *stock*) for that year. Stationary source air pollution control capital costs were amortized over 20 years; mobile source air pollution control costs were amortized over 10 years.[16] All capital expenditures were annualized using a five percent, inflation-adjusted rate of interest. Additionally, annualized costs were calculated using discount rates of three and seven percent to determine the sensitivity of the cost results to changes in the discount rate. Table 4 summarizes costs annualized at three, five, and seven percent, as well as annual expenditures.

Total expenditures over the 1973-1990 period, discounted to 1990 using a five percent (net of inflation) discount rate, amount to 628 billion dollars (in 1990 dollars). Discounting the annualized cost stream to 1990 (with both annualization and discounting procedures using a five percent rate) gives total costs of 523 billion dollars (in 1990 dollars). Aggregate annualized costs are less than expenditures because the annualization procedure spreads some of the capital cost beyond 1990.[17]

Indirect Effects of the CAA

Through changing production costs, CAA implementation induced changes in consumer good prices, and thus in the size and composition of economic output. The Project Team used a general equilibrium macroeconomic model to assess the extent of such second-order effects. This type of model is useful because it can capture the feedback effects of an action. In the Section 812 macroeconomic modeling exercise, the feedback effects arising from expenditure changes were captured, but the analogous effects arising from improvements in human health were not captured by the model. For example, the macroeconomic model results do not reflect the indirect economic effects of worker productivity improvements and medical expenditure savings caused by the CAA. Consequently, the macroeconomic modeling exercise provides limited and incomplete information on the type and potential scale of indirect economic effects.

The effects estimated by the macroeconomic model can be grouped into two broad classes: sectoral impacts (i.e., changes in the composition of economic output), and aggregate effects (i.e., changes in the degree of output or of some measure of human welfare). The predicted sectoral effects were used as inputs to the emissions models as discussed in Chapter 2. In general, the estimated second-order macroeconomic effects were small relative to the size of the U.S. economy. See Appendix A for more detail on data sources, analytical methods, and results for the macroeconomic modeling performed for this assessment.

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[16] Although complete data are available only for the period 1973-1990, *Cost of Clean* includes capital expenditures for 1972. Those capital expenditure data have been used here. Therefore, amortized costs arising from 1972 capital investments *are included* in the 1973-1990 annualized costs, even though 1972 costs are not otherwise included in the analysis. Conversely, some capital expenditures incurred in the 1973-1990 period are not reflected in the 1973-1990 annualized costs -- those costs are spread through the following two decades, thus falling outside of the scope of this study (e.g., only one year of depreciation and interest expense is included for 1989 capital expenditures). Similarly, benefits arising from emission reductions realized after 1990 as a result of capital investments made during the 1970 to 1990 period of this analysis are not included in the estimates of benefits included in this report.

[17] This adjustment is required because many 1970 to 1990 investments in control equipment continue to yield benefits beyond 1990. Annualization of costs beyond 1990 ensures that the costs and benefits of any particular investment are properly scaled and matched over the lifetime of the investment.

Sectoral Impacts

The CAA had variable compliance impacts across economic sectors. The greatest effects were on the largest energy producers and consumers, particularly those sectors which relied most heavily on consumption of fossil fuels (or energy generated from fossil fuels). In addition, production costs increased more for capital-intensive industries than for less capital-intensive industries under the control scenario due to a projected increase in interest rates. The interest rate increase, which resulted in an increase in the cost of capital, occurred under the control scenario because CAA-mandated investment in pollution abatement reduced the level of resources available for other uses, including capital formation.

Generally, the estimated difference in cost impacts under the control and no-control scenarios for a particular economic sector was a function of the relative energy-intensity and capital-intensity of that sector. Increased production costs in energy- and capital-intensive sectors under the control scenario were reflected in higher consumer prices, which resulted in reductions in the quantity of consumer purchases of goods and services produced by those sectors. This reduction in consumer demand under the control scenario led, ultimately, to reductions in output and employment in those sectors. The sectors most affected by the CAA were motor vehicles, petroleum refining, and electricity generation. The electricity generation sector, for example, incurred a two to four percent increase in consumer prices by 1990, resulting in a three to five and a half percent reduction in output. Many other manufacturing sectors saw an output effect in the one percent range.

Some other sectors, however, were projected to increase output under the control scenario. Apart from the pollution control equipment industry, which was not separately identified and captured in the macroeconomic modeling performed for this study, two example sectors for which output was higher and prices were lower under the control scenario are food and furniture. These two sectors showed production cost and consumer price reductions of one to two percent relative to other industries under the control scenario, resulting in output and employment increases of similar magnitudes.

Aggregate Effects

As noted above, the control and no-control scenarios yield different estimated mixes of investment. In particular, the control scenario was associated with more pollution control capital expenditure and less consumer commodity capital expenditure. As a result, the growth pattern of the economy under the control scenario differed from the no-control scenario. Under the control scenario, the macroeconomic model projected a rate of long-run GNP growth about one twentieth of one percent per year lower than under the no-control scenario. Aggregating these slower growth effects of the control scenario over the entire 1970 to 1990 period of this study results, by 1990, in a level of GNP one percent (or approximately $55 billion) lower than that projected under the no-control scenario.

Although small relative to the economy as a whole, the estimated changes in GNP imply that the potential impact of the CAA on the economy by 1990 was greater than that implied by expenditures ($19 billion in 1990) or annualized costs ($26 billion in 1990, annualized at five percent). Discounting the stream of 1973-1990 GNP effects to 1990 gives an aggregate impact on production of 1,005 billion dollars (in 1990 dollars discounted at five percent). Of that total, $569 billion represent reductions in household

consumption, and another $200 billion represent government consumption, for an aggregate effect on U.S. consumption of goods and services equal to 769 billion dollars. Both the aggregate GNP effects and aggregate consumption effects exceed total 1973-1990 expenditures ($628 billion) and annualized costs ($523 billion, with all dollar quantities in $1990, discounted at five percent).

Changes in GNP (or, even, changes in the national product account category "consumption") do not necessarily provide a good indication of changes in social welfare. Social welfare is not improved, for example, by major oil tanker spills even though measured GNP is increased by the "production" associated with clean-up activities. Nevertheless, the effects of the CAA on long-term economic growth would be expected to have had some effect on economic welfare. One of the characteristics of the macroeconomic model used by the Project Team is its ability to estimate a measure of social welfare change which is superior to GNP changes. This social welfare measure estimates the monetary compensation which would be required to offset the losses in consumption (broadly defined) associated with a given policy change. The model reports a range of results, with the range sensitive to assumptions regarding how cost impacts are distributed through society. For the CAA, the model reports an aggregate welfare effect of 493 billion to 621 billion dollars (in 1990 dollars), depending on the distributional assumptions used. This range does not differ greatly from the range of results represented by 1973-1990 expenditures, compliance costs, and consumption changes.

Uncertainties and Sensitivities in the Cost and Macroeconomic Analysis

The cost and macroeconomic analyses for the present assessment relied upon survey responses, EPA analyses, and a macroeconomic simulation model. Although the Project Team believes that the results of the cost and macroeconomic analyses are reasonably reliable, it recognizes that every analytical step is subject to uncertainty. As noted at the beginning of this chapter, explicit and implicit assumptions regarding hypothetical technology development paths are crucial to framing the question of the cost impact of the CAA. In addition, there is no way to verify the accuracy of the survey results used; alternative, plausible cost analyses exist that arrive at results that differ from some of the results derived from EPA analyses; and it is not clear how the use of a general equilibrium macroeconomic model affects the accuracy of macroeconomic projections in a macroeconomy characterized by disequilibrium. For many factors engendering uncertainty, the degree or even the direction of bias is unknown. In several areas, nevertheless, uncertainties and/or sensitivities can be identified that may bias the results of the analysis.

Productivity and Technical Change

An important component of the macroeconomic model used by the Project Team is its treatment of technical change and productivity growth. Three factors associated with productivity and technical change have been identified which may bias the results of the macroeconomic simulation: (1) the long-run effects of reducing the "stock" of technology, (2) the possible "chilling" effect of regulations on innovation and technical change, and (3) the role of endogenous productivity growth within the macroeconomic model.

The macroeconomic model projected a decrease in the growth of GNP as a result of CAA compliance. Decreased growth was due not only to decreased capital investment, but also to decreased factor

productivity. The annual decrement in productivity can be thought of as a reduction of the stock of available technology. That reduction in stock could be expected to affect macroeconomic activity after 1990, as well as during the 1973-1990 period studied by the Project Team. Thus, to the extent that this effect exists, the Project Team has underestimated the macroeconomic impact of the CAA by disregarding the effect of 1973-1990 productivity change decrements on post-1990 GNP.

Some economists contend that regulations have a "chilling" effect on technological innovation and, hence, on productivity growth. The macroeconomic model allowed policy-induced productivity change throught the mechanism of price changes and resultant factor share changes. To the extent that additional policy-induced effects on productivity growth exist, the Project Team has underestimated the impact of the CAA on productivity growth during the 1973-1990 period, and, thus, has underestimated macroeconomic impacts during the 1973-1990 period and beyond.

The macroeconomic model allowed productivity growth to vary with changes in prices generated by the model. This use of "endogenous" productivity growth is not universal in the economic growth literature -- that is, many similar macroeconomic models do not employ analogous forms of productivity growth. The Project Team tested the sensitivity of the model results to the use of endogenous productivity growth. If the model is run without endogenous productivity growth, then the predicted macroeconomic impacts (GNP, personal consumption, etc.) of the CAA are reduced by approximately 20 percent. That is, to the extent that use of endogenous productivity growth in the macroeconomic model is an inaccurate simulation technique, then the Project Team has overestimated the macroeconomic impact of the CAA.

Discount Rates

There is a broad range of opinion in the economics profession regarding the appropriate discount rate to use in analyses such as the current assessment. Some economists believe that the appropriate rate is one that approximates the social rate of time preference -- that is, the rate of return at which individuals are willing to defer consumption to the future. A three percent rate would approximate the social rate of time preference (all rates used here are "real", i.e., net of price inflation impacts). Others believe that a rate that approximates the opportunity cost of capital (e.g., seven percent or greater) should be used.[18] A third school of thought holds that some combination of the social rate of time preference and the opportunity cost of capital is appropriate, with the combination effected either by use of an intermediate rate or by use of a multiple-step procedure employing the social rate of time preference as the "discount rate," but still accounting for the opportunity cost of capital.

[18] Some would argue that use of the opportunity cost of capital approach would be inappropriate in the current assessment if the results of the macroeconomic modeling (such as GNP) were used as the definition of "cost," since the macro model already accounts for the opportunity cost of capital. The appropriate rate would then be the social rate of time preference.

The Project Team elected to use an intermediate rate (five percent), but recognizes that analytical results aggregated across the study period are sensitive to the discount rate used. Consequently, all cost measures are presented at three and seven percent, as well as the base case five percent. Table 5, 36 summarizes major cost and macroeconomic impact measures expressed in constant 1990 dollars, and discounted to 1990 at rates of three, five, and seven percent.

Table 5. Compliance Cost, GNP, and Consumption Impacts Discounted to 1990 ($1990 billions)

	3%	5%	7%
Expenditures	520	628	761
Annualized Costs	417	523	657
GNP	880	1005	1151
Household Consumption	500	569	653
HH and Gov't Consumption	676	769	881

Exclusion of Health Benefits from the Macroeconomic Model

The macroeconomic modeling exercise was designed to capture the second-order macroeconomic effects arising from CAA compliance expenditures. Those predicted second-order effects are among the factors used to drive the emissions estimates and, ultimately, the benefits modeled for this assessment. The *benefits* of the CAA, however, would also be expected to induce second-order macroeconomic effects. For example, increased longevity and decreased incidence of non-fatal heart attacks and strokes would be expected to improve macroeconomic performance measures. The structure of the overall analysis, however, necessitated that these impacts be excluded from the macroeconomic simulation.

The first-order CAA beneficial effects *have* been included in the benefits analysis for this study, including measures that approximate production changes (e.g., income loss due to illness, or lost or restricted work days; income loss due to impaired cognitive ability; and income loss due to reduced worker production in certain economic sectors). These measures are analogous to compliance expenditures in the cost analysis. The second-order benefits impacts, which would result from price changes induced by CAA-related benefits, have not been estimated. It is likely that the estimated adverse second-order macroeconomic impacts would have been reduced had the impact of CAA benefits been included in the macroeconomic modeling exercise; however, the magnitude of this potential upward bias in the estimate of adverse macroeconomic impact was not quantitatively assessed.

3 Emissions

This chapter presents estimates of emissions reductions due to the Clean Air Act (CAA) for six criteria air pollutants. Reductions are calculated by estimating, on a sector-by-sector basis, the differences in emissions between the control and no-control scenarios. While the relevant years in this analysis are 1970 through 1990, full reporting of emissions was only made for the 1975 to 1990 period since 1970 emission levels are, by assumption, identical for the two scenarios. The criteria pollutants for which emissions are reported in this analysis are: total suspended particulates (TSP),[19] carbon monoxide (CO), volatile organic compounds (VOC), sulfur dioxide (SO_2), nitrogen oxides (NO_x), and Lead (Pb).

The purpose of the present study is to estimate the differences in economic and environmental conditions between a scenario reflecting implementation of historical CAA controls and a scenario which assumes that no additional CAA-related control programs were introduced after 1970. Because of the focus on differences in –rather than absolute levels of– emissions between the scenarios, the various sector-specific emission models were used to estimate both the control and no-control scenario emission inventories. This approach ensures that differences between the scenarios are not distorted by differences between modeled and actual historical emission estimates.[20]

Despite the use of models to estimate control scenario emission inventories, the models used were configured and/or calibrated using historical emissions estimates. The control scenario utility emissions estimates, for example, were based on the ICF CEUM model which was calibrated using historical emissions inventory data.[21] In other cases, such as the EPA Emissions Trends Report (*Trends*) methodology[22] used to estimate industrial process emissions, historical data were used as the basis for control scenario emissions with little or no subsequent modification. Nevertheless, differences in model

[19] In 1987, EPA replaced the earlier TSP standard with a standard for particulate matter of 10 microns or smaller (PM_{10}).

[20] By necessity, emission models must be used to estimate the hypothetical no-CAA scenario. If actual historical emissions data were adopted for the control scenario, differences between the monitoring data and/or models used to develop historical emission inventories and the models used to develop no-control scenario emission estimates would bias the estimates of the differences between the scenarios.

[21] See ICF Resources, Inc., "Results of Retrospective Electric Utility Clean Air Act Analysis - 1980, 1985 and 1990," September 30, 1992, Appendix C.

[22] EPA, 1994a: U.S. Environmental Protection Agency, "National Air Pollutant Emission Trends, 1900-1993," EPA-454/R-94-027, Office of Air Quality Planning and Standards, Research Triangle Park, NC, October 1994.

selection, model configuration, and macroeconomic input data[23] result in unavoidable, but in this case justifiable, differences between national total historical emission estimates and national total control scenario emission estimates for each pollutant. Comparisons between no-control, control, and official EPA Trends Report historical emissions inventories are presented in Appendix B.[24]

To estimate no-control scenario emissions, sector-specific historical emissions are adjusted based on changes in the following two factors: (1) growth by sector predicted to occur under the no-control scenario; and (2) the exclusion of controls attributable to specific provisions of the CAA.

To adjust emissions for economic changes under the no-control scenario, activity levels that affect emissions from each sector were identified. These activity levels include, for example, fuel use, industrial activity, and vehicle miles traveled (VMT). The Jorgenson-Wilcoxen (J/W) general equilibrium model was used to estimate changes in general economic conditions, as well as sector-specific economic outcomes used as inputs to the individual sector emission models.[25]

The specific outputs from the J/W model used in this analysis are the percentage changes in gross national product (GNP), personal consumption, and output for various economic sectors under the control and no-control scenario for the years 1975, 1980, 1985, and 1990.[26] The sectors for which the results of the J/W model are used include: industrial processes, electric utilities, highway vehicles, industrial boilers, and the commercial/residential sector. For the off-highway sector, economic growth was not taken into account as there was no direct correspondence between J/W sectors and the off-highway vehicle source category activity.

In addition to adjusting for economic activity changes, any CAA-related control efficiencies that were applied to calculate control scenario emissions were removed for the no-control scenario. In most instances, emissions were recalculated based on 1970 control levels.

Uncertainty associated with several key modeling inputs and processes may contribute to potential errors in the emission estimates presented herein. Although the potential errors are likely to contribute in only a minor way to overall uncertainty in the estimated monetary benefits of the Clean Air Act, the most significant emission modeling uncertainties are described at the end of this chapter.

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[23] The Jorgenson/Wilcoxen macroeconomic model outputs were used to configure both the control and no-control scenario emission model runs. While this satisfies the primary objective of avoiding "across model" bias between the scenarios, the macroeconomic conditions associated with the control scenario would not be expected to match actual historical economic events and conditions. To the extent actual historical economic conditions are used to estimate official historical emission inventories, conformity between these historical emissions estimates and control scenario emission estimates would be further reduced.

[24] In general, these comparisons show close correspondence between control scenario and *Trends* estimates with the largest differences occurring for VOC and CO emissions. The *Trends* report VOC estimates are generally higher than the control scenario estimates due primarily to the inclusion of Waste Disposal and Recycling as a VOC source in the *Trends* report. This inconsistency is of no consequence since Waste Disposal and Recycling sources were essentially uncontrolled by the historical CAA and therefore do not appear as a difference between the control and no-control scenarios. The higher CO emission estimates in the *Trends* Report are primarily associated with higher off-highway vehicle emissions estimates. Again, since off-highway emissions do not change between the control and no-control scenario in the present analysis, this inconsistency is of no consequence.

[25] For example, the change in distribution of households by income class predicted by the J/W model was used as input to the transportation sector model system. Changes in household income resulted in changes in vehicle ownership and usage patterns which, in turn, influence VMT and emissions. (See Pechan, 1995, p. 43).

[26] For details regarding the data linkages between the J/W model and the various emission sector models, see Pechan (1995).

Sector-Specific Approach

The approaches used to calculate emissions for each sector vary based on the complexity of estimating emissions in the absence of CAA controls, taking economic activity levels and CAA regulations into account. For the off-highway vehicle and industrial process sectors, a relatively simple methodology was developed. The approaches used for the highway vehicles, electric utilities, industrial boilers, and commercial/residential sectors were more complex because the J/W model does not address all of the determinants of economic activity in these sectors that might have changed in the absence of regulation. The approaches by sector used to estimate emissions for the two scenarios are summarized in Table 6, and

draft

are described in more detail in Appendix B.

Summary of Results

Table 6. Summary of Sector-Specific Emission Modeling Approaches.

Sector	Modeling Approach
On-Highway Vehicles	Modeled using ANL's TEEMS; adjusted automobile emission estimates by changes in personal travel and economic activity in the without CAA case. Truck VMT was obtained from the Federal Highway Administration (FHWA). MOBILE5a emission factors were used to calculate emissions. Lead emission changes from gasoline were estimated by Abt Associates based on historical gasoline sales and the lead content of leaded gasoline in each target year.
Off-Highway Vehicles	ELI analysis based on Trends methods. Recalculated historical emissions using 1970 control efficiencies from Trends. No adjustment was made to activity levels in the without the CAA case.
Electric Utilities	ICF's Coal and Electric Utility Model (CEUM) used to assess SO_2, NO_x, and TSP emission changes. Electricity sales levels were adjusted with results of the J/W model. The Argonne Utility Simulation Model (ARGUS) provided CO and VOC results. Changes in activity levels were adjusted with results of the J/W model. Lead emissions were calculated based on energy consumption data and Trends emission factors and control efficiencies.
Industrial Combustion	ANL industrial boiler analysis for SO_2, NO_x, and TSP using the Industrial Combustion Emissions (ICE) model. VOC and CO emissions from industrial boilers were calculated based on Trends methods; recalculated using 1970 control efficiencies. Lead emissions calculated for boilers and processes based on Trends fuel consumption data, emission factors, and 1970 control efficiencies.
Industrial Processes	ELI analyzed industrial process emissions based on Trends methods. Adjusted historical emissions with J/W sectoral changes in output, and 1970 control efficiencies from Trends. Lead emissions calculated for industrial processes and processes based on Trends fuel consumption data, emission factors, and 1970 control efficiencies.
Commercial / Residential	ANL's Commercial and Residential Simulation System (CRESS) model was used.

Figure 5 compares the total estimated sulfur dioxide emission from all sectors under the control and no-control scenarios over the period from 1975 to 1990. Figures 6, 7, 8, 9, and 10 provide similar comparisons for NO_x, VOCs, CO, TSP, and Lead (Pb) respectively.

Additional tables presented in Appendix B provide further breakdown of the emissions estimates by individual sector. The essential results are characterized below. For most sectors, emission levels under the control scenario were substantially lower than levels projected under the no-control scenario.

The CAA controls that affected SO_2 emitting sources had the greatest proportional effect on industrial process emissions, which were 60 percent lower in 1990 than they would have been under the no-control scenario. SO_2 emissions from electric utilities and industrial boilers were each nearly 40 percent lower in 1990 as a result of the controls. In terms of absolute tons of emission reductions, controls on electric utilities account for over 10 million of the total 16 million ton difference between the 1990 control and no-control scenario SO_x emission estimates.

CAA regulation of the highway vehicles sector led to the greatest percent reductions in VOC and NO_x. Control scenario emissions of these pollutants in 1990 were 66 percent and 47 percent lower, respectively, than the levels estimated under the no-control scenario. In absolute terms, highway vehicle VOC controls account for over 15 million of the roughly 17 million ton difference in control and no-control scenario emissions.

Differences between control and no-control scenario CO emissions are

Figure 5. Control and No-control Scenario Total SO_x Emission Estimates.

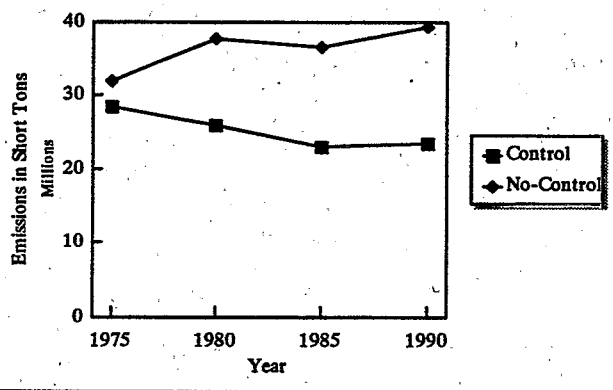


Figure 6. Control and No-control Scenario Total NO_x Emission Estimates.

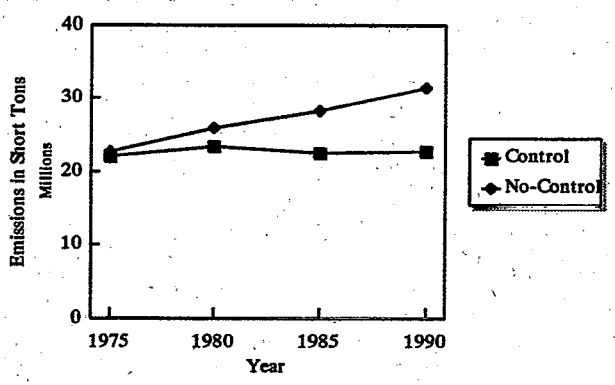
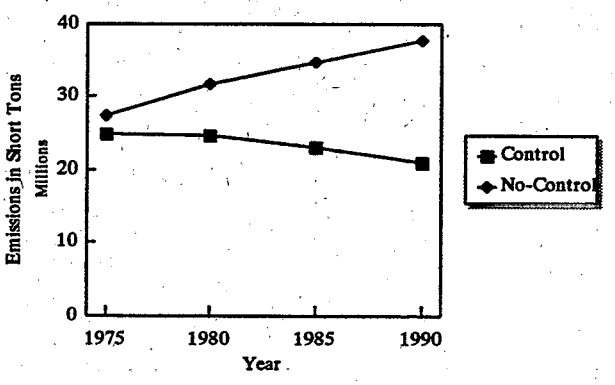


Figure 7. Control and No-control Scenario Total VOC Emission Estimates.



also most significant for highway vehicles. In percentage terms, highway vehicle CO emissions were 56 percent lower in 1990 under the control scenario than under the no-control scenario. Industrial process CO emission estimates under the control scenario were about half the levels projected under the no-control scenario. Of the roughly 89 million ton difference in CO emissions between the two scenarios, 84 million tons are attributable to highway vehicle controls and the rest is associated with reductions from industrial process emissions.

For TSP, the highest level of reductions on a percentage basis was achieved in the electric utilities sector. TSP emissions from electric utilities were 93 percent lower in 1990 under the control scenario than projected under the no-control scenario. TSP emissions from industrial processes were also significantly lower on a percentage basis under the control scenario, with the differential reaching 76 percent by 1990.

This is not an unexpected result as air pollution control regulations in the 1970's focused on solving the visible particulate problems from large fuel combustors. In terms of absolute tons, electric utilities account for over 5 million of the 16 million ton difference between the two scenarios and industrial processes account for almost 10 million tons.

The vast majority of the difference in lead emissions under the two scenarios is attributable to reductions in burning of leaded gasoline. By 1990, reductions in highway vehicle emissions account for 221 thousand of the total 234 thousand ton difference in lead

Figure 8. Control and No-control Scenario Total CO Emission Estimates.

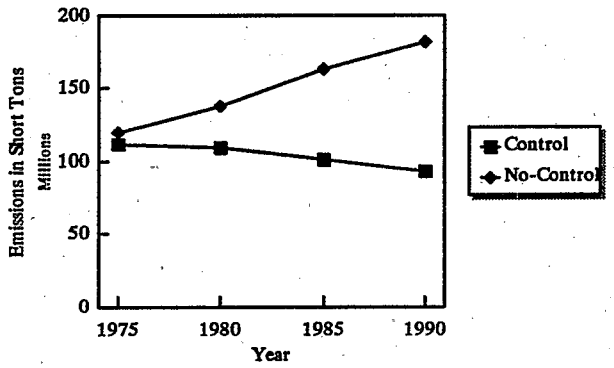


Figure 9. Control and No-control Scenario Total TSP Emission Estimates.

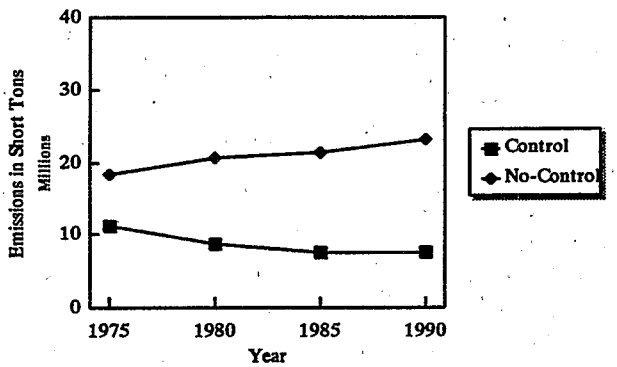
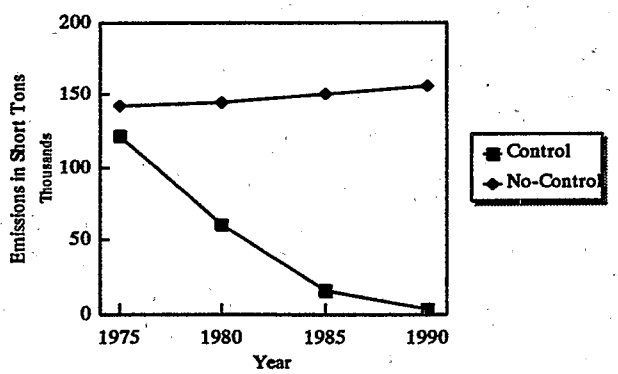


Figure 10. Control and No-control Scenario Total Pb Emission Estimates.



emissions. As shown in more detail in Appendix B, airborne lead emissions from all sectors were virtually eliminated by 1990.

As described in the following chapter and in Appendix C, these emissions inventories were used as inputs to a series of air quality models. These air quality models were used to estimate air quality conditions under the control and no-control scenarios.

Uncertainty in the Emissions Estimates

The emissions inventories developed for the control and no-control scenarios reflect at least two major sources of uncertainty. First, potential errors in the macroeconomic scenarios used to configure the sector-specific emissions model contribute to uncertainties in the emissions model outputs. Second, the emissions models themselves rely on emission factors, source allocation, source location, and other parameters which may be erroneous.

An important specific source of potential error manifest in the present study relates to hypothetical emission rates from various sources under the no-control scenario. Emission rates from motor vehicles, for example, would have been expected to change during the 1970 to 1990 period due to technological changes not directly related to implementation of the Clean Air Act (e.g., advent of electronic fuel injection, or EFI). However, the lack of emissions data from vehicles with EFI but without catalytic converters compelled the Project Team to use 1970 emission factors throughout the 1970 to 1990 period for the no-control scenario. Although this creates a potential bias in the emissions inventories, the potential errors from this and other uncertainties in the emissions inventories are considered unlikely to contribute significantly to overall uncertainty in the monetary estimates of Clean Air Act benefits. This conclusion is based on the demonstrably greater influence on the monetary benefit estimates of uncertainties in other analytical components (e.g., concentration-response functions). A list of the most significant potential errors in the emissions modeling, and their significance relative to overall uncertainty in the monetary benefit estimate, are presented in Table 7.

Table 7. Uncertainties Associated with Emissions Modeling.

Potential Source of Error	Direction of Potential Bias in Estimate of Emission Reduction Benefits	Significance Relative to Key Uncertainties in Overall Monetary Benefit Estimate
Use of 1970 motor vehicle emission factors for no-control scenario without adjustment for advent of Electronic Fuel Injection (EFI) and Electronic Ignition (EI).	Overestimate.	Unknown, but likely to be minor due to overwhelming significance of catalysts in determining emission rates.
Use of ARGUS for utility CO and VOC rather than CEUM.	Unknown.	Negligible.
Use of historical fuel consumption to estimate 1975 SO_2, NO_x, TSP utility emissions.	Unknown.	Negligible.
Adoption of assumption that utility unit inventories remain fixed between the control and no-control scenarios.	Overestimate.	Unknown, but likely to be small since the CAA had virtually no effect on costs of new coal-fired plants built prior to 1975 and these plants comprise a large majority of total coal-fired capacity operating in the 1970 to 1990 period. (See ICF CEUM Report, p. 7).

4

Air Quality

Air quality modeling is the crucial analytical step which links emissions to changes in atmospheric concentrations of pollutants which affect human health and the environmental. It is also one of the more complex and resource-intensive steps, and contributes significantly to overall uncertainty in the bottom-line estimate of net benefits of air pollution control programs. The assumptions required to estimate hypothetical no-control scenario air quality conditions are particularly significant sources of uncertainty in the estimates of air quality change, especially for those pollutants which are not linearly related to changes in associated emissions. Specific uncertainties are described in detail at the end of this chapter.

The key challenges faced by air quality modelers attempting to translate emission inventories into air quality measures involve modeling of pollutant dispersion and atmospheric transport, and modeling of atmospheric chemistry and pollutant transformation. These challenges are particularly acute for those pollutants which, rather than being directly emitted, are formed through secondary formation processes. Ozone is the paramount example since it is formed in the atmosphere through complex, nonlinear chemical interactions of precursor pollutants, particularly volatile organic compounds (VOCs) and nitrogen oxides (NO_x). In addition, atmospheric transport and transformation of gaseous sulfur dioxide and nitrogen oxides to particulate sulfates and nitrates, respectively, contributes significantly to ambient concentrations of fine particulate matter. In addition to managing the complex atmospheric chemistry relevant for some pollutants, air quality modelers also must deal with uncertainties associated with variable meteorology and the spatial and temporal distribution of emissions.

Given its comprehensive nature, the present analysis entails all of the aforementioned challenges, and involves additional complications as well. For many pollutants which cause a variety of human health and environmental effects, the concentration-response functions which have been developed to estimate those effects require, as inputs, different air quality indicators. For example, adverse human health effects of particulate matter are primarily associated with the respirable particle fraction;[27] whereas household soiling is a function of total suspended particulates, especially coarse particles. It is not enough, therefore, to simply provide a single measure of particulate matter air quality. Even for pollutants for which particle size and other characteristics are not an issue, different air quality indicators are needed which reflect different periods of cumulative exposure (i.e., "averaging periods"). For example, 3-month growing season averages are needed to estimate effects of ozone on yields of some agricultural crops, whereas

[27] Particles with an aerometric diameter of less than or equal to 10 microns.

adverse human health effect estimates require ozone concentration profiles based on a variety of short-term averaging periods.[28]

Fortunately, in responding to the need for scientifically valid and reliable estimation of air quality changes, air quality modelers and researchers have developed a number of highly sophisticated atmospheric dispersion and transformation models. These models have been employed for years supporting the development of overall federal clean air programs, national assessment studies, State Implementation Plans (SIPs), and individual air toxic source risk assessments. Some of these models, however, require massive amounts of computing power. For example, completing the 160 runs of the Regional Acid Deposition Model (RADM) required for the present study required approximately 1,080 hours of CPU time on a Cray-YMP supercomputer at EPA's Bay City Supercomputing Center.

Given the resource-intensity of many state-of-the-art models, the Project Team was forced to make difficult choices regarding where to invest the limited resources available for air quality modeling. With a mandate to analyze all of the key pollutants affected by historical Clean Air Act programs, to estimate all of the significant endpoints associated with those pollutants, and to do so for a 20 year period covering the entire continental U.S., it was necessary to use simplified approaches for most of the pollutants to be analyzed. In several cases related to primary emissions —particularly sulfur dioxide (SO_2), nitrogen oxides (NO_x), and carbon monoxide (CO)— simple "roll-up model" strategies were adopted based on the expectation that changes in emissions of these pollutants would be highly correlated with subsequent changes in air quality.[29] Significant pollutants involving secondary atmospheric formation, nonlinear formation mechanisms, and/or long-range transport were analyzed using the best air quality model which was affordable given time and resource limitations. These models, discussed in detail in Appendix C, included the Ozone Isopleth Plotting with Optional Mechanism-IV (OZIPM4) model for urban ozone; various forms of the above-referenced RADM model for background ozone, acid deposition, sulfate, nitrate, and visibility effects in the eastern U.S.; and the SJVAQS/AUSPEX Regional Modeling Adaptation Project (SARMAP) Air Quality Model (SAQM) for rural ozone in California agricultural areas. In addition, a linear scaling approach was developed and implemented to estimate visibility changes in large southwestern U.S. urban areas.

By adopting simplified approaches for some pollutants, the air quality modeling step adds to the overall uncertainties and limitations of the present analysis. The limited expanse and density of the U.S. air quality monitoring network and the limited coverage by available air quality models of major geographic areas[30] further constrain the achievable scope of the present study. Under these circumstances, it is important to remember the extent and significance of gaps in geographic coverage for key pollutants when considering the overall results of this analysis. Key uncertainties are summarized at the end of this chapter in Table 8. More extensive discussion of the caveats and uncertainties associated with the air quality modeling step is presented in Appendix C. In addition, information regarding the specific air quality models used, the characteristics of the historical monitoring data used as the basis for the control

[28] For example, ozone concentration-response data exists for effects associated with 1-hour, 2.5-hour, and 6.6-hour exposures.

[29] It is important to emphasize that the correlation expected is between changes in emissions and changes in air quality. Direct correlations between the absolute emissions estimates and empirical air quality measurements used in the present analysis may not be strong due to expected inconsistencies between the geographically local, monitor-proximate emissions densities affecting air quality data.

[30] For example, the regional oxidant models available for the present study do not cover some key Midwestern states, where human health, agricultural crop, and other effects from ozone may be significant.

scenario profiles, pollutant-specific modeling strategies and assumptions, references to key supporting documents, and important caveats and uncertainties are also presented in Appendix C.

General Methodology

The general methodological approach taken in this analysis starts with the assumption that actual historical air quality will be taken to represent the control scenario. This may seem somewhat inconsistent with the approach taken in earlier steps of the analysis, which used modeled macroeconomic conditions as the basis for estimating macroeconomic effects and emissions. However, the central focus of the overall analysis is to estimate the difference in cost and benefit outcomes between the control and no-control scenarios. It is consistent with this central paradigm to use actual historical air quality data as the basis for estimating how air quality might have changed in the absence of the Clean Air Act.

The initial step, then, for each of the five non-lead (Pb) criteria pollutants[31] was to compile comprehensive air quality profiles covering the entire analytical period from 1970 to 1990. The source for these data was EPA's Aerometric Information Retrieval System (AIRS), which is a publicly accessible database of historical air quality data. The vast number of air quality observations occurring over this twenty year period from the thousands of monitors in the U.S. indicates the need to represent these observations by statistical distributions. As documented in detail in the supporting documents covering SO_2, NO_x, CO, and ozone,[32] both lognormal and gamma distributional forms were tested against actual data to determine the form which provided the best fit to the historical data.[33] Based on these tests, one or the other statistical distribution was adopted for the air quality profiles developed for each pollutant. In addition to reducing the air quality data to a manageable form, this approach facilitated transformations of air quality profiles from one averaging period basis to another.

Once the control scenario profiles based on historical data were developed, no-control scenarios were derived based on the results of the various air quality modeling efforts. Again, the focus of the overall analysis is to isolate the difference in outcomes between the control and no-control scenarios. The no-control scenario air quality profiles were therefore derived by adjusting the control scenario profiles upward (or downward) based on an appropriate measure of the difference in modeled air quality outcomes. To illustrate this approach, consider a simplified example where the modeled concentration of Pollutant A under the no-control scenario is 0.12 ppm, compared to a modeled concentration under the control scenario of 0.10 ppm. An appropriate measure of the difference between these outcomes, whether it is the 0.02 ppm difference in concentration or the 20 percent percentage differential, is then used to ratchet up the control case profile to derive the no-control case profile. Generally, the modeled differential is applied across the entire control case profile to derive the no-control case profile. As described below in the individual sections covering particulate matter and ozone, however, more refined approaches are used where necessary to take account of differential outcomes for component species (i.e., particulate matter), long-range transport, and background levels of pollutants.

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[31] Lead (Pb), the sixth criteria pollutant, is analyzed separately. The ability to correlate emissions directly with blood lead levels obviates the need for using air quality modeling as an intermediate step toward estimation of exposure.

[32] See SAI SO_2, NO_x and CO Report (1994) and SAI Ozone Report (1995).

[33] The statistical tests used to determine goodness of fit are described in the SAI reports.

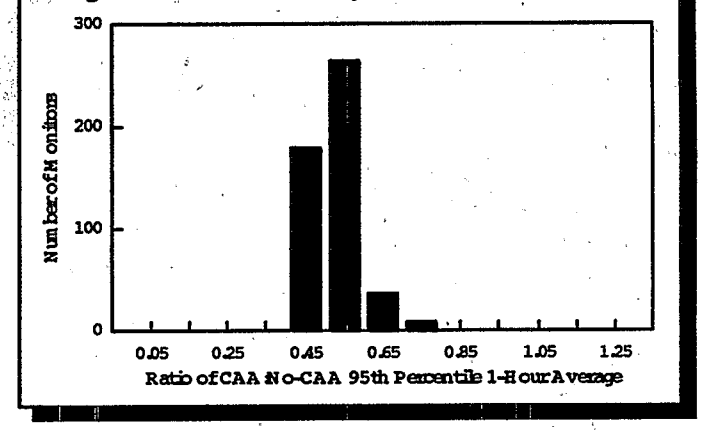
Sample Results

The results of the air quality modeling effort include a vast array of monitor-specific air quality profiles for fine particulates (PM_{10}), total suspended particulates (TSP),[34] SO_2, NO_2, NO, CO, and ozone; RADM grid cell-based estimates of sulfur and nitrogen deposition; and estimates of visibility degradation for eastern U.S. RADM grid cells and southwestern U.S. urban areas. All of these data were transferred to the effects modelers for use in configuring the human health, welfare, and ecosystem physical effects models. Given the massive quantity and intermediate nature of the air quality data, they are not exhaustively reported herein.[35] To provide the reader with some sense of the magnitude of the difference in modeled air quality conditions under the control and no-control scenarios, some illustrative results for 1990 are presented in this chapter and in Appendix C. In addition, maps depicting absolute levels of control and no-control scenario acid deposition and visibility are presented to avoid potential confusion which might arise through examination of percent change maps alone.[36]

Carbon Monoxide

Figure 11 provides an illustrative comparison of 1990 control versus no-control scenario CO concentrations, expressed as a frequency distribution of the ratios of 1990 control to no-control scenario 95th percentile 1-hour average concentrations at individual CO monitors. Consistent with the emission changes underlying these air quality results, CO concentrations under the control scenario tend to be about half those projected under the no-control scenario, with most individual monitor ratios ranging from about 0.40 to 0.60 percent, and a few with ratios in the 0.60 to 0.80 range.

Figure 11. Frequency Distribution of Estimated Ratios for 1990 Control to No-control Scenario 95th Percentile 1-Hour Average CO Concentrations, by Monitor.



In considering these results, it is important to note that CO is essentially a "hot spot" pollutant, meaning that higher concentrations tend to be observed in localized areas of relatively high emissions. Examples of such areas include major highways, major intersections, and tunnels. Since CO monitors tend

[34] PM data are reported as county-wide values for counties with PM monitors and a sufficient number of monitor observations.

[35] The actual air quality profiles, however, are available on disk from EPA. See Appendix C for further information.

[36] Large percentage changes can result from even modest absolute changes when they occur in areas with good initial (e.g., control scenario) air quality. Considering percentage changes alone might create false impressions regarding absolute changes in air quality in some areas. For example, Appendix C discusses in detail two such cases: the Upper Great Lakes and Florida-Southeast Atlantic Coast areas, which show high percentage changes in sulfur deposition and visibility.

to be located in order to monitor the high CO concentrations observed in such locations, one might suspect that using state-wide emissions changes to scale air quality concentration estimates at strategically located monitors might create some bias in the estimates. However, the vast majority of ambient CO is contributed from on-highway vehicles. In addition, the vast majority of the change in CO emissions between the control and no-control scenario occurs due to catalyst controls on highway vehicles. Since CO hot spots result primarily from highway vehicles emissions, controlling such vehicles would mean CO concentrations would be commensurately lowered at CO monitors. While variability in monitor location relative to actual hot spots and other factors raise legitimate concerns about assuming ambient concentrations are correlated with emission changes at any given monitor, the Project Team believes that the results observed provide a reasonable characterization of the aggregate change in ambient CO concentrations between the two scenarios.

Sulfur Dioxide

As for CO, no-control scenario SO_2 concentrations were derived by scaling control scenario air quality profiles based on the difference in emissions predicted under the two scenarios. Unlike CO, SO_2 is predominantly emitted from industrial and utility sources. This means that emissions, and the changes in emissions predicted under the two scenarios, will tend to be concentrated in the vicinity of major point sources. Again, while state-wide emissions changes are used to scale SO_2 concentrations between the scenarios, these state-wide emission changes reflect the controls placed on these individual point sources. Therefore, the Project Team again considers the distribution of control to no-control ratios to be a reasonable characterization of the aggregate results despite the uncertainties associated with estimation of changes at individual monitors.

Figure 12. Frequency Distribution of Estimated Ratios for 1990 Control to No-control Scenario 95th Percentile 1-Hour Average SO_2 Concentrations, by Monitor.

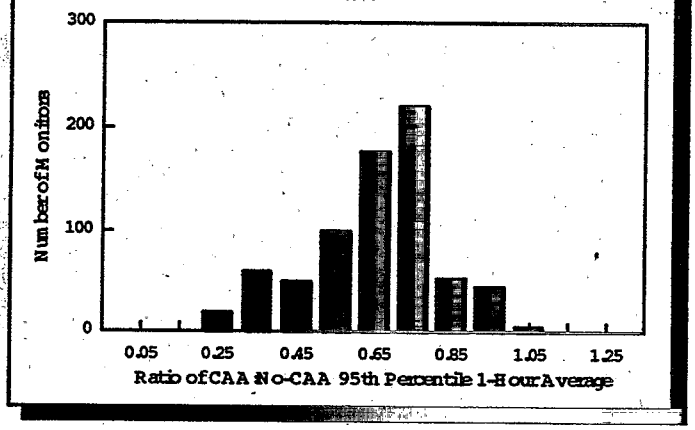


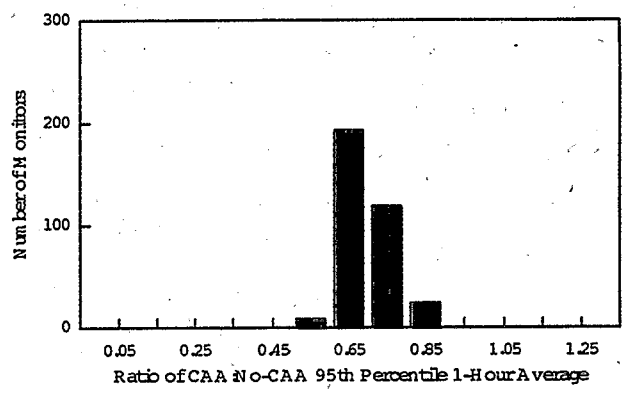
Figure 12 provides a histogram of the predicted control to no-control ratios for SO_2 which is similar to the one presented for CO. The results indicate that, on an overall basis, SO_2 concentrations were reduced by about one-third. The histogram also shows a much wider distribution of control to no-control ratios for individual monitors than was projected for CO. This result reflects the greater state to state variability in SO_2 emission changes projected in this analysis. This greater state to state variability in turn is a function of the variable responses of SO_2 point sources to historical CAA control requirements.[37] This source-specific variability was not observed for CO because controls were applied relatively uniformly on highway vehicles.

[37]Figure 12 indicates that six monitors were projected to have higher SO_2 concentrations for 1990 under the control scenario than under the no-control scenario. All six of these monitors are located in Georgia, a state for which higher 1990 utility SO_2 emissions are projected in the control scenario due to increased use of higher-sulfur coal. The projected increase in overall Georgia utility consumption of higher sulfur coal under the control case is a result of increased competition for the low-sulfur southern Appalachian coal projected to occur under the control scenario.

Nitrogen Dioxide

Results for NO_2 are presented in Figure 13. These results are similar to the results observed for CO, and for a similar reason: the vast majority of change in NO_2 emissions between the two scenarios is related to control of highway vehicle emissions. While baseline emissions of NO_2 from stationary sources may be significant, these sources were subject to minimal controls during the historical period of this analysis. On an aggregated basis, overall NO_2 concentrations are estimated to be roughly one-third lower under the control scenario than under the no-control scenario.

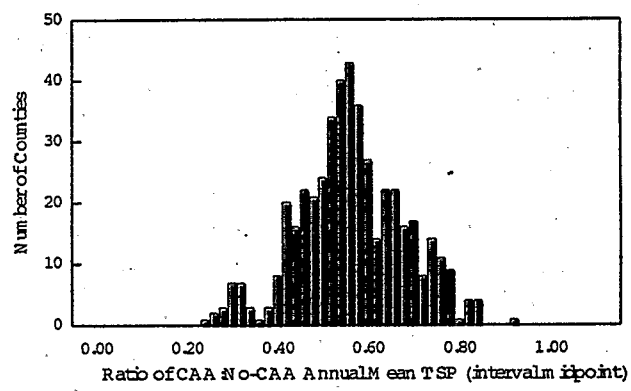
Figure 13. Frequency Distribution of Estimated Ratios for 1990 Control to No-control Scenario 95th Percentile 1-Hour Average NO_2 Concentrations, by Monitor.



Particulate Matter

An indication of the difference in outcomes for particulate matter between the two scenarios is provided by Figure 14. This graph shows the distribution of control to no-control ratios for annual mean TSP in 1990 for those counties which both had particulate monitors and a sufficient number of observations from those monitors.[38] While the distribution of results is relatively wide, reflecting significant county to county variability in ambient concentration, on a national aggregate basis particulate matter concentrations under the control scenario were just over half the level projected under the no-control scenario. The significant county to county variability observed in this case reflects point source-specific controls on particulate matter precursors, especially SO_2, and the effects of long-range transport and transformation.

Figure 14. Distribution of Estimated Ratios for 1990 Control to No-Control Annual Mean TSP Concentrations, by Monitored County.



Ozone

Urban Ozone

Figure 15 presents a summary of the results of the 1990 OZIPM4 ozone results for all 147 of the modeled urban areas. In this case, the graph depicts the distribution of ratios of peak ozone concentrations

[38] Given the relative importance of particulate matter changes to the bottom line estimate of CAA benefits, and the fact that a substantial portion of the population lives in unmonitored counties, a methodology was developed to allow estimation of particulate matter benefits for these unmonitored counties. This methodology was based on the use of regional air quality modeling to interpolate between monitored counties. It is summarized in Appendix C and described in detail in the SAI PM Interpolation Report (1996).

estimated for the control and no-control scenarios. While the vast majority of simulated peak ozone concentration ratios fall below 1.00, eight urban areas show lower simulated peak ozone for the no-control scenario than for the control scenario. For these eight urban areas, emissions of precursors were higher under the no-control scenario; however, the high proportion of ambient NO_x compared to ambient non-methane organic compounds (NMOCs) in these areas results in a decrease in net ozone production in the vicinity of the monitor when NO_x emissions increase.[39]

Rural Ozone

Figures 16 and 17 present frequency distributions for control to no-control ratios of average ozone-season daytime ozone concentrations at rural monitors as simulated by RADM and SAQM, respectively.

Both the RADM and SAQM results indicate relatively little overall change in rural ozone concentrations. This is primarily because reductions in ozone precursor emissions were concentrated in populated areas.

Acid Deposition

Figure 18 is a contour map showing the estimated percent increase in sulfur deposition under the no-control scenario relative to the control scenario for 1990. Figure 19 provides comparable information for nitrogen deposition.

These results show that acid deposition rates increase significantly under the no-control scenario, particularly in the Atlantic Coast area and in the vicinity of states for which relatively large increases in emissions

Figure 15. Distribution of Estimated Ratios for 1990 Control to No-control OZIPM4 Simulated 1-Hour Peak Ozone Concentrations, by Urban Area.

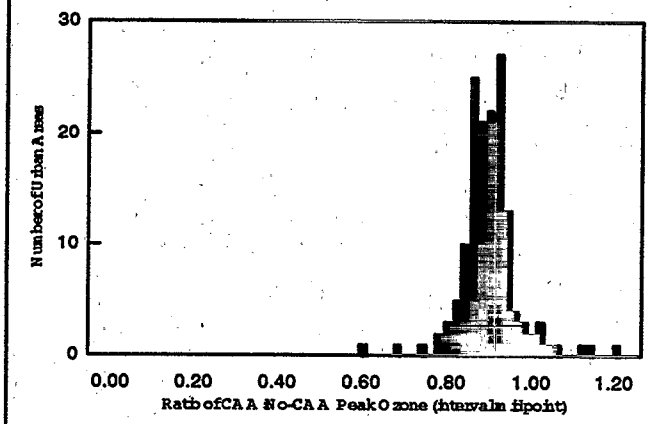


Figure 16. Distribution of Estimated Ratios for 1990 Control to No-control RADM Simulated Daytime Average Ozone Concentrations, by RADM Grid Cell.

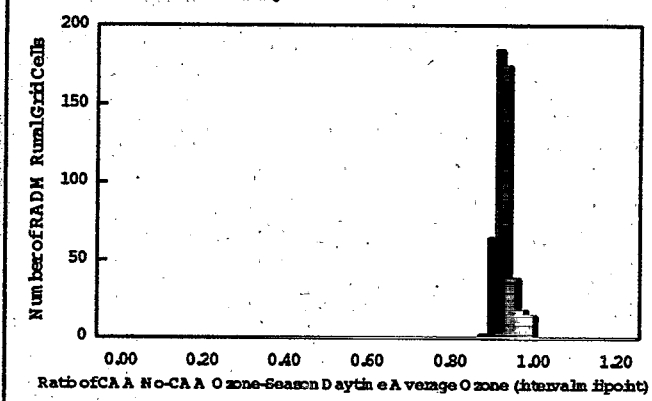
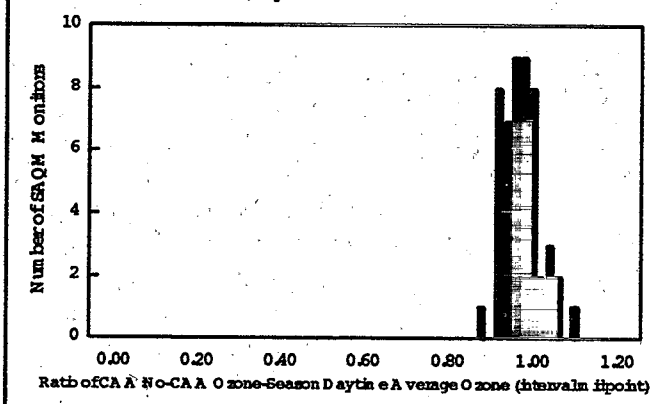


Figure 17. Distribution of Estimated Ratios for 1990 Control to No-control SAQM Simulated Daytime Average Ozone Concentrations, by SAQM Monitor.



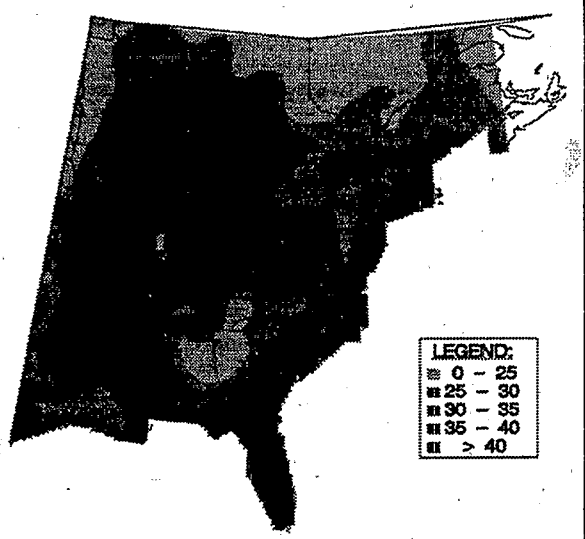
[39] Over an unbounded geographic area, NO_x reductions generally decrease net ozone production.

(i.e., Kentucky, Florida, Michigan, Mississippi, Connecticut, and Florida).

In the areas associated with large increases in sulfur dioxide emissions, rates of sulfur deposition increase to greater than or equal to 40 percent. The high proportional increase in these areas reflects both the significant increase in acid deposition precursor emissions in upwind areas and the relatively low deposition rates observed under the control scenario.[40]

Along the Atlantic Coast, 1990 nitrogen deposition rates increase by greater than or equal to 25 percent under the no-control scenario. This is primarily due to the significant increase in mobile source nitrogen oxide emissions along the major urban corridors of the eastern seaboard.

Figure 18. RADM-Predicted Percent Increase in Total Sulfur Deposition (Wet + Dry) Under the No-control Scenario.

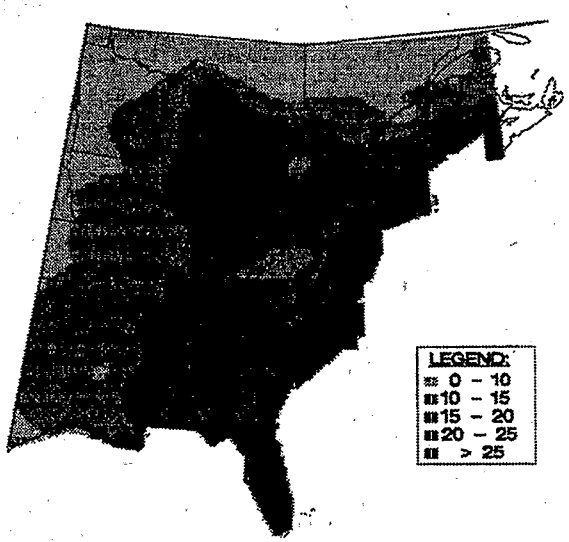


Visibility

The difference in modeled 1990 control and no-control scenario visibility conditions projected by the RADM/EM for the eastern U.S. is depicted by the contour map presented in Figure 21. This figure shows the increase in modeled annual average visibility degradation, in DeciView[41] terms, for 1990 when moving from the control to the no-control scenario. Since the DeciView metric is based on perceptible changes in visibility, these results indicate noticeable deterioration of visibility in the eastern U.S. under the no-control scenario.

Visibility changes in 30 southwestern U.S. urban areas were also estimated using emissions scaling techniques. This analysis also found significant, perceptible changes in visibility between the two scenarios. Details of this analysis, including the specific outcomes for the 30 individual urban areas, are presented in Appendix C.

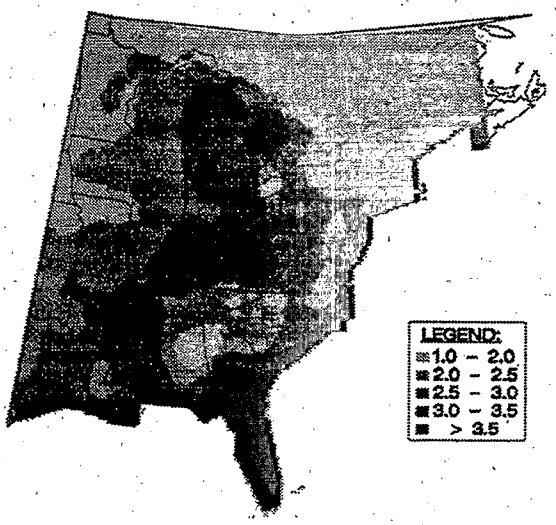
Figure 19. RADM-Predicted Percent Increase in Total Nitrogen Deposition (Wet + Dry) Under the No-control Scenario.



[40] Even small changes in absolute deposition can yield large percentage changes when initial absolute deposition is low. See Appendix C for further discussion of this issue.

[41] The DeciView Haze Index (dV) is a relatively new visibility indicator aimed at measuring visibility changes in terms of human perception. It is described in detail in Appendix C.

Figure 20. RADM-Predicted Increase in Visibility Degradation, Expressed in DeciViews, for Poor Visibility Conditions (90th Percentile) Under the No-control Scenario.



Uncertainty prevades the projected changes in air quality presented in this study. These uncertainties arise due to potential inaccuracies in the emissions inventories used as air quality modeling inputs and due to potential errors in the structure and parameterization of the air quality models themselves. In addition, an important limitation of the present study is the lack of available data and/or modeling results for some pollutants in some regions of the country (e.g., visibility changes in western U.S. Class I areas such as the Grand Canyon). The inability to provide comprehensive estimates of changes in air quality due to the Clean Air Act creates a downward bias in the monetary benefit estimates.

The most important specific sources of uncertainty are presented in Table 8, and are described further in Appendix C. While the list of potential errors presented in Table 8 is not exhaustive, it incorporates the uncertainties with the greatest potential for contributing to error in the monetary benefit estimates. Overall, the uncertianties in the estimated change in air quality are considered small relative to uncertainties contributed by other components of the analysis.

Table 8. Key Uncertainties Associated with Air Quality Modeling.

Potential Source of Error	Direction of Potential Bias in Estimate of Air Quality Benefits	Significance Relative to Key Uncertainties in Overall Monetary Benefit Estimate
Use of OZIPM4 model, which does not capture long-range and night-time transport of ozone. Use of a regional oxidant model, such as UAM-V, would mitigate errors associated with neglecting transport.	Underestimate.	Significant, but probably not major. Overall average ozone response of 15% to NO_x and VOC reductions of approximately 30% and 45%, respectively. Even if ozone response doubled to 30%, estimate of monetized benefits of CAA will not change very much. Significant benefits of ozone reduction, however, could not be monetized.
Use of early biogenic emission estimates in RADM to estimate rural ozone changes in the eastern 31 states.	Underestimate.	Probably minor. Errors are estimated to be within -15% to +25% of the ozone predictions.
Use of proxy pollutants to scale up some particulate species in some areas. Uncertainty is created to the extent species of concern are not perfectly correlated with the proxy pollutants.	Unknown.	Potentially significant. Given the relative importance of the estimated changes in fine particle concentrations to the monetized benefit estimate, any uncertainty associated with fine particles is potentially significant. However, the potential error is mitigated to some extent since proxy pollutant measures are applied to both scenarios.
Use of state-wide average emission reductions to configure air quality models. In some cases, control programs may have been targeted to problem areas, so using state-wide averages would miss relatively large reductions in populated areas.	Underestimate.	Probably minor.
Exclusion of visibility benefits in Class I areas in the Southwestern U.S.	Underestimate.	Probably minor. No sensitivity analysis has been performed; however, monetized benefits of reduced visibility impairment in the Southwest would probably not significantly alter the estimate of monetized benefits.

Table 8 (con't). Key Uncertainties Associated with Air Quality Modeling.

Potential Source of Error	Direction of Potential Bias in Estimate of Air Quality Benefits	Significance Relative to Key Uncertainties in Overall Monetary Benefit Estimate
Lack of model coverage in western 17 states for acid deposition.	Underestimate.	Probably minor. No sensitivity analysis has been performed; however, monetized benefits of reduced acid deposition in the 17 western states would probably not significantly alter the estimate of monetized benefits.
Use of spatially and geographically aggregated emissions data to configure RADM. Lack of available day-specific meteorological data results in inability to account for temperature effects on VOCs and effect of localized meteorology around major point sources.	Unknown.	Potentially significant. Any effect which might influence the direction of long-range transport of fine particulates such as sulfates and nitrates could significantly influence the estimates of total monetized benefits of the CAA.
Use of late-1980's monitoring data to derive the PM_{10}:TSP ratios needed to develop full PM profiles. Since early control programs focused significantly on combustion sources, using recent year PM_{10}:TSP ratios may lead to underestimation of early period controls of the fine particle fraction.	Underestimate.	Potentially significant. Any underestimation of the baseline contribution of fine particles to total PM would lead to an underestimation of fine particle controls, which are a relatively significant contributor to the monetized benefit estimate.
Use of constant concentration for organic aerosols between the two scenarios. Holding organic aerosol concentrations fixed omits the effect of changes in this constituent of fine particulate matter.	Underestimate.	Probably minor, because (a) nitrates were also held fixed and nitrates and organic aerosols move in opposite directions so the exclusion of both mitigates the effect of omitting either, (b) sulfates are by far the dominant species in the eastern U.S., and (c) larger errors would be introduced by using emissions scaling to estimate changes in organic aerosols since most organic aerosols are caused by biogenic emissions which don't change between the scenarios.
Unavailability of ozone models for rural areas outside the RADM and SAQM domains.	Underestimate.	Probably minor. Misses potential human health, welfare, and ecological benefits of reducing rural ozone in agricultural and other rural areas; however, ozone changes are likely to be small given limited precursor reductions in rural areas. RADM control:no-control ratios are in fact, relatively small.

5

Physical Effects

Human Health and Welfare Effects Modeling Approach

This chapter identifies and, where possible, estimates the principal health and welfare benefits enjoyed by Americans due to improved air quality resulting from the CAA. Health benefits have resulted from avoidance of air pollution-related health effects, such as premature mortality, respiratory illness, and heart disease. Welfare benefits accrued where improved air quality averted damage to measurable resources, including agricultural production and visibility. The analysis of physical effects required a combination of three components: air quality, population, and health or welfare effects. As structured in this study, the 3-step process involved (1) estimating changes in air quality between the control and no-control scenarios, (2) estimating the human populations and natural resources exposed to these changed air quality conditions, and (3) applying a series of concentration-response equations which translated changes in air quality to changes in physical health and welfare outcomes for the affected populations.

Air Quality

The Project Team first estimated changes in concentrations of criteria air pollutants between the control scenario, which at this step was based on historical air quality, and the no-control scenario. Air quality improvements resulting from the Act were evaluated in terms of both their temporal distribution from 1970 to 1990 and their spatial distribution across the 48 conterminous United States. Generally, air pollution monitoring data provided baseline ambient air quality levels for the control scenario. Air quality modeling was used to generate estimated ambient concentrations for the no-control scenario. A variety of modeling techniques was applied, depending on the pollutant modeled. These modeling approaches and results are summarized in Chapter 4 and presented in detail in Appendix C.

Population

Health and some welfare benefits resulting from air quality improvements were distributed to individuals in proportion to the reduction in exposure. Predicting individual exposures, then, was a necessary step in estimating health effects. Evaluating exposure changes for the present analysis required not only an understanding of where air quality improved as a result of the CAA, but also how many individuals were affected by varying levels of air quality improvements. Thus, a critical component of the benefits analysis required that the distribution of the U.S. population nationwide be established.

Three years of U.S. Census data were used to represent the geographical distribution of U.S. residents: 1970, 1980, and 1990. Population data was supplied at the census block group level, with approximately 290,000 block groups nationwide. Allocating air quality improvements to the population for the other target years of this study – 1975 and 1985 – necessitated interpolation of the three years of population data. Linear interpolation was accomplished for each block group in order to maintain the variability in growth rates throughout the country.

Health and Welfare Effects

Benefits attributable to the CAA were measured in terms of the avoided incidence of physical health effects and measured welfare effects. To quantify such benefits, it was necessary to identify concentration-response relationships for each effect being considered. As detailed in Appendix D, such relationships were derived from the published science literature. In the case of health effects, concentration-response functions combined the air quality improvement and population distribution data with estimates of the number of fewer individuals that suffer an adverse health effect per unit change in air quality. By evaluating each concentration-response function for every monitored location throughout the country, and aggregating the resulting incidence estimates, it was possible to generate national estimates of incidence under the control and no-control scenarios.

In performing this step of the analysis, the Project Team discovered that it was impossible to estimate all of the health and welfare benefits which have resulted from the Clean Air Act. While scientific information was available to support estimation of some effects, many other important health and welfare effects could not be estimated. Furthermore, even though some physical effects could be quantified, the state of the science did not support assessment of the economic value of all of these effects. Table 10 shows the health effects for which quantitative analysis was prepared, as well as some of the health effects which could not be quantified in the analysis. Table 11 provides similar information for selected welfare effects.

While the 3-step analytical process described above was applied for most pollutants, health effects for lead were evaluated using a different methodology. Gasoline as a source of lead exposure was addressed separately from conventional point sources. Instead of using ambient concentrations of lead resulting from use of leaded gasoline, the concentration-response functions linked changes in lead releases directly to changes in the population's mean blood lead level. The amount of leaded gasoline used each year was directly related to mean blood lead levels using a relationship described in the 1985 Lead Regulatory Impact Analysis (U.S. EPA, 1985). Health effects resulting from exposure to point sources of atmospheric lead, such as industrial facilities, were considered using the air concentration distributions modeled around these point sources. Concentration-response functions were then used to estimate changes in blood lead levels in nearby populations.

Most welfare effects were analyzed using the same basic 3-step process used to analyze health effects, with one major difference in the concentration-response functions used. Instead of quantifying the relationship between a given air quality change and the number of cases of a physical outcome, welfare effects were measured in terms of the avoided resource losses. An example is the reduction in agricultural crop losses resulting from lower ambient ozone concentrations under the control scenario. These agricultural benefits were measured in terms of net economic surplus.

Table 10. Human Health Effects of Criteria Pollutants.

Pollutant	Quantified Health Effects	Unquantified Health Effects	Other Possible Effects
Ozone	Respiratory symptoms Minor restricted activity days Respiratory restricted activity days Hospital admissions Asthma attacks Changes in pulmonary function Chronic Sinusitis & Hay Fever	Increased airway responsiveness to stimuli Centroacinar fibrosis Inflammation in the lung	Immunologic changes Chronic respiratory diseases Extrapulmonary effects (e.g., changes in structure, function of other organs)
Particulate Matter/ TSP/ Sulfates	Acute bronchitis Hospital admissions Mortality Chronic bronchitis Chest illness Respiratory symptoms Minor restricted activity days All restricted activity days Days of work loss Moderate or worse asthma status (asthmatics)	Changes in pulmonary function	Chronic respiratory diseases other than chronic bronchitis Inflammation in the lung
Carbon Monoxide	Hospital Admissions Decreased time to onset of angina	Behavioral effects	Other cardiovascular effects Developmental effects
Nitrogen Oxides	Respiratory illness	Increased airway responsiveness	Decreased pulmonary function Inflammation in the lung Immunological changes
Sulfur Dioxide	In exercising asthmatics: Respiratory symptoms	In exercising asthmatics: Changes in pulmonary function Combined responses of respiratory symptoms and pulmonary function changes	Respiratory symptoms in non-asthmatics Hospital admissions
Lead	Hypertension Non-fatal coronary heart disease Non-fatal strokes Mortality IQ loss effect (on lifetime earnings) IQ loss effects on special education needs Neonatal mortality due to decreased gestational age	Reproductive effects Fetal effects from maternal exposure Other neurological and metabolic effects Other cardiovascular effects	Cancer

Another important welfare effect is the benefit accruing from improvements in visibility under the control scenario. Again, a slightly different methodological approach was used to evaluate visibility improvements. Visibility changes were a direct output of the models used to estimate changes in air quality. The models provided estimates of changes in light extinction, which were then translated mathematically into various specific measures of perceived visibility change. These visibility change

Table 11. Selected Welfare Effects of Criteria Pollutants.

Pollutant	Quantified Welfare Effects	Unquantified Welfare Effects
Ozone	Changes in crop yields (for 7 crops) Decreased worker productivity	Changes in other crop yields Materials damage Effects on forests Effects on wildlife
Particulate Matter/ TSP/ Sulfates	Household soiling Visibility	Other materials damage Effects on wildlife
Nitrogen Oxides	Visibility	Crop losses due to acid deposition Materials damage due to acid deposition Effects on fisheries due to acidic deposition Effects on forests
Sulfur Dioxide	Visibility	Crop losses due to acid deposition Materials damage due to acid deposition Effects on fisheries due to acidic deposition Effects on forests

measures were then combined with population data to estimate the economic value of the visibility changes. Other welfare effects quantified in terms of avoided resource losses include household soiling damage by PM_{10} and decreased worker productivity due to ozone exposure. The results of the welfare effects analysis are found in Chapter 6 and in Appendices D and F.

Because of a lack of available concentration-response functions (or a lack of information concerning affected populations), ecological effects were not quantified for this analysis. However, Appendix E provides discussion of many of the important ecological benefits which may have accrued due to historical implementation of the CAA.

Key Analytical Assumptions

Several important analytical assumptions affect the confidence which can be placed in the results of the physical effects analysis. The most obvious concerns the choices which have to be made among competing scientific studies. The Project Team relied on the most recent available, published scientific literature to ascertain the relationship between air pollution and human health and welfare effects. The choice of studies, and the uncertainties underlying those studies, also created uncertainties in the results. For example, to the extent the published literature may collectively overstate the effects of pollution, EPA's analysis will overstate the effects of the CAA. Such outcomes may occur because scientific research which fails to find significant relationships is less likely to be published than research with positive results. On the other hand, history has shown that it is highly likely that scientific understanding of the effects of air pollution will improve in the future, resulting in discovery of previously unknown effects. Important examples of this phenomenon are the substantial expected health and welfare benefits of reductions in lead and ambient particulate matter, both of which have been shown in recent studies to impose more severe effects than scientists previously believed. To the extent the present analysis misses effects of air pollution

that have not yet been subject to adequate scientific inquiry, the analysis may understate the effects of the CAA.

Because these resultant uncertainties were caused by the inadequacy of currently available scientific information, there is no compelling reason to believe that the results of the present analysis are biased in a particular direction. Some significant uncertainties, however, may have arisen from interpretation of model results, underlying data, and supporting scientific studies. These assumptions and uncertainties are characterized in this report to allow the reader to understand the degree of uncertainty and the potential for misestimation of results. In addition, the overall results are presented in ranges to reflect the aggregate effect of uncertainty in key variables. A quantitative assessment of some of the uncertainties in the present study is presented in Chapter 7. In addition, the key uncertainties in the physical effects modeling step of this analysis are summarized in Table 16 at the end of this chapter. The remainder of this section discusses a few of the most important modeling procedures and associated assumptions.

Mapping Populations to Monitors

As noted above, the Project Team's method of calculating benefits of air pollution reductions required a correlation of air quality data changes to exposed populations. For pollutants with monitor-level data (i.e., SO_2, O_3, NO_2, CO), it was assumed that all individuals were exposed to air quality changes estimated at the nearest monitor. For PM_{10}, historical air quality data were available at the county level. All individuals residing in a county were assumed to be exposed to that county's PM_{10} air quality.[42]

Many counties did not contain particulate matter air quality monitors or did not have a sufficient number of monitor observations to provide reliable estimates of air quality. For those counties, the Project Team conducted additional analyses to estimate PM_{10} air quality changes during the study period. For counties in the eastern 31 states, the grid cell-specific sulfate particle concentrations predicted by the RADM model were used to provide a scaled interpolation between monitored counties. For counties outside the RADM domain, an alternative method based on state-wide average concentrations was used. With this supplemental analysis, estimates were developed of the health effects of the CAA on almost the entire continental U.S. population. Compliance costs incurred in Alaska and Hawaii were included in this study, but the benefits of historical air pollution reductions were not. In addition, the CAA yielded benefits to Mexico and Canada which were not captured in this study.

[42] In some counties and in the early years of the study period, particulate matter was monitored as TSP rather than as PM_{10}. In these cases, PM_{10} was estimated by applying TSP:PM_{10} ratios derived from historical data. This methodology is described in Appendix C.

Air quality monitors are more likely to be found in high pollution areas rather than low-pollution areas. Consequently, mapping population to the nearest monitor *regardless of the distance to that monitor* almost certainly results in an overstatement of health impacts due to air quality changes for those populations. The Project Team conducted a sensitivity analysis to illustrate the importance of the "mapping to nearest monitor" assumption. For comparison to the base case, which modeled exposure for the 48 state population, Table 12 presents the percentage of the total 48-state population covered in the "50 km" sensitivity scenario. For most pollutants in most years, 25 percent or more of the population resided more than 50 km from an air quality monitor (or in a county without PM_{10} monitors). Estimated health benefits are approximately linear to population covered -- that is, if the population modeled for a pollutant in a given year in the sensitivity analysis is 25 percent smaller than the corresponding population modeled in the base case, then estimated health benefits are reduced by roughly 25 percent in the sensitivity case. This sensitivity analysis demonstrates that limiting the benefits analysis to reflect only those living within 50 km of a monitor or within a PM-monitored county would lead to a substantial underestimate of the historical benefits of the CAA. Since these alternative results may have led to severely misleading comparisons of the costs and benefits of the Act, the Project Team decided to adopt the full 48-state population estimate as the central case for this analysis despite the greater uncertainties and potential biases associated with estimating exposures from distant monitoring sites.

Table 12. Percent of Population (of the Continental US) within 50km of a monitor (or in a County with PM monitors), 1970-1990.

	Pollutant				
Year	PM_{10}	O_3	NO_2	SO_2	CO
1975	79%	56%	53%	65%	67%
1980	80%	71%	59%	73%	68%
1985	75%	72%	61%	73%	68%
1990	68%	74%	62%	71%	70%

Choice of Study

As noted above, the Project Team estimated health and welfare effects of improved air quality through the use of concentration-response (CR) functions derived from the peer-reviewed scientific literature. For some health endpoints, however, several functions were available, and significant differences were found among the CR functions in implied health effects. For example, 19 CR functions were available correlating excess premature mortality to short-term exposure to particulate matter (PM). It is not unusual for two equally-reputable studies to differ by a factor of three or four in implied health impact. The difference in implied health effects across studies can be considered an indication of the degree of scientific uncertainty associated with measurement of that health effect.

Where more than one acceptable study was available, the Project Team used CR functions from *all* relevant studies to infer health effects. That is, the health effect implied by each study is reported (see Appendix D), and a range of reported results for a particular health endpoint can be interpreted as a measure of the uncertainty of the estimate.

Variance Within Studies

Even where only one CR function was available for use, the uncertainty associated with application of that function to estimate physical outcomes can be evaluated quantitatively. Health effects studies provided "best estimates" of the relationship between air quality changes and health effects, and a measure of the statistical uncertainty of the relationship. In this analysis, the Project Team used simulation modeling

techniques to evaluate the overall uncertainty of the results given uncertainties within individual studies, across studies examining a given endpoint, and in the economic valuation coefficients applied to each endpoint. The analysis estimating aggregate quantitative uncertainty is presented in Chapter 7.

Health Effects Modeling Results

This section provides a summary of the differences in health effects estimated under the control and no-control scenarios. Because the differences in air quality between the two scenarios generally increased from 1970 to 1990, and the affected population grew larger during that period, the beneficial health effects of the CAA increased steadily during the 1970 to 1990 period. More detailed results are presented in Appendix D.

Mortality

The most serious human health impact of air pollution is an increase in incidences of premature mortality. Increased levels of lead (Pb) in the bloodstream and increases in various measures of ambient particulate matter have each been correlated with elevated mortality rates. For presentation purposes in this analysis, the expected reduction in mortality rates due to CAA-induced air quality improvements have been expressed as avoided "excess premature mortalities per year." For Pb, separate analyses were conducted for men, women, and children, and then aggregated to present a total effect for Pb (see Appendix G for detailed information on methods, sources, and results of the Pb mortality analysis). For PM_{10}, two types of studies were used as sources of concentration/response functions. Table 13 presents estimated avoided excess premature mortalities for 1990 only, with the mean estimate and 90 percent confidence interval. See Appendix D for more detail on results implied by individual epidemiological studies, and on the temporal pattern of impacts.[43]

Short-Term Exposure Studies

A substantial body of published health science literature recognizes a correlation between elevated PM concentrations (either PM_{10}, or a related measure such as TSP) and increased mortality rates. Many of the studies evaluate changes in mortality rates several days after a period of elevated PM concentrations. In general, strong correlations have been found. These "short-term exposure" or "episodic" studies are unable to address two important issues: The degree to which the observed excess mortalities are "premature," and the degree to which mortality rates are correlated with long-term exposure to elevated PM concentrations (that is, exposures over many years rather than a few days).

Because the episodic mortality studies evaluate the mortality rate impact a only few days after a high-pollution event, it is likely that many of the "excess mortality" cases represented individuals who were already suffering impaired health, and for whom the high-pollution event represented an exacerbation of an already serious condition. Based on the episodic studies only, however, it is unknown how many of the victims would have otherwise lived only a few more days or weeks, or how many would have recovered to enjoy many years of a healthy life in the absence of the high-pollution event. For the purpose of cost-

[43] Earlier years are estimated to have had fewer excess premature mortalities.

benefit analysis, it can be important to determine whether a pollution event reduces the average lifespan by several days or by many years. Although the episodic mortality studies do not provide an estimate of the expected life years lost (nor do they address the health status of victims), some *have* evaluated the age of the excess premature mortality cases, and have estimated that 80 to 85 percent of the victims are age 65 or older.

In addition to causing short-term health problems, air pollution (measured by elevated PM_{10} concentrations) can cause longer-term health problems that may lead to premature mortality. Such long-term changes in susceptibility to premature mortality in the future will be missed by efforts to correlate premature mortalities with near-term episodes of elevated pollution concentrations. Consequently, excess premature mortality estimates based on the results of the "episodic" mortality studies will underestimate the effect of long-term elevated pollution concentrations on mortality rates.

Long-Term Exposure Studies

Several health effects studies have correlated long-term exposure to criteria air pollutants (measured by PM_{10} concentrations, or a related measure such as TSP concentration) with annual mortality rates. These studies are able to avoid some of the weaknesses of the episodic studies. In particular, by investigating changes in annual (rather than daily) mortality rates, the long-term studies do not predict most cases of excess premature mortality where mortality is deferred for only a few days; also, the long-term studies are able to discern changes in mortality rates due to long-term exposure to elevated air pollution concentrations. Other characteristics, however, make this class of studies imperfectly suited for the present analysis. Most importantly, the concentration response functions inferred from the long-term exposure studies may overstate health effects of elevated air pollution concentrations if pollution exposures from before the study period affect health status during the study period.

Avoided Excess Premature Mortality Estimates

The number of premature mortalities avoided in 1990 (only) estimated by the health benefits model is presented in Table 13. The model reports a range of results for each health endpoint. Here, the fifth percentile, mean, and ninety-fifth percentile estimates are used to characterize the distribution. One study (Pope *et al.*, 1995) was used as the source for a concentration-response function for long-term exposure. Concentration-response data inferred from nineteen studies were used to estimate short-term exposure effects. The long-term and short-term results are not additive -- they should be viewed as representing a plausible range of avoided premature mortalities. The estimates of premature mortalities avoided due to reduced lead exposure *are* additive with the PM-related short-term and long-term exposure estimates.

Table 13. Criteria Pollutants Health Benefits -- Distributions of 1990 Avoided Premature Mortalities (thousands of cases reduced) for 48 State Population

		Annual Cases Avoided (thousands)		
Endpoint	Pollutant	5th %ile	Mean	95th %ile
Mortality (long-term exposure)	PM-10	145	238	341
Mortality (short-term exposure)	PM-10	55	58	61
Mortality (Lead exposure)	Lead	8	24	58

Non-Fatal Health Impacts

The health benefits model reports non-fatal health effects estimates similarly to estimates of premature mortalities: as a range of estimates for each quantified health endpoint, with the range dependent on the quantified uncertainties in the underlying concentration-response functions. The range of results for 1990 only is characterized in Table 14 with fifth percentile, mean, and ninety-fifth percentile estimates. All estimates are expressed as thousands of new cases avoided in 1990. "Lost IQ Points" represent the aggregate number of points (in thousands) across the population affected by lead concentrations in 1990. All "Hospital Admissions" estimates are in thousands of admissions, regardless of the length of time spent in the hospital. "Shortness of breath" is expressed as thousands of days: that is, one "case" represents one child experiencing shortness of breath for one day. Likewise, "Restricted Activity Days" and "Work Loss Days" are expressed in person-days.

Table 14. Criteria Pollutants Health Benefits – Distributions of 1990 Non-Fatal Avoided Incidence (thousands of cases reduced) for 48 State Population

Endpoint	Pollutant(s)	Annual Cases Avoided (thousands)		
		5th %ile	Mean	95th %ile
Chronic Bronchitis	PM-10	491	671	895
Non-Fatal Lead-induced Ailments				
Lost IQ Points	Lead	7,600	10,400	13,100
IQ < 70	Lead	40	49	59
Hypertension	Lead	9,700	12,800	15,600
Coronary Heart Disease	Lead	5	36	105
Atherothrombotic brain infarction	Lead	0	7	18
Initial cerebrovascular accident	Lead	0	9	26
Hospital Admissions				
All Respiratory	PM-10 & O3	74	88	104
COPD + Pneumonia	PM-10 & O3	56	63	71
Ischemic Heart Disease	PM-10	8	19	31
Congestive Heart Failure	PM-10 & CO	28	39	50
Other Respiratory-Related Ailments				
Children				
Shortness of breath, days	PM-10	15,000	68,000	129,000
Acute Bronchitis	PM-10	0	8,700	21,600
Upper & Lower Resp. Symptoms	PM-10	6,200	9,500	12,300
Adults				
Any of 19 Acute Symptoms	PM-10 & O3	25,000	137,000	252,000
All				
Asthma Attacks	PM-10 & O3	860	1,710	2,580
Increase in Respiratory Illness	NO2	4,800	9,900	14,200
Any Symptom	SO2	26	267	759
Restricted Activity and Work Loss Days				
MRAD	PM-10 & O3	8,030	8,940	9,930
Work Loss Days (WLD)	PM-10	1,390	1,610	1,830

The following additional welfare benefits were quantified directly in economic terms: household soiling damage, visibility, decreased worker productivity, and agricultural benefits (measured in terms of net surplus).

Other Physical Effects

Human health impacts of criteria pollutants dominate quantitative analyses of the effects of the CAA, in part because the scientific bases for quantifying air quality and physical effect relationships are most advanced for health effects. The CAA yielded other benefits, however, which are important even though they were sometimes difficult or impossible to quantify fully given currently available scientific and applied economic information.

Ecological Effects

Beyond the intrinsic value of preserving natural aquatic and terrestrial ecosystems and the life they support, protection of ecosystems from the adverse effects of air pollution can yield significant benefits to human welfare. The historical reductions in air pollution achieved under the CAA probably led to significant improvements in the health of ecosystems and the myriad ecological services they provide. For example, improvements in water quality stemming from a reduction in acid deposition-related air pollutants such as SO_x and NO_x likely yielded significant benefits to human welfare through enhancements in commercial and recreational fishing, wildlife viewing, maintenance of biodiversity, nutrient cycling, and maintenance and improvement of drinking water quality. Increased growth and productivity of U.S. forests were likely enhanced due to reduced concentrations of ambient ozone. Protection of forest resources results in benefits associated with increased timber production, improved outdoor recreation (e.g., hunting, camping), and ecological services such as nutrient cycling and carbon sequestration. These improvements in ecological conditions have not been quantified in this assessment. For a fuller discussion of the possible ecological effects of the CAA, see Appendix E.

Quantified Agricultural Effects

Quantification of the effects of the CAA on agriculture was limited to effects on seven crops. Changes in crop yields between the two scenarios were less than one percent in 1990 for barley, corn, soybeans, and sorghum. Changes in peanut and cotton yields were estimated in the four to seven percent range, and winter wheat yield changes were estimated at one-half of one percent to eight percent. These ranges reflect usage of alternate exposure-response functions. Exposure-response relationships and cultivar mix reflect historical patterns and do not account for possible substitution of more ozone-resistant cultivars in the no-control scenario. Thus, these are likely overestimates of the actual effect of the CAA on crop yields. It is unclear whether that likely overestimate is greater than or less than the underestimate of benefits due to exclusion of all other crops from the quantitative analysis.

Effects of Air Toxics

In addition to control of criteria pollutants, the Clean Air Act resulted in control of some air toxics — defined as non-criteria pollutants which can cause adverse effects to human health and the environment. Control of these pollutants resulted both from incidental control due to criteria pollutant programs and

specific controls targeted at air toxics through the National Emission Standards for Hazardous Air Pollutants (NESHAPs) under Section 112 of the Act.

Air toxics are capable of producing a wide variety of effects. Table 15 presents the range of potential human health and ecological effects which can occur due to air toxics exposure. For several years, the primary focus of risk assessments and control programs designed to reduce air toxics has been cancer. According to present EPA criteria, there are over 100 known or suspected carcinogens. EPA's 1990 Cancer Risk study indicated that as many as 1,000 to 3,000 cancers annually may be attributable to the air toxics for which assessments were available (virtually all of this estimate came from assessments of about a dozen well-studied pollutants).[44]

In addition to cancer, these pollutants can cause a wide variety of health effects, ranging from respiratory problems to reproductive and developmental effects. There has been considerably less work done to assess the magnitude of non-cancer effects from air toxics, but one survey study has shown that some pollutants are present in the atmosphere at reference levels that have caused adverse effects in animals.[45]

Emissions of air toxics can also cause adverse health effects via non-inhalation exposure routes. Persistent bioaccumulating pollutants, such as mercury and dioxins, can be deposited into water or soil and subsequently taken up by living organisms. The pollutants can biomagnify through the food chain and exist in high concentrations when consumed by humans in foods such as fish or beef. The resulting exposures can cause adverse effects in humans, and can also disrupt ecosystems by affecting top food chain species.

Finally, there are a host of other potential ecological and welfare effects associated with air toxics, for which very little exists in the way of quantitative analysis. Toxic effects of these pollutants have the potential to disrupt both terrestrial and aquatic ecosystems and contribute to adverse welfare effects such as fish consumption advisories in the Great Lakes.[46]

Unfortunately, the effects of air toxics emissions reductions could not be quantified for the present study. Unlike criteria pollutants, there was relatively little monitoring data for available air toxics, and that which exists covered only a handful of pollutants. Emissions inventories were very limited and inconsistent, and air quality modeling has only been done for a few source categories. In addition, the scientific literature on the effects of air toxics was generally much weaker than that available for criteria pollutants.

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[44] U.S. EPA, Cancer Risk from Outdoor Exposure to Air Toxics. EPA-450/1-90-004f. Prepared by EPA/OAR/OAQPS.

[45] U.S. EPA, "Toxic Air Pollutants and Noncancer Risks: Screening Studies," External Review Draft, September, 1990.

[46] U.S. EPA, Office of Air Quality Planning and Standards. "Deposition of Air Pollutants to the Great Waters, First Report to Congress," May 1994. EPA-453/R-93-055.

Limitations in the underlying data and analyses of air toxics led the Project Team to exclude the available quantitative results from the primary analysis of CAA costs and benefits. The estimates of cancer incidence benefits of CAA air toxics control which were developed, but ultimately rejected, are presented in Appendix H. Also found in Appendix H is a list of research needs identified by the Project Team which, if met, would enable at least a partial assessment of air toxics benefits in future section 812 studies.

Table 15. Health and Welfare Effects of Hazardous Air Pollutants.

Effect Category	Quantified Effects	Unquantified Effects	Other Possible Effects
Human Health	Cancer Mortality - nonutility stationary source - mobile source	Cancer Mortality - utility source - area source Noncancer effects - neurological - respiratory - reproductive - hematopoietic - developmental - immunological - organ toxicity	
Human Welfare		Decreased income and recreation opportunities due to fish advisories Odors	Decreased income resulting from decreased physical performance
Ecological		Effects on wildlife Effects on plants Ecosystem effects Loss of biological diversity	Effects on global climate
Other Welfare		Visibility Building Deterioration	Loss of biological diversity

Table 16. Uncertainties Associated with Physical Effects Modeling.

Potential Source of Error	Direction of Potential Bias in Physical Effects Estimate	Significance Relative to Key Uncertainties in Overall Monetary Benefit Estimate
Extrapolation of health effects to populations distant from monitors (or monitored counties in the case of PM).	Probable overestimate.	Probably minor. In addition, this adjustment avoids the underestimation which would result by estimating effects for only those people living near monitors. Potential overestimate may result to the extent air quality in areas distant from monitors is significantly better than in monitored areas. This disparity should be quite minor for regional pollutants, such as ozone and fine particulates.
Choice of CR function (i.e., "across-study" uncertainties)	Unknown.	Significant. The differences in implied physical outcomes estimated by different underlying studies are large.
Uncertainty associated with CR functions derived from each individual study (i.e., "within study" uncertainty)	Unknown.	Probably minor.
Exclusion of potential substitution of ozone-resistant cultivars in agriculture analysis.	Overestimate.	Insignificant, given small relative contribution of quantified agricultural effects to overall quantified benefit estimate.
Exclusion of other agricultural effects (crops, pollutants)	Underestimate.	Unknown, possibly significant.
Exclusion of effects on wetland and aquatic ecosystems and forests.	Underestimate.	Unknown, possibly significant.
No quantification of materials damage	Underestimate	Unknown, possibly significant.

6

Economic Valuation

Estimating the reduced incidence of physical effects represents a valuable measure of health benefits for individual endpoints; however, to compare or aggregate benefits across endpoints, the benefits must be monetized. Assigning a monetary value to avoided incidences of an individual effect permits a summation, in terms of dollars, of monetized benefits realized as a result of the CAA, and allows that summation to be compared to the cost of the CAA.

For the present analysis of health and welfare benefits, valuation estimates were obtained from the economic literature, and are reported in dollars per case reduced for health effects and dollars per unit of avoided damage for welfare effects.[47] Similar to estimates of physical effects provided by health studies, each of the monetary values of benefits applied in this analysis is reported in terms of mean value and a probability distribution around the mean estimate. The statistical form of the probability distribution used for the valuation measures varies by endpoint. For example, while the estimate of the dollar value of an avoided premature mortality is described by the Weibull distribution, the estimate for the value of a reduced case of acute bronchitis is assumed to be uniformly distributed between a minimum and maximum value.

Willingness-to-Pay Estimates

In benefit-cost analysis, the dollar value of an environmental effect on a person is the dollar amount that is needed to compensate the person to make him or her just as well off as he or she would have been absent the effect. Theoretically, the dollar amount required to compensate a person for exposure to an adverse effect should be about the same as the dollar amount a person is willing to pay to avoid the effect. Thus, economists speak of "willingness-to-pay" (WTP) as the appropriate measure of the value of avoiding an adverse effect. For example, the value of an avoided respiratory symptom would be a person's WTP to avoid that symptom.[48]

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[47] The literature reviews and valuation estimate development process is described in detail in Appendix I and in the referenced supporting reports.

[48] The "compensation principle" is used implicitly for cost calculations as well. In the case of the CAA, compliance cost represents the compensation required to elicit voluntary provision of labor and capital services, supplies, etc., to meet CAA requirements.

For most goods, WTP can be observed by examining actual market transactions. For example, if a gallon of bottled drinking water sells for one dollar, it can be observed that at least some persons are willing to pay one dollar for such water. For goods that are not exchanged in the market, such as most environmental "goods," valuation is not so straightforward. Nevertheless, value may be inferred from observed behavior, such as through estimation of the WTP for mortality risk reductions based on observed sales and prices of safety devices such as smoke detectors. Alternatively, surveys may be used in an attempt to elicit directly WTP for an environmental improvement.

Wherever possible, this analysis uses estimates of the WTP of the U.S. population to avoid an environmental effect as the value of avoiding that effect. In some cases, such estimates are not available, and the cost of mitigating or avoiding the effect is used as a rough estimate of the value of avoiding the effect. For example, if an effect results in hospitalization, the avoided medical costs were considered as a possible estimate of the value of avoiding the effect. Finally, where even the "avoided cost" estimate is not available, the analysis relies on other available methods to provide a rough approximation of WTP. As noted above, this analysis uses a range of values for most environmental effects, or endpoints. Table 17 provides a summary of the mean unit value estimates used in the analysis. The full range of values can be found in Appendix I.

Table 17. Health and Welfare Effects Unit Valuation (1990 dollars).

Endpoint	Pollutant	Valuation (mean est.)	
Mortality	PM_{10} & Pb	$4,800,000	per case
Chronic Bronchitis	PM_{10}	$587,500	per case
IQ Changes			
Lost IQ Points	Pb	$3,400	per case
IQ < 70	Pb	$41,800	per case
Hypertension	Pb	$681	per case
Hospital Admissions			
Coronary Heart Disease	Pb	$10,300	per case
Atherothrombotic brain infarction	Pb	$10,100	per case
Initial cerebrovascular accident	Pb	$7,600	per case
Ischemic Heart Disease	PM_{10}	$10,300	per case
Congestive Heart Failure	PM_{10}	$8,300	per case
Respiratory	PM_{10} & O_3	$7,400	per case
Respiratory Illness and Symptoms			
Acute Bronchitis	PM_{10}	$45	per case
Asthma Attacks	PM_{10} & O_3	$32	per case
Acute Respiratory Symptoms	PM_{10}, O_3, NO_2, SO_2	$24	per case
Upper Respiratory Illness	PM_{10}	$19	per case
Lower Respiratory Illness	PM_{10}	$12	per case
Shortness of Breath	PM_{10}	$5	per day
Work Loss Days	PM_{10}	$83	per day
Restricted Activity Days	PM_{10} & O_3	$38	per day
Welfare Benefits			
Visibility	deciview	$149	per % visual
	East	$117	range chge.
	West		
Household Soiling	PM_{10}	$3	per household per PM_{10} change
Decreased Worker Productivity	O_3	$73	per day
Agriculture (Net Surplus)	O_3	Est. Change in Econ. Surplus	

Mortality

Some forms of air pollution increase the probability that individuals will die prematurely. The concentration-response functions for mortality used in this analysis express this increase in mortality risk as cases of "excess premature mortality" per time period (e.g., per year). It is not possible to compensate the victims; thus, it is not possible to "value" the lives of individual victims in a benefit-cost sense. One can, however, determine the WTP of individuals to accept

relatively small reductions in mortality risk (conversely, one can determine the compensation required for individuals to accept small increases in mortality risk). Since air pollution affects mortality rates by changing the mortality probability (per time period) for many individuals, it is standard practice in cost-benefit analyses to value this effect of air pollution by examining the WTP for small reductions in mortality risk. For expository purposes, this valuation is expressed as "dollars per mortality avoided," even though the actual valuation is based on small changes in mortality risk.[49]

The mortality risk valuation estimate used in this study is based on an analysis of 26 policy-relevant value-of-life studies (see Table 19). Five of the 26 studies are contingent valuation (CV) studies, which directly solicit WTP information from subjects; the rest are wage-risk studies, which base WTP estimates on estimates of the additional compensation demanded in the labor market for riskier jobs. The Project Team used the best estimate from each of the 26 studies to construct a distribution of mortality risk valuation estimates for the Section 812 study. A Weibull distribution, with a mean of $4.8 million and standard deviation of $3.24 million, provided the best fit to the 26 estimates. There is considerable uncertainty associated with this approach, however, which is discussed in detail later in this chapter.

Table 18. Summary of Mortality Valuation Estimates (millions of $1990)

Study	Type of Estimate	Valuation (millions 1990$)
Kneisner and Leeth (1991) (US)	Labor Market	0.6
Smith and Gilbert (1984)	Labor Market	0.7
Dillingham (1985)	Labor Market	0.9
Butler (1983)	Labor Market	1.1
Miller and Guria (1991)	Cont. Value	1.2
Moore and Viscusi (1988a)	Labor Market	2.5
Viscusi, Magat, and Huber (1991b)	Cont. Value	2.7
Gegax et al. (1985)	Cont. Value	3.3
Marin and Psacharopoulos (1982)	Labor Market	2.8
Kneisner and Leeth (1991) (Australia)	Labor Market	3.3
Gerking, de Haan, and Schulze (1988)	Cont. Value	3.4
Cousineau, Lacroix, and Girard (1988)	Labor Market	3.6
Jones-Lee (1989)	Cont. Value	3.8
Dillingham (1985)	Labor Market	3.9
Viscusi (1978, 1979)	Labor Market	4.1
R.S. Smith (1976)	Labor Market	4.6
V.K. Smith (1976)	Labor Market	4.7
Olson (1981)	Labor Market	5.2
Viscusi (1981)	Labor Market	6.5
R.S. Smith (1974)	Labor Market	7.2
Moore and Viscusi (1988a)	Labor Market	7.3
Kneisner and Leeth (1991) (Japan)	Labor Market	7.6
Herzog and Schlottman (1987)	Labor Market	9.1
Leigh and Folson (1984)	Labor Market	9.7
Leigh (1987)	Labor Market	10.4
Gaten (1988)	Labor Market	13.5

SOURCE: Viscusi, 1992

Survey-Based Values

Willingness-to pay for environmental improvement is often elicited through survey methods (such as the "contingent valuation" method). Use of such methods in this context is controversial within the economics profession. In general, economists prefer to infer WTP from observed behavior. There are times when such inferences are impossible, however, and some type of survey technique may be the only means of eliciting WTP. Economists' beliefs regarding the reliability of such survey-based data cover a broad spectrum, from unqualified acceptances of the results of properly-conducted surveys to outright rejections of all survey-based valuations.

[49] For example, if the WTP for a one percent reduction in mortality risk is $50,000, then the "benefit per life saved" would be [$50,000 x 100 =] $5 million.

In this analysis, unit valuations which rely exclusively on the contingent valuation method are chronic bronchitis, respiratory-related ailments, minor restricted activity days, and visibility. As indicated above, the value derived for excess premature mortality stems from 26 studies, of which five use the contingent valuation method. These five studies are within the range of the remaining 21 labor market studies. All five report mortality valuations lower than the central estimate used in this analysis. Excluding the contingent valuation studies from the mortality valuation estimate would yield a central estimate approximately ten percent higher than the 4.8 million dollar value reported above. The endpoints with unit valuations based exclusively on contingent valuation account for approximately 30 percent of the present value of total monetized benefits. Most of the CV-based benefits are attributable to avoided cases of chronic bronchitis.

Chronic Bronchitis

The WTP estimate used for avoided chronic bronchitis is based on studies that elicited from respondents a willingness to pay for chronic bronchitis in terms of a willingness to trade the risk of chronic bronchitis for the risk of a fatal automobile accident (a risk-risk tradeoff).[50] The valuation of a change in the risk of a fatal automobile accident is then applied to infer a valuation for a case of chronic bronchitis avoided. Using this method, four unit values have been suggested:[51] (1) a mean value of $883,000; (2) a median value of $457,000 which captures the skewness of the response distribution; (3) $210,000, based on the mean value, with an adjustment for the severity of the chronic bronchitis case; and (4) a value of $800,000 derived from the mean risk-risk response, but adjusting for the skewness of automobile mortality valuation by using the *median* value for automobile mortality. For this analysis, the central estimate of the value of avoiding a case of chronic bronchitis is taken to be the mean of the four suggested estimates, which is $587,500.

The uncertainty surrounding this estimate is also derived from the range of estimates summarized above. It is assumed that individuals are willing to pay one of the four valuation estimates to avoid chronic bronchitis. Without further information, it is not possible to weight the proportion of individuals that would select each alternative value. Therefore, four discrete values, each with equal probability of selection, describe the uncertainty distribution associated with valuing this endpoint.

Respiratory-Related Ailments

In general, the valuations assigned to the respiratory-related ailments listed in Table 17 represent a combination of willingness to pay estimates for individual symptoms which comprise each ailment. For example, a willingness to pay estimate to avoid the combination of specific upper respiratory symptoms defined in the concentration-response relationship measured by Pope et al. (1991) is not available. However, while that study defined upper respiratory symptoms as one suite of ailments (runny or stuffy nose; wet cough; and burning, aching, or red eyes), the valuation literature reported individual WTP estimates for three closely matching symptoms (head/sinus congestion, cough, and eye irritation). The available WTP estimates were therefore used as a surrogate to the values for the precise symptoms defined in the concentration-response study.

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[50] Viscusi et al. (1991) and Krupnick and Cropper (1992).

[51] *Review of Clean Air Act Section 812 Retrospective Study of Costs and Benefits*, Report of the Advisory Council on Clean Air Compliance Analysis (EPA-SAB-ACCACA-96-003), June 1996.

To capture the uncertainty associated with the valuation of respiratory-related ailments, this analysis incorporated a range of values reflecting the fact that an ailment, as defined in the concentration-response relationship, could be comprised of just one symptom or several. At the high end of the range, the valuation represents an aggregate of WTP estimates for several individual symptoms. The low end represents the value of avoiding a single mild symptom.

Minor Restricted Activity Days

An individual suffering from a single severe or a combination of pollution-related symptoms may experience a Minor Restricted Activity Day (MRAD). Krupnick and Kopp (1988) argue that mild symptoms will not be sufficient to result in a MRAD, so that WTP to avoid a MRAD should exceed WTP to avoid any single mild symptom. On the other hand, WTP to avoid a MRAD should not exceed the WTP to avoid a work loss day (which results when the individual experiences more severe symptoms). No studies are reported to have estimated WTP to avoid a day of minor restricted activity. Instead, this analysis uses an estimate derived from WTP estimates for avoiding combinations of symptoms which may result in a day of minor restricted activity ($38 per day). The uncertainty range associated with this value extends from the highest value for a single symptom to the value for a work loss day. Furthermore, the distribution acknowledges that the actual value is likely to be closer to the central estimate than either extreme.

Visibility

The value of avoided visibility impairment is derived from various studies of the WTP to improve visibility. These studies estimated WTP values in different cities in the U.S. The values used to monetize the measured reductions in visibility were found to vary between the Eastern and Western United States ($149 and $117 per percent change in visual range, respectively).

Avoided Cost Estimates

For some health effects, WTP estimates are not available, and the Project Team instead used "costs avoided" as a substitute for WTP. Avoided costs were used to value the following endpoints: hypertension, hospital admissions, and household soiling.

Hypertension and Hospital Admissions

Avoided medical costs and lost work time were used to value hypertension (high blood pressure) and hospital admissions (this includes hospital admissions for respiratory ailments as well as heart disease, heart attacks, and strokes) . This analysis considers medical costs to be the sum of physician charges, medication costs, and hospitalization costs. Lost work time is valued at the rate estimated for a work loss day (see discussion below). Based on this approach, the value per year of avoiding a case of hypertension is taken to equal the sum of medical costs per year plus work loss costs per year; the resulting value is $681 per case per year. For health effects which lead to hospitalization, the analysis uses the sum of medical costs (per case) and work loss costs ($6,131 to $10,308 depending on the ailment for which hospitalization is required).

Presumably, willingness-to-pay to avoid the effects (and treatment) of hypertension would reflect the value of pain and suffering, and the value placed on dietary changes, etc. Likewise, the value of a health

effect that would require hospitalization or doctor's care would include pain and suffering and lost leisure time, in addition to medical costs and lost work time. Consequently, the valuations for these endpoints used in this analysis likely represent lower-bound estimates of the true social values for avoiding such health effects.

Household Soiling

This analysis values benefits for this welfare effect by considering the avoided costs of cleaning houses due to particulate matter soiling. The Project Team's estimate reflects the household's annual cost of cleaning per $\mu g/m^3$ particulate matter ($2.52). Considered in this valuation are issues such as the nature of the particulate matter, and the proportion of households likely to do the cleaning themselves. Since the avoided costs of cleaning used herein do not reflect the loss of leisure time (and perhaps work time) incurred by those who do their own cleaning, the valuation function likely underestimates true WTP to avoid additional soiling.

Other Valuation Estimates

Changes in Children's IQ

One of the major effects of lead exposure is permanently impaired cognitive development in children. No ready estimates of society's WTP for improved cognitive ability are currently available. Two effects of IQ decrements can be monetized, however: reductions in expected lifetime income, and increases in societal expenditures for compensatory education. These two effects almost certainly understate the WTP to avoid impaired cognitive development in children, and probably should be considered lower bound estimates. In the absence of better estimates, however, the Project Team has assumed that the two monetized effects represent a useful approximation of WTP.

The effect of IQ on expected lifetime income comprises a direct and an indirect effect. The direct effect is drawn from studies that estimate, all else being equal, the effect of IQ on income. The indirect effect is through education: there is a correlation between IQ and years of education, and between education and expected future income. The two effects together imply a change in expected lifetime income (discounted to the present at 5 percent) of $3,400 per IQ point change. The calculated effect overstates net income change because it does not account for the cost of incremental education. If one were to include expected incremental education cost in the calculation, the indirect effect would be lessened somewhat. Given that the effect of IQ on lifetime earnings is almost certainly an underestimate of the WTP to avoid impaired cognitive development in children, the Project Team did not reduce the estimate to account for expected incremental education cost.

In this analysis, it is assumed that part-time compensatory education is required for all children with IQ less than 70. The Project Team assumed that the WTP to avoid cases of children with IQ less than 70 can be approximated by the cost ($41,800 per child) of part-time special education in regular classrooms from grades one through twelve (as opposed to independent special education programs), discounted to the present at 5 percent. See Appendix G for more detail on valuation methods and data sources for IQ effects and other lead-related health impacts.

Work Loss Days and Worker Productivity

For this analysis, it was assumed that the median daily 1990 wage income of 83 dollars was a reasonable approximation of WTP to avoid a day of lost work. Although a work loss day may or may not affect the income of the worker, depending on the terms of employment, it does affect economic output and is thus a cost to society. Conversely, avoiding the work loss day is a benefit.

A decline in worker productivity has been measured in outdoor workers exposed to ozone. Reduced productivity is measured in terms of the total daily income of the average worker engaged in strenuous outdoor labor, estimated at 73 dollars per day (U.S. Bureau of the Census, 1990).

Agricultural Benefits

Similar to the other welfare effects, the agricultural benefits analysis estimated benefits in dollars per unit of avoided damage, based on estimated changes in crop yields predicted by an agricultural sector model. This model incorporated agricultural price, farm policy, and other data for each year. Based on expected yields, the model estimated the production levels for each crop, and the economic benefits to consumers, and to producers, associated with these production levels. To the extent that alternative exposure-response relationships were available, a range of potential benefits was calculated.

Valuation Uncertainties

The Project Team attempted to handle most valuation uncertainties explicitly and quantitatively by expressing values as distributions (see Appendix I for a complete description of distributions employed), using a Monte-Carlo simulation technique to apply the valuations to physical effects (see Chapter 7) with the mean of each valuation distribution equal to the "best estimate" valuation. This approach does not, of course, guarantee that all uncertainties have been adequately characterized, nor that the valuation estimates are unbiased. It is entirely possible that the actual WTP to avoid an air pollution-related impact is outside of the range of estimates used in this analysis. Nevertheless, the Project Team believes that the distributions employed are reasonable approximations of the ranges of uncertainty, and that there is no compelling reason to believe that the mean values emloyed are systematically biased (except for the IQ-related and avoided cost-based values, both of which probably underestimate WTP).

One particularly important area of uncertainty is valuation of mortality risk reduction. As noted below (see Chapter 7), changes in mortality risk are a very important component of aggregate benefits, and mortality risk valuation is an extremely large component of the quantified uncertainty. Consequently, any uncertainty concerning mortality risk valuation beyond that addressed by the quantitative uncertainty assessment (i.e., that related to the Weibull distribution with a mean value of $4.8 million) deserves note. One issue merits special attention: uncertainties and possible biases related to the "benefits transfer" from the 26 valuation source studies to valuation of reductions in PM-related mortality rates.

Mortality Risk Benefits Transfer

Although each of the mortality risk valuation source studies (see Table 19) estimated the average WTP for a given reduction in mortality risk, the degree of reduction in risk being valued varied across studies and is not necessarily the same as the degree of mortality risk reduction estimated in this analysis. The

transferability of estimates of the value of a statistical life from the 26 studies to the Section 812 benefit analysis rests on the assumption that, within a reasonable range, WTP for reductions in mortality risk is linear in risk reduction. For example, suppose a study estimates that the average WTP for a reduction in mortality risk of 1/100,000 is 50 dollars, but that the actual mortality risk reduction resulting from a given pollutant reduction is 1/10,000. If WTP for reductions in mortality risk is linear in risk reduction, then a WTP of 50 dollars for a reduction of 1/100,000 implies a WTP of 500 dollars for a risk reduction of 1/10,000 (which is ten times the risk reduction valued in the study). Under the assumption of linearity, the estimate of the value of a statistical life does not depend on the particular amount of risk reduction being valued.

Although the particular amount of mortality risk reduction being valued in a study may not affect the transferability of the WTP estimate from the study to the benefit analysis, the characteristics of the study subjects and the nature of the mortality risk being valued in the study could be important. Certain characteristics of both the population affected and the mortality risk facing that population are believed to affect the average WTP to reduce risk. The appropriateness of the mean of the WTP estimates from the 26 studies for valuing the mortality-related benefits of reductions in pollutant concentrations therefore depends not only on the quality of the studies (i.e., how well they measure what they are trying to measure), but also on (1) the extent to which the subjects in the studies are similar to the population affected by changes in air pollution and (2) the extent to which the risks being valued are similar.

The substantial majority of the 26 studies relied upon are wage-risk (or labor market) studies. Compared with the subjects in these wage-risk studies, the population most affected by air pollution-related mortality risk changes is likely to be, on average, older and probably more risk averse. Some evidence suggests that approximately 85 percent of those identified in short-term ("episodic") studies who die prematurely from PM-related causes are over 65.[52] The average age of subjects in wage-risk studies, in contrast, would be well under 65.

The direction of bias resulting from the age difference is unclear. It could be argued that, because an older person has fewer expected years left to lose, his or her WTP to reduce mortality risk would be less than that of a younger person. This hypothesis is supported by one empirical study, Jones-Lee et al. (1985), which found WTP to avoid mortality risk at age 65 to be about 90 percent of what it is at age 40. On the other hand, there is reason to believe that those over 65 are, in general, more risk averse than the general population, while workers in wage-risk studies are likely to be less risk averse than the general population. Although the list of 26 studies used here excludes studies that consider only much-higher-than-average occupational risks, there is nevertheless likely to be some selection bias in the remaining studies – that is, these studies are likely to be based on samples of workers who are, on average, more risk-loving than the general population. In contrast, older people as a group exhibit more risk averse behavior.

There is substantial evidence that the income elasticity of WTP for health risk reductions is positive (although there is uncertainty about the exact value of this elasticity). Individuals with higher incomes (or greater wealth) should, then, be willing to pay more to reduce risk, all else equal, than individuals with lower incomes or wealth. One could therefore argue that because the elderly are relatively wealthy, the affected population is also wealthier, on average, than are the wage-risk study subjects, who tend to be middle-aged (on average) blue-collar workers. It is possible, however, that among the elderly it is largely the poor elderly who are most vulnerable to air pollution-related mortality risk (e.g., because of generally

[52] See Schwartz and Dockery (1992), Ostro et al. (1995), and Chestnut (1995).

poorer health care). If this is the case, the average wealth of those affected by a reduction in pollutant concentrations relative to that of subjects in wage-risk studies is uncertain.

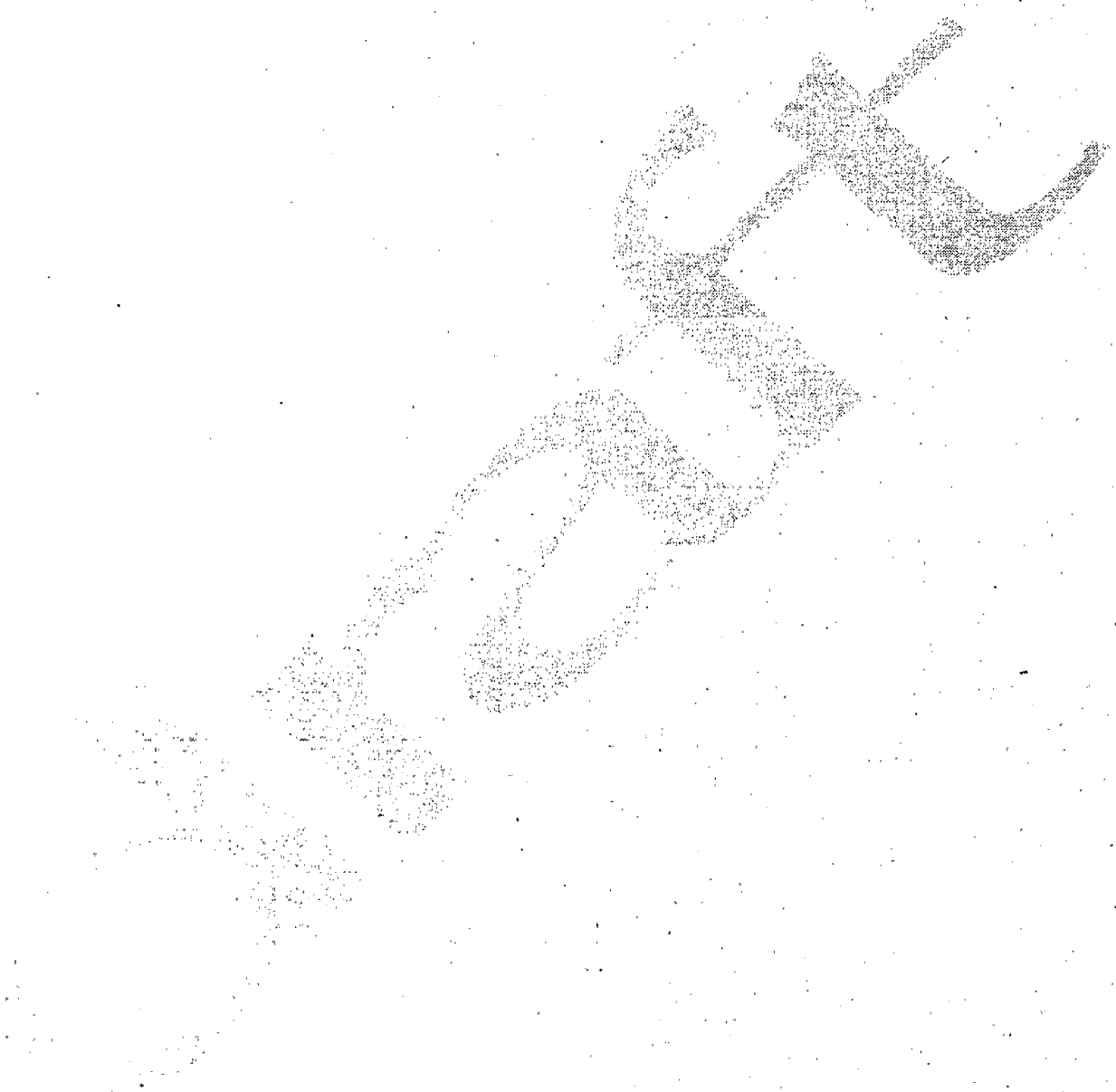
Although there may be several ways in which job-related mortality risks differ from air pollution-related mortality risks, the most important difference may be that job-related risks are incurred voluntarily whereas air pollution-related risks are incurred involuntarily. There is some evidence (see, for example, Violette and Chestnut, 1983) that people will pay more to reduce involuntarily incurred risks than risks incurred voluntarily. If this is the case, WTP estimates based on wage-risk studies may be downward biased estimates of WTP to reduce involuntarily incurred air pollution-related mortality risks.

Finally, another important difference related to the nature of the risk may be that workplace mortality risks tend to involve sudden, catastrophic events, whereas air pollution-related risks tend to involve longer periods of disease and suffering prior to death. Some evidence suggests that WTP to avoid a risk of a protracted death involving prolonged suffering and loss of dignity and personal control is greater than the WTP to avoid a risk (of identical magnitude) of sudden death. To the extent that the mortality risks addressed in this assessment are associated with longer periods of illness or greater pain and suffering than are the risks addressed in the valuation literature, the WTP measurements employed in the present analysis would reflect a downward bias.

The potential sources of bias in an estimate of WTP to reduce the risk of air pollution-related mortality based on wage-risk studies are summarized in Table 19. The need to adjust wage-risk-based WTP estimates downward because of the likely upward bias introduced by the age discrepancy has received significant attention (Chestnut, 1995; IEc, 1992). If the age difference were the only difference between the population affected by air quality changes and the subjects in the wage-risk studies, there might be some justification for trying to adjust the valuation for an excess premature mortality downward. Even in this case, however, the degree of the adjustment would be unclear. There is good reason to suspect, however, that there are biases in both directions, as shown in the table above. Because in each case the extent of the bias is unknown, the overall direction of bias in the valuation estimate is similarly unknown.

Table 19. Estimating Mortality Risk Based on Wage-Risk Studies: Potential Sources and Likely Direction of Bias.

Factor	Likely Direction of Bias in WtP Estimate
Age	Uncertain, perhaps upward
Degree of Risk Aversion	Downward
Income	Downward, if the elderly affected are a random sample of the elderly; Unclear, if the elderly affected are poor.
Risk Perception: Voluntary vs. Involuntary	Downward
Catastrophic vs Protracted Death	Uncertain, perhaps downward



7

Aggregate Results and Uncertainty

This chapter presents a summary of the monetary benefits of the CAA from 1970 to 1990, compares these with the corresponding costs, and explores some of the major sources of uncertainty in the benefits estimates. Monetized economic benefits for the 1970 to 1990 period were derived by applying the unit valuations discussed in Chapter 6 to the stream of physical effects estimated by the method documented in Chapter 5. One goal was to retain the uncertainty associated with the health and welfare effects estimates and the uncertainty associated with the unit valuations.

Two sources of variability describe the uncertainty in the concentration-response (CR) relationships. The first is the statistical uncertainty associated with the CR relationships reported in the health and welfare effects literature. The studies quantifying relationships between air pollution levels and an adverse health or welfare effect typically report a statistical distribution along with a mean value of the concentration-response estimate. The distribution captures the inherent uncertainty in the reported estimate. The second source of uncertainty lies in the choice of studies. Different published results reported in the scientific literature typically do not report identical findings; in some instances the differences are substantial. These differences reflect variability of the concentration-response relationship across studies and are captured in the overall uncertainty by the method to combine physical effects with unit valuations below. Finally, the uncertainty in the unit valuations is captured by the range of values derived for each quantified endpoint. For example, the Weibull distribution surrounding the $4.8 million mean estimate of the value of an excess premature mortality reflects the uncertainty associated with this health effect.

The Project team adopted a Monte Carlo approach to evaluate the combined uncertainties of the quantified health and welfare effects estimates and the unit values associated with each effect. This approach permitted the uncertainty associated with each component of the analysis to be retained through the process of monetizing and aggregating benefits. The first step in the Monte Carlo technique was to select randomly a study from the set of available studies for a particular health effect. Each study provides an estimate of the CR function and its standard deviation. In the second step, a normal statistical distribution was derived based on the mean and standard deviation of the coefficient associated with the selected study. A specific value for the coefficient was then randomly selected from the derived normal distribution. This two-step procedure was repeated to produce a large number of estimates of the concentration-response relationship for the given endpoint. Because the value of the coefficient varied from iteration to iteration due to the two random selection procedures described above, the predicted health effects varied as well. The Monte Carlo method thus produced a distribution of predicted health effect outcomes due to the change in air quality between the control and no-control scenarios.

Concurrently, the Monte Carlo technique was used to estimate the range of monetary benefits associated with the avoided health and welfare effects. The monetary values assigned to each health or welfare endpoint were also measured with considerable uncertainty, which was represented by the distribution of monetary values derived for each endpoint. Each iteration of the Monte Carlo assessment picked a specific monetary value of the endpoint from the assigned distribution of values. The random draws of the health or welfare effect incidence and the monetary value of that effect were multiplied to yield an estimate of monetary benefits for changes in the given effect for that Monte Carlo iteration. Repeating the process many times generated a distribution of estimated monetary benefits by endpoint. Combining the results for the individual endpoints using the Monte Carlo procedure yielded a range of total estimated monetary benefits for each target year (1975, 1980, 1985 and 1990). This technique enabled a representation of uncertainty in current scientific and economic opinion in these benefits estimates.

Aggregate Monetary Benefits

Table 20 presents monetary benefits for each quantified and monetized health and welfare endpoint (or group of endpoints), aggregated from 1970 to 1990. The mean estimate resulting from the Monte Carlo simulation is presented, along with the measured credible range (upper and lower fifth percentiles of the distribution). Aggregating the stream of monetary benefits across years involved compounding the stream of monetary benefits estimated for each year to the 1990 present value (using a 5 percent discount rate).

Since Monte Carlo modeling was accomplished only for the four target years, estimates for intervening years were interpolated based on the trend of benefits estimated by combining only the central estimates of the concentration-response relationship for each endpoint with the central estimate of the unit valuation. While the monetary benefits estimated using these "point" estimate values did not attempt to capture uncertainty, they did provide an approximation of the temporal trend of expected benefits associated with the year by year changes in air quality. The mean values and the credible range for each endpoint in the intervening years were calculated based on the ratios of the 5th percentile, mean, and 95th percentiles to the point estimate in the target years. These ratios (which were fairly consistent) were linearly interpolated and applied to the point estimate of benefits in the intervening years. For example, point estimates of monetary benefits were generated for 1976 through 1979 based on interpolation of the air quality estimates for those years. The Monte Carlo results were used to calculate ratios of the 5th percentile, mean, and 95th percentile to the point estimate for the target years 1975 and 1980. Linearly interpolating these ratios and applying them to the point estimates of benefits in the intervening years, yielded a stream of benefits for all years described by the mean values and credible ranges. Finally, the monetary benefits for each year were adjusted to their equivalent 1990 value and summed.

Table 20. Present Value of 1970 to 1990 Monetized Benefits by Endpoint Category for 48 State Population and Total Monetized Benefits (billions of $1990, discounted to 1990 at 5 percent)

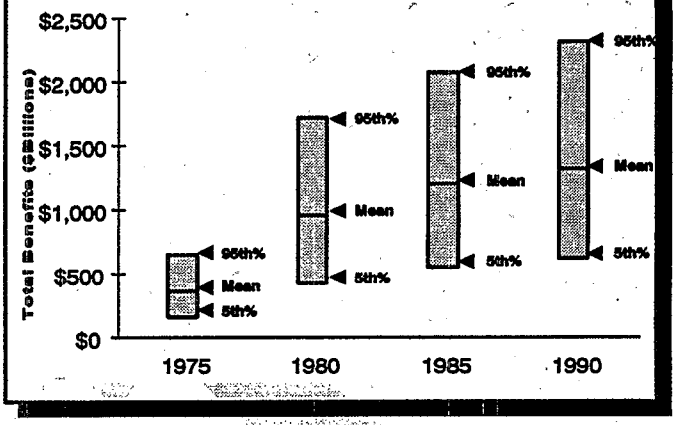
Endpoint	Pollutant(s)	Present Value		
		5th %ile	Mean	95th %ile
Mortality	PM-10	$1,991	$13,542	$30,968
Mortality	Pb	$125	$1,550	$4,096
Chronic Bronchitis	PM-10	$2,143	$7,156	$12,613
IQ (Lost IQ Pts. + Children w/ IQ<70)	Pb	$299	$466	$656
Hypertension	Pb	$77	$99	$120
Hospital Admissions	PM-10, O3, Pb, & CO	$18	$24	$33
Respiratory-Related Symptoms, Restricted Activity, & Decreased Productivity	PM-10, O3, NO2, SO2	$31	$76	$149
Soiling Damage	PM-10	$7	$76	$196
Visibility	particulates	$55	$72	$91
Agriculture (Net Surplus)	O3	$11	$23	$35
TOTAL (Billions of 1990-value dollars)		$10,500	$23,000	$40,600

Table 20 offers a comparison of benefits by endpoint. When aggregating benefits across endpoints care must be taken to avoid double-counting. In this analysis, double-counting was avoided by assuming that any endpoints which measured overlapping effects were treated as alternatives (i.e., they were not treated as additive). To generate benefit estimates for the endpoint categories shown, the Project Team had to combine certain endpoints to avoid double-counting. In particular, the mortality estimate associated with PM_{10} represents a pooling of the estimates derived from the short-term time series studies and the long-term cohort study, applying equal weights to each study *type*. The potential for double-counting also arises in the Hospital Admissions endpoint. This endpoint includes hospital admissions for all respiratory ailments, which is treated as an alternative to the benefits estimated for COPD and Pneumonia admissions combined. Likewise, since their definitions of symptoms overlap, acute bronchitis is treated as an alternative to the combination of upper and lower respiratory symptoms. All these effects are combined into the Respiratory-Related Symptoms, Restricted Activity, & Decreased Productivity endpoint category in Table 115. Finally, the potential for double-counting of the effects of restricted activity was addressed by combining benefits estimates of "restricted activity days" of various degrees of severity into a single benefit category.

Table 20 also reports the estimated total national monetized benefits attributed in this analysis to the CAA from 1970 to 1990. The benefits, valued in 1990 dollars, range from $10.5 to $40.6 trillion with a central estimate of $23 trillion. This analysis depended on the Monte Carlo technique to aggregate monetary benefits across endpoints. For each of several thousand iterations, a random draw of the monetized benefits for each endpoint was selected from the distributions summarized in Table 22 and the individual endpoint estimates were then summed. This resulted in the distribution of total national monetized benefits reported above.

The temporal pattern of benefits during the 1970 to 1990 period is related to the difference in emissions between the control and no-control scenarios, and magnified by population growth during that period. As illustrated by Figure 21, quantified annual benefits increased steadily during the study period, with the greatest increases occurring during the late 1970s. The mean estimate of quantified annual benefits grew from 364 billion dollars (expressed as inflation-adjusted 1990 dollars) to 957 billion dollars in 1980, 1,204 billion dollars in 1985, and 1,318 billion dollars in 1990.

Figure 21. Monte Carlo Simulation Model Results for Target Years (in billions of 1990 dollars)



Benefits and Costs

Table 21 presents summary quantitative results for the retrospective assessment. Annual results are presented for four individual years, with all dollar figures expressed as inflation-adjusted, 1990 dollars. The final column sums the stream of costs and benefits from 1970 to 1990, discounted (i.e., compounded) to 1990 at 5 percent. "Monetized benefits" are the mean estimates reported by the Monte Carlo benefits model. "Net Benefits" are monetized benefits less annualized costs for each year.

Table 21. Monetized Annual Benefits and Costs, 1970-1990 (in billions of 1990-value dollars).

	1975	1980	1985	1990	PV
Monetized Benefits	364	957	1,204	1,318	23,000
Annualized Costs (5%)	14	21	25	26	523
Net Benefits	350	936	1,179	1,292	22,477
Benefit/Cost ratio	26/1	46/1	48/1	51/1	44/1

Notes: PV=1990 present value reflecting compounding of costs and benefits from 1971 to 1990 at 5 percent.

Several measures of "cost" are available for use in this analysis (see Chapter 2). The Project Team employs "annualized cost" because it measures cost in a fashion most analogous to the benefits estimation method. An alternative measure, "compliance expenditure," is a reasonable cost measure. Some capital expenditures, however, generate a benefit stream beyond the period of the analysis (i.e., beyond 1990). Those post-1990 benefits are not, in general, included in the benefit estimates presented above. The annualization procedure reduces the bias introduced by the use of capital expenditures by spreading the cost of the capital investment over its expected life, then counting as a "cost" only those costs incurred in the 1970 to 1990 period.

The macroeconomic analysis employed for this analysis (see Chapter 2) indicates that compliance expenditures induce significant second-order effects, and it can be argued that those effects should be included in a comprehensive cost analysis. Benefits resulting from compliance expenditures should also induce second-order macroeconomic effects (which would, one would expect, partly or completely offset the estimated second-order adverse effects induced by compliance expenditures). Due to the sequencing of the analytical steps in this assessment, it was not practical to estimate the second-order cost and benefit impacts induced by the estimated health and welfare benefits. Because second-order impacts of benefits are

not estimated, the Project Team refrained from choosing as the primary cost measure one that included second-order impacts, and instead employed "annualized costs."

Quantified Uncertainty and Sensitivities

Some degree of uncertainty is associated with every stage of the section 812 analysis, from data inputs for compliance cost estimation to economic valuation of physical effects. For some factors, both the degree of uncertainty and the direction of any potential bias are unknown; for some other factors, no employable quantitative estimates are available, even though one would expect a non-zero result (for example, quantitative estimates are unavailable for some human health effects, some human welfare effects, and all ecological effects). Some important quantified factors, nevertheless, lend themselves to quantified uncertainty assessments and/or sensitivity analyses. Some of those factors are addressed in this section.

Quantified Uncertainty in the Benefits Analysis

As previously indicated, a Monte Carlo simulation method was used to aggregate the benefit estimates generated by this assessment. The two primary sources of quantified uncertainty captured by the benefits model are the variability in CR relationship (both in terms of statistical variance reported for each source study, and of differences in CR relationships found in alternative studies), and the distribution of plausible economic valuations[53] of the quantified endpoints. The combination of variability in CR relationships and in valuation estimates (along with changes in other point estimated variables such as population and air quality) resulted in a distribution of benefit estimates for each year of the period of analysis. In this assessment, the benefits distribution is described as the mean value and a ninety percent probability interval around the mean.

Figure 22. Distribution of 1990 Monetized Benefits of CAA (in billions of 1990 dollars)

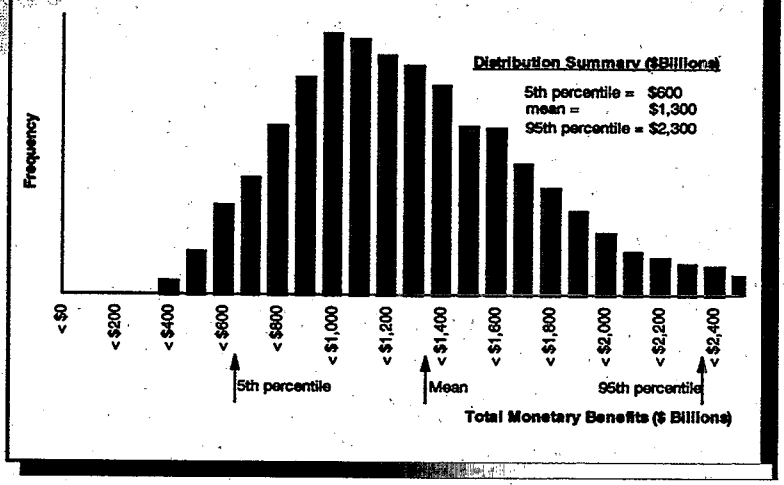


Figure 22 depicts the distribution of monetary benefits for 1990 (similar distributions were generated for other years in the analysis period). The solid vertical bars in the figure represent the relative frequency of a given result in the 1990 Monte Carlo analysis. The largest bar, located above the "<$1,000", indicates that more Monte Carlo iterations generated monetized benefits of $900 billion to $1 trillion than in any other $100 billion range bin. This tallest bar, however, does not represent the expected value of the estimate for total monetized benefit for 1990; rather, the mean value of the entire distribution, calculated to be $1.3 trillion, provides this value. The ninety percent confidence interval, a summary description of the spread of a distribution, is also noted in the figure.

[53] Valuation and uncertainty distributions are described in Appendix I.

Annual benefit distributions were calculated or inferred for the 1971-1990 period. Table 22 summarizes the calculated benefit ranges for 1975, 1980, 1985, and 1990, and notes the benefit/cost ratios implied by the benefit ranges. The fifth percentile present value computation is the sum of the 1990 present values of the annual fifth percentile estimates. The distribution of benefits changes little (except in scale) from year-to-year: The mean estimate is somewhat greater than twice the fifth percentile estimate, and the ninety-fifth percentile estimate is somewhat less than twice the mean estimate. The distribution shape changes little across years because the variables which induce the distribution spread (i.e., CR function variability and economic valuation variability) are unchanged from year-to-year. Some variability is induced by changes in relative pollutant concentrations over time, which then change the relative impact of individual CR functions.

Table 22. Quantified Uncertainty Ranges for Monetized Annual Benefits and Benefit/Cost Ratios, 1970-1990 (in billions of 1990-value dollars).

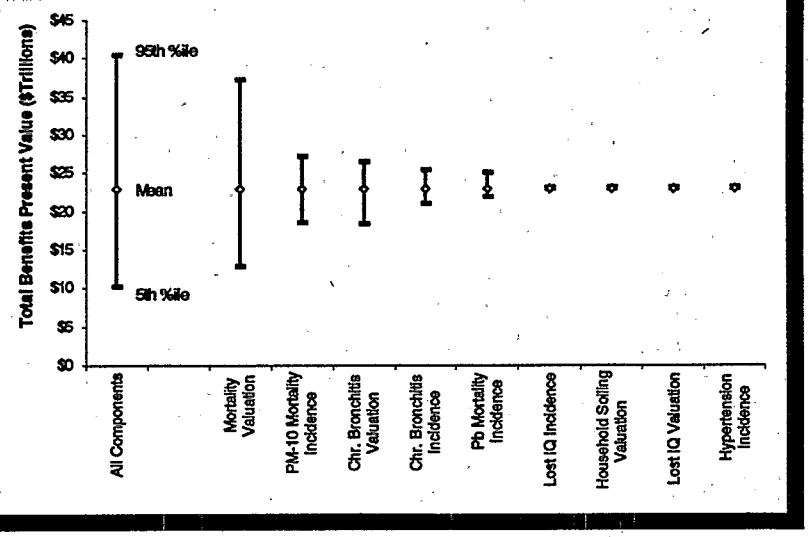
	1975	1980	1985	1990	PV
Monetized Benefits					
5th percentile	161	431	558	623	10,500
Mean estimate	364	957	1,204	1,318	23,000
95th percentile	655	1,710	2,075	2,317	40,600
Annualized Costs (5%)	14	21	25	26	523
Benefit/Cost ratio					
5th percentile	12/1	21/1	22/1	24/1	20/1
Mean estimate	26/1	46/1	48/1	51/1	44/1
95th percentile	47/1	81/1	83/1	89/1	78/1

Notes: PV=1990 present value reflecting compounding of costs and benefits from 1971 to 1990 at 5 percent.

Sources of Quantified Uncertainty

The estimated uncertainty ranges for each endpoint category summarized in Table 20 reflect the measured uncertainty associated with both avoided incidence and economic valuation. The Project Team conducted a sensitivity analysis to determine the variables with the greatest contribution to the quantified uncertainty range. The results of the uncertainty analysis are illustrated in Figure 23.

Figure 23. Uncertainty Ranges Deriving From Individual Uncertainty Factors



In this uncertainty analysis, all the inputs to the Monte Carlo uncertainty analysis are held constant (at their mean values), allowing only one variable -- for example, the economic valuation of mortality -- to vary across the range of that variable's uncertainty. The sensitivity analysis then isolates how this single source of variability contributes to the variation in estimated total benefits. The first uncertainty bar represents the credible range associated with the total monetary benefits of the Clean Air Act, as reported above. This captures the multiple uncertainties in the quantified benefits estimation. The

rest of the uncertainty bars represent the quantified uncertainty ranges generated by single variables. The most important contributor to aggregate quantified uncertainty is mortality valuation and incidence, followed by chronic bronchitis valuation and incidence.

Potential Sensitivities in the Cost-Benefit Analysis

Some uncertainties in the present analysis lend themselves to explicit quantitative analysis. In this section, two analytical variables considered *ex ante* to have potentially significant influences on the outcome of this analysis are evaluated. In addition, the potential for bias due to excluded second-order cost and benefit impacts is discussed.

Discount Rates

Typically, the choice of discount rate has an important effect on the results of a multiyear benefit cost analysis. In this assessment, the discount rate affects only three factors: IQ-related benefits estimates (especially estimates of changes in discounted lifetime income), annualized costs (i.e., amortized capital expenditures), and compounding of all costs and benefits to 1990. Table 23 summarizes the effect of alternative discount rates on the "best estimate" results of this analysis. Because monetized benefits exceed costs for all years in the analysis period, net benefits increase as the discount rate increases. Because the annual B/C ratio increases as one moves from 1970 toward 1990 (see Table 21 above), benefit cost ratios decline as the discount rate increases (because earlier periods are given greater weight). Overall, the cost/benefit results of the assessment appear to be generally insensitive to the choice of discount rate.

Table 23. Effect of Alternative Discount Rates on Present Value of Total Monetized Benefits/Costs for 1970 to 1990 (in trillions of 1990 dollars).

	Discount rate		
	3%	5%	7%
Mean Estimated Benefits	19.9	23.0	26.7
Annualized Costs	0.4	0.5	0.7
Net Benefits	19.5	22.5	26.0
Benefit/Cost ratio	48/1	44/1	41/1

PM-Related Mortality Valuation

The economic benefits of avoided premature mortality associated with reduced concentrations of non-lead criteria pollutants (using PM_{10} as a surrogate measure) represent the largest source of benefits quantified in this assessment, and the PM-mortality incidence and valuation estimates are the largest sources of quantified uncertainty in the Monte Carlo uncertainty analysis. The Project Team conducted two additional analyses to test the sensitivity of the assessment results to changes in the mortality valuation method. Both sensitivity analyses are discussed in greater detail in Appendix I.

Eighty to eighty-five percent of the cases of premature mortality estimated by some "short-term exposure" mortality studies involve persons 65 years of age or older. The age profile of the subjects in the studies used as sources for the mortality valuation estimates is quite different, with an average subject's age of 40 years or so. If WTP to avoid changes in mortality risk is correlated with age, then an adjustment to the unit value used in this assessment would be appropriate (at least for mortality estimates based on the short-term exposure epidemiological studies). Some have suggested that the WTP for reduced mortality risk of persons over the age of 65 is approximately 75 percent of the WTP of the remainder of the population. As discussed in Chapter 6 (and in greater detail in Appendix I), EPA believes that the state of the science has not yet advanced sufficiently to provide broadly convincing evidence in support of age-

specific adjustments to the mortality valuation estimates used as the central case results of this analysis. Nevertheless, for illustrative purposes, the Project Team adjusted the mean mortality estimation using the above method, which results in an adjusted benefits estimate that is approximately 79 percent of the central case estimate (see Table 24).

A second alternative analysis involves changing the approach taken for both the physical effects and valuation estimates. Instead of expressing changes in mortality risk as changes in the expected number of premature mortalities per year, this method would transform changes in mortality risk to changes in expected life-span. Thus, increased air pollution levels would result in "life years lost" (LYL) rather than "excess premature mortalities." Similarly, the estimates of WTP to avoid mortality risk would be converted to estimates of WTP to avoid a lost life-year. However, both the epidemiological science and economic valuation science are inadequately advanced to allow confident employment of the LYL method. Nevertheless, to test the potential magnitude of the effect that the LYL approach might have on assessment results, the Project Team attempted to infer LYL from the long-term PM exposure epidemiological study, and applied a crude estimate of WTP to avoid a LYL (see Appendix I for greater detail on analytical method).

Table 24. Results of Mortality Benefits Sensitivity Analyses, Monetized Benefits for 1970 to 1990 (in trillions of 1990 dollars, discounted at 5 percent).

	Benefits	
Benefit Estimation Method	PM	Tot.
Central analysis method ($4.8M/case)	13.5	23.0
79% adjustment method	10.7	20.2
LYL: single relative risk	12.4	22.9
LYL: age-specific relative risk	9.5	19.0

Note: The first two methods employ averages of the results of long-term exposure and short-term exposure studies for mortality incidences. The LYL method employs only the results of the long-term exposure study.

Table 24 summarizes the results of the two sensitivity analyses. Estimated 1970 to 1990 benefits from PM-related mortality alone and total assessment benefits are reported, in 1990 dollars discounted to 1990 at 5 percent. Two separate LYL results are reported; the difference between the two is the product of alternative assumptions used to infer LYL from the results of the epidemiological study. Note that the LYL results are based on the single long-term PM exposure study used for this assessment, while the central case and "79 percent adjusted" results are based on premature mortality incidence results averaged across the long-term exposure study and the pooled short-term exposure studies.

Second-Order Macroeconomic Impacts

As just mentioned in the section on benefits and costs, the cost measure used for this assessment does not include second-order adverse macroeconomic impacts arising from compliance expenditures, even though the magnitude of those impacts were estimated by the Project Team (see Chapter 2 and Appendix A). It is unclear, however, whether consideration of indirect impacts would increase or decrease overall cost impacts if one were to include second-order impacts arising from estimated benefits. For illustration, Table 25 presents 1970 to 1990 compliance costs and some benefit measures which, one would expect, should induce second-order impacts of similar magnitudes. The direction of possible second-order impacts is unclear.

Several of the health and welfare benefits estimated in this assessment are expressed as "avoided costs." In the aggregate, businesses and households would view these avoided costs (if unavoided) as being essentially identical in effect to compliance expenditures, tax treatment of capital expenditures aside (for example, a business should be financially indifferent between incurring 100 dollars of additional pollution abatement compliance cost versus 100 dollars of additional health insurance cost). In this assessment, benefits estimated as "avoided cost" are about ½ as large as compliance costs over the 1970 to 1990 period. On this evidence alone, one would expect to find that second-order impacts from benefits mitigate, but would not completely offset, second-order impacts arising from compliance expenditures.

Table 25. Compliance Costs, Some Avoided Costs, and Productivity Improvements, 1970-1990 (in billions of $1990).

	PV[a]	%[b]
Annualized Compliance Costs (5%)	523	
Avoided Cost estimates[c]	275	53%
IQ-related effects (productivity)	466	89%
All avoided cost and productivity effects	741	142%

Notes:
a. PV=1990 present value reflecting compounding of costs and benefits from 1971 to 1990 at 5%.
b. %= percentage of annualized cost estimate.
c. Includes hypertension, hospital admissions, respiratory-related symptoms, RADs, decreased worker productivity, and soiling damage (from Table 20). Does not include avoided costs associated with other health endpoints (e.g., chronic bronchitis)

A second category of avoided cost was calculated. Some IQ-related benefits (here estimated as increased expected lifetime income) are avoided societal costs insofar as they represent avoided decreases in future production (here referred to as a "productivity" effect). Here, these benefits are presented separately from the "avoided cost" estimates for two reasons. First, they differ conceptually from the other avoided cost categories, and it is not so clear (when considering impact on second-order effects) that they are analogous to compliance cost. Second, the IQ effects calculations differ from the other benefit calculations, since they are lifetime benefits discounted to the time of exposure. That is, unlike other benefit categories and the annualized cost estimates, some of the estimated IQ-related benefits actually accrue after 1990. Only some of the IQ-related benefits, then, would accrue during the 1970 to 1990 period and would then induce second-order macroeconomic effects which would offset the compliance cost-induced effects. Calculated IQ-related benefits are roughly 90 percent of compliance cost estimates; some portion of the total would offset compliance cost-induced second-order macroeconomic effects. It is unclear whether this result, combined with the avoided cost impact discussed above, would be enough to completely offset the second-order impacts of compliance costs.

Of course, the other benefit categories would also be expected to induce second-order effects (e.g., improved longevity and decreased chronic bronchitis incidence would affect labor supply). Furthermore, although not counted here as "avoided cost," the benefit estimates for all other health effects include *some* amount of "avoided cost." In some cases (chronic bronchitis, for example), one would expect the "avoided cost" component of the benefits estimate to be substantial relative to aggregate compliance cost.

Finally, environmental regulation can, in some instances, spur rather than impair technological innovation. Technological improvements motivated by regulatory requirements, such as more efficient stack scrubbers and low emission diesel engines, can yield beneficial secondary economic effects through energy savings, increased export earnings, recovery of valuable materials (e.g., solvents), and reduced waste disposal costs.

It is unclear at this time given current evidence, however, whether the net effect on the cost estimates of including second-order macroeconomic impacts would be positive or negative. In any case, even if only the adverse second-order economic effects were included, the resulting increase in the cost estimates would

not significantly affect the bottom line outcome that the monetizable benefits of 1970 to 1990 clean air programs exceeded costs by a substantial margin.

Appendix A: Cost and Macroeconomic Modeling

The purpose of this appendix is to describe in detail the estimation of direct compliance costs associated with the CAA and the effect of those expenditures on U.S. economic conditions from 1970 to 1990. The first section of this appendix describes the dynamic, general equilibrium macroeconomic model used to examine economy-wide effects. Two broad categories of models were considered for use in the assessment: Macroeconomic forecasting models (e.g., the Data Resources Inc. model of the U.S. economy), and general equilibrium models (e.g., Hazilla and Kopp [1990], and Jorgenson and Wilcoxen [1990a]). The project team selected the Jorgenson-Wilcoxen (J/W) general equilibrium model of the United States for this analysis (Jorgenson and Wilcoxen [1990a]). There are two main reasons for choosing a dynamic general equilibrium approach: To capture both the direct and indirect economic effects of environmental regulation, and to capture the long-run dynamics of the adjustment of the economy. The general equilibrium framework enabled the project team to assess shifts in economic activity between industries, including changes in distributions of labor, capital, and other production factors within the economy, and changes in the distribution of goods and services.

The second section describes the data sources for direct compliance expenditures and presents estimates of historical air pollution control expenditures. These estimates are derived primarily from EPA's 1990 report entitled "Environmental Investments: The Cost of a Clean Environment"[54] (hereafter referred to as *Cost of Clean*). Specific adjustments to the *Cost of Clean* stationary source and mobile source O&M data needed to adapt these data for use in the present study are also described. These adjusted expenditure estimates represent the compliance cost data used as inputs to the J/W model to determine macroeconomic effects.

The final section presents a summary of the direct expenditure data, presents direct costs in a form that can be compared to the benefits estimates found elsewhere in the study, and discusses indirect effects arising from compliance expenditures estimated by the macroeconomic model. The indirect effects reported by the model are sectoral impacts and changes in aggregate measures of economic activity such as household consumption and gross national product. These indirect effects are second-order impacts of compliance expenditures -- a parallel modeling exercise to estimate second-order economic impacts arising from the benefits of compliance (e.g., increased output as a result of improved longetivity or fewer workdays lost as a result of non-fatal heart attacks) has not been attempted.

Macroeconomic Modeling

EPA analyses of the costs of environmental regulations typically quantify the direct costs of pollution abatement equipment and related operating and maintenance expenses. However, this approach does not fully account for all of the broader economic consequences of reallocating resources to the production and use of pollution abatement equipment. A general equilibrium, macroeconomic model could, in theory, capture the complex interactions between sectors in the economy and assess the full economic cost of air

[54] Environmental Investments: The Cost of a Clean Environment, Report of the Administrator of the Environmental Protection Agency to the Congress of the United States, EPA-230-11-90-083, November 1990.

pollution control. This would be particularly useful for assessing regulations that may produce significant interaction effects between markets. Another advantage of a general equilibrium, macroeconomic framework is that it is internally consistent. The consistency of sectoral forecasts with realistic projections of U.S. economic growth is ensured since they are estimated within the context of a single model.[55] This contrasts with typical EPA analyses that compile cost estimates from disparate sectoral and partial equilibrium models.

The economic effects of the CAA may be over- or underestimated, if general equilibrium effects are ignored, to the extent that sectors not directly regulated are affected. For example, it is well known that the CAA imposed significant direct costs on the energy industry. Economic sectors not directly regulated will nonetheless be affected by changes in energy prices. However, an examination of the broader effects of the CAA on the entire economy might reveal that the CAA also led to more rapid technological development and market penetration of environmentally "clean" renewable sources of energy (e.g., photovoltaics). These effects would partially offset adverse effects on the energy industry, and lead to a different estimate of the total economic cost to society of the CAA.

The significance of general equilibrium effects in the context of any particular analysis is an empirical question. Kokoski and Smith (1987) used a computable general equilibrium model to demonstrate that partial-equilibrium welfare measures can offer reasonable approximations of the true welfare changes for large exogenous changes. In contrast, the results of Jorgenson and Wilcoxen (1990a) and Hazilla and Kopp (1990) suggest that total pollution abatement in the U.S. has been a major claimant on productive resources, and the effect on long-run economic growth may be significant. Again, such conclusions must be considered in light of the limitations of general equilibrium models.

Choice of Macroeconomic Model

The adequacy of any model or modeling approach must be judged in light of the policy questions being asked. One goal of the present study is to assess the effects of clean air regulations on macroeconomic activity. Two broad categories of macroeconomic models were considered for use in the assessment: short run, Keynesian models and long-run, general equilibrium models.

Recognizing that structural differences exist between the models, one needs to focus in on the particular questions that should be answered with any particular model. The Congressional Budget Office (1990) noted:

> "Both the [Data Resources Incorporated] DRI and the IPCAEO models show relatively limited possibilities for increasing energy efficiency and substituting other goods for energy in the short run... Both models focus primarily on short-term responses to higher energy prices, and *neither is very good at examining how the structure of the economy could change in response to changing energy prices.* The [Jorgenson-Wilcoxen] model completes this part of the picture..."[56]

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[55] In the present study, both benefits and costs are driven off of the same macroeconomic projections from the Jorgenson/Wilcoxen model, to ensure that the estimates are based on a consistent set of economic assumptions.

[56] The Congressional Budget Office report (1990) refers to an older (1981) version of the Jorgenson model, not the current (1988) version. The approach to long-run dynamics differs between the two models. The newer Jorgenson-Wilcoxen model contains both the capital accumulation equation and the capital asset pricing equation. The 1981 version of the model contained only the capital accumulation equation.

One strategy for assessing the macroeconomic effects of the CAA would be to use a DRI-type model in conjunction with the Jorgenson-Wilcoxen model to assess both the long-term effects and the short-run transitions, in much the same way that the Congressional Budget Office used these models to assess the effects of carbon taxes. However, because of significant difficulties in trying to implement the DRI model in a meaningful way, the project team chose to focus on the long-run effects of the CAA. Structural changes (e.g., changes in employment in the coal sector due to the CAA) can be identified with the Jorgenson-Wilcoxen model.

Overview of the Jorgenson-Wilcoxen Model

The discussion below focuses on those characteristics of the Jorgenson-Wilcoxen model that have important implications for its use in the assessment of environmental regulations (see Table 26). The J/W model is a detailed dynamic general equilibrium model of the U.S. economy designed for medium run analysis of regulatory and tax policy (Jorgenson and Wilcoxen [1990a]). It provides projections of key macroeconomic variables, such as GNP and aggregate consumption, as well as energy flows between economic sectors. As a result, the model is particularly useful for examining how the structure of the economy could change in response to changes in resource prices. For the purpose of this study, it has five key features: a detailed treatment of production and consumption, parameters estimated econometrically form historical data, an endogenous model of technical change, a rigorous representation of saving and investment, and free mobility of labor and capital between industries.

Table 26. Key Distinguishing Characteristics of the Jorgenson-Wilcoxen Model.

- Dynamic, general equilibrium, macroeconomic model of the U.S. economy.
- Econometrically estimated using historic data.
- Free mobility of a single type of capital and labor between industries.
- Detailed treatment of production and consumption.
- Rigorous representation of savings and investment.
- Endogenous model of technical change.
- Does not capture unemployment, underemployment, or the costs of moving capital from one industry to another.

The first two features, industry and consumer detail and econometric estimation, allow the model to capture the effects of the CAA at each point in time for given levels of technology and the size of the economy's capital stock. A detailed treatment of production and consumption is important because the principal effects of the Clean Air Act fell most heavily on a handful of industries. The J/W model divides total U.S. production into 35 industries which allows the primary economic effects of the CAA to be captured. Econometric estimation is equally important because it ensures that the behavior of households and firms in the model is consistent with the historical record.

The model's second two features —its representations of technical change and capital accumulation— complement the model's intratemporal features by providing specific information on how the Act affected technical change and the accumulation of capital. Many analyses of environmental regulations overlook or ignore intertemporal effects but these effects can be very important. Jorgenson and Wilcoxen (1990a) suggests that the largest cost of all U.S. environmental regulations together was that the regulations reduced the rate of capital accumulation.

The model's last feature, free mobility of a single type of capital and a single type of labor, is important because it limits the model's ability to measure the short run costs of changes in policy. J/W is a full-

employment model that describes the long-run dynamics of transitions from one equilibrium to another. Capital and labor are both assumed to be freely mobile between sectors (that is, they can be moved from one industry to another at zero cost) and to be fully used at all times. Over the medium to long run, this is a reasonable assumption, but in the short run it is too optimistic. In particular, the model will understate the short run costs of a change in policy because it does not capture unemployment, underemployment, or the costs of moving capital from one industry to another. A single rate of return on capital exists that efficiently allocates the capital in each period among sectors. Similarly, a single equilibrium wage rate allocates labor throughout the economy.

Table 27. Definitions of Industries Within the J/W Model.

Industry Number	Description
1	Agriculture, forestry, and fisheries
2	Metal mining
3	Coal mining
4	Crude petroleum and natural gas
5	Nonmetallic mineral mining
6	Construction
7	Food and kindred products
8	Tobacco manufacturers
9	Textile mill products
10	Apparel and other textile products
11	Lumber and wood products
12	Furniture and fixtures
13	Paper and allied products
14	Printing and publishing
15	Chemicals and allied products
16	Petroleum refining
17	Rubber and plastic products
18	Leather and leather products
19	Stone, clay, and glass products
20	Primary metals
21	Fabricated metal products
22	Machinery, except electrical
23	Electrical machinery
24	Motor vehicles
25	Other transportation equipment
26	Instruments
27	Miscellaneous manufacturing
28	Transportation and warehousing
29	Communication
30	Electric utilities
31	Gas utilities
32	Trade
33	Finance, insurance, and real estate
34	Other services
35	Government enterprises

Structure of the Jorgenson-Wilcoxen Model

The J/W model assesses a broad array of economic effects of environmental regulations. Direct costs are captured as increased expenditures on factors of production --capital, labor, energy and materials-- that the various industries must make to comply with the regulations, as well as additional out-of-pocket expenditures that consumers must make. Indirect costs are captured as general equilibrium effects that occur throughout the economy as the prices of factors of production change (e.g., energy prices). Also, the rate of technological change can respond to changes in the prices of factors of production, causing changes in productivity (Jorgenson and Fraumeni, 1981).

The model is divided into four major sectors: the business, household, government, and rest-of-the-world sectors. The business sector is further subdivided into 35 industries (see Table 27).[57] Each sector produces a primary product, and some produce secondary products. These outputs serve as inputs to the production processes of the other industries, are used for investment, satisfy final demands by the household and government sectors, and are exported. The model also allows for imports from the rest of the world.

[57] The 35 industries roughly correspond to a two-digit SIC code classification scheme.

The Business Sector

The model of producer behavior allocates the value of output of each industry among the inputs of the 35 commodity groups, capital services, labor services, and noncompeting imports. Output supply and factor demands of each sector are modeled as the results of choices made by wealth maximizing, price taking firms which are subject to technological constraints. Firms have perfect foresight of all future prices and interest rates. Production technologies are represented by econometrically estimated cost functions that fully capture factor substitution possibilities and industry-level biased technological change.

Capital and energy are specified separately in the factor demand functions of each industry. The ability of the model to estimate the degree of substitutability between factor inputs facilitates the assessment of the effect of environmental regulations. A high degree of substitutability between inputs implies that the cost of environmental regulation is low, while a low degree of substitutability implies high costs of environmental regulation. Also, different types of regulations lead to different responses on the part of producers. Some regulations require the use of specific types of equipment. Others regulations restrict the use of particular factor inputs; for example, through restrictions on the combustion of certain types of fuels. Both of these effects can change the rate of productivity growth in an industry through changes in factor prices.

The Household Sector

In the model of consumer behavior, consumer choices between labor and leisure and between consumption and saving are determined. A system of individual, demographically defined household demand functions are also econometrically estimated. Household consumption is modeled as a three stage optimization process. In the first stage households allocate lifetime wealth to full consumption in current and future time periods to maximize intertemporal utility. Lifetime wealth includes financial wealth, discounted labor income, and the imputed value of leisure. Households have perfect foresight of future prices and interest rates. In the second stage, for each time period full consumption is allocated between goods and services and leisure to maximize intratemporal utility. This yields an allocation of household's time endowment between the labor market (giving rise to labor supply and labor income) and leisure time and demands for goods and services. In the third stage, personal consumption expenditures are allocated among capital, labor, noncompeting imports and the outputs of the 35 production sectors to maximize a subutility function for goods consumption. As with the business sector, substitution possibilities exist in consumption decisions. The model's flexibility enables it to capture the substitution of nonpolluting products for polluting ones that may be induced by environmental regulations. Towards this end, purchases of energy and capital services by households are specified separately within the consumer demand functions for individual commodities.

It is important to be clear regarding the notions of labor supply and demand within the J/W model, and what is meant by "employment" throughout this report. Labor demands and supplies are represented as quality-adjusted hours denominated in constant dollars. The labor market clears in each period; the quantity of labor services offered by households is absorbed fully by the economy's producing sectors. However, inferences regarding the number of persons employed require information on labor quality and work-hours per person over time and across simulations. Neither of these are explicitly modeled.

The Government Sector

The behavior of government is constrained by exogenously specified budget deficits. Government tax revenues are determined by exogenously specified tax rates applied to appropriate transactions in the

business and household sectors. Levels of economic activity in these sectors are endogenously determined. Capital income from government enterprises (determined endogenously), and nontax receipts (given exogenously), are added to tax revenues to obtain total government revenues. Government expenditures adjust to satisfy the exogenous budget deficit constraint.

The Rest-of-the-World Sector

The current account balance is exogenous, limiting the usefulness of the model to assess trade competitiveness effects. Imports are treated as imperfect substitutes for similar domestic commodities and compete on price. Export demands are functions of foreign incomes and ratios of commodity prices in U.S. currency to the exchange rate. Import prices, foreign incomes, and tariff policies are exogenously specified. Foreign prices of U.S. exports are determined endogenously by domestic prices and the exchange rate. The exchange rate adjusts to satisfy the exogenous constraint on net exports.

Environmental Regulation, Investment, and Capital Formation

Environmental regulations have several important effects on capital formation. At the most obvious level, regulations often require investment in specific pieces of pollution abatement equipment. If the economy's pool of savings were essentially fixed, the need to invest in abatement equipment would reduce, or crowd out, investment in other kinds of capital on a dollar for dollar basis. On the other hand, if the supply of savings were very elastic then abatement investments might not crowd out other investment at all. In the J/W model, both the current account and government budget deficits are fixed exogenously so any change in the supply of funds for domestic investment must come from a change in domestic savings. Because households choose consumption, and hence savings, to maximize a lifetime utility function, domestic savings will be somewhat elastic. Thus, abatement investment will crowd out other investment, although not on a dollar for dollar basis.

The J/W assumption that the current account does not change as a result of environmental regulation is probably unrealistic, but it is not at all clear that this biases the crowding out effects in any particular direction. By itself, the need to invest in abatement capital would tend to raise U.S. interest rates and draw in foreign savings. To the extent this occurred, crowding out would be reduced. At the same time, however, regulation reduces the profitability of domestic firms. This effect would tend to lower the return on domestic assets, leading to a reduced supply of foreign savings which would exacerbate crowding out. Which effect dominates is an empirical question beyond the scope of this study.

In additional to crowding out ordinary investment, environmental regulation also has a more subtle effect on the rate of capital formation. Regulations raise the prices of intermediate goods used to produce new capital. This leads to a reduction in the number of capital goods which can be purchased with a given pool of savings. This is not crowding out in the usual sense of the term, but it is an important means by which regulation reduces capital formation.[58]

[58] Wilcoxen (1988) suggests that environmental regulation may actually lead to a "crowding in" phenomenon. Wilcoxen examined the effects of regulation at the firm level, and introduced costs into the model related to the installation of capital. He found that when firms shut down their plants to install environmental capital, they take account of the adjustment costs and often concurrently replace other older capital equipment. This effect, however, is not captured in the current version of the Jorgenson-Wilcoxen model.

The General Equilibrium

The J/W framework contains intertemporal and intratemporal models (Jorgenson and Wilcoxen [1990c]). In any particular time period, all markets clear. This market clearing process occurs in response to any changes in the levels of variables that are specified exogenously to the model. The interactions among sectors determine, for each period, aggregate domestic output, capital accumulation, employment, the composition of output, the allocation of output across different household types, and other variables.

The model also produces an intertemporal equilibrium path from the initial conditions at the start of the simulation to the stationary state. (A stationary solution for the model is obtained by merging the intertemporal and intratemporal models.) The dynamics of the J/W model have two elements: An accumulation equation for capital, and a capital asset pricing equation. Changes in exogenous variables cause several adjustments to occur within the model. First, the single stock of capital is efficiently allocated among all sectors, including the household sector. Capital is assumed to be perfectly malleable and mobile among sectors, so that the price of capital services in each sector is proportional to a single capital service price for the economy as a whole. The value of capital services is equal to capital income. The supply of capital available in each period is the result of past investment, i.e., capital at the end of each period is a function of investment during the period and capital at the beginning of the period. This capital accumulation equation is backward-looking and captures the effect of investments in all past periods on the capital available in the current period.

The capital asset pricing equation specifies the price of capital services in terms of the price of investment goods at the beginning and end of each period, the rate of return to capital for the economy as a whole, the rate of depreciation, and variables describing the tax structure for income from capital. The current price of investment goods incorporates an assumption of perfect foresight or rational expectations. Under this assumption, the price of investment goods in every period is based on expectations of future capital service prices and discount rates that are fulfilled by the solution of the model. This equation for the investment goods price in each time period is forward-looking.[59]

One way to characterize the J/W model —or any other neoclassical growth model— is that the short-run supply of capital is perfectly inelastic, since it is completely determined by past investment. However, the supply of capital is perfectly elastic in the long run. The capital stock adjusts to the time endowment, while the rate of return depends only on the intertemporal preferences of the household sector.

A predetermined amount of technical progress also takes place that serves to lower the cost of sectoral production. Finally, the quality of labor is enhanced, giving rise to higher productivity and lower costs of production.

Given all of these changes, the model solves for a new price vector and attains a new general equilibrium. Across all time periods, the model solves for the time paths of the capital stock, household consumption, and prices. The outcomes represent a general equilibrium in all time periods and in all markets covered by the J/W model.

[59] The price of capital assets is also equal to the cost of production, so that changes in the rate of capital accumulation result in an increase in the cost of producing investment goods. This has to be equilibrated with the discounted value of future rentals in order to produce an intertemporal equilibrium. The rising cost of producing investment is a cost of adjusting to a new intertemporal equilibrium path.

Configuration of the No-control Scenario

One of the difficulties in describing the no-control scenario is ascertaining how much environmental regulation would have been initiated by state and local governments in the absence of a federal program. It may reasonably be argued that many state and local governments would have initiated their own control programs in the absence of a federal role. This view is further supported by the fact that many states and localities have, in fact, issued rules and ordinances which are significantly more stringent and encompassing than federal minimum requirements. However, it may also be argued that the federal CAA has motivated a substantial number of stringent state and local control programs.

Specifying the range and stringency of state and local programs that would have occurred in the absence of the federal CAA would be almost entirely speculative. For example, factors which would complicate developing assumptions about stringency and scope of unilateral state and local programs include: (I) the significance of federal funding to support state and local program development; (ii) the influence of more severe air pollution episodes which might be expected in the absence of federally-mandated controls; (iii) the potential emergence of pollution havens, as well as anti-pollution havens, motivated by local political and economic conditions; (iv) the influence of federally-sponsored research on the development of pollution effects information and control technologies; and (v) the need to make specific assumptions about individual state and local control levels for individual pollutants to allow estimation of incremental reductions attributable to federal control programs.

Another complication associated with the no-control scenario is the treatment of air pollution control requirements among the major trading partners of the U.S. Real-world manifestation of a no-control scenario would imply that public health and environmental goals were not deemed sufficiently compelling by U.S. policymakers. Under these conditions, major trading partners of the U.S. in Japan, Europe, and Canada may well reach similar policy conclusions. Simply put, if the U.S. saw no need for air pollution controls, there is little reason to assume other developed industrial countries would have either. In this case, some of the estimated economic benefits of reducing or eliminating air pollution controls in the U.S. would not materialize because U.S. manufacturers would not necessarily gain a production cost advantage over foreign competitors. However, like the question of state and local programs in the absence of a federal program, foreign government policies under a no-control scenario would be highly speculative.

Given the severity of these confounding factors, the only analytically feasible assumptions with respect to the no-control scenario are that (a) no new control programs would have been initiated after 1970 by the states or local governments in the absence of a federal role, and (b) environmental policies of U.S. trading partners remain constant regardless of U.S. policy.

Elimination of Compliance Costs in the No-Control Case

Industries that are affected by environmental regulations can generally respond in three ways: (I) with process changes (e.g., fluidized bed combustion); (ii) through input substitution (e.g., switching from high sulfur coal to low sulfur coal); and (iii) end-of-pipe abatement (e.g., the use of electrostatic precipitation to reduce the emissions of particulates by combustion equipment).[60] Clean air regulations have typically led

[60] Regulation may also affect the rate of investment, and change the rate of capital accumulation.

to the latter two responses, especially in the short run. End-of-pipe abatement is usually the method of choice for existing facilities, since modifying existing production processes can be costly. This approach is also encouraged by EPA's setting of standards based on the notion of "best available technology" (Freeman, 1978).

All three possible responses may lead to: (I) unanticipated losses to equity owners; (ii) changes in current output; and (iii) changes in long-run profitability. If firms were initially maximizing profits, then any of the above three responses will increase its costs. Fixed costs of investment will be capitalized immediately. This will result in a loss to owners of equity when regulations are introduced. As far as firms are concerned, this is just like a lump sum tax on sunk capital. Such effects will not affect growth or efficiency. However, regulations could also change marginal costs and therefore current output. In addition, they could change profits (i.e., the earnings of capital), and thus affect investment. Both of these effects will reduce the measured output of the economy.

On the consumption side, environmental regulations change consumers' expectations of their lifetime wealth. In the no-control scenario of this assessment, lifetime wealth increases. This causes an increase in consumption. In fact, with perfect foresight, consumption rises more in earlier time periods. This also results in a change in savings.

Capital Costs - Stationary Sources

To appropriately model investment in pollution control requires a recognition that the CAA had two different effects on capital markets. First, CAA regulations led to the retrofitting of existing capital stock in order to meet environmental standards. In the no-control scenario, these expenditures do not occur. Instead, the resources that were invested in pollution abatement equipment to retrofit existing sources are available to go to other competing investments. Thus, at each point in time, these resources might go to investments in capital in the regulated industry, or may go into investments in other industries, depending upon relative rates of return on those investments. This will affect the processes of capital formation and deepening.

Second, the CAA placed restrictions on new sources of emissions. When making investment decisions, firms take into account the additional cost of pollution abatement equipment. Effectively, the "price" of investment goods is higher because more units of capital are required to produce the same amount of output. In the no-control scenario, there are no restrictions on new sources and hence no requirements for pollution control expenditures. Effectively, the "price" of investment goods is lower. Thus, at each point in time, investors are faced with a lower price of investment goods. This results in a different profile for investment over time.

Operating and Maintenance Costs - Stationary Sources

In addition to purchasing pollution abatement equipment, firms incurred costs to run and maintain the pollution abatement equipment. In the no-control scenario, resources used to pay for these operating and maintenance (O&M) costs are freed up for other uses. The model assumes that the resources required to run and maintain pollution control equipment are in the same proportions as the factor inputs used in the underlying production technology. For example, if 1 unit of labor and 2 units of materials are used to produce 1 unit of output, then one-third of pollution control O&M costs are allocated to labor and two-thirds are allocated to materials. These adjustments were introduced at the sector level. O&M expenditures are exclusive of depreciation charges and offset by any recovered costs.

Capital Costs - Mobile Sources

Capital costs associated with pollution control equipment were represented by changing costs for motor vehicles (sector 24) and other transportation equipment (sector 26). Prices (unit costs) were reduced in proportion to the value of the pollution control devices contained in cars, trucks, motorcycles, and aircraft.

Operating and Maintenance - Mobile Sources

Prices for refined petroleum products (sector 16) were changed to reflect the resource costs associated with producing unleaded and reduced lead gasoline (fuel price penalty), the change in fuel economy for vehicles equipped with pollution control devices (fuel economy penalty), and the change in fuel economy due to the increased fuel density of lower leaded and no lead gasoline (fuel economy credit). Third, inspection and maintenance costs and a maintenance credit associated with the use of unleaded and lower leaded (i.e., unleaded and lower leaded gasoline is less corrosive, and therefore results in fewer muffler replacements, less spark plug corrosion, and less degradation of engine oil) were represented as changes in prices for other services (sector 34).

Direct Compliance Expenditures Data

Sources of Cost Data

Cost data for this study are derived primarily from the 1990 *Cost of Clean* report. EPA publishes cost data in response to requirements of the Clean Air and Clean Water Acts. The following subsections describe *Cost of Clean* data in detail, as well as adjustments made to the data and data from other sources.

Cost of Clean Data

EPA is required to compile and publish public and private costs resulting from enactment of the Clean Air Act and the Clean Water Act. The 1990 *Cost of Clean* report presents estimates of historical pollution control expenditures for the years 1972 through 1988 and projected future costs for the years 1989 through 2000. This includes federal, state, and local governments as well as the private sector. Estimates of capital costs, operation and maintenance (O&M) costs, and total annualized costs for five categories of environmental media, including air, water, land, chemical, and multi-media, are presented. It should be noted that these estimates represent direct regulatory implementation and compliance costs rather than social costs. The *Cost of Clean* relied on data from two governmental sources, the EPA and the U.S. Department of Commerce (Commerce).

EPA Data

EPA expenditures were estimated from EPA budget justification documents.[61] Estimates of capital and operating costs resulting from new and forthcoming regulations were derived from EPA's Regulatory

[61] The main source of data for EPA expenditures is the *Justification of Appropriation Estimates for Committee on Appropriations.*

Impact Analyses (RIAs). RIAs have been prepared prior to the issuance of all major regulations since 1981. Finally, special analyses conducted by EPA program offices or contractors were used when other data sources did not provide adequate or reliable data.

Commerce Data

Data collected by Commerce were used extensively in the *Cost of Clean* for estimates of historical pollution control expenditures made by government agencies other than EPA and by the private sector. Two Commerce agencies, the Bureau of Economic Analysis (BEA) and the Bureau of the Census (Census), have collected capital and operating costs for compliance with environmental regulations since the early 1970's. Commerce is, in fact, the primary source of original survey data for environmental regulation compliance costs. Commerce publishes a number of documents that report responses to surveys and comprise most of the current domain of known pollution abatement and control costs in the United States, including:

- A series of articles entitled "Pollution Abatement and Control Expenditures" published annually in the *Survey of Current Business* by BEA (BEA articles);
- A series of documents entitled "Pollution Abatement Costs and Expenditures" published annually in the *Current Industrial Reports* by Census (PACE reports); and
- A series of documents entitled *Government Finances* published annually by Census (Government Finances).

BEA articles contain data derived from a number of sources, including two key agency surveys —the "Pollution Abatement Costs and Expenditures Survey" (PACE Survey) and the "Pollution Abatement Plant and Equipment Survey" (PAPE Survey)— which are conducted annually by Census for BEA. Data have been reported for 1972 through 1987.[62]

PACE reports have been published annually since 1973 with the exception of 1987. Figures for 1987 were estimated on the basis of historical shares within total manufacturing. These reports contain expenditure estimates derived from surveys of about 20,000 manufacturing establishments. Pollution abatement expenditures for air, water and solid waste are reported by state and Standard Industrial Code (SIC) at the four-digit level. According to Census, surveys conducted since 1976 have not included establishments with fewer than 20 employees because early surveys showed that they contributed only about 2 percent to the pollution estimates while constituting more than 10 percent of the sample size.

Each year Census conducts a survey of state, local, and county governments; and survey results are published in *Government Finances*. Census asks government units to report revenue and expenditures, including expenditures for pollution control and abatement.

Non-EPA Federal expenditures were estimated from surveys completed by federal agencies detailing their pollution control expenditures, which are submitted to BEA. Private sector air pollution control expenditures, as well as state and local government air pollution expenditures, were taken from BEA articles.

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[62] The most recent BEA article used as a source for air pollution control costs in the Cost of Clean was "Pollution Abatement and Control Expenditures, 1984-87" in *Survey of Current Business*, June 1989.

Stationary Source Cost Data

Capital Expenditures Data

Capital expenditures for stationary air pollution control are made by factories and electric utilities for plant and equipment that abate pollutants through end-of-line (EOL) techniques or that reduce or eliminate the generation of pollutants through changes in production processes (CIPP). For the purposes of this report EOL and CIPP expenditures are aggregated.[63] Table 28 summarizes capital expenditures for stationary air pollution control, categorized as "nonfarm business" or "government enterprise" expenditures.

Nonfarm business capital expenditures consist of plant and equipment expenditures made by 1) manufacturing companies and 2) privately and cooperatively owned electric utilities and 3) other nonmanufacturing companies. "Government enterprise" is, according to BEA, an agency of the government whose operating costs, to a substantial extent, are covered by the sale of goods and services. Here, government enterprise means specifically government enterprise electric utilities. Government enterprise capital expenditures are pollution abatement expenditures made by publicly owned electric utilities.[64]

Table 28. Estimated Capital and O&M Expenditures for Stationary Source Air Pollution Control (millions of current dollars).

	Nonfarm Business		Government Enterprise	
Year	Cap.[a]	O&M[b]	Cap.[c]	O&M[d]
1973	2,968	1,407	82	29
1974	3,328	1,839	104	56
1975	3,914	2,195	102	45
1976	3,798	2,607	156	58
1977	3,811	3,163	197	60
1978	3,977	3,652	205	72
1979	4,613	4,499	285	106
1980	5,051	5,420	398	148
1981	5,135	5,988	451	135
1982	5,086	5,674	508	141
1983	4,155	6,149	422	143
1984	4,282	6,690	416	147
1985	4,141	6,997	328	189
1986	4,090	7,116	312	140
1987	4,179	7,469	277	130
1988	4,267	7,313	243	161
1989	4,760	7,743	235	173
1990	4,169	8,688	226	154

Sources:
a. Non-farm capital expenditures for 1973-87 are from *Cost of Clean*, Table B-1, line 2.
b. Non-farm O&M expenditures for 1973-85 are from *Cost of Clean*, Table B-1, line 8.
c. Government enterprise capital expenditures for 1973-87 are from *Cost of Clean*, Table B-9, line 1.
d. Government enterprise O&M expenditures for 1973-85 are from *Cost of Clean*, Table B-9, line 5.
All other reported expenditures are EPA estimates.

Operation and Maintenance Expenditures Data

Stationary source O&M expenditures are made by manufacturing establishments, private and public electric utilities, and other nonmanufacturing businesses to operate air pollution abatement equipment. O&M expenditures for electric utilities are made up of two parts: 1) expenditures for operating air pollution equipment and 2) the additional expenditures associated with switching to alternative fuels that have lower sulfur content (fuel

[63] Survey respondents to the Census annual Pollution Abatement Surveys report the difference between expenditures for CIPP and what they would have spent for comparable plant and equipment without pollution abatement features. Disaggregated capital expenditures by private manufacturing establishments can be found in annual issues of Census reports.

[64] BEA calculates these expenditures using numbers obtained from Energy Information Agency (EIA) Form 767 on steam-electric plant air quality control.

differential). Expenditures to operate air pollution abatement equipment are for the collection and disposal of flyash, bottom ash, sulfur and sulfur products, and other products from flue gases.[65] O&M expenditures are net of depreciation and payments to governmental units, and are summarized in Table 28. O&M data were disaggregated to the two digit SIC level for use in the macroeconomic model.

For both capital and O&M expenditures, historical survey data were not available for each year through 1990 prior to publication of *Cost of Clean*. For the purpose of the 812 analysis, EPA projected 1988 to 1990 capital expenditures and 1986 to 1990 O&M expenditures. Those projections were used in the macroeconomic simulation, and have been retained as cost estimates to ensure consistency between the macroeconomic results and the direct cost estimates. Since completion of the macroeconomic modeling, however, BEA has published expenditure estimates through 1990. A comparison of more recent BEA estimates with the EPA projections used in the 812 analysis can be found in the "Uncertainties in the Cost Analysis" section, below.

Recovered Costs

"Recovered costs" are costs recovered (i.e., revenues realized) by private manufacturing establishments through abatement activities. According to instructions provided to survey participants by Census, recovered costs consist of 1) the value of materials or energy reclaimed through abatement activities that were reused in production and 2) revenue that was obtained from the sale of materials or energy reclaimed through abatement activities. Estimates of recovered costs were obtained from the PACE reports and are summarized in Table 29. In this analysis, recovered costs were removed from total stationary source air pollution control O&M costs -- that is, net O&M cost in any year would be O&M expenditures (see Table 28) less recovered costs. Recovered cost data were disaggregated to the two digit SIC level for use in the macroeconomic model.

[65] Farber, Kit D. and Gary L. Rutledge, "Pollution Abatement and Control Expenditures: Methods and Sources for Current-Dollar Estimates," Unpublished paper, Bureau of Economic Analysis, U.S. Department of Commerce, October 1989.

Mobile Source Data

Costs of controlling pollution emissions from motor vehicles were estimated by calculating the purchase price and O&M cost premiums associated with vehicles equipped with pollution abatement controls over the costs for vehicles not equipped with such controls. These costs were derived using EPA analyses, including EPA RIAs, the *Cost of Clean*, and other EPA reports.[66] This Appendix summarizes the Section 812 mobile source compliance cost estimates and provides references to published data sources where possible. Further information on specific methods, analytical steps, and assumptions can be found in McConnell *et al.* (1995),[67] which provides a detailed description of the Section 812 mobile source cost estimation exercise (referred to as *Cost of Clean* (1993, upublished)) and compares the method and results to other similar analyses (including *Cost of Clean* (1990)).

Table 29. Estimated Recovered Costs for Stationary Source Air Pollution Control (millions of current dollars).

Year	PACE*	Estimated
1972		248
1973		199
1974		296
1975		389
1976		496
1977		557
1978		617
1979	750	750
1980	862	862
1981	1,000	997
1982	858	857
1983	822	822
1984	866	870
1985	767	768
1986	860	867
1987		987
1988	1,103	1,107
1989		1,122
1990		1,256

* Air cost recovered as reported in PACE

Source: "Pollution Abatement Costs and Expenditures" published annually in the Current Industrial Reports by Census.

[66] A complete listing of sources used in calculating mobile source capital and operating expenditures can be found in *Environmental Investments: The Cost of a Clean Environment*, Report of the Administrator of the Environmental Protection Agency to the Congress of the United State, EPA-230-11-90-083, November 1990.

[67] *Evaluating the Cost of Compliance with Mobile Source Emission Control Requirements: Retrospective Analysis*, Resources for the Future Discussion Paper, 1995.

Capital Expenditures Data

Capital expenditures for mobile source emission control are associated primarily with pollution abatement equipment on passenger cars, which comprise the bulk of all mobile sources of pollution. These capital costs reflect increasingly stringent regulatory requirements and improvements in pollution control technologies over time. Each of the following devices have been used at one time or another dating back to the Clean Air Act Amendments of 1965: air pumps, exhaust-gas recirculation valves, high altitude controls, evaporative emissions controls, and catalysts. The cost estimates for each component were computed on a per-vehicle basis by engineering cost analyses commissioned by EPA. The resulting per-vehicle capital costs were multiplied by vehicle production estimates to determine annual capital costs. Table 30 summarizes mobile source capital costs.

Table 30. Estimated Capital and Operation and Maintenance Expenditures for Mobile Source Air Pollution Control (millions of current dollars).

Year	Capital[a]	O&M[b]
1973	276	1,765
1974	242	2,351
1975	1,570	2,282
1976	1,961	2,060
1977	2,248	1,786
1978	2,513	908
1979	2,941	1,229
1980	2,949	1,790
1981	3,534	1,389
1982	3,551	555
1983	4,331	-155
1984	5,679	-326
1985	6,387	337
1986	6,886	-1,394
1987	6,851	-1,302
1988	7,206	-1,575
1989	7,053	-1,636
1990	7,299	-1,816

Sources:
a. Capital exp.: *Cost of Clean*, Tables C-2 to C-9, line 3 on each; Tables C-2Ato C-9A, line 10 on each; converted from $1986 to current dollars.
b. O&M exp.: EPA analyses based on sources and methods in: *Costs and Benefits of Reducing Lead in Gasoline: Final Regulatory Impact Analysis*, U.S. Environmental Protection Agency, Office of Policy Analysis, EPA-230-05-85-006, February 1985; and *Cost of Clean*.

Operation and Maintenance Expenditures Data

Costs for operation and maintenance of emission abatement devices include the costs of maintaining pollution control equipment plus the cost of vehicle inspection/maintenance programs. Operating costs per vehicle were multiplied by total vehicles in use to determine annual cost. Mobile source O&M costs are made up of three factors: 1) fuel price penalty, 2) fuel economy penalty, and 3) inspection and maintenance program costs as described below. These costs are mitigated by cost savings in the form of maintenance economy and fuel density economy. Table 31 summarizes mobile source O&M expenditures and cost savings by categories, with net O&M costs summarized above in Table 30. The following sections describe the components of the mobile source O&M cost estimates.

Fuel Price Penalty

Historically, the price of unleaded fuel has been several cents per gallon higher than the price of leaded fuel. CAA costs were calculated as the difference between the cost of making unleaded gasoline and leaded gasoline with lower lead levels and the cost of making only leaded gasoline with a lead content set at pre-regulatory levels. These cost estimates were developed using a linear programming model of the refinery industry. Prices of crude oil and other unfinished oils, along with the prices of refinery outputs, were adjusted annually according to price indices for imported crude oil over the period of analysis. The relative shares of leaded and unleaded gasoline and the average lead content in leaded gasoline also were

adjusted adjusted annnually according to the historical record.

These estimates may tend to understate costs due to a number of biases inherent in the analysis process. For example, the refinery model was allowed to optimize process capacities in each year. This procedure is likely to understate costs because regulatory requirements and market developments cannot be perfectly anticipated over time. This procedure resulted in estimates that are about ten percent less than estimates in other EPA reports.[68] However, new process technologies that were developed in the mid-1980s were not reflected in either the base case or regulatory case runs. It is reasonable to expect that regulatory requirements would have encouraged development of technologies at a faster rate than would have occurred otherwise.

Fuel Economy Penalty

The fuel economy penalty benefit is the cost associated with the increased/decreased amount of fuel used by automobiles with air pollution control devices (all else being equal). An assumption that can be made is that the addition of devices, such as catalytic converters, decrease automobile fuel efficiency.[69] If this assumption is true, air pollution control devices increase the total fuel cost to consumers. An alternative assumption is that the use of catalytic converters has increased fuel economy. This increase has been attributed in large measure to the feedback mechanism built into three-way catalytic converters.[70] Under this assumption, the decrease in total fuel cost to consumers is considered a benefit of the program.

Table 31. O&M Costs and Credits (millions of current dollars).

Year	Fuel Price Penalty	Fuel Econ. Penalty	Net I & M*	Total Costs
1973	91	1700	-26	1765
1974	244	2205	-98	2351
1975	358	2213	-289	2282
1976	468	2106	-514	2060
1977	568	1956	-738	1786
1978	766	1669	-1527	908
1979	1187	1868	-1826	1229
1980	1912	1998	-2120	1790
1981	2181	1594	-2386	1389
1982	2071	1026	-2542	555
1983	1956	628	-2739	-155
1984	2012	313	-2651	-326
1985	3057	118	-2838	337
1986	2505	-40	-3859	-1394
1987	2982	-158	-4126	-1302
1988	3127	-210	-4492	-1575
1989	3476	-318	-4794	-1636
1990	3754	-481	-5089	-1816

* Inspection and maintenance costs less fuel density savings and maintenance savings.

Sources: All results are presented in Jorgenson *et al.* (1993), pg. A.17. FPP results are based on a petroleum refinery cost model run for the retrospective analysis. FEP and Net I&M are based on data and methods from *Costs and Benefits of Reducing Lead in Gasoline: Final Regulatory Impact Analysis*, U.S. Environmental Protection Agency, Office of Policy Analysis, EPA-230-05-85-006, February 1985; and *Cost of Clean* (1990). Specific analytic procedures are summarized in McConnell *et al.* (1995).

For the purposes of this study, sensitivity analyses were performed using data presented in the *Cost of Clean* report. These analyses were conducted to evaluate the significance of assumptions about the relationship between mile per gallon (MPG) values for controlled automobiles and MPG values for uncontrolled cars. Based on results of these and other analyses, fuel economy was assumed to be equal for

[68] Costs and Benefits of Reducing Lead in Gasoline: Final Regulatory Impact Analysis, U.S. Environmental Protection Agency, Office of Policy Analysis, EPA-230-05-85-006, February 1985.

[69] Memo from Joel Schwartz (EPA/OPPE) to Joe Somers and Jim DeMocker dated December 12, 1991, and entitled "Fuel Economy Benefits." Schwartz states that since this analysis is relative to a no Clean Air Act baseline, not a 1973 baseline, fuel economy benefits are not relevant. In the absence of regulation, tuning of engines for maximum economy would presumably be optimal in the base case as well.

[70] Memo from Joseph H. Somers, EPA Office of Mobile Sources, to Anne Grambsch (EPA/OPPE) and Joel Schwartz (EPA/OPPE) entitled "Fuel Economy Penalties for Section 812 Report," December 23, 1991.

controlled and uncontrolled vehicles from 1976 onward. This may bias the cost estimates although in an unknown direction.

Inspection and Maintenance Programs

Inspection and maintenance programs are administered by a number of states. Although these programs are required by the Clean Air Act, the details of administration were left to the discretion of state or local officials. The primary purpose of inspection and maintenance programs is to identify cars that require maintenance —including cars that 1) have had poor maintenance, 2) have been deliberately tampered with or had pollution control devices removed, or 3) have used leaded gasoline when unleaded is required— and force the owners of those cars to make necessary repairs or adjustments.[71] Expenditures for inspection and maintenance were taken from the *Cost of Clean*.

Beneficial effects of the mobile source control program associated with maintenance and fuel density were also identified. These cost savings were included in this study as credits to be attributed to the mobile source control program. Credits were estimated based on an EPA study,[72] where more detailed explanations may be found.

Maintenance Credits

Catalytic converters require the use of unleaded fuel, which is less corrosive than leaded gasoline. On the basis of fleet trials, the use of unleaded or lower leaded gasoline results in fewer muffler replacements, less spark plug corrosion, and less degradation of engine oil, thus reducing maintenance costs. Maintenance credits account for the majority of the direct (non-health) economic benefits of reducing the lead concentration in gasoline.

Fuel Density Credits

The process of refining unleaded gasoline increases its density. The result is a gasoline that has higher energy content. Furthermore, unleaded gasoline generates more deposits in engine combustion chambers, resulting in slightly increased compression and engine efficiency. Higher energy content of unleaded gasoline and increased engine efficiency from the used of unleaded gasoline yield greater fuel economy and therefore savings in refining, distribution, and retailing costs.

Other Direct Cost Data

The *Cost of Clean* report includes several other categories of cost that are not easily classified as either stationary source or mobile source expenditures. Federal and state governments incur air pollution abatement costs; additionally, federal and state governments incur costs to develop and enforce CAA regulations. Research and development expenditures by the federal government, state and local governments, and (especially) the private sector can be attributed to the CAA. These data are summarized by year in Table 32.

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[71] Walsh, Michael P., "Motor Vehicles and Fuels: The Problem," *EPA Journal*, Vol. 17, No. 1, January/February 1991, p. 12.

[72] Schwartz, J., *et al. Costs and Benefits of Reducing Lead in Gasoline: Final Regulatory Impact Analysis*, U. S. Environmental Protection Agency, Economic Analysis Division, Office of Policy Analysis, February 1985.

Unlike the other private sector expenditure data used for this analysis, the survey data used as a source for private sector R&D expenditures cannot be disaggregated into industry-specific expenditure totals. Consequently, private sector R&D expenditures are omitted from the macroeconomic modeling exercise (the macro model is industry-specific). The R&D expenditures are, however, included in aggregate cost totals used in the benefit-cost analysis.

Table 32. Other Air Pollution Control Expenditures (millions of current dollars).

Year	Abatement		Regulations and Monitoring		Research and Development			Total
	Fed.[a]	State & Local[b]	Fed.[c]	State & Local[d]	Private[e]	Fed.[f]	State & Local[g]	
1973	47	0	50	115	492	126	6	836
1974	56	0	52	131	520	100	7	866
1975	88	1	66	139	487	108	8	897
1976	105	1	69	135	562	131	6	1,009
1977	106	1	80	161	675	144	7	1,174
1978	90	0	93	183	805	146	8	1,325
1979	103	0	100	200	933	105	7	1,448
1980	95	0	122	207	851	130	5	1,410
1981	85	0	108	226	798	131	0	1,348
1982	87	0	93	230	761	126	2	1,229
1983	136	4	88	239	691	133	6	1,297
1984	115	14	101	250	665	165	4	1,314
1985	98	12	103	250	775	247	3	1,488
1986	67	14	106	307	833	217	4	1,548
1987	80	15	110	300	887	200	2	1,594
1988	65	10	120	320	934	220	1	1,670
1989	70	12	130	360	984	230	2	1,788
1990	71	13	133	343	749	231	2	1,542

Sources:
a. Federal government abatement expenditures: 1973-82, "Pollution Abatement and Control Expenditures", Survey of Current Business (BEA) July 1986 Table 9 line 13; 1983-87, BEA June 1989 Table 7 line 13; 1988-90, BEA May 1995 Table 7 line 13.
b. State and local abatement expenditures: 1973-87, *Cost of Clean*, Table B-9 line 2; 1988-90, BEA May 1995 Table 7 line 14.
c. Federal government "regs/monitoring" expenditures: 1973-82, BEA July 1986, Table 9 line 17; 1983-87, BEA June 1989 Table 6 line 17; 1988-90, BEA May 1995 Table 7 line 17.
d. State and local government "regs/monitoring" expenditures: 1973-87, *Cost of Clean*, Table B-9 line 3; 1988-90, BEA May 1995 Table 7 line 18.
e. Private sector R&D expenditures: 1973-86, BEA May 1994 Table 4 (no line #) [total R&D expenditures in $1987 are converted to current dollars using the GDP price deflator series found elsewhere in this Appendix – netting out public sector R&D leaves private sector expenditures]; 1987-90, BEA May 1995 Table 7 line 20.
f. Federal government R&D expenditures: 1973-82, BEA July 1986 Table 9 line 21; 1983-87, BEA June 1989 Table 6 line 21; 1988-90, BEA May 1995, Table 7 line 21.
g. State and local government R&D expenditures: 1973-87, *Cost of Clean*, Table B-9 line 4; 1988-90, BEA May 1995 Table 7 line 22.

The *Cost of Clean* and the series of articles "Pollution Abatement and Control Expenditures" in the Survey of Current Business (various issues) are the data sources for "Other Air Pollution Control Expenditures." State and local expenditures through 1987 are found in *Cost of Clean*; 1988-90 expenditures are from more recent issues of the Survey of Current Business (BEA). Federal government expenditures are from BEA (various issues). Private R&D expenditures were reported in *Cost of Clean*. Since publication of *Cost of Clean*, however, BEA has revised its private sector R&D expenditure series (BEA, 1994 and 1995). Since private R&D expenditures were not included in the macroeconomic modeling exercise, the revised series can be (and has been) used without causing inconsistency with other portions of the 812 analysis.

Assessment Results

Direct Expenditures and Costs

Compliance with the CAA imposed direct costs on businesses, consumers, and governmental units, and triggered other expenditures such as governmental regulation and monitoring costs and expenditures for research and development by both government and industry. As shown in Table 33, annual direct CAA compliance expenditures – including R&D, etc.– over the period from 1973 to 1990 were remarkably stable, ranging from $20 billion to $25 billion in inflation-adjusted 1990 dollars. This is equal to approximately ⅓ of one percent of total domestic output during that period, with the percentage falling from ½ of one percent of total output in 1973 to ¼ of one percent in 1990.

Table 33. Estimated Annual CAA Compliance Expenditures ($millions).

Year	Capital[a]	O&M and "Other"[b]	Total	GDP Price Defl.	Total Exp. ($1990)
1973	3,326	3,838	7,164	41.3	19,635
1974	3,674	4,816	8,490	44.9	21,405
1975	5,586	5,030	10,616	49.2	24,425
1976	5,915	5,238	11,153	52.3	24,139
1977	6,256	5,626	11,882	55.9	24,062
1978	6,695	5,340	12,035	60.3	22,593
1979	7,839	6,532	14,371	65.5	24,837
1980	8,398	7,906	16,304	71.7	25,741
1981	9,120	7,863	16,983	78.9	24,367
1982	9,145	6,812	15,957	83.8	21,555
1983	8,908	6,612	15,520	87.2	20,148
1984	10,377	6,955	17,332	91.0	21,560
1985	10,856	8,243	19,099	94.4	22,903
1986	11,288	6,543	17,831	96.9	20,831
1987	11,307	6,904	18,211	100.0	20,615
1988	11,716	6,462	18,178	103.9	19,805
1989	12,048	6,946	18,994	108.5	19,817
1990	11,707	7,312	19,019	113.2	19,019
average expenditure, 1973-1990 ($1990) =					22,081

a. Capital expenditures are the sum of stationary source (private sector plus government enterprise) and mobile source capital expenditures.
b. The sum of stationary source O&M, mobile source O&M, anatement, regulation and monitoring, and R&D expenditures.

Although useful for many purposes, a summary of direct annual expenditures is not the best cost measure to use when comparing costs to benefits. Capital expenditures are investments, generating a stream of benefits (and opportunity cost) over the life of the investment. The appropriate accounting technique to use for capital expenditures in a cost/benefit analysis is to *annualize* the expenditure – i.e., spread the cost over the useful life of the investment, applying a discount rate to account for the time value of money.

For this cost/benefit analysis, all capital expenditures have been annualized at three, five, and seven percent (real) rates of interest. Therefore, "annualized" costs reported for any given year are equal to O&M expenditures (plus R&D, etc., expenditures) plus amortized capital costs (i.e., depreciation plus interest costs associated with the pre-existing capital *stock*) for that year. Stationary source air pollution control capital costs are amortized over 20 years; mobile source air pollution control costs are amortized over 10 years. Table 34 summarizes annual expenditures and annualized costs.

Due to data limitations, the cost analysis for this CAA retrospective starts in 1973, missing costs incurred in 1970-72. *Cost of Clean*, however, includes capital expenditures for 1972. In this analysis, amortized costs arising from 1972 capital investments *are included* in the 1973 to 1990 annualized costs,

even though 1972 costs are not otherwise included in the analysis. Conversely, only a portion of the (e.g.) 1990 capital expenditures are reflected in the 1990 annualized costs -- the remainder of the costs are spread through the following two decades, which fall outside of the scope of this study (similarly, benefits arising from emission reductions in, e.g., 1995 caused by 1990 capital investments are not captured by the benefits analysis).

Table 34. Compliance Expenditures and Annualized Costs, 1973 to 1990 ($1990 millions).

		Annualized Costs		
Year	Expend.	at 3%	at 5%	at 7%
1973	19,635	10,957	11,042	11,134
1974	21,405	13,231	13,435	13,655
1975	24,425	13,314	13,638	13,988
1976	24,139	14,123	14,611	15,139
1977	24,062	15,253	15,904	16,608
1978	22,593	14,963	15,776	16,653
1979	24,837	17,309	18,282	19,331
1980	25,741	19,666	20,812	22,046
1981	24,367	19,590	20,905	22,321
1982	21,555	18,643	20,125	21,720
1983	20,148	19,095	20,734	22,498
1984	21,560	20,133	21,909	23,819
1985	22,903	22,516	24,447	26,523
1986	20,831	21,109	23,161	25,364
1987	20,615	22,072	24,237	26,562
1988	19,805	22,012	24,281	26,719
1989	19,817	22,916	25,288	27,836
1990	19,019	23,598	26,066	28,717

The stream of costs from 1973 to 1990 can be expressed as a single cost number by *discounting* all costs to a common year. In this analysis, all costs and benefits are discounted to 1990 (in addition, all costs and benefits are converted to 1990-value dollars, removing the effects of price inflation). There is a broad range of opinion in the economics profession regarding the appropriate discount rate to use in analyses such as this. Some economists believe that the appropriate rate is one that approximates the social rate of time preference -- three percent, for example (all rates used here are "real", i.e., net of price inflation impacts). Others believe that a rate that approximates the opportunity cost of capital (e.g., seven percent or greater) should be used. A third school of thought holds that some combination of the social rate of time preference and the opportunity cost of capital is appropriate, with the combination effected either by use of an intermediate rate or by use of a multiple-step procedure which uses the social rate of time preference as the "discount rate," but still accounts for the cost of capital. The 812 Project Team determined to use a range of discount rates (three, five, and seven percent) for the analysis.

Expenditures and annualized costs discounted to 1990 are found on Table 5,35. Expenditures are discounted at all three rates; annualized costs are discounted at the rate corresponding to that used in the annualization procedure (i.e., the "annualized at 3%" cost stream is discounted to 1990 at three percent). The final row presents the result of an explicit combination of two rates: Capital costs are annualized at seven percent, then the entire cost stream is discounted to 1990 at three percent.

Table 35. Costs Discounted to 1990 ($1990 millions)

	3%	5%	7%
Expenditures	520,475	627,621	760,751
Annualized Costs	416,804	522,906	657,003
Annualized at 7%	476,329		

Indirect Effects of the CAA

In addition to imposing direct compliance costs on the economy, the CAA induced indirect economic effects, primarily by changing the size and composition of consumption and investment flows. Although this analysis does not add these indirect effects to the direct costs and include them in the comparison to benefits, they are important to note. This

section summarizes the most important effects of the CAA, as estimated by the J/W macroeconomic simulation.

Table 36. Differences in Gross National Product Between the Control and No-control Scenarios.

Year	Nominal % Change	Real % Change
1973	-0.09	0.26
1974	-0.18	0.27
1975	-0.10	0.44
1976	-0.00	0.49
1977	-0.10	0.54
1978	-0.16	0.56
1979	-0.16	0.63
1980	-0.14	0.69
1981	-0.14	0.73
1982	-0.19	0.74
1983	-0.19	0.78
1984	-0.17	0.84
1985	-0.12	0.95
1986	-0.14	0.98
1987	-0.15	1.01
1988	-0.20	1.00
1989	-0.21	0.99
1990	-0.18	1.00

GNP and Personal Consumption

Under the no-control scenario, the level of GNP increases by one percent in 1990 relative to the control case (see Table 36). During the period 1973 to 1990, the percent change in real GNP rises monotonically from 0.26 percent to 1.0 percent. The increase in the level of GNP is attributable to a rapid accumulation of capital, which is driven by changes in the price of investment goods. The capital accumulation effect is augmented by a decline in energy prices relative to the base case. Lower energy prices that correspond to a world with no CAA regulations decreases costs and increases real household income, thus increasing consumption.

Removing the pollution control component of new capital is equivalent to lowering the marginal price of investment goods. Combining this with the windfall gain of not having to bring existing capital into compliance leads to an initial surge in the economy's rate of return, raising the level of real investment. The investment effects are summarized in Figure 24, which is found at the end of this appendix. More rapid (ordinary) capital accumulation leads to a decline in the rental price of capital services which, in turn, stimulates the demand for capital services by producers *and* consumers. The capital rental price reductions also serve to lower the prices of goods and services and, so, the overall price level. Obviously, the more capital intensive sectors exhibit larger price reductions.[73] The price effects from investment changes are compounded by the cost reductions associated with releasing resources from the operation and maintenance of pollution control equipment and by the elimination of higher prices due to regulations on mobile sources.

To households, no-control scenario conditions are manifest as an increase in permanent future real earnings which supports an increase in real consumption in all periods and, generally, an increase in the demand for leisure (see Table 38). Households marginally reduce their offer of labor services as the income effects of higher real earnings dominate the substitution effects of lower goods prices. The increase in consumption is dampened by an increase in the rate of return that produces greater investment (and personal savings).

[73] Not surprisingly, at the industry level, the principal beneficiaries in the long run of eliminating the costs associated with air pollution abatement are the most heavily regulated industries. The largest changes in industry prices and outputs occur in the motor vehicles industry. Other industries that benefit significantly from the elimination of environmental controls are refined petroleum products, electric utilities, and other transportation equipment. Turning to manufacturing industries, metal mining and the primary metals have the largest gains in output from elimination of air pollution controls.

Finally, technical change is a very important aspect of the supply-side adjustments under the no-control scenario. Lower factor prices increase the endogenous rates of technical change in those industries that are factor-using. Lower rental prices for capital benefit the capital-using sectors, lower materials prices benefit the materials-using sectors, and lower energy prices benefit the energy-using sectors. On balance, a significant portion of the increase in economic growth is attributable to accelerated productivity growth. Under the no-control scenario, economic growth averages 0.05 percentage points higher over the interval 1973 to 1990. The increased availability of capital accounts for 60 percent of this increase while faster productivity growth accounts for the remaining 40 percent. Thus, the principal effect arising from the costs associated with CAA initiatives is to slow the economy's rates of capital accumulation and productivity growth.

As with the cost and expenditure data presented above, it is possible to present the stream of GNP and consumption changes as single values by discounting the streams to a single year. Table 37 summarizes the results of the discounting procedure, and also includes discounted expenditure and annualized cost data for reference. Accumulated (and discounted to 1990) losses to GNP over the 1973 to 1990 period were half again as large as expenditures during the same period, and approximately twice as large as annualized costs. Losses in household consumption were approximately as great as annualized costs.

Table 37. GNP and Consumption Impacts Discounted to 1990 ($1990 billions)

	3%	5%	7%
Expenditures	520	628	761
Annualized Costs	417	523	657
GNP	880	1005	1151
Household Consumption	500	569	653
HH and Gov't Consumption	676	769	881

Source: Expenditures and annualized costs from above; macroeconomic impacts from Jorgenson et al. (1993), Table 4.1

Although they have value as descriptors of the magnitude of changes in economic activity, neither GNP nor consumption changes are perfect measures of changes in social welfare. A better measure is Equivalent Variations (EVs), which measure the change in income that is equivalent to the change in (lifetime) welfare due to removal of the CAA. As part of its macroeconomic exercise, EPA measured the EVs associated with removal of the CAA. Elimination of CAA compliance costs (disregarding benefits) represents a welfare gain of $493 billion to $621 billion, depending on assumptions used in the analysis.[74] This result does not differ greatly from the range of results represented by expenditures, anualized costs, and consumption changes.

Prices

One principal consequence of the Clean Air Act is that it changes prices. The largest price reductions accrue to the most heavily regulated industries which are the large energy producers and consumers (see Table 39). But these are also the most capital intensive sectors and it is the investment effects that are the dominant influences in altering the course of the economy. Focusing on energy prices, under the no-control scenario the price of coal in 1990 declines by 1.3 percent, refined petroleum declines by 3.03 percent, electricity from electric utilities declines by 2.75 percent, and the price of natural gas from gas utilities declines by 1.2 percent. The declining price of fossil fuels induces substitution toward fossil fuel energy sources and toward energy in general. Total Btu consumption also increases.

[74] Jorgenson et al., 1993.

Sectoral Effects: Changes in Prices and Output by Industry

At the commodity level, the effect of the CAA varies considerably. Figure 25, found at the end of this appendix, shows the changes in the supply price of the 35 commodities measured as changes between the no-control case and the control-case for 1990.

In 1990, the largest change occurs in the price of motor vehicles (commodity 24), which declines by 3.8 percent in the no-control case. Other prices showing significant effects are those for refined petroleum products (commodity 16) which declines by 3.0 percent, and electricity (commodity 30) which declines 2.7 percent. Eight of the remaining industries have decreases in prices of 1.0 to 1.4 percent under the no-control scenario. The rest are largely unaffected by environmental regulations, exhibiting price decreases between 0.3 and 0.8 percent.

To assess the intertemporal consequences of the CAA, consider the model's dynamic results and the adjustment of prices between 1975 and 1990. Initially, in 1975, the biggest effect is on the price of output from petroleum refining (sector 16), which declines by 4.3 percent. But by 1990, the price of petroleum refining is about 3.0 percent below control scenario levels. In contrast, the price of motor vehicles (sector 24) is about 2.4 percent below baseline levels in 1975, but falls to about 3.8 percent below baseline levels in 1990.

Table 38. Difference in Personal Consumption Between the Control and No-Control Scenarios.

Year	Nominal % Change	Real % Change
1973	-0.02	0.33
1974	-0.01	0.43
1975	-0.10	0.24
1976	-0.10	0.39
1977	-0.10	0.54
1978	-0.09	0.63
1979	-0.11	0.68
1980	-0.12	0.71
1981	-0.13	0.74
1982	-0.12	0.81
1983	-0.13	0.85
1984	-0.15	0.86
1985	-0.19	0.88
1986	-0.19	0.94
1987	-0.19	0.98
1988	-0.17	1.03
1989	-0.17	1.04
1990	-0.18	1.01

The price changes affect commodity demands, which in turn determine how industry outputs are affected. Figure 26, found at the end of this appendix, shows percentage changes in quantities produced by the 35 industries for the years 1975, 1980, 1985 and 1990. As noted earlier, the principal beneficiaries under the no-control scenario are the most heavily regulated industries: motor vehicles, petroleum refining, and electric utilities.

In 1990, the motor vehicle sector (sector 24) shows the largest change in output, partly due to the fact that the demand for motor vehicles is price elastic. Recall that the largest increase in prices also occurred in the motor vehicles sector. The 3.8 percent reduction in prices produces an increase in output of 5.3 percent relative to the base case.

Significant output effects are also seen in the petroleum refining sector (sector 16) with a 3.2 percent increase, in electricity (sector 30) with a 3.0 percent increase, and in other transportation equipment (sector 25) with a 1.6 percent increase. The large gains in output for these industries are mostly due to the decline in their prices. In manufacturing, the sectors exhibiting the most significant output effects are metal mining (sector 2) with a 2.0 percent increase, and primary metals (sector 20) with a 1.8 percent increase. Twenty of the remaining industries exhibit increase in output of less than 0.9 percent after pollution controls are removed.

While most sectors increase output under the no-control scenario, a few sectors decline in size in the absence of air pollution controls. The most notable of these are food and kindred products (sector 7) which decline by 0.5 percent, furniture and fixtures (sector 12) which decline by 0.6 percent, and rubber and plastic products (sector 17) which decline by 0.3 percent. These sectors are among the least capital intensive, so the fall in the rental price of capital services has little effect on the prices of outputs. Buyers of the commodities produced by these industries face higher relative prices and substitute other commodities in both intermediate and final demand. The rest of the sectors are largely unaffected by environmental regulations.

Table 39. Percentage Difference in Energy Prices Between the Control and No-control Scenarios.

Year	Coal	Refined Petroleum	Electric Utilities	Gas Utilities
1973	-0.44	-5.99	-2.11	-0.32
1974	-0.47	-4.84	-2.53	-0.44
1975	-0.42	-4.28	-2.19	-0.31
1976	-0.57	-3.83	-2.12	-0.44
1977	-0.74	-3.43	-2.22	-0.59
1978	-0.86	-3.28	-2.39	-0.68
1979	-0.91	-2.92	-2.81	-0.71
1980	-0.94	-2.76	-2.97	-0.69
1981	-0.97	-2.50	-2.76	-0.71
1982	-0.98	-2.42	-2.63	-0.77
1983	-1.09	-2.35	-2.58	-0.85
1984	-1.12	-2.26	-2.49	-0.91
1985	-1.21	-2.89	-2.62	-0.97
1986	-1.27	-3.35	-2.69	-1.12
1987	-1.31	-3.50	-2.78	-1.18
1988	-1.30	-3.61	-2.75	-1.19
1989	-1.31	-3.45	-2.74	-1.19
1990	-1.30	-3.03	-2.75	-1.20

Changes in Employment Across Industries

The effect of the CAA on employment presents a much more complicated picture. Although Jorgenson-Wilcoxen is a full-employment model and cannot be used to simulate unemployment effects, it is useful for gaining insights about changes in the patterns of employment across industries. Percentage changes in employment by sector for 1990 are presented in Figure 27, which is located at the end of this appendix.

For 1990, the most significant changes in the level of employment relative to the control scenario occur in motor vehicles (sector 24) which increases 1.2 percent, other transportation equipment (sector 25) which increases 0.8 percent, electric utilities (sector 30) which increases 0.7 percent, and primary metals (sector 20) which increases 0.6 percent. The level of employment is higher relative to the control case in 10 other industries.

For a few sectors, the no-control scenario results in changes in real wages that cause *reductions* in employment. The most notable reductions in employment under the no-control scenario occur in: tobacco manufacturing (sector 8), which declines 1.2 percent; furniture and fixtures (sector 12), which declines 0.8 percent; rubber and plastic products (sector 17), which declines 0.8 percent; food and kindred products (sector 7), which declines 0.7 percent, stone; clay and glass products (sector 19), which declines 0.6 percent; and instruments (sector 26), which declines 0.6 percent. These sectors are generally those in which the level of output was lower in 1990 relative to the control scenario, since they are among the least capital intensive and the fall in the rental price of capital services has little effect on the prices of outputs. Buyers of the commodities produced by these industries face higher relative prices and substitute other commodities in both intermediate and final demand. It is interesting to note that several of the least capital intensive sectors experience insignificant employment effects in the short run (1975) under the no-control scenario, but increasingly adverse effects over the 20-year period of analysis. These include food and kindred products, furniture and fixtures, rubber and plastic products, stone, clay and glass products, and instruments.

Examination of the transition of employment in the economy from the initial equilibrium to 1990 reveals that the employment effects of the CAA on motor vehicles, transportation equipment, electric utilities, and primary metals persist over the entire period of analysis. Employment varies from: an increase of 1.7 percent in 1975 to 1.2 percent in 1990 in motor vehicles; from an increase of 0.7 in 1975 to 0.8 percent in 1990 in transportation equipment; from an increase of 1.2 percent in 1975 to 0.7 percent in 1990 in electric utilities; and from an increase of 0.8 percent in 1975 to 0.6 percent in 1990.

Figure 24. Percent Difference in Real Investment Between Control and No-control Scenarios.

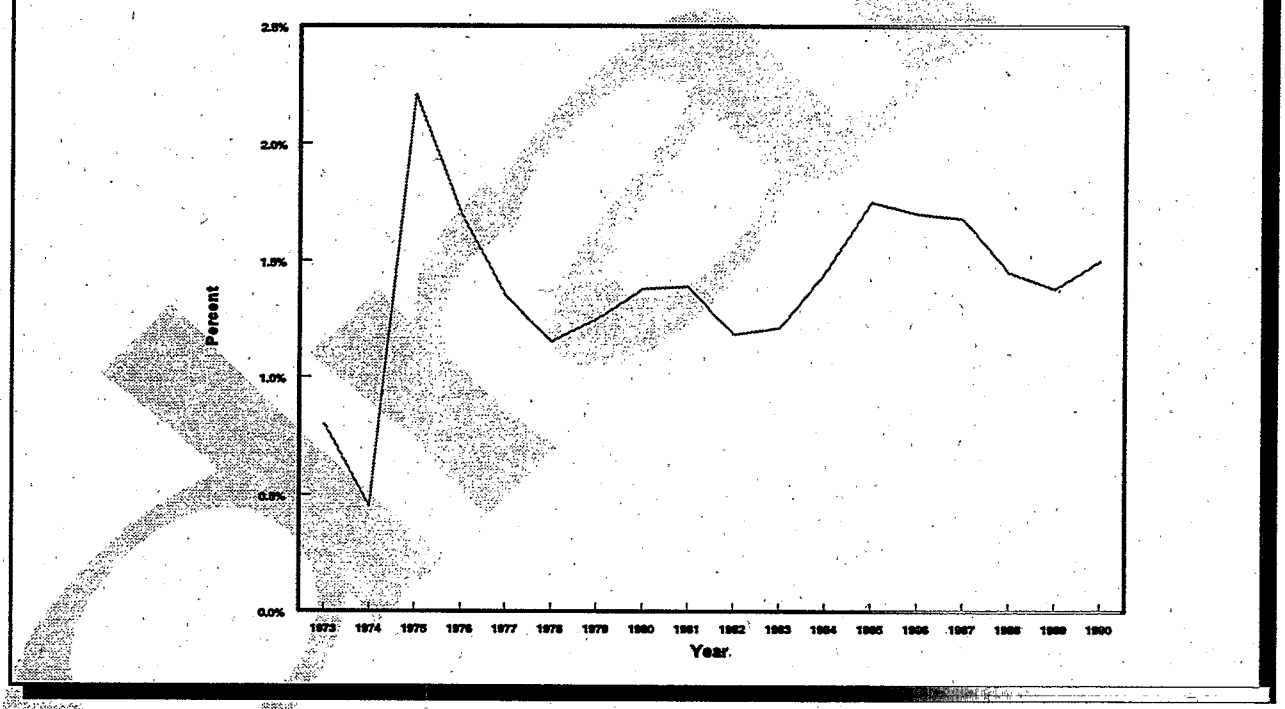


Figure 25. Percent Difference in Price of Output by Sector Between Control and No-control Scenario for 1990.

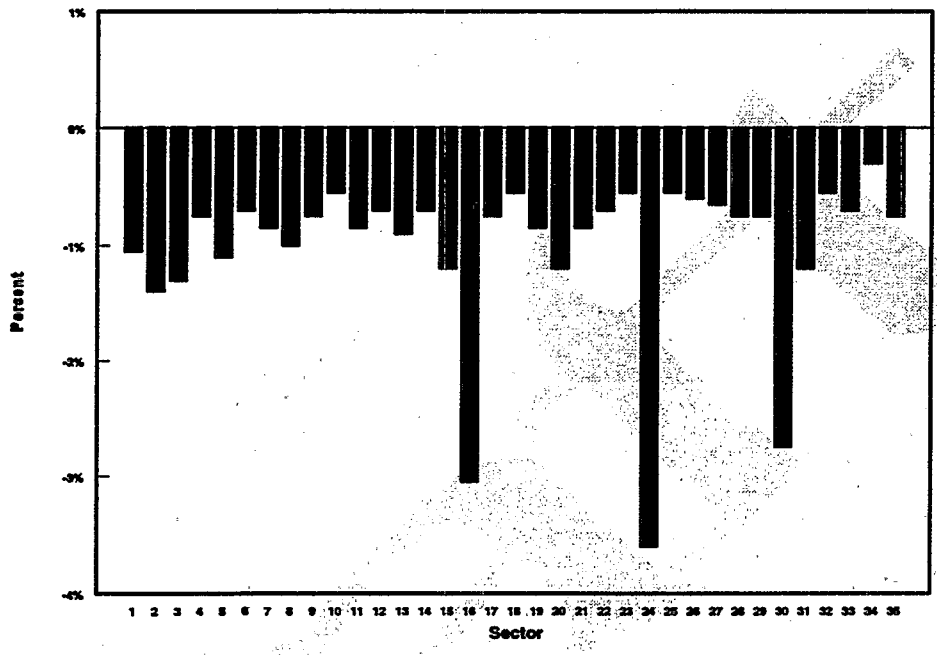


Figure 26. Percent Difference in Quantity of Output by Sector Between Control and No-control Scenario for 1990.

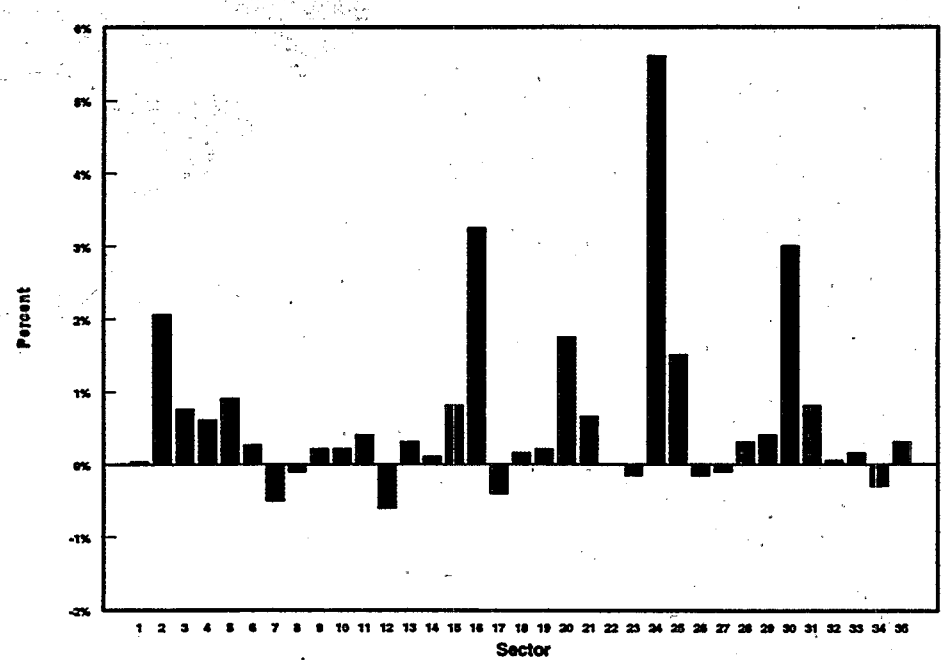
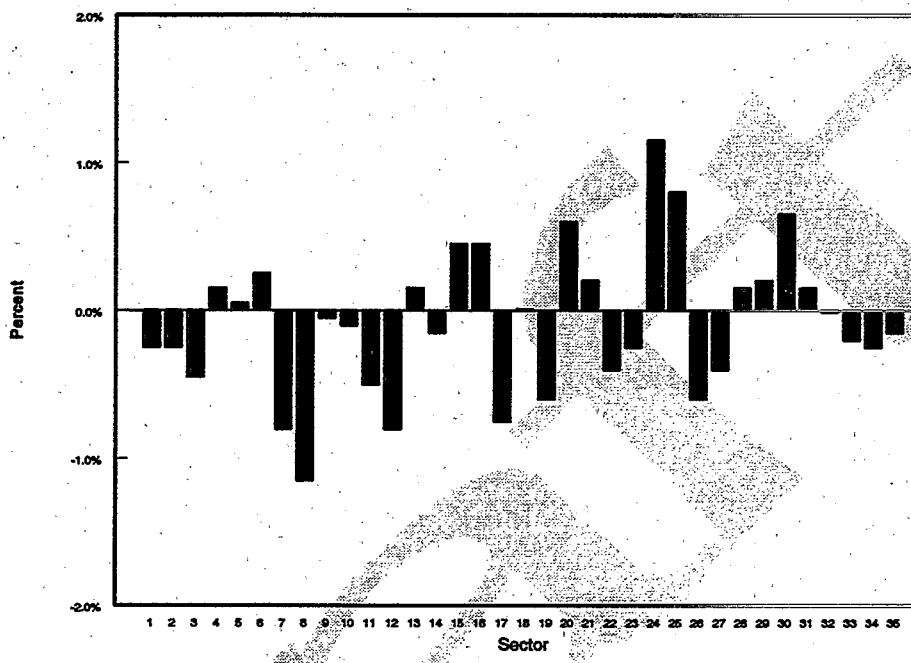


Figure 27. Percent Difference in Employment by Sector Between Control and No-control Scenario for 1990.



Uncertainties in the Cost Analysis

Potential Sources of Error in the Cost Data

Because of the importance of the *Cost of Clean* data for this assessment, the project team investigated potential sources of error due to the use of industry's self-reported costs of compliance with air pollution abatement requirements. Concerns about the accuracy of responses include (1) misreporting by firms in response to federal agency surveys, and (2) omission of important categories of compliance cost from the data collected or reported by these federal agencies.[75] Table 40 contains a summary of the results of the analysis.

[75] Memorandum from Industrial Economics, Incorporated to Jim DeMocker (EPA/OAR) dated 10/16/91 and entitled "Sources of Error in Reported Costs of Compliance with Air Pollution Abatement Requirements."

Table 40. Potential Sources of Error and Their Effect on Total Costs of Compliance.

Source of Error	Effect on Capital Costs	Effect on O&M Costs
Lack of Data at Firm Level	Under-reported Percent Unknown	Under-reported Percent Unknown
Misallocation of Costs:		
Inclusion of OSHA and Other Regulatory Costs	Over-reported Percent Unknown	Over-reported Percent Unknown
Exclusion of Solid Waste Disposal Costs Related to Air Pollution Abatement	—	Under-reported Percent Unknown
Exclusion of Costs:		
Exclusion of Private R&D Expenses	—	Under-reported by 14 to 17%. (varies by year)
Exclusion of Energy Use by Pollution Abatement Devices[a]	—	Under-reported by 1 to 3% (varies by year)
Exclusion of Depreciation Expenses[a]	—	Under-reported by 1 to 2% (varies by year)
Exclusion of Recovered Costs	—	Over-reported by 1% Plus
Omission of Small Firms	Under-reported by 1 to 2%	Under-reported by 1 to 2%
NET EFFECT	Under-reported	Under-reported

[a] Energy outlays *are* part of the data on O&M costs and depreciation expenses *are not*. Accordingly, in the J/W model, energy outlays are considered along with other operating expenditures in terms of their impacts on unit costs. Depreciation is represented fully in the capital accumulation process, as the undepreciated capital stock at the beginning of any period gives rise to the flow of capital services available to producers and consumers.

Source: Industrial Economics, Incorporated, memorandum to Jim DeMocker, EPA/OAR, "Sources of Error in Reported Costs of Compliance with Air Pollution Abatement Requirements," October 16, 1991.

The major finding from this analysis indicates that total O&M costs are likely to be under-reported due to exclusion of private research and development (R&D) expenditures. Note, however, that although these costs were excluded from those used for the macroeconomic modeling, they were included in the overall direct cost estimate of the CAA; see "Other Direct Costs," above. These costs are excluded from the macromodeling because they cannot be disaggregated by industry and, more importantly, because there is no information on what was purchased or obtained as a result of these expenditures.

Based on the need indicated by the IEc review, modifications to the BEA data were made to remedy some of the biases noted above. In particular, recovered costs for stationary source air pollution, e.g. sulfur

removed using scrubbers that is then sold in the chemical market, have been accounted for in the data set used in the model runs.

Mobile Source Costs

For the Section 812 analysis, EPA used the best available information on the estimated cost of mobile source air pollution control. Several other sources of cost estimates exist, however, including a cost series produced by the Department of Commerce Bureau of Economic Analysis (BEA), which is summarized in Table 41. The BEA estimates differ significantly from EPA estimates, particularly with respect to estimates of capital costs and the "fuel price penalty" associated with the use of unleaded gasoline.

Table 41. BEA Estimates of Mobile Source Costs.

Year	Capital Exp.	Net I&M*	Fuel Price Penalty	Fuel Economy Penalty
1973	1,013	1,104		697
1974	1,118	1,380	5	1,180
1975	2,131	1,520	97	1,344
1976	2,802	1,420	309	1,363
1977	3,371	1,289	701	1,408
1978	3,935	1,136	1,209	1,397
1979	4,634	931	1,636	1,792
1980	5,563	726	2,217	2,320
1981	7,529	552	2,996	2,252
1982	7,663	409	3,518	1,876
1983	9,526	274	4,235	1,582
1984	11,900	118	4,427	1,370
1985	13,210	165	4,995	1,133
1986	14,368	(331)	4,522	895
1987	13,725	(453)	3,672	658
1988	16,157	(631)	3,736	420
1989	15,340	(271)	1,972	183
1990	14,521	(719)	1,370	(55)

* Inspection and maintenance costs less fuel density savings and maintenance savings.

EPA's capital cost estimates are based on estimates of the cost of equipment required by mobile source regulations. BEA's estimates are based on survey data from the Bureau of Labor Statistics (BLS) that measures the increase in the per-automobile cost (relative to the previous model year) due to pollution control and fuel economy changes for that model year. The difference in approach is significant: BEA's annual capital cost estimates exceed EPA's by a factor of (roughly) two. EPA may underestimate costs to the extent that engineering cost estimates of components exclude design and development costs for those components. The BLS estimates add the incremental annual costs to all past costs to derive total current-year costs. Such an approach overestimates costs to the extent that it fails to account for cost savings due to changes in component mixes over time.

Some mobile source pollution control devices required the use of unleaded fuel. Unleaded gasoline is more costly to produce than is leaded gasoline, and generally has a greater retail price, thus imposing a cost on consumers. EPA estimated the "fuel price penalty" by using a petroleum refinery cost model to determine the expected difference in production cost between leaded and unleaded gasoline. BEA's "fuel price penalty" was the difference between the retail price of unleaded gasoline and that of leaded gasoline.

A detailed description of the data sources, analytic methods, and assumptions that underlie the EPA and BEA mobile source cost estimates can be found in McConnell (1995).

Endogenous Productivity Growth in the Macro Model

For each industry in the simulation, the JW model separates price-induced changes in factor use from changes resulting strictly from technical change. Thus, simulated productivity growth for each industry has two components: (a) an exogenous component that varies over time, and (b) an endogenous component that varies with policy changes. Some reviewers have noted that, although not incorrect, use of endogenous productivity growth is uncommon in the economic growth literature. EPA conducted a sensitivity run of the J-W model, setting endogenous growth parameters to zero (i.e., removing endogenous productivity growth from the model).[76]

Endogenous productivity growth is an important factor in the JW model. For example, for the period 1973 to 1990, removal of the endogenous productivity growth assumptions reduces household income by 2.9%-3.0% (depending on whether one uses a world with CAA or one without CAA as the baseline). In comparison, removal of CAA compliance costs results in a 0.6%-0.7% change in household income (depending on whether one uses, as a baseline, a world with or one without endgenous productivity growth). That is, use of the endogenous productivity growth assumption has four to five times the impact of that of CAA compliance costs.

Although very important to the simulated growth of the economy within any policy setting, the endogenous productivity growth assumption is less important across policy settings. Under the base (i.e., "with endogenous productivity growth") scenario, the aggregate welfare effect (measured as EVs, see above) of CAA compliance costs and indirect effects is estimated to be 493 billion to 621 billion dollars (in 1990-value dollars). If one removes the endogenous productivity growth assumption, the aggregate welfare effect declines to the range 391 billion to 494 billion dollars (in 1990-value dollars) [Jorgenson et al., 1993, pg. 6-15], a reduction of about 20 percent.

Stationary Source Cost Estimate Revisions

As noted above on page 83, the costs used for stationary sources in the macro-modeling (and retained in this cost analysis) were projected for several years in the late 1980s. Since that time, BEA has released historical expenditure estimates for those years based on survey data. A comparison of the expenditure series can be found in Table 42. Apparently, EPA's projections overestimated stationary source compliance expenditures by approximately $2 billion per year for the period 1987 to 1990. Since expenditures from all sources are estimated to be $18 billion -$19 billion (current dollars) per year during 1987 to 1990, this implies that EPA has overestimated compliance expenditures by more than ten

Table 42. Comparison of EPA and BEA Stationary Source Expenditure Estimates (millions of current dollars)

	Private sector		Gov't. Enterprise		Total
Year	capital	O&M	capital	O&M	Expend.
	EPA Estimates				
1986	4,090	7,116	312	140	11,658
1987	4,179	7,469	277	130	12,055
1988	4,267	7,313	243	161	11,984
1989	4,760	7,743	235	173	12,911
1990	4,169	8,688	226	154	13,237
	BEA Estimates				
1986	4,090	7,072	312	182	11,656
1987	3,482	5,843	246	141	9,712
1988	3,120	6,230	121	161	9,632
1989	3,266	6,292	229	152	9,939
1990	4,102	6,799	200	154	11,255

"Recovered Costs" are not included in this table.
Sources for "BEA Estimates": for 1986, "Pollution Abatement and Control Expenditures," Survey of Current Business (BEA) June 1989, Table 7; for 1987-90, BEA May 1995, Table 8.

[76] For greater detail, see Jorgenson et al., 1993.

percent during this period. Although a substantial overstatement for those years, the $2 billion per year overestimate would have little impact (probably less than two percent) on the discounted present value in 1990 dollars of the 1973 to 1990 expenditure stream.

Amortization Period for Stationary Source Plant and Equipment

In developing annualized costs, stationary source capital expenditues were amortized over a twenty-year period. That is, it was assumed that plant and equipment would depreciate over twenty years. It is possible that stationary source plant and equipment has, on average, a useful lifetime significantly greater than twenty years. The Project Team tested the sensitivity of the cost analysis results to changes in stationary source capital amortization periods.

Table 43 presents total annualized compliance costs assuming a 40-year amortization period for stationary source capital expenditures (all other cost components are unchanged from the base analysis). All costs are in 1990-value dollars, ad three alternative discount rates are used in the annualization period. Table 44 presents the results discounted to 1990, and compared to the base case results (i.e., using a twenty-year amortization period). Doubling the amortization period to 40 years decreases the 1990 present value of the 1973 to 1990 cost stream by approximately 40 billion dollars. This represents a change of six percent to nine percent, depending on the discount rate employed.

Table 43. Annualized Costs Assuming 40-Year Stationary Source Capital Amortization Period, 1973 to 1990 ($1990 millions).

	Annualized Costs		
Year	at 3%	at 5%	at 7%
1973	10,801	10,899	11,008
1974	12,875	13,108	13,366
1975	12,751	13,121	13,532
1976	13,338	13,891	14,504
1977	14,263	14,996	15,807
1978	13,778	14,690	15,695
1979	15,936	17,024	18,220
1980	18,091	19,368	20,771
1981	17,809	19,272	20,880
1982	16,670	18,316	20,123
1983	16,941	18,759	20,754
1984	17,836	19,803	21,960
1985	20,079	22,213	24,551
1986	18,544	20,809	23,288
1987	19,384	21,772	24,387
1988	19,203	21,706	24,446
1989	19,989	22,604	25,467
1990	20,546	23,268	26,247

Table 44. Effect of Amortization Periods on Annualized Costs Discounted to 1990 (billions of $1990)

	Discount rate		
	3%	5%	7%
20-yr amortization period	417	523	657
40-yr amortization period	379	483	617

Cost and Macroeconomic Modeling References

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Kokoski, Mary F., and V. Kerry Smith, "A General Equilibrium Analysis of Partial-Equilibrium Welfare Measures: The Case of Climate Change," *American Economic Review*, Vol. 77, No. 3, June 1987, 331-341.

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Verleger, Philip K., Jr., "Clean Air Regulation and the L.A. Riots," *The Wall Street Journal*, Tuesday, May 19, 1992, A14.

Wilcoxen, Peter J., *The Effects of Environmental Regulation and Energy Prices on U.S. Economic Performance*, Doctoral thesis presented to the Department of Economics at Harvard University, Cambridge, MA, December 1988.

Appendix B: Emissions Modeling

This appendix provides additional detail regarding the methodologies used to estimate control and no-control scenario emissions from each of the six principal emission sectors: industrial combustion, industrial processes, electric utilities, on-highway vehicles, off-highway vehicles, and commercial/residential sources. This appendix is adapted from the draft report "The Impact of the Clean Air Act on 1970 to 1990 Emissions; Section 812 Retrospective Analysis," March 1, 1995 by Pechan Associates, which distills the methodologies and results associated with the emission modeling efforts by Argonne National Laboratory (ANL), ICF Resources Incorporated (ICF), Abt Associates (Abt), and the Environmental Law Institute (ELI).

The control scenario emission results are similar, but not identical, to official EPA historical emission estimates provided by the EPA National Air Pollutant Emission Trends Reports.[77] Comparisons between the current estimates and the *Trends* data for SO_2, NO_x, VOC, and CO are presented in Figures 28, 29, 31, and 32, respectively. No comparison between the particulate matter (PM) emission estimates developed for this analysis and the estimates generated for the EPA *Trends* Report are presented, however, since *Trends* estimates are presented for only PM_{10}, a subset of the TSP measures developed for this analysis. More detailed tables providing emission estimates by sector and by target year for TSP, SO_2, NO_x, VOC, CO, and Lead are presented in Tables 61, 62, 63, 64, 65, and 66, respectively, at the end of this appendix.

Figure 28. Comparison of Control, No-control, and *Trends* SO_x Emission Estimates.

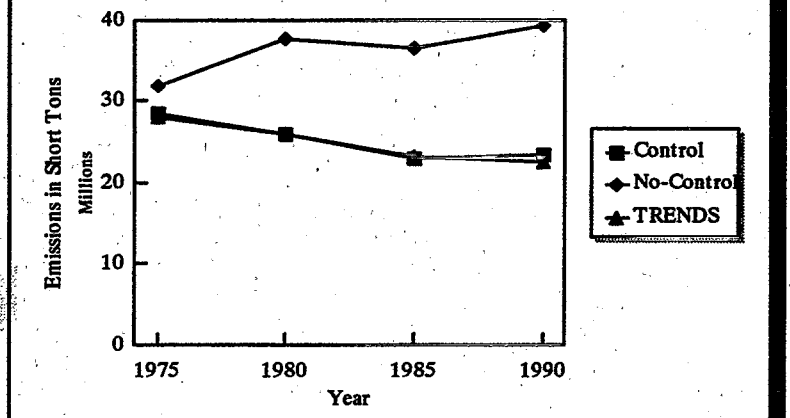
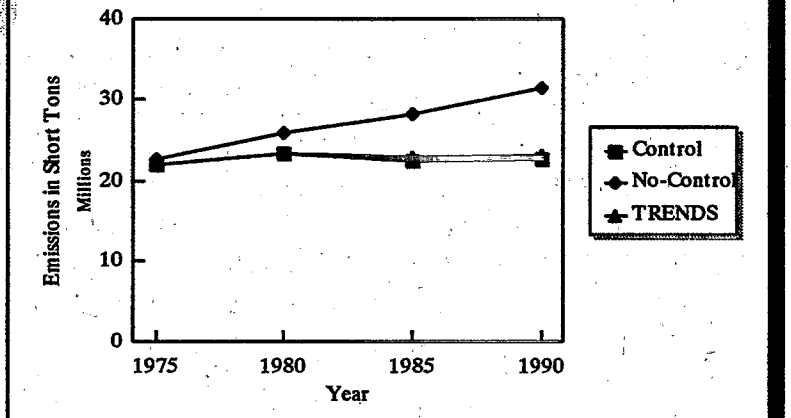


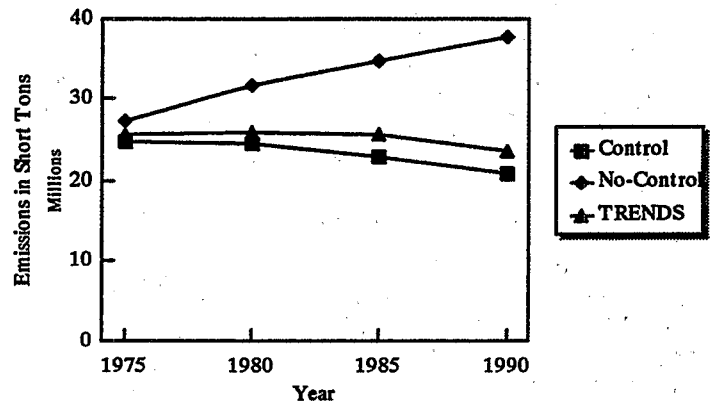
Figure 29. Comparison of Control, No-control, and *Trends* NO_x Emission Estimates.



[77] EPA/OAQPS, "National Air Pollutant Emission Trends 1900 - 1994," EPA-454/R-95-011, October 1995.

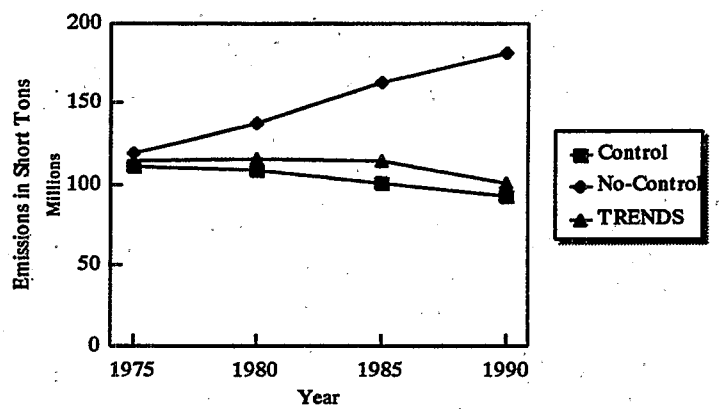
Though the EPA *Trends* and the present study emission profiles are similar to each other, they should not be expected to match precisely. This is because the emission estimates developed for the present study are based on modeled macroeconomic and emission sector conditions. Even though the macroeconomic and sector models themselves are constructed and calibrated using historical data, modeled replications of historical trends would not be expected to precisely capture actual historical events and conditions which affect emissions. Relying on modeled historical scenarios is considered reasonable for the present analysis since its purpose is to estimate the differences between conditions with and without the CAA. Comparing actual historical emissions with modeled no-control emissions would lead to an inconsistent basis for comparisons between scenarios. Using models for both scenarios allows potential model biases to essentially cancel out.

Figure 30. Comparison of Control, No-control, and *Trends* VOC Emission Estimates.



In general, however, these comparisons show close correspondence between control scenario and *Trends* estimates with the largest differences occurring for VOC and CO emissions. The *Trends* report VOC estimates are generally higher than the control scenario estimates due to the inclusion of Waste Disposal and Recycling as a VOC source in the *Trends* report. This inconsistency is of no consequence since Waste Disposal and Recycling sources were essentially uncontrolled by the historical CAA and therefore do not appear as a difference between the control and no-control scenarios. The higher CO emission estimates in the *Trends* Report are primarily associated with higher off-highway vehicle emissions estimates. Again, since off-highway emissions do not change between the control and no-control scenario in the present analysis, this inconsistency is of no consequence.

Figure 31. Comparison of Control, No-control, and *Trends* CO Emission Estimates.



The following sections of this appendix summarize the methodologies used to model control and no-control scenario emissions for each of the six major emission sectors. Additional details can be found in the supporting documents listed in the References section of this appendix.

Industrial Boilers and Processes

For the purposes of the retrospective analysis, the industrial sector was divided into two components: (1) boilers; and (2) industrial processes and process heaters. The factors affecting emissions from these two source types are different, and, as a result, separate methods were used to calculate control and no-control scenario emissions in each of the target years. To analyze the change in emissions from industrial boilers, ANL used the ICE model (Hogan, 1988). This model was developed under the auspices of NAPAP to forecast State-level fuel choice and emissions from conventional, steam raising, industrial boilers. For the retrospective analysis of industrial processes and fuel use emissions from process heaters, ELI used the EPA *Trends* methods and the ANL MSCET data base (EPA, 1991; Kohout, 1990). The *Trends* report contains estimates of national emissions for a variety of industrial sources for the time period of interest. The MSCET data base provided the spatial distribution used to calculate State-level emissions.

The distinction between industrial boilers and non-boiler industrial processes was necessitated by the structure of the CAA regulations and by the factors affecting emission levels from these two source types. Boilers are regulated differently from processes and process heaters. Emissions from industrial processes are primarily a function of levels of industrial activity. The emissions from fuel combustion, however, are a function of energy use and fuel choice as well as industrial activity. Fossil fuel emissions in the absence of the CAA are not proportional to industrial output, since the level of energy use is a decision variable for the firm in its production process. Therefore, in the ICE model simulations used to estimate no-control scenario boiler emissions, the level (and type) of energy use were determined first, and then the effects of emission regulation were taken into account.

Overview of Approach

Industrial Boilers

ICE model inputs include fuel prices, total fossil boiler fuel demand by industry type, and environmental control costs. The outputs of the ICE model were SO_2, NO_x, and TSP emissions by State, industry, and boiler size class. The model runs in 5-year increments and has a current base year of 1985.

The boiler demand input data were required at the State level. The following seven industry types were included in the ICE model: Standard Industrial Classification (SIC) codes 20, 22, 26, 28, 29, 33, and "other manufacturing." ANL's approach assumed that industrial boiler fuel use occurs only in the manufacturing sector. Fuel price data in each of the target years was required at the Federal Region level. Prices by grade of coal and petroleum product, such as sulfur content and heating value, were used by the model. These prices were used to determine the cost of compliance and to determine emissions when the regulations are not binding.

Control costs were computed by engineering subroutines in the model. These costs were used by the ICE model's fuel choice component to determine the effect of CAA-related costs on the market share of a particular fuel. This fuel choice decision only applies to new industrial boilers, since the cost of existing emission controls are not in the ICE data base and fuel choice is not re-evaluated for existing boilers.

Industrial Processes and In-Process Fuel Combustion

The calculation of historical emissions from industrial processes uses EPA *Trends* methods to estimate national emissions for the analysis years, then allocates these emissions to States using the State shares from the MSCET data base.

MSCET uses a variety of methods to estimate historical emissions for the various industrial sectors. For industrial process emissions, MSCET is based on historical data on industrial activity to allocate emissions based on the State level distribution of the polluting activities. The State level distribution and benchmark is based on the 1985 NAPAP Inventory (EPA, 1989). This approach implies that the MSCET data corresponds directly to the 1985 NAPAP Inventory, and that, for any State, the sum of the emissions from Source Classification Codes (SCCs) that comprise the MSCET industry sector are equal to the MSCET data for that State and sector. Data from *Trends* are used by MSCET to provide information on changes in the aggregate level of control for years other than the 1985 benchmark. Since no direct correspondence existed between the *Trends* data and MSCET, a relationship was developed to link MSCET sectors to *Trends* industry categories to industry categories in the J/W model, which was used to change activity levels for the no-control scenario.

Table 45 shows the relationship between the sector definition used by MSCET, *Trends*, and the J/W model. The mapping from MSCET to J/W and *Trends* is used to provide the changes in aggregate activity and emission control for the calculation of no-control scenario emissions.

Establishment of Control Scenario Emissions

Energy use and corresponding emissions were broken down between boilers and non-boiler industrial processes. The latter category includes furnaces, kilns, internal combustion engines (e.g., compressors), and other non-steam types of process heat. The focus of this analysis is on boiler emissions, which were subject to increasingly stringent regulations over the 1970 to 1990 period. (Emissions from some types of industrial processes were also regulated, but regulation of non-boiler sources was targeted on the emissions from the industrial process itself, not on its fuel combustion) For this study, ANL assumed that only boiler fuel use is affected by emission regulations. The non-steam boiler portion of industrial fuel use is not directly affected by the CAA. This portion of the emissions may be affected indirectly by changes in industry activity level and fuel consumption. The emissions from non-boiler industrial processes were calculated separately by ELI.

Control Scenario Boiler Emissions

Control scenario boiler SO_2, NO_x, and TSP emissions were calculated by the ICE model. The MSCET data base provided an estimate of historical emissions for total fossil fuel combustion by industry. Since MSCET does not identify the two required components of boiler and non-boiler emissions, ANL defined the residual of the ICE model control scenario and MSCET as the non-boiler or in-process fuel use emissions. For the relevant study period, MSCET provided a control scenario

Table 45. Correspondence Between Process Emissions Categories Used by MSCET, *Trends*, and J/W Industrial Sectors and Identifier Codes.

MSCET Category	MSCET Code	*Trends* Industry Category	JW Code	JW Industry Category
Food Proc. and Agric. Operations	FOODAG	Cattle Feed Lots (0211)	1	Agriculture/forestry/fisheries
		Cotton Ginning (0724)		
		Feed and Grain Milling (204)		
		Grain Elevators (4421,5153)		
Mining Operations	MINING	Metallic Ore Mining (10)	2	Metal Mining
		Coal Mining (1211)		
Oil and Gas Extraction	OILGAS	Crude Oil Production, Storage, and Transfer (1211,4463)	3	Oil & Gas Extraction
		Natural Gas Production (1311)		
Mining Operations	MINING	Crushed Stone (142)	5	Nonfuel mining
		Sand and Gravel (144)		
		Clays (145)		
		Potash/Phosphate Rock (1474,1475)		
Degreasing	DEGRS	Degreasing	NA	Manufacturing
Misc. Industrial Processes	MISIND	Adhesives		
Indus. Organic Solvent Use, Misc.	SOLV	Other Organic Solvent Use		
		Solvent Extraction Processes		
Surface Coating	SRFCT	Surface Coating	NA	Durable Goods
Misc. Industrial Processes	MISIND	Lumber and Plywood (24)	11	Lumber & Wood Products
Cement Production	CEMNT	Cement (3241)	19	Stone, Clay, & Glass Products
Glass Manufacturing	GLASS	Glass (321,322)	19	Stone, Clay, & Glass Products
Lime Manufacturing	LIME	Concrete, Lime, Gypsum (327)	19	Stone, Clay, & Glass Products
		Lime (3274)		
Mineral Products Processing	MINRL	Clay Sintering (3295)	19	Stone, Clay, & Glass Products
		Brick and Tile (3251)		
Iron and Steel Production	IRNST	Iron and Steel (3312)	20	Primary Metal Industries
		Ferroalloys (3313)		
		Iron and Steel Foundries (332)		
Other Primary Metals Smelting	OTHMET		20	Primary Metal Industries

(continued)

Table 45 (continued). Correspondence Between Process Emissions Categories Used by MSCET, *Trends*, and J/W Industrial Sectors and Identifier Codes.

MSCET Category	MSCET Code	*Trends* Industry Category	JW Code	JW Industry Category
Primary Aluminum Smelting	PALUM	Primary Aluminum (3334)	20	Primary Metal Industries
Primary Copper Smelting	PCOPR	Primary Copper (3331)	20	Primary Metal Industries
Primary Lead and Zinc Smelting	PLDZC	Primary Lead and Zinc (3332,3333)	20	Primary Metal Industries
Other Sec. Metal Smelting and Refining	SECMET	Primary Nonferrous Smelters (333)	20	Primary Metal Industries
Other Sec. Metal Smelting and Refining	SECMET	Secondary Nonferrous Smelters (334,336)	20	Primary Metal Industries
Secondary Lead Refining	SLEAD	Secondary Lead (3341)	20	Primary Metal Industries
Food Proc. and Agric. Operations	FOODAG	Food and Beverages (20)	7	Food & Kindred Products
Misc. Industrial Processes	MISIND	Textiles (22)	9	Textile Mill Products
Paper and Pulp Mills Operations	PAPER	Pulp Mills (261,262)	13	Paper & Allied Products
Misc. Industrial Processes	MISIND	Graphic Arts (27)	14	Printing & Publishing
Printing Operations	PRINT		14	Printing & Publishing
Organic Chemicals Manufacture	ORGCM	Organic Chemicals (286)	15	Chemicals & Allied Products
		Carbon Black (2895)		
Other Chemicals Manufacture	OTHCM	Ammonia (2873)	15	Chemicals & Allied Products
		Nitric Acid (2873)		
		Chemicals (28)		
		Sulfuric Acid (2819)		
Petroleum Refining	PTREF	Petroleum Refining (2911)	16	Petroleum & Coal Products
		Asphalt Paving and Roofing (295)	16	Petroleum & Coal Products
Plastics Production	PLAST	Plastics (2821,3079)	17	Rubber and plastic products
Rubber and Misc. Plastics Manufacture	RUBR	Rubber Tires (3011)	17	Rubber and plastic products

estimate of total boiler and non-boiler emissions, which was used to calculate the control scenario State-level boiler emissions based on a special run of the ICE model.[78]

In order to use ICE to model the historical emissions path, it was necessary to construct a new ICE model base year file and new user input file so that the model could begin its calculations from 1975 conditions. Construction of the base year file is discussed first. The construction was completed in two stages, using two different data sources. The user input file has several elements, including energy prices and historical boiler fuel use. The description of the construction of the control scenario user input file follows the discussion of the ICE base year file. The model base year file provided the energy use in boilers and corresponding emission control regulations (State Implementation Plans –SIPs– for example) by several categories. These categories include:

- State;
- Industry group (one of seven);
- Fuel type (natural gas, distillate or residual fuel oil, and coal);
- Boiler size class (MMBTU/hr, one of eight categories);
- Utilization rate (one of five categories); and
- Air quality control region (AQCR).

For the purposes of ANL's analysis, only the first three categories were assumed to vary. In other words, for each State, industry, and fuel type combination, the distribution of boiler size, utilization rate, and AQCR was assumed to be constant. Over time, however, changes in the aggregate composition of State, industry, and fuel type would cause corresponding changes in the aggregate composition of the other three characteristics. As mentioned previously, the current base year file was 1985. The retrospective analysis required a 1975 base year. Because of data limitations, the approach to construct a new base year was achieved in the following two steps: the construction of a 1980 interim base year file from the 1985 file, and then the construction of the 1975 file from the interim 1980 file.

Estimates of boiler fossil fuel consumption in 1980 for each State and major fuel type were provided by Hogan (Hogan, 1988). These estimates are based on the assumption that the industry mix, size, utilization, and AQCR distribution within a State are constant. Through assuming this relationship, the 1985 ICE base year was scaled to match the data for 1980, thus forming the 1980 interim base year data.

To construct the 1975 base year file, the assumption of a constant industry mix for a State and fuel type was no longer necessary, since detailed data on each industry for 1980 and 1975 were available from PURchased Heat And Power (PURHAPS) model data files (Werbos, 1983). These PURHAPS data files were derived from the Annual Survey of Manufactures: Fuels and Electric Energy Purchased for Heat and Power (DOC, 1991). The available data in these files were for *total* fuel use not *boiler* fuel use. To make use of these data, it was necessary to assume that the fraction of fuel used in boilers, for any given State and industry, remained constant from 1975 to 1980. To the extent that the fraction of boilers' heat versus process heat applications is a function of the specific industrial production process, this assumption is reasonable.

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[78] MSCET does not provide State-level estimates of TSP, while ICE does. To estimate total regional TSP from fuel combustion, the *Trends* model was employed. These national emissions estimates were allocated to the States based on the State-level shares of TSP from the NAPAP inventory.

Based on the assumption of constant boiler fuel fraction of total fuel use, the ratio of 1975 to 1980 energy use for each State, industry, and fuel type was applied to the corresponding record of the 1980 interim base year file to produce 1975 base year files.

Control Scenario Industrial Process Emissions

To estimate boiler emissions of sulfur oxides (SO_x), NO_x, and VOC from industrial processes, data from *Trends* were used. The percentage change in national emissions by *Trends* category was applied to the appropriate sector from MSCET to obtain State-level emissions. In some cases there are several categories in *Trends* that match directly with MSCET categories (see Table 45). In these cases, the *Trends* sectors were aggregated and the percentage change was computed. It was assumed that the level of control in each industry sector implied by *Trends* was uniform across States. The changes in emissions in each State are not equal to those at the national level, since the industry composition in each State varies.

Development of Economic Driver Data for the Control Scenario - Industrial Boilers and Processes

The results of the J/W model were the primary source of activity in the ICE model driver data. These results were also used by ELI to produce the national results for industrial processes from Trends. Both ICE and Trends use the forecasted change in industrial activity that results under the no-control scenario. These data were in the form of industry specific changes in energy consumption and industrial output, for boilers and industrial processes.

Economic Driver Data for Industrial Boiler Approach

Using the 1975 base year file as a starting point, the ICE model estimated fuel choice and emissions based on a user input file containing total boiler energy demand and regional energy prices. The 1975, interim 1980, and original 1985 base year files contained the required information on energy demand for each industry group and State, so the data in these three files were aggregated across fuel type, and other boiler characteristics (for example, size). These aggregated data provided the energy demand for three of the target years. Since 1990 State-level data on energy use by industry group were not available at the time of the study, the NAPAP base case forecast for the ICE model for 1990 was used to provide the demand data for this year.

The user input file for ICE also requires a price input for each target year. These prices were input by Federal Region for distillate oil, 4 grades of residual oil (by sulfur content), natural gas, and 11 grades of coal (by sulfur content and coal rank, i.e., bituminous and sub-bituminous). Prices for 1985 and 1990 were obtained from the NAPAP base case user input file. The prices for 1975 and 1980 are from U.S. Department of Energy (DOE) data on State-level industrial energy prices (DOE, 1990). Regional prices of natural gas, distillate oil, steam coal, and residual oil were constructed by aggregating expenditures across States within each region and dividing by total British thermal unit (BTU) consumption for the years 1975, 1980, and 1985. Since prices by sulfur content grade are not reported by this DOE source, ANL assumed that the sulfur premium implied by the 1985 ICE model input file was proportional to the average price.

Based on this assumption, the ratio of the regional coal and residual oil price in 1975 and 1980 to the 1985 price was applied to the 1985 price in the ICE model base case file for each grade of fuel. To provide additional consistency between the NAPAP analysis and ANL's study, the distillate oil and natural gas prices were benchmarked to the 1985 ICE model prices as well.

One possible inconsistency arises using this procedure. The residual oil and natural gas markets are closely linked, particularly for industrial customers. These markets, specifically the gas market, underwent tremendous changes over the study period. To model the effect of these structural changes on the sulfur premiums in residual oil would require a detailed oil and gas supply model that was beyond the scope of this project. Moreover, the CAA regulations themselves create the potential for sulfur premiums. This potential effect of the CAA was not captured, though, because of the assumption of proportional fuel sulfur premiums on residual fuel oil. The relationship between market driven sulfur premiums in the coal market and the CAA was given additional consideration in this analysis through the use of an explicit coal supply model.

The J/W data for industrial energy consumptions was supplied in the form of percentage change in cost shares. In order to compute the percentage change in the quantity of energy used, ANL used the following identity:

$$\ln\left(\frac{P_E \times E}{P_Q \times Q}\right) = \ln(P_E) + \ln(E) - \ln(P_Q \times Q), \text{ or} \quad (1)$$

$$\ln\left(\frac{P_E \times E}{P_Q \times Q}\right) - \ln(P_E) + \ln(P_Q \times Q) = \ln(E) \quad (2)$$

The percentage change in E is the percentage change in cost share, minus the change in price, plus the change in value of shipments. These calculations were performed for each energy type and industry sector in the J/W model. The ICE model requires total fuel use, so the fuel specific percentages were weighted by historical fuel consumption to produce an aggregate change in fuel consumption to apply to the ICE model input data files.[79]

ICE also uses energy prices to simulate boiler fuel choices. The control scenario forecasts of energy prices in ICE were adjusted based on the percentage changes in energy prices, by coal, oil and natural gas. This implicitly assumes that the oil and coal fuel sulfur premiums, by region, are proportional to the average national price. To test this assumption for the coal market, additional

[79]ICE uses six of the manufacturing industries from the JW model directly. The remaining industries' percentage changes were weighted to produce the "other" category.

modeling of the coal prices was performed using the coal market component of the ARGUS model.

It is possible that in some regions low sulfur coal prices to the industrial sector may be lower than the national average. This was not found to be the case. For example, in 1990, delivered regional industrial coal prices change by less than two-thirds of one percent. In most cases, the percentage change was near zero. This result appears to occur because of the highly regional nature of the coal market. While the artificial demand for low sulfur coal may fall, power plants near low sulfur coal reserves now find it advantageous to buy this local coal, which raises the price back to an equilibrium level near to that of the control scenario. This is even more likely to be true of industrial delivered prices, since industrial prices are more affected by transportation costs than are the utility prices. No additional ICE modeling was performed.

Economic Driver Data for the Industrial Process Approach

The J/W model was also used to account for activity level changes in the calculation of industrial process emissions under the no-control scenario. The correspondence between Trends, MSCET, and the J/W model was used to apply changes in industrial activity in each target year to each industrial process.

No-control Scenario Emissions

Industrial Boiler Emissions of SO_2, NO_x, and TSP

The CAA imposed different regulations, SIPs, and New Source Performance Standards (NSPS) that apply to industrial boilers of varying size. The primary effect of CAA regulations on industrial boilers was simulated by defining the Air Quality Control Region (AQCR), the resulting SIPs, and subsequent NSPS for boilers. The industrial boiler SIP regulations were included in the ICE base year file discussed in the previous section. Since the ICE model estimates new boiler emissions for each target year, the boiler NSPS are input through the ICE user files. Industrial NSPS were implemented in two phases. The 1971 regulations are imposed for the study years 1975 and 1980. The 1984 NSPS revisions are imposed in the study years 1985 and 1990. For the no-control scenario, ANL set the SIPs and NSPS to a flag that indicated "no regulation."

Industrial Boiler Emissions of CO and VOC

Two of the criteria pollutants emitted by industrial fuel combustors, CO and VOC, were not included as outputs of the ICE model. Therefore, CO and VOC emissions were analyzed separately using *Trends* methods. Control scenario CO and VOC emissions were taken directly from *Trends*.

To estimate CO and VOC emissions from industrial combustion for the no-control scenario, fuel use for industrial manufacturing was adjusted, reflecting fuel consumption changes estimated by the J/W model. These changes in the level of fuel consumption by industrial combustion were also used in ANL's ICE boiler model. Changes in industrial combustion fuel use by manufacturing in the no-control scenario

are reported in Table 46. These estimates represent an average of several sectors, which were developed by ANL as part of the modeling process for ICE.

No-control scenario emissions were computed using 1970 emission factors. Since there were no add-on controls for industrial combustion VOC and CO emissions, it was not necessary to adjust the no-control scenario for changes in control efficiency.

Emission estimates were regionalized using State-level emissions data from industrial boilers recorded in MSCET. For the control scenario estimates, VOCs were regionalized using the MSCET State-level shares for industrial fuel combustion. In the no-control scenario, the State-level shares were held constant. The control scenario emissions of CO were regionalized using the control scenario NO_x emissions from the ICE model. This approach assumes that CO emissions are consistent with NO_x emissions. The no-control scenario CO emission estimates from industrial combustion sources were regionalized using no-control NO_x emission estimates from industrial combustion sources.

Industrial Process Emissions

A wide range of controls were imposed on industrial processes. These emission limits are embodied in the assumptions of control efficiencies in the *Trends* model. Data on national no-control scenario emissions from industrial processes were provided by EPA. These data were combined with MSCET to produce regional-level results.

Lead Emissions

Estimates of lead emissions from industrial boilers and industrial processes were completed by Abt Associates. The methods used for calculating lead emissions from industrial processes and industrial boilers were similar. The starting point was the TRI, which provides air toxics emissions data for manufacturing facilities with more than 10 employees. To estimate lead emissions from industrial boilers and processes, 1990 facility-level lead emissions data were extracted from the TRI. These data were then adjusted to create estimates of lead emissions from industrial sources under the control and no-control scenarios for each of the target years. For the control scenario, lead emissions for 1975, 1980, and 1985 were obtained by extracting an emission factor and a control efficiency for each lead-emitting industrial process in the *Trends* data base. These emission factors and control efficiencies were multiplied by the economic activity data for each year for each process as reported in *Trends* to yield estimated control scenario emissions by industrial process. Each industrial process was assigned a code to correspond with energy

Table 46. Fuel Use Changes Between Control and No-control Scenarios.

Year	Fuel Type	Fuel Use Changes
1975	Coal	-.0042
	Oil	+.0311
	Gas	-.0064
1980	Coal	-.0061
	Oil	+.0107
	Gas	-.0095
1985	Coal	-.0061
	Oil	+.0089
	Gas	-.0097
1990	Coal	-.0079
	Oil	+.0091
	Gas	-.0099

consumption data by industrial process compiled in the National Energy Accounts (NEA) by the Bureau of Economic Analysis, and emissions were summed over all processes to obtain a total for each target year.

For consistency with the other emission estimates in this analysis, industrial process no-control scenario lead emissions were adjusted for changes in industrial output, and for changes in emissions per unit of output due to control technology applications. Changes in industrial output were accounted for using results from the J/W model. Lead-emitting industrial processes in the *Trends* data base were assigned to a J/W sector. For each sector, the percentage change in economic output was used to adjust the economic activity data for that process from the *Trends* data base. These adjusted economic output figures were used with the 1970 emission factors and control efficiencies to derive the estimated no-control scenario lead emissions for each industrial process in each target year. The process-level emissions were then aggregated to the NEA-code level as in the control scenario.

The lead emission estimates from industrial processes, by NEA code, were used to derive percentage changes in emissions under the control and no-control scenarios by NEA code for application to the TRI emissions data. Since TRI data are reported by SIC code, NEA codes were "mapped" to the appropriate SIC codes, and then the percentage change for each NEA code was used to represent the percentage change for all SIC codes covered by that NEA code.

To calculate lead emissions from industrial boilers, Abt Associates developed estimates of lead emissions from industrial combustion under the CAA for each of the target years. The *Trends* data base contains national aggregate industrial fuel consumption data by fuel type. For each fuel type, the fuel consumption estimate was disaggregated by the share of that fuel used by each NEA industrial category. The *Trends* data base also contains emission factors for industrial fuel use, by fuel type, as well as control efficiencies. The lead emissions from industrial combustion for each NEA category were derived by multiplying the fuel-specific combustion estimate for each NEA category by the emission factor and control efficiency for that fuel type. The result was emissions of lead by NEA code and by fuel type. Emissions from all fuel types were then summed by NEA code. The NEA data were used to disaggregate the industrial fuel consumption figures, based on the assumption that the ICE are the same among all industries covered by a given NEA code.

To estimate no-control scenario lead emissions, the macroeconomic effect of the CAA and the change in emissions per unit of output that resulted from specific pollution control mandates of the CAA were both taken into account. As in the control scenario, the national aggregate industrial fuel consumption estimate by fuel type was disaggregated by the share of that fuel used by each NEA industrial category. The fuel use was then adjusted in two ways: some NEA codes were specifically modeled by the ICE model, and for the remaining NEA codes, J/W percentage changes in fuel use were applied. These fuel use estimates were then combined with the 1970 emission factors and control efficiencies for industrial combustion by fuel type from the *Trends* data base to obtain no-control scenario combustion-related lead emissions from industrial boilers by NEA code. These estimates of total lead emissions by NEA codes were matched to SIC codes, and then to the data in the TRI data base. This approach assumed that an average emission value was assigned to all reporting TRI facilities in a given SIC code.

Off-Highway Vehicles

The off-highway vehicle sector includes all transportation sources that are not counted as highway vehicles. Therefore, this sector includes marine vessels, railroads, aircraft, and off-road internal combustion engines and vehicles. As a whole, off-highway vehicle emissions are a relatively small fraction of total national anthropogenic emissions.

Overview of Approach

The process used by ELI to determine the national level of emissions from the off- highway transportation sector is similar to the procedure outlined above for industrial processes. To estimate the emissions of criteria air pollutants from these sources under the no-control scenario, the historical activity levels were held constant, rather than attempting to calculate a new no-control scenario level of off-highway vehicle activity. This assumption was necessary since the off-highway activity indicators (amount of fuel consumed, and landing and take-off cycles for aircraft) do not have direct correspondence with a given J/W category. The national no-control scenario emissions of criteria air pollutants from these sources were simply derived by recalculating emissions using 1970 emission factors.

Development of Control Scenario

To estimate control scenario emissions, the analysis relied on *Trends* methods, using historical activity indicators, emission factors, and control efficiencies. Essentially, the estimates of off-highway emissions under the control scenario represent the historical estimates from the *Trends* data base.

No-control Scenario Emissions Estimates

The calculation of off-highway emissions for the no-control scenario required the *Trends* data to be adjusted to reflect changes in controls and economic activity in each of the target years. Linking source activity changes with economic activity for this section is not straightforward. The economic activity data for off-highway engines and vehicles are expressed either in terms of amount of fuel consumed, or in terms of landing and take-off cycles for aircraft. Neither of these off-highway activity indicators has a direct correspondence with a given J/W sector, making the sort of direct linkage between *Trends* categories and J/W sectoral outputs that was used for industrial processes inappropriate.

In the absence of a link between the economic factors that are determinants of emissions from this sector and the available economic activity forecasts, the no-control scenario emissions of criteria air pollutants from off-highway mobile sources were estimated based on the same historical activity levels used for the control scenario. Although there were changes in sectoral output and personal income that might have had an effect on off-highway vehicle usage, these changes were deemed to be small and not likely to have a major effect on the emissions from this sector.

Emission factors for each of the off-highway sources were also held constant at 1970 levels to calculate no-control scenario emissions for each target year. The national emissions of criteria air pollutants from these sources were then recalculated using 1970 emission factors.

National and State-Level Off-Highway Emission Estimates

Table 47 summarizes national-level emission estimates for off-highway sources. The emission estimates derived from using the methodology discussed above yielded results that seem counter-intuitive. The emissions from off-highway sources, in particular the emissions from aircraft, are lower in the no-control scenario than those projected for the control scenario for most pollutants. This is a result of calculating emissions using 1970 emission factors, since the 1970 emission factors for aircraft are lower than the aircraft emission factors in later years.

ELI identified several potential sources of uncertainty in the emission estimates for this sector. First, the assumption that the total level of off-highway vehicle fuel consumption is constant between the two scenarios may be flawed. Second, the use of 1970 emission factors in the no-control scenario may fail to capture significant changes in technology. These technological changes are implicitly captured in the control scenario and it is possible that these technological changes may also have occurred under a no-control scenario.

Table 47. Difference in Control and No-control Scenario Off-Highway Mobile Source Emissions.

		1975	1980	1985	1990
TSP	Control Scenario:	268.6	281.1	268.7	280.9
	No-Control Scenario:	260.8	268.8	261.2	266.9
	Percentage Increase:	-3%	-4%	-3%	-4%
NO_x	Control Scenario:	1,987.6	2,176.7	2,077.5	2,085.9
	No-Control Scenario:	1,974.6	2,150.5	2,042.7	2,058.9
	Percentage Increase:	-1%	-1%	-2%	-1%
SO_x	Control Scenario:	364.6	531.1	406.4	392.5
	No-Control Scenario:	363.2	528.6	403.0	386.9
	Percentage Increase:	0%	0%	-1%	-1%
CO	Control Scenario:	8,512.8	8,101.4	7,881.9	8,079.0
	No-Control Scenario:	8,511.0	8,071.2	7,880.2	8,077.7
	Percentage Increase:	0%	0%	0%	0%
VOCs	Control Scenario:	1,374.9	1,370.8	1,334.8	1,405.0
	No-Control Scenario:	1,385.9	1,416.1	1,388.6	1,485.8
	Percentage Increase:	1%	3%	4%	6%

Note: Emission estimates are expressed in thousands of short tons. Percentage increase is the differential between scenarios divided by the Control Scenario projection.

One possible response to the biases created by the use of 1970 emission factors for all years in the no-control scenario is to test how results might differ if the emission factors used for the control scenario, which would include technological change, were also used for the no-control scenario. However, using this treatment of emission factors, the emissions projections from the adopted methodology from non-highway sources in the no-control scenario would be identical to the emissions projections under the control scenario. The reason for this is that the economic activity levels were not adjusted for the calculation of emissions under the no-control scenario.

In order to disaggregate the national data to a State level, the methodology used the MSCET data base, which is described starting on page 107. Emissions of VOC, SO_x, and NO_x were regionalized using the State-level shares from the MSCET methodology. The emissions of TSP were regionalized by using the State-level shares for SO_x reported by MSCET, and the emissions of CO were regionalized using the State-level shares for NO_x, also reported by MSCET. The potential bias that this introduces is likely to be small, due to the relative homogeneity of off-highway vehicle emission sources. As with regionalization of industrial process emissions, the State-level shares are held constant between the two scenarios. To the extent that the distribution of economic activity between States was not constant over the period of the analysis, holding State-level emission shares constant may bias the results, although the direction and magnitude of the potential bias is unknown.

On-Highway

This section addresses the highway vehicle portion of the transportation sector. Highway vehicle emissions depend on fuel type, vehicle type, technology, and extent of travel. Emissions from these vehicles have been regulated through Federal emission standards and enforced through in-use compliance programs, such as State-run emission inspection programs. Vehicle activity levels are related to changes in economic conditions, fuel prices, cost of regulations, and population characteristics. Emissions are a function of vehicle activity levels and emission rates per unit activity.

TEEMS was employed by ANL to analyze the transportation sector. The modeling system links several models, disaggregate and aggregate, to produce State-level estimates of criteria pollutants. The system is subdivided into two modules: an activity/energy module and an emissions module. Each module contains multiple models. TEEMS has been documented in several reports and papers (Mintz and Vyas, 1991; Vyas and Saricks, 1986; Saricks, 1985). It has been used for several policy analyses and assessment studies for DOE and NAPAP. This section presents an overview of the approach used to conduct the analysis of the transportation sector. Also included in this section is a summary of the methodology used by Abt Associates to estimate changes in lead emissions from highway vehicles in each target year.

Overview of Approach

TEEMS has two modules: an activity/energy module and an emissions module. The activity/energy module calculates emissions based on: (1) personal travel; (2) goods movement; and (3) other transportation activity inputs.

Personal Travel

Personal travel activity and resulting fuel consumption were calculated for each target year using procedures that disaggregate households by demographic and economic attributes. Economic driver data, developed from U.S. Government data and macroeconomic model(s) of the domestic economy, formed the basis for household disaggregation. Modeling procedures were employed by ANL to project movement of households between various attribute classes, and vehicle holdings were projected in terms of the number and type of vehicles held by each household type. National totals were then developed by aggregating the vehicle holding estimates for each household type, accounting for the number of households of that type.

Travel estimates, in terms of VMT, were calculated using the same approach, and based on the VMT of each household type. The basis for household transportation activity projection has been empirically established through analysis of the 1983-84 Nationwide Personal Transportation Survey (NPTS) (FHWA, 1986; Mintz and Vyas, 1991). VMT are projected using this empirical relationship, and estimates of the elasticity of VMT to vehicle operating cost are then made. Energy consumption was estimated in each target year using VMT, shares of VMT by vehicle type, and exogenously developed vehicle characteristics.

The following three models and an accounting procedure were employed to develop target year personal travel activity projections:

1. The first model projected the target year distribution of households by their attributes. This model employed an iterative proportional fitting (IPF) technique and projected the number of households in each cell of the household matrix - each of which is defined by various categories within six household attributes.

2. The second model projected changes in vehicle ownership resulting from changes in income and cost of vehicle operation. The model applied estimated ownership changes to each target year household matrix such that the control values within each of the household attributes, excepting vehicle ownership, remained unchanged.

3. The third model estimated the composition of household vehicle fleet by type (cars and trucks), size, technology, and fuel.

4. An accounting procedure applied VMT per vehicle to vehicle ownership in each combination of household attributes. VMT and energy consumption were accumulated by vehicle type, size, and fuel.

Each of these models is described separately in the following subsections.

Iterative Proportional Fitting (IPF)

This IPF model modified a control scenario matrix of household counts. A household matrix was developed from the 1983 NPTS data and upgraded to the year 1985 using published aggregate data. The procedure used in constructing the 1985 household matrix has been documented elsewhere (Appendix B of Mintz and Vyas, 1991). The matrix is defined by six attributes: (1) residential location (central city, suburb, rural); (2) household income; (3) age of householder; (4) household size; (5) number of drivers; and (6) number of vehicles. The household matrix has 3,072 cells, some of which are illogical (such as 1 person, 2 drivers). Illogical cells were replaced with zeros.

Household shares within each attribute in each target year were developed exogenously using data from the Bureau of the Census and selected macroeconomic model runs. The projected total of households and shares of households in each category of an attribute were supplied to the IPF model. The model modified the control scenario household matrix to match the specified shares and total number of households.

The IPF model treated household distribution within each attribute as a set of vectors. These vectors were scaled to match the specified shares and household total. Following the initial scaling, a gradual

scaling technique was used to move in the direction of the target shares. The scaling process was repeated until closure was achieved for all attribute classes. Since vehicle ownership levels were estimated by the vehicle ownership model (described in the next section), shares within the sixth household attribute (number of vehicles held) were not specified, leaving it uncontrolled. This flexibility of an uncontrolled attribute helped to facilitate the model operation. The number of households in each class of vehicle ownership within the output matrix represents distribution of households using the control scenario (1985) relationship of vehicle ownership to other household attributes.

Vehicle Ownership Projection (VOP)

The VOP model projected the changes in vehicle ownership resulting from changes in the number of licensed drivers, disposable personal income, and annual fuel cost of vehicle operation. The model is based on historical household ownership rates. A target per-driver ownership rate was computed using disposable income and fuel cost. This target rate represented desired ownership if income and fuel cost were the only determinants. A parameter representing ownership responsibilities such as acquisition effort, disposal effort, parking requirements, and other indirect aspects was applied to adjust this target. The new ownership rate was used to estimate the number of household vehicles.

The household matrix created by the IPF model was revised to match the projected household vehicle ownership. Household shares within the first five attributes remain constant while those within the sixth attribute (i.e., number of vehicles) were variable. A deviation measure was defined and its value for each class within the first five attributes was minimized. A set of simultaneous equations was solved using Lagrangian multipliers.

Projection of Vehicle Fleet Composition

The composition of household vehicles was projected for each household matrix cell using a vehicle choice model called the Disaggregate Vehicle Stock Allocation Model (DVSAM). Vehicles are defined by type (auto, light truck), size (small, mid-size, full-size auto; small pickup, small utility/minivan, standard pickup, large utility/standard van; or any other size classification), fuel (gasoline, diesel, methanol, ethanol, or compressed natural gas), and technology (stratified charge, direct injection, electric, fuel cell, or Brayton).

The model computed vehicle composition based on an individual vehicle's utility to households and household needs. A menu of vehicles classified by the previously mentioned vehicle attributes was supplied to the model. The menu specified characteristics of each vehicle available to households. Vehicles were characterized by price, operating cost, seating capacity, curb weight, and horsepower. These variables formed the basis for computing "utility" (analogous to consumer satisfaction). The household matrix provided demographic and economic attributes which, when combined with vehicle usage in miles, define household needs. Vehicle usage (VMT) was computed as a function of income, number of drivers, and number of vehicles. A logit model was applied to compute vehicle ownership shares. Several model enhancements facilitated modeling of limited range vehicles, and representation of supply constraints and/or regulated market penetration.

Activity/Energy Computation

An accounting procedure was applied to compute personal travel activity in terms of VMT by vehicle type. Control scenario VMT per vehicle estimates for each cell in the household matrix were developed from the 1983 NPTS. These rates were adjusted within the procedure on the basis of changes in average vehicle operating cost per mile for each cell. The vehicle composition projection model computes ownership shares and share-weighted change in vehicle operating cost. Elasticity values were applied to this change.

ANL assumed that VMT per vehicle remained nearly unchanged for a household matrix cell over time (with the exception of the effect of changes in vehicle operating cost). In other words, variation of VMT across household types is far greater than within household types. VMT per household vehicle remained stable during the period from 1977 to 1984 (Klinger and Kuzmyak, 1986). Some increases were observed in recent years, which were attributed to lower fuel prices and increased household income (DOC, 1991; FHWA, 1992). (A portion of the increase could be attributed to the method of computing average VMT per vehicle.) The assumption that VMT per vehicle for each cell remained nearly constant and was elastic relative to vehicle operating cost is reasonable. As households move from one cell of the matrix to another, they "acquire" the VMT per vehicle rate of that cell. Thus, this approach accounted for changes in VMT per vehicle due to increased household affluence, increased rate of driver licensing, changes in fuel price, and changes in vehicle technology.

Goods Movement

Energy and activity demand resulting from movement of 24 aggregate categories of commodities is estimated by this subcomponent of the TEEMS activity module. Changes in commodity demand/production were provided by growth indexes by two-digit SIC generated by a macro model. A model that projects shifts in mode shares among truck, rail, marine, air, and pipeline modes was used, followed by a procedure to compute ton miles of travel for each mode, VMT by fuel type for trucks, and energy consumption by operation type for non-highway modes. The model used 1985 control scenario data, which were compiled from railroad waybill sample and publications, waterborne commerce publications, transportation statistics, and other sources. The procedure used in developing the 1985 control scenario freight data has been documented in an ANL report (Appendix A of Mintz and Vyas, 1991).

This goods movement model was not used for this retrospective analysis because of funding and time constraints. A procedure to estimate truck VMT by fuel type was employed in its place. Published historical VMT values (FHWA, 1988; 1992) were used along with VMT shares by fuel and truck type from Truck Inventory and Use Surveys (TIUS) (DOC, 1981; 1984; 1990).

Other Transportation Activities

The activity/energy module also has other models for developing activity and energy use projections for air, fleet automobiles, and bus modes. Fleet automobile activity estimates from an earlier study (Mintz and Vyas, 1991) were used while other modes were not analyzed.

Lead Emissions

Estimates of lead emissions in the transportation sector were developed by Abt Associates based on changes in reductions of lead in gasoline. This estimation required the estimates of lead in gasoline consumed over the period from 1970 to 1990 and the amount of lead content in gasoline that would have been consumed in the absence of the CAA. These values were calculated using the quantity of both leaded and unleaded gasoline sold each year and the lead concentration in leaded gasoline in each target year. Data on annual gasoline sales were taken from a report by ANL that presented gasoline sales for each State in each target year. For the control scenario, data on the fraction of gasoline sales represented by leaded gasoline were used. For the no-control scenario, all of the gasoline sold was assumed to be leaded. Data on the lead content of gasoline was obtained from ANL for 1975 through 1990. For 1970 through 1975, the analysis assumed that the 1974 lead content was used.

Estimation of No-control Scenario Emissions

TEEMS emissions projections were carried out by ANL in the following three steps:

1. Development of emission factors;
2. Allocation of highway activity to States; and
3. Development of highway pollutant estimates.

The following subsections describe the procedures used for computing highway vehicle emissions.

Development of Emission Factors

EPA's MOBILE5a Mobile Source Emission Factor model was used to provide all of the highway vehicle emission factors used to estimate 1975 to 1990 emission rates (EPA, 1994b). Documentation of the MOBILE5a model is found in the User's Guide for the MOBILE5 model.[80]

Although the actual emission factors used by ANL are not documented in either the original ANL TEEMS model report or in the Pechan summary report, the Project Team provided direction that defined the emission factors to be used. For the control scenario, ANL was directed to use the official EPA emission factors prevailing at the time for each target year. For example, the official EPA emission factor being used in 1980 for on-highway vehicle NO_x was to be used to estimate 1980 control scenario on-highway vehicle NO_x emissions. For the no-control scenario, the official EPA emission factors used to estimate emissions in 1970 were to be used throughout the 1970 to 1990 period.

It is important to note that using the 1970 on-highway vehicle emission factors to estimate no-control scenario emissions for the entire 1970 to 1990 period may bias scenario emission differentials upward. This is because it is possible that technological changes to on-highway vehicles unrelated to CAA compliance strategies may have yielded incidental reductions in emissions. However, EPA Office of Mobile Sources (EPA/OMS) experts indicate that the two major technological changes in vehicles

[80] EPA/OAR/OMS, "User's Guide to MOBILE5," EPA-AA-AQAB-94-01, May 1994; see also 58 FR 29409, May 20, 1993.

occurring during the period of the analysis –electronic ignition and electronic fuel injection– would have yielded negligible emission reductions in the absence of catalytic converters.[81]

Another potential bias is introduced by assuming the CAA had no substantial effect on vehicle turnover. However, two factors render this potential bias negligible. First and foremost, under the no-control scenario retired vehicles would be replaced by new but equally uncontrolled vehicles. Second, no-control scenario vehicle use is greater in terms of VMT per year. This means no-control scenario vehicles would reach the end of their service lives earlier, offsetting to some extent the alleged incentive to retire vehicles earlier due to costs imposed by CAA control requirements.

Allocation of Highway Activity to States

TEEMS' activity module generated national activity and energy estimates. These activity totals were allocated to States through a regionalization algorithm that used time series data on historical highway activity shares by State. A trend extrapolation methodology was used that stabilizes shifts after 5 years in the future. For the retrospective analysis, historical highway activity shares for each target year were developed using data published by the Federal Highway Administration (FHWA) (FHWA, 1988; 1992).

Development of Highway Pollutant Estimates

Highway emission estimates were calculated in both scenarios for each target year using VMT estimates generated by TEEMS and emission factors from MOBILE5a. Control scenario activity levels were adjusted for the no-control scenario using economic forecasts and historical data.

Control Scenario Emissions Calculation

Control scenario data for the transportation sector were compiled from several sources. Household counts and shares of households by six attributes were obtained from various editions of the *Statistical Abstracts* of the United States. Household income information was obtained from the control scenario run of the J/W model. Fuel prices were obtained from the *Annual Energy Review* (DOE, 1992) while vehicle fuel economy and aggregate VMT per vehicle were obtained from *Highway Statistics* (FHWA, 1988; 1992). Table 48 lists data sources for the control scenario run.

Table 49 shows household shares prepared for the IPF model. The total number of households increased from 63.4 million in 1970 to 93.3 million in 1990. A gradual shift from rural to urban was observed with movement to suburbs within urban areas. The effect of economic downturns in 1975 and 1980 was an increase in share for the lowest income category; more households moved to the highest income group from 1970 to 1990, while the lower middle income group share expanded and the upper middle income share declined. The rate of household formation was high during the 1970's, which resulted in increases in smaller and younger households. The trend in younger households reversed after 1980 as household formation slowed. Average household size dropped from 3.2 in 1970 to 2.67 in 1990.

[81] Telephone conversation between Jim DeMocker, EPA/OAR and EPA/OMS/Ann Arbor Laboratory staff (date unknown). Nevertheless, the Project Team did consider reviewing emission factors for European automobiles to attempt to estimate no-control scenario emission factors for 1975 through 1990 reflecting the use of electronic fuel injection and electronic ignition but no catalytic converter. However, the Project Team concluded that differences in fuel/air mix ratios used in Europe would probably obscure any differences in emission rates attributable to the use of electronic fuel injection and electronic ignition.

The number of licensed drivers increased throughout the analysis period as more and more young people were licensed to drive.

Data for the VOP model included disposable income per capita, fuel price, overall personal vehicle fuel economy, and annual usage in terms of VMT. Table 50 shows these data for each year in the analysis period.

Data preparation for the model that projected household vehicle composition was limited to characterization of existing technology vehicles. Seven vehicle size and type combinations were characterized for 1975 and 1980 while one vehicle, minivan/small utility, was added for 1985 and 1990. Control scenario vehicle characteristics are tabulated in Table 51. TEEMS' activity and energy computation procedure was executed to produce personal vehicle travel and energy consumption estimates.

Commercial truck travel was not modeled but, historical data published by the FHWA (FHWA, 1987; 1991) were used. FHWA publishes truck travel by three categories: 1) 2-axle, 4-tire trucks; 2) single unit trucks; and 3) combination trucks. All 2-axle, 4-tire trucks were treated as light-duty trucks. VMT by personal light trucks were subtracted from the published totals to arrive at commercial light truck VMT. Diesel truck VMT shares of total VMT were obtained from TIUS (DOC, 1981; 1984; 1990). TIUS data were also used to split VMT by single unit and combination trucks. All combination trucks were assumed to be the heaviest, class 7 and class 8, while single unit trucks could be of any size class – 3 through 8. Gasoline and diesel VMT totals were developed for these heavy-duty trucks and were kept constant for the control and no-control scenarios.

Table 48. Sources of Data for Transportation Sector Control Scenario Activity Projection.

Data Item	Model	Source
Household total, population, household shares by four attributes (location, income, age of head, and household size).	IPF	Statistical Abstract of the United States, editions 96th, 98th, 103rd, 104th, 108th, and 113th.
Household shares by number of drivers.	IPF	Statistical Abstracts and FHWA Highway Statistics provided total drivers. The *with CAA* distribution of households trended.
Personal and Disposable income.	VOP	JW model output and Statistical Abstracts.
Vehicle fleet on-road fuel economy.	VOP DVSAM	FHWA Highway Statistics.
Fuel Prices	VOP DVSAM	Energy Information Administration's (EIA) Annual Energy Review.
Vehicle Price	DVSAM	Ward's Automotive Yearbooks 1975-1983, Automotive News Market Data Book 1985.

IPF - Iterative Proportional Fitting
VOP - Vehicle Ownership Projection
DVSAM - Disaggregate Vehicle Stock Allocation Model
FHWA - Federal Highway Administration
EIA - Energy Information Administration

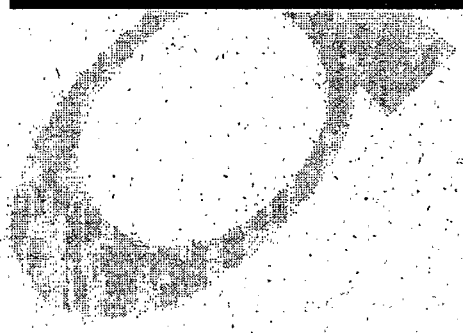

Table 49. Distribution of Households by Demographic Attributes for Control Scenario.

Household (Million)	63.4	71.1	80.8	86.8	93.3
Population (Million)	204.0	215.5	227.2	237.9	249.5
Attribute	**Household Percentage, by Year**				
	1970	1975	1980	1985	1990
Location					
Central City	33.2	32.0	31.9	31.6	31.4
Suburbs	33.6	36.0	37.0	38.1	38.3
Rural	33.2	32.0	31.1	30.3	30.3
Income (1990 $)*					
<$13,000	25.9	26.5	26.6	25.9	25.5
$13,000 - $33,000	34.0	37.2	37.4	37.7	38.0
$33,000 - $52,500	27.6	22.7	22.4	22.2	22.2
>$52,500	12.5	13.6	13.6	14.2	14.3
Age of Householder (YR)					
<35	25.4	29.1	31.1	29.3	27.4
35 - 44	18.6	16.7	17.3	20.1	22.1
45 - 64	36.3	34.0	31.2	29.6	29.0
> = 65	19.7	20.2	20.4	21.0	21.5
Household Size					
1	17.2	19.5	22.7	23.7	24.6
2	29.0	30.7	31.3	31.6	32.2
3 - 4	33.0	33.0	33.2	33.5	32.8
> = 5	20.8	16.8	12.8	11.2	10.4
Licensed Drivers					
0	9.1	8.5	8.1	7.2	6.6
1	27.8	27.3	27.0	26.2	26.0
2	48.1	49.2	50.5	52.5	53.5
> = 3	15.0	15.0	14.4	14.1	13.9

Note: *Approximated to 1990 dollars.

Table 50. Economic and Vehicle Usage Data for Vehicle Ownership Projection - Control Scenario.

Year	Disposable Income per Capita (84 $)	Fuel Price (84 $)/Gallon	Miles/Gallon	VMT/Vehicle
1970	7,597	0.92	13.5	10,143
1971	7,769	0.88	13.5	10,246
1972	7,990	0.84	13.4	10,350
1973	8,436	0.84	13.3	10,184
1974	8,270	1.06	13.4	9,563
1975	8,340	1.03	13.5	9,729
1976	8,553	1.02	13.5	9,833
1977	8,742	1.01	13.8	9,936
1978	9,070	0.97	14.0	10,143
1979	9,154	1.21	14.4	9,522
1980	9,052	1.53	15.5	9,212
1981	9,093	1.55	15.9	9,212
1982	9,050	1.38	16.7	9,419
1983	9,239	1.27	17.1	9,419
1984	9,691	1.20	17.8	9,550
1985	9,881	1.09	18.2	9,568
1986	10,139	0.88	18.3	9,672
1987	10,174	0.88	19.2	10,090
1988	10,564	0.86	19.9	10,100
1989	10,713	0.90	20.3	9,819
1990	10,903	1.00	20.8	9,780

Table 51. Control Scenario Personal Characteristics.*

Vehicle Type and Size (Seats)	1975 Curb Weight (lb)	1975 Engine Power (hp)	1975 Fuel Economy (mpg)	1980 Curb Weight (lb)	1980 Engine Power (hp)	1980 Fuel Economy (mpg)
Automobile						
Small (2-4)	2,770	91	17.2	2,535	83	19.6
Compact (4)	3,625	115	14.6	3,335	105	16.9
Mid-size (5)	4,140	128	13.3	3,730	116	15.1
Large (6)	4,900	155	12.2	4,840	153	13.3
Light truck						
Std. truck	4,530	141	11.2	4,455	143	12.6
Compact	3,745	108	14.2	3,580	99	15.9
Std. Van/Std.	5,010	145	9.9	4,975	144	11.4
Utility (11-15)						
Minivan/Small Utility (7-8)						

Vehicle Type and Size (Seats)	1985 Curb Weight (lb)	1985 Engine Power (hp)	1985 Fuel Economy (mpg)	1990 Curb Weight (lb)	1990 Engine Power (hp)	1990 Fuel Economy (mpg)
Automobile						
Small (2-4)	2,225	75	22.7	2,135	75	24.9
Compact (4)	2,775	90	19.3	2,595	90	22.0
Mid-size (5)	3,180	108	16.8	3,050	108	19.5
Large (6)	3,975	135	14.6	3,705	130	17.1
Light truck						
Std. truck	4,160	132	13.1	4,000	128	14.1
Compact	3,495	90	17.2	3,360	90	18.9
Std. Van/Std.	4,920	142	12.4	4,765	138	12.9
Utility (11-15)						
Minivan/Small Utility (7-8)	4,125	101	16.7	3,910	108	18.2

Note: *Average for all vehicles of each type and size.

No-control Scenario Emissions

The control scenario data were modified to reflect no-control scenario emissions using economic changes predicted by the J/W model, EPA, and ANL. The J/W model predicted a slight loss of employment and drop in GNP in terms of nominal dollars. However, the lower rate of inflation coincided with a real GNP rise. ANL's information from the model did not include any indexes for converting nominal income to real income. ANL assumed real income changes to be similar to those of real GNP and modified household shares by income classes accordingly. The model also predicted a slight drop in refined petroleum price beginning in 1973. The predicted drop was the largest (5.35 percent) in 1973, reached the lowest level (2.16 percent) in 1984, then increased to a second peak (3.44 percent) in 1988, and dropped again from 1989 to 1990. Since these changes were inconsistent with historical patterns of leaded and unleaded gasoline price change, ANL developed an estimate of changes in fuel price resulting from the cost of removal of lead from gasoline and other infrastructure costs involved with distributing a new grade of fuel. Subsequently, EPA provided a set of fuel costs for use in the analysis. Both ANL and EPA fuel prices followed a similar pattern, although their magnitudes differed. The no-control scenario was analyzed with EPA fuel prices. ANL also established a relationship with cost of regulation/emission control technology, and the effect of costs on vehicle price and fuel economy directly from the EPA publication *Cost of Clean Environment* (EPA, 1990). These changes were used in the analysis.

The IPF model was executed for target years 1975, 1980, 1985, and 1990 using a set of revised household shares by income class. Table 52 shows the revised shares. Comparing Table 52 no-control scenario shares with those in Table 49 for the control scenario, there seems to be a slight shift away from travel by the lowest income group and toward the middle income groups.

The vehicle ownership projection model was executed for the above four target years using the data listed in Table 53. Changes in fleet characteristics are summarized in Table 54.

Table 52. Distribution of Households by Income Class for No-control Scenario.

	Household Shares (%), by Year			
Attribute	**1975**	**1980**	**1985**	**1990**
Income (1990 $)*				
<$13,000	26.3	26.2	25.3	24.7
$13,000-33,000	37.3	37.6	38.4	38.4
$33,000-52,000	22.8	22.6	22.0	22.6
>$52,000	13.6	13.6	14.3	14.3

Note: *Approximated to 1990 dollars.

Table 53. Economic and Vehicle Usage Data for Vehicle Ownership Projection – No-control Scenario.

Year	Disposable Income per Capita (84 $)	Fuel Price (84 $)/Gallon	Miles/ Gallon	VMT/Vehicle
1970	7,597	0.91	13.5	10,143
1971	7,769	0.88	13.5	10,247
1972	7,990	0.83	13.4	10,353
1973	8,463	0.84	13.3	10,189
1974	8,297	1.06	13.4	9,569
1975	8,406	1.02	13.5	9,736
1976	8,600	1.01	13.5	9,854
1977	8,795	1.01	13.8	9,963
1978	9,126	0.96	14.0	10,174
1979	9,216	1.19	14.4	9,557
1980	9,114	1.51	15.5	9,234
1981	9,158	1.53	16.0	9,234
1982	9,116	1.36	16.8	9,447
1983	9,312	1.25	17.2	9,450
1984	9,775	1.18	17.9	9,582
1985	9,976	1.06	18.3	9,607
1986	10,244	0.84	18.4	9,738
1987	10,282	0.86	19.4	10,201
1988	10,676	0.83	20.1	10,214
1989	10,827	0.88	20.5	9,902
1990	11,019	0.97	21.0	9,849

Note: The effect of reductions in vehicle price and vehicle operating cost, and increases in fuel economy and horsepower were reflected in the menu of the vehicle choice model (DVSAM). Vehicle weight and seating capacity were kept unchanged from the *with CAA* run. Table IV-7 shows the changes in various vehicle attributes.

Table 54. Percent Changes in Key Vehicle Characteristics Between the Control and No-control Scenarios.

	1975			1980		
Vehicle	**Price**	**mpg**	**HP**	**Price**	**mpg**	**HP**
Small Auto	-2.35	0.01	0.59	-2.76	0.22	1.81
Compact Auto	-2.35	0.01	0.59	-2.76	0.22	1.81
Midsize Auto	-2.35	0.01	0.59	-2.76	0.22	1.81
Large Auto	-2.35	0.01	0.59	-2.76	0.22	1.81
Small Truck	-1.30	0.01	0.59	-2.71	0.22	1.81
Std Truck	-1.30	0.01	0.59	-2.71	0.22	1.81
Std Van/Util	-1.30	0.01	0.59	-2.71	0.22	1.81
M Vn/Sm Utility						

	1985			1990		
Vehicle	**Price**	**mpg**	**HP**	**Price**	**mpg**	**HP**
Small Auto	-3.25	0.62	2.20	-2.94	0.95	2.77
Compact Auto	-3.25	0.62	2.20	-2.94	0.95	2.77
Midsize Auto	-3.25	0.62	2.20	-2.94	0.95	2.77
Large Auto	-3.25	0.62	2.20	-2.94	0.95	2.77
Small Truck	-2.53	0.62	2.20	-2.58	0.95	2.77
Std Truck	-2.53	0.62	2.20	-2.58	0.95	2.77
Std Van/Util	-2.53	0.62	2.20	-2.58	0.95	2.77
M Vn/Sm Utility	-2.53	0.62	2.20	-2.58	0.95	2.77

Note: *Average change for each vehicle size and type combination.

Utilities

The electric utility industry retrospective analysis was prepared using two different utility simulation models. ICF utilized its CEUM to estimate control and no-control scenario emissions for SO_2, PM, and NO_x in each of the target years. ANL's ARGUS model was used to estimate electric utility CO and VOC emissions for the same period. This mix of modeling approaches was used because, while CEUM was determined to be a better tool for examining fuel shifts that were affected by the CAA than ARGUS, the CEUM model was not initially set-up to evaluate CO or VOC emissions. Although CEUM can be (and eventually was) configured to provide emission estimates for pollutants other than SO_2, NO_x, and PM, ARGUS was already configured to provide VOC and CO emissions. However, it should also be noted that VOC and CO emissions from utilities are quite low, as efficient fuel combustion reduces both pollutants. Thus, for this sector, the presence or absence of the CAA would not produce any different VOC or CO control techniques. VOC and CO emission rates for this sector differ primarily based on the fuel and boiler type. Therefore, a simpler modeling approach was judged to be acceptable and appropriate for these two pollutants. This chapter presents the methodology used to estimate utility emissions under the control and no-control scenario using the CEUM and ARGUS models. The method used by Abt Associates to estimate lead emissions from utilities is also presented.

Overview of Approach

The CEUM model uses industry capacity data and specific unit-by-unit characteristics, operating costs data, electricity demand estimates under the control and no-control scenario, and historical fuel prices to estimate SO_2, TSP, and NO_x emissions for 1980, 1985, and 1990. Changes in electric utility emissions, costs, and regional coal production were developed using ICF's CEUM with a calibration to historical electricity generation, fuel use, and emissions. The ARGUS model, which was used by ANL to estimate utility VOC and CO emissions, is driven by operating costs, industry capacity and generation data, demand for coal, and unit-level operating characteristics. The J/W model is used to incorporate predicted changes in electricity demand under the no-control scenario. Finally, Abt Associates relied upon energy use data, the *Trends* data base, and the Interim 1990 Inventory to calculate utility lead emissions based on coal consumption. The approaches used by each of these three contractors are discussed individually in the following sections.

Establishment of Control Scenario Emissions

A common feature of the approaches taken by ICF and ANL was to identify conditions that are inputs to the CEUM and ARGUS models, respectively, in the control scenario. Later in the analysis, these variables were revised to reflect no-control scenario conditions. The next section discusses the specific assumptions used in the CEUM analysis.

Key Assumptions in the Development of the ICF Analysis

At EPA's direction, ICF made several assumptions in conducting this analysis for purposes of consistency with other ongoing EPA efforts assessing the effects of the CAA. These include the

macroeconomic assumptions regarding the effects of the CAA on economic growth, or more specifically, electricity demand, developed from other EPA commissioned efforts. Each is described briefly below.

Pollution Control Equipment Costs

Only limited actual data were available for this analysis on the historical capital and operating costs of pollution control equipment. Accordingly, for this analysis, the actual capital and operating costs of scrubbers were estimated using EPA scrubber cost assumptions adjusted to reflect actual data from a survey of scrubbed power plants with scrubbers installed during the 1970s and early 1980s. For those power plants with actual survey data, actual capital costs were used. For other pre-1985 scrubbers, ICF relied on the average costs from the survey data. For particulate control equipment (primarily electrostatic precipitators, or ESPs), costs were estimated based on limited actual data, and a 1980 Electric Power Research Institute (EPRI) study of ESP and baghouse costs. Based on this information, ESPs were estimated to cost an average of $50 per kilowatt (in 1991$). The development of more detailed data on actual power plant pollution control costs was beyond the scope of ICF's analysis. ICF concluded that such an effort would not significantly change the national or regional cost estimates developed by its approach.

Electricity Demand and Fuel Prices

Consistent with other EPA ongoing analyses, ICF assumed that the CAA resulted in a reduction in electricity demand of 3.27 percent in 1980, 2.77 percent in 1985, and 2.97 percent in 1990. Also consistent with these studies, ICF assumed that natural gas prices and oil prices would not be affected by the CAA. Coal prices were estimated to change in line with increases and decreases in demand for specific coal supplies (and consistent with ICF's detailed modeling of coal supply and demand). The average prices of all residual oils consumed were also estimated to change due to a greater use of more expensive lower sulfur residual oils under the CAA.

Coal, Nuclear, Hydro, and Oil/Gas Capacity

At EPA's direction, ICF's approach was based on the assumption that no changes in the amount of nuclear, coal, hydro, or oil/gas stream or combined cycle capacity would be built or in place in 1980, 1985, or 1990. Given that the driving factors associated with the actual decisions to build new baseload capacity were not based solely on economics but entailed financial, regulatory, and political factors as well, the actual effect of the CAA on these build decisions is very uncertain. To the extent that more coal-fired power plants would be built and fewer oil/gas-fired power plants constructed, the actual emission reductions associated with the CAA would be greater than those estimated by ICF, while the estimated costs of the CAA would be greater (because fewer, lower-cost, coal-fired power plants would be on line under the CAA). However, the CAA had virtually no effect on the costs of constructing new coal-fired power plants that came on line prior to about 1975 and a relatively moderate cost effect on coal-fired power plants that came on line through the early 1980s (since these power plants were not required to install scrubbers). Since a large majority of coal-fired power plant capacity came on line prior to 1975, ICF concluded that the effect of the CAA on the amount of total coal-fired capacity was not expected to be very large.

Natural Gas Consumption

The analysis assumed that the amount of natural gas consumed under the no-control scenario could not exceed the actual amount of consumption in 1980, 1985, and 1990. In part, because of natural gas price regulation and the oil price shocks of the 1970s, natural gas was often unavailable to electric utilities in the early 1980s. Since the CAA is relatively unrelated to the questions of supply availability and price regulation of natural gas, ICF assumed that no additional gas supplies would be available if the CAA had never been adopted. It is possible, however, that in the absence of the CAA, industrial and commercial users of natural gas would have used more oil or coal. To the extent that this would have occurred, there would have been more natural gas supplies available to the electric utility sector. This increase in supply would have resulted in an increase in the estimated costs of the CAA, and a corresponding decrease in the estimated emission reductions. ICF concluded, however, that this effect would not be very significant.

State and Local Environmental Regulations

At EPA's direction, ICF assumed that there would be no State and local emission limits or other emission control requirements under the no-control scenario. Accordingly, ICF assumed that there would be no SO_2, NO_x, or TSP emission limits under the no-control scenario and that all scrubbers, NO_x controls, and ESPs/baghouses (at coal-fired power plants) were installed as a result of the CAA. (The more limited amount of particulate control equipment installed at oil-fired plants was assumed to have been installed prior to the passage of the CAA.) In the case of particulate control equipment, some ESPs and other equipment were installed at coal plants prior to the 1970 CAA. To the extent that this is the case, the estimates of the costs of meeting the CAA have been overstated. ICF concluded, however, that the amount of such capacity was not substantial.

Retirement Age

The analysis assumed that unit retirement age was constant between the control and no-controls scenarios. Adoption of this assumption might bias the emission reduction estimates upward to the extent turnover rates of older (and presumably higher-emitting) units may be slower under the control scenarios, because more significant CAA control requirements focused on new units. However the vast majority of existing coal and oil capacity was built after 1950 and it is generally acknowledged that a relatively short technical plant lifetime would be about 40 years. As such, even if the no-control scenarios resulted in no life-extension activity, there would be virtually no effect over the 1970 to 1990 timeframe of the analysis.[82]

ICF 1975 Control Scenario Emissions

The 1975 emissions under both scenarios were calculated differently than emissions in 1980, 1985, and 1990. In calculating or estimating 1975 SO_2 emissions for the control scenario (i.e., "actual" 1975), the weighted average emission rates at the State level, in the year 1975 were estimated, based on plant level average sulfur content of fuel deliveries from Federal Energy Regulatory Commission (FERC) Form 423 and assumed AP-42 sulfur retention in ash. These weighted average emission rates were then applied to actual State-level electric utility fuel consumption in the year 1975 (DOE, 1991). In the case of NO_x emissions, first, an estimate of Statewide NO_x emissions in the year 1975 was derived based on the use of

[82]This conclusion is supported by R. Nelson, T. Tietenberg and M. Donihue, "Differential Environmental Regulation: The Effects on Electric utility Capital Turnover and Emissions," Review of Economics and Statistics, Vol. 75, no. 2 (May 1993), pp. 368-373, which indicates that a delay in utility retirement due to new source regulation exists but that it has only a small effect on emissions.

the same NO_x emission rates, by fuel type, as developed for the 1980 no-control scenario modeling runs. These emission rates were specific to the fuel type (coal, oil, or natural gas). These Statewide NO_x emission rates or factors were then applied to actual fuel consumed by electric utilities in the year 1975, in order to obtain estimated "actual" 1975 emissions. As before, the fuel consumption at a State level was derived from the *State Energy Data Report* (DOE, 1991). ICF calculated the weighted average heat content (BTU/lb) by State from the 1975 FERC Form 423 data and used these figures with the TSP emission factors (lbs/ton) to derive emission rates by State (lbs/MMBTU). These emission rates were then applied to 1975 fuel consumption estimates obtained from the *State Energy Data Report*. For the control scenario 1975 estimates, ICF used the 1975 factors.

For the remaining target years, ICF used the results of CEUM runs that provided fuel consumption figures in 1980, 1985, and 1990, respectively. Emissions were then calculated using the appropriate emission factors for each year.

ARGUS Modeling Assumptions

The portion of the electric utility sector analysis conducted by ANL with the ARGUS model is described in this subsection. ARGUS contains four major components: BUILD, DISPATCH, the Emissions and Cost Model, and the Coal Supply and Transportation Model (CSTM). An overview of ARGUS can be found in Veselka *et al* (1990). Only the DISPATCH and CSTM modules were used for the present analysis. A brief description of the ARGUS components used in this analysis is found in the following subsections.

DISPATCH Module

The DISPATCH module contains a probabilistic production-cost model called the Investigation of Costs and Reliability in Utility Systems (ICARUS). This module calculates reliability and cost information for a utility system. ICARUS represents detailed, unit-by-unit operating characteristics such as fuel cost, forced outage rate, scheduled maintenance, heat rate, and fixed and variable operating and maintenance (O&M) costs. These components are used to efficiently compute system reliability (such as loss-of-load probability and unserved energy) and production costs.

The input data required by ICARUS include monthly load duration curves, annual peak demands, and, for both new and existing units, unit sizes, capital costs, fixed and variable O&M costs, fuel types and costs, heat rates, scheduled maintenance, and equivalent forced outage rates. The output from ICARUS includes annual summaries of capacity, generation, cost, and reliability for the entire generating system.

CSTM Module

The CSTM module determines the least-cost combination, on a per BTU basis, of coal supply sources and transportation routes for each demand source. First, it estimates coal market prices based on regional demands for coal from all economic sectors. To generate market prices, CSTM estimates regional coal production patterns and coal transportation routes. The CSTM input data are grouped into three major categories: demand, supply, and transportation. CSTM uses supply curves from the Resource Allocation and Mine Costing (RAMC) Model (DOE, 1982). Every region has a separate curve for one or more of the 60 different coal types that may be produced in that region. CSTM modifies the original RAMC supply curve by dividing the single RAMC curve into two curves, one representing deep mines and the other

representing surface mines, but still uses the same ranges for heating values and mine prices that define the supply curves in RAMC. Prices fluctuate as a result of different mining methods, size of mining operations, reserve characteristics, and depletion effects.

The transportation data defines the network that connects 32 coal supply origins with 48 demand centers. Transportation cost is affected by distance, terrain, congestion, variable fuel costs, cost escalators for fuels and facility upgrades, and competition. CSTM first computes the production cost for each coal supply region and coal type. It then matches supply sources with transportation routes to find the lowest delivered costs.

Coal demand for a particular region is based on the amount, geographic region, economic sector, and range of coal types. There are 44 domestic demand regions. CSTM allows demand to be met by one, or a combination of, different supply regions.

The ARGUS input data for existing units are based on the Argonne Power Plant Inventory (APPI). APPI is a data base of operating and planned generating units in the United States that was current through 1988 at the time of ANL's analysis. This data base is updated annually based on information in the regional North American Electric Reliability Council (NERC) reports, reports from the Energy Information Administration (EIA), and other sources. Unit operating characteristics (fixed O&M, variable O&M, heat rate, forced outage rate, and scheduled maintenance) are based on regional data as defined in the EPRI report on regional systems and other historic data (EPRI, 1981).

ANL used the 1988 inventory to generate a 1990 inventory. The 1990 inventory was then used to generate a separate unit inventory for the target years 1975, 1980 and 1985. The target year inventories were generated by removing units whose on-line year was greater than the target year, from their respective inventory. The regional capacity totals in these preliminary inventories were tabulated by major fuel category (nuclear, coal, oil and gas steam) and compared to the regional historic NERC totals. This review identified capacity differences, especially in 1975 and 1980 inventories. The original plan was to add phantom units to match the regional historic totals. However, based on the need for State-level emissions, it was decided that a more thorough review of the unit inventories was required.

ANL's detailed review included an examination of the nuclear and coal units greater than 100 megawatt equivalent (MWe) in each target year. Missing units, with the appropriate unit size and State code, were added so that the regional totals were comparable. The availability of coal units was based on the on-line year of the unit as reported in the EIA report *Inventory of Power Plants in the United States* (DOE, 1986). The coal units were also checked against the EIA Cost and Quality Report (EIA, 1985) to verify the existence of flue gas desulfurization (FGD) systems in each of the target years. The nuclear unit inventories were verified with the EIA report *An Analysis of Nuclear Power Plant Operating Costs* (DOE, 1988). The review also included oil and gas steam units greater than 100 MWe. The total capacity of the oil and gas steam units were compared because many units switched primary fuel from oil to gas during the relevant time period. The oil and gas units were compared to historic inventories based on information provided by Applied Economic Research. In addition to thermal generation, the hydro and exchange energy was reviewed. For each target year, the hydro generation and firm purchase and sale capacity data was adjusted to reflect the historic levels. These two components, hydro and firm purchase and sales, are accounted for first in the loading order. If these variables are overestimated, there will be less generation from coal units. Likewise, if they are underestimated, there will be too much coal generation. The hydro

and firm purchases and sales can vary significantly from year to year because of weather conditions and other variables. Therefore, it was important that they be accurately represented.

No-control Scenario Emissions

In order to calculate utility emissions under the no-control scenario, inputs to both the CEUM and ARGUS models were adjusted to reflect no-control scenario conditions. The changes made to each model's base year input files are discussed separately in the following sections.

ICF Estimates of SO_2, TSP, and NO_x Emissions in the No-control Scenario

As described on page 136, ICF utilized a different methodology to calculate 1975 emission estimates. Rather than relying on the use of detailed modeling runs, ICF based the 1975 emission estimation on historic fuel consumption and sulfur content data in 1975. This subsection first outlines the process used to calculate no-control scenario emissions in 1975 and then presents the methods used for the remaining target years.

1975 Utility SO_2, NO_x, and TSP Emissions

To develop State-level no-control scenario utility SO_2 emissions, ICF developed no-control scenario SO_2 emission rates. A reasonable surrogate for these emission rates is SO_2 rates just prior to the implementation of the SIPs under the CAA. ICF developed 1972 rates (based on the earliest year available for FERC Form 423) and compared these with 1975 rates. In each State, the greater of 1972 or 1975 rates was used in the calculation of SO_2 emissions in the absence of the CAA. To develop State-level no-control scenario SO_2 emissions, no-control scenario fuel consumption data were needed. ICF assumed that the demand for electricity in 1975 would be 2.73 percent higher than the actual energy sales in 1975. This assumption is identical to the no-control scenario electricity demand projections derived from the J/W projections. For the purpose of this analysis, it was further assumed that this increment in demand would have been met in 1975 from the oil and coal-fired plants in each State. The increase in consumption of these fuels was assumed to be in the same proportion as their share in the 1975 total energy mix for electricity generation in that State. It was assumed that the generation of nuclear, gas-fired, and other electricity generation would not change. A sensitivity case without an assumed electricity demand change was also calculated. (The sensitivity analysis results are presented later in this appendix, starting on page 142.)

For NO_x emissions under the no-control scenario, it was also assumed that the 1975 electricity sales would have been 2.73 percent higher than was the case in 1975. No-control scenario TSP emissions in 1975 were based on national emission rate numbers from EPA that were converted to pounds per million BTU using the average energy content of fuels in each State. No-control scenario TSP emissions were calculated based on 1970 emission factors (Braine, Kohli, and Kim, 1993).

1980, 1985, and 1990 Utility Emissions

For 1980, 1985, and 1990, ICF calculated no-control scenario emissions based on fuel consumption figures from the CEUM runs, and 1970 emission factors from EPA.

Electric utility SO_2 emission estimates are approximately 10 million tons (or about 38 percent) lower by 1990 under the control scenario than under the no-control scenario. Most of this estimated difference results from the imposition of emission limits at existing power plants through the SIPs under the 1970 CAA. Most of these SIPs were effective by 1980 (with some not fully effective until 1985). Most of the additional reductions that occurred during the 1980s were the result of the electric utility NSPS, which required the installation of 70 to 90 percent SO_2 removal control equipment.

By contrast, electric utility NO_x emission estimates under the control scenario are only about 1.2 million tons, or 14 percent, lower than under the no-control scenario by 1990. This occurs because, under the implementation of the 1970 CAA, only a few existing power plants were subject to NO_x emission limits. Virtually all of the estimated reductions are the result of NO_x NSPS, which generally required moderate reductions at power plants relative to uncontrolled levels. In addition, electricity demand is estimated to be about 3 percent lower under the control scenario. This decrease reduces the utilization of existing power plants and also contributes to lower NO_x emissions (and other pollutants as well).

Electric utility annualized costs (levelized capital, fuel, and O&M) are estimated to be $0.2 billion lower in 1980, $1.5 billion higher in 1985, and $1.9 billion higher in 1990 under the control scenario. Note, however, that this reflects the effects of two offsetting factors: (1) the *higher* utility compliance costs associated with using lower sulfur fuels, and the increased O&M and capital costs associated with scrubbers and particulate control equipment; and (2) *lower* utility generating costs (fuel, operating and capital costs) associated with lower electricity demand requirements. In 1980, the increase in fuel costs due to higher generation requirements (under the no-control scenario), was larger than the decrease in capital and O&M costs and thus yielded a cost increase over the control case.

However, lower electricity demand for the utility sector would translate into higher costs in other sectors (as electricity substitutes are used). This effect was captured to some extent by the original J/W macroeconomic modeling conducted for the present analysis.

Average levelized U.S. electricity rate estimates are approximately 3 percent higher under the control scenario during the 1980s. Note that year by year, electric utility revenue requirements and capital expenditures (not estimated by ICF) would be estimated to have increased by a greater percentage particularly in the 1970s and early 1980s as incremental capital expenditures for scrubbers and ESPs were brought into the rate base.

Significant shifts in regional coal production are estimated to have occurred between the control and no-control scenarios. High sulfur coal producing regions such as Northern Appalachia and the Midwest/Central West are estimated to have lower production under the control scenario, while lower sulfur coal producing regions such as Central and Southern Appalachia are estimated to have higher coal production.[83]

[83] At EPA's direction, ICF's analysis did not estimate the effect of shifts in non-utility coal consumption on regional coal production, nor did it consider the possibility that fewer new coal powerplants might have been built due to the CAA as discussed earlier. Both of these factors could result in a greater estimated change in total U.S. coal production than estimated herein although the difference is not likely to be very significant.

ARGUS No-control Scenario

Regional fuel prices, for the thermal units, were based on historic information from the EIA Form 423 data for the year 1977, 1980 and 1985. The 1977 data was used for 1975. Fixed and variable O&M costs were adjusted from the 1988 level, and all cost data were converted to 1985 dollars.

Table 55. J/W Estimates of Percentage Increases in National Electricity Generation Under No-control Scenario.

Year	Percentage Increase
1975	2.7%
1980	3.3%
1985	2.8%
1990	3.0%

The load data were based on regional historic NERC data for each of the target years. The shapes of the monthly load duration curves are the result of modifications based on the data in the EPRI report on regional systems (EPRI, 1981). The shapes were modified to match the projected 1988 monthly load factors for the NERC regions. These load shapes were held constant for all years.

The actual peak-loads were selected from historic information and used with the existing load duration curves. The system was dispatched so that the calculated generation could be compared with historic data. Discrepancies were resolved by adjusting the peak load so that the annual generation was on target. This procedure was repeated for each of the target years.

The electric utilities were expected to have an increase in generation as identified by the J/W data. Table 55 identifies the increase in national level generation by year. The national level increase in generation was applied to each power pool.

In addition to load changes, coal units with FGD equipment were modified. These units had their FGD equipment removed along with a 3 percent decrease in heat rate, a 2 percentage point decrease in forced outage rate, and a 50 percent decrease in their fixed and variable O&M costs. These changes were incorporated into the ARGUS model for each of the target years. Model runs were then conducted to arrive at estimates of VOC and CO emissions in the no-control scenario.

Estimation of Lead Emissions from Utilities

In order to estimate Pb emissions from electric utilities in each of the target years, data from three different sources were used. Energy use data for the control and no-control scenarios were obtained from the national coal use estimates prepared for the Section 812 analysis by ICF (Braine and Kim, 1993). The *Trends* data base provided emission factors and control efficiencies, and the Interim 1990 Inventory identified utility characteristics. The ICF data bases provided the amount of coal consumed for both the control and no-control scenarios in each of the target years. A correspondence between the Interim Inventory and the ICF data base was achieved through the plant name variable. Using emission factors for lead and control efficiencies for electric utilities, estimates of lead emissions per plant per year were calculated. These factors were obtained from the *Trends* data base. It was assumed that pollution control on coal-burning power plants under the no-control scenario would be the same as the pollution control level in 1970. Therefore, the control efficiency from 1970 is used as the basis for the no-control case.

CEUM Sensitivity Case

In addition to comparing actual (control scenario) historical costs and emissions with the higher electricity demand under the no-control scenario, ICF also evaluated emissions in a sensitivity case without the CAA or no-control scenario with the same electricity demand (versus the no-control scenarios with higher demand). The purpose of this sensitivity analysis was to isolate the incremental electric utility compliance costs and reductions in emissions associated with the CAA from the lower resulting generation costs and emissions due to lower estimated electricity demand under the CAA. The incremental effects of the CAA when compared with this case indicate:

- Estimated reductions in emissions due to the CAA are somewhat lower if measured against the sensitivity case without the CAA with the same electricity demand than the emissions without the CAA with lower demand. This occurs because lower electricity demand under the no-control scenario sensitivity results in lower utilization of existing coal and oil plants which, in turn, results in lower emissions. As noted above, in some sense, the changes in emissions represent the effects of electric utility compliance actions under the CAA, absent the effect of lower resultant demand for electricity.

- When measured against the sensitivity case without the CAA (with the same electricity demand), electric utility annualized costs are estimated to have increased by about $5 to $6 billion during the 1980 to 1990 period. This reflects the following cost factors: (1) higher annualized capital costs associated primarily with scrubbers and ESPs installed by electric utilities to comply with the CAA; (2) higher O&M costs associated with the additional air pollution control equipment; and (3) higher fuel costs associated with using lower sulfur coal and oil in order to meet the emission limit requirements of the CAA.

Commercial/Residential

The Commercial and Residential Simulation System (CRESS) model was developed by ANL as part of the Emissions and Control Costs Integrated Model Set and used in the NAPAP assessment (*Methods for Modeling Future Emissions and Control Costs, State of Science and Technology, Report 26*) (McDonald and South, 1984). CRESS is designed to project emissions for five pollutants: SO_x, NO_x, VOC, TSP, and CO. The CRESS output is aggregated into residential and commercial subsectors related to both economic activity and fuel use. The introductory material provided in this appendix about CRESS describes the base year as being 1985. It appears in this way because CRESS was originally developed to operate using the 1985 NAPAP Emission Inventory as its base year data set. For the five pollutants reported by CRESS, emission estimates are provided for the following sectors:

- Commercial/institutional
  - coal, including point and area categories of anthracite and bituminous boilers;
  - liquid fuel, including boiler and space heating uses of residual, distillate, LPG, and other fuels;
  - natural gas boilers, space heaters, and internal combustion engines;
  - wood used in boilers and space heaters; and
  - other mixed or unclassified fuel use.

- Residential
  - coal, including area sources of anthracite and bituminous;
  - liquid fuel, composed of distillate and residual oil;
  - natural gas; and
  - wood.

- Miscellaneous
  - waste disposal, incineration, and open burning; and
  - other, including forest fires, managed and agricultural burning, structural fires, cutback asphalt paving, and internal combustion engine testing.

In addition, VOC emissions are projected for these source categories:

- Service stations and gasoline marketing;

- Dry-cleaning point and area sources; and

- Other solvents, including architectural surface coating, auto-body refinishing, and consumer/commercial solvent use.

This section describes the use of CRESS to estimate control and no-control scenario emissions from the commercial/residential sector.

Control Scenario Emissions

For the NAPAP assessment, 1985 CRESS output corresponded to the 1985 NAPAP Inventory (EPA, 1989), which served as the benchmark for any projections. The design of CRESS is such that emissions by NAPAP SCC are input for each State, then projected to future years by scaling them to economic data such as energy demand. In estimating emissions, differences in emission controls associated with new,

replacement, and existing equipment are taken into account where such differences are considered significant. The basic modeling approach is shown in the following equation:

$$Q_{t,b} = \left(\frac{Q_0}{E_0}\right), b \times \left(\frac{D_t}{D_0}\right) \times \sum^{T} (f_{t,j} \times E_{t,j}) \quad (3)$$

where:

Q = emissions in year t or the base year, year 0

E = emission factor for the source category b in the base year, or for a subcategory j subject to controls in year t (this takes into account changes in emission rates that may occur as a result of emission regulations or technology changes)

D = driver data indicating activity levels in the base and future years

f = fraction of total activity in year t differentially affected by emission controls

The calculations are carried out in two subroutines, one for SO_2, NO_x, TSP and CO, and one for VOC.

Typically SO_2, NO_x, TSP, and CO emissions are projected by multiplying the 1985 NAPAP SCC data or base year data by the ratio of the driver data (activity level) value in the projection year to its value in the base year. Because there are few controls on SO_x or NO_x emissions from the sources covered by CRESS, projected emissions for most sectors are proportional to the expected activity levels. Thus,

$$Q_t = Q_0 \times \left(\frac{D_t}{D_0}\right) \quad (4)$$

There are a few source types, such as commercial/institutional boilers, for which emission controls are mandated. These are modeled by multiplying the 1985 emission data by the ratio of the controlled emission factor to the base-year emission factor. Emission factors for each source type are weighted by the proportion of base year activity in each subsector to which controls are expected to apply.

$$Q_{t,b} = Q_0 \left[g_{t,b} + \left(\frac{E_{t,n}}{E_{0,b}}\right) \times (g_{t,r} + g_{t,n})\right] \quad (5)$$

where:

g = the fraction of base-year activity accounted for by existing source b, replacement source r, or new source n in year t

The effective emission factor ($E_{t,n}$) for the sector is calculated by weighing the portions of sectoral emissions subject to NSPS controls and those likely to continue at existing levels. An appropriate Internal Revenue Service-based rate at which new equipment replaces existing sources is applied to each sector in the model. This is done to estimate how emissions might change as older sources are retired and replaced by new sources that emit at lower rates.

The SO_x/NO_x/TSP/CO subroutine varies in new and replacement emission-source fractions subject to NSPS controls. These fractions are applied to the emission-source replacement rates. In addition, ratios for new source emission factors are varied by State. However, emission ratios for any pollutant/source type combination do not vary over the projection period.

The VOC estimation methodology is similar, but allows variation in emission factors over time. Emission ratios are calculated from files of replacement and existing source emission factors weighted by the replacement rate for each sector and new source factors by State. These are input for each 5-year projection interval. For most source categories, VOC controls are not envisioned, and the 1985 NAPAP emissions for the category are simply scaled proportionally to changes in the driver (activity level) data.

For sources to which controls apply, a variation on the following equation is employed:

$$Q_{t,b} = \left(\frac{Q_0}{E_0}, b\right) \times (E_{t,b} + g_{t,n} \times E_{t,n}) \qquad (6)$$

In equation 7, the emission factors for new and existing sources are effectively weighted by the proportion of total activity in year t to which controls apply.

In using CRESS for the CAA retrospective analysis, the base year was 1975. CRESS requires emissions information by State and NAPAP source category as input. Since detailed information on emission levels for 1975 by NAPAP source category were not available, the data were developed from a combination of sources. The procedure for calculating 1975 emissions based on the 1985 NAPAP inventory is described below. The emissions module uses these initial values in conjunction with activity estimates to project control and no-control scenario emissions.

Emissions Data

Since the starting point for the analysis was 1975, emissions data by State and SCC for SO_2, NO_x, VOC, TSP, and CO were required. Available emissions information for this year was not at the level of detail needed by CRESS. The 1985 NAPAP Inventory, which contains the necessary level of detail, in conjunction with information from EPA's *National Air Pollutant Emission Estimates, 1940-1990* (*Trends*) and ANL's MSCET, was used to construct an emissions inventory for 1975. The model then uses these emissions as a benchmark for the analysis.

The method for constructing the 1975 emissions data base was consistent for all pollutants; however, two different sources of emissions data were necessary in order to obtain time series information on all pollutants. MSCET contains monthly State-level emission estimates from 1975 to 1985 by emission

source group for SO_2, NO_x, and VOC. Therefore, MSCET information was used for SO_2, NO_x, and VOC, while *Trends* data were used for TSP and CO. Emission source groups from MSCET were matched with 1985 NAPAP Inventory SCCs. The MSCET methodology is benchmarked to the 1985 NAPAP Inventory and uses time series information from *Trends* in conjunction with activity information to estimate State-level emissions for SO_2, NO_x, and VOC. Although the level of detail contained in the NAPAP Inventory could not be preserved because of the aggregation needed to match with MSCET emissions sources, MSCET provided the State-level spatial detail required by CRESS.

Once the 1985 emissions by SCC and State from the 1985 NAPAP Inventory were matched with emission source groups and States from the MSCET data base, an estimate of 1975 emissions was computed by multiplying the 1985 NAPAP Inventory emissions value by the ratio of 1975 MSCET emissions to 1985 MSCET emissions. Ratios were computed and applied for each combination of State, pollutant, and MSCET emission source group.

This method of constructing an emissions inventory for 1975 utilizes the State estimates from MSCET, thus capturing the spatial shifts that occurred over the analysis period. It is assumed that NAPAP provides the most reliable point and area source information in terms of the level of 1985 emissions (which is also the assumption of the MSCET methodology). Note that if there were a 1-to-1 correspondence between MSCET and NAPAP, this method would be equivalent to using the MSCET methodology directly for constructing 1975 emission levels.

A similar method was used for TSP and CO, but since these pollutants are not included in MSCET, the *Trends* ratio of 1975 to 1985 emissions for these two pollutants was used. Thus, for TSP and CO, all States were assumed to have experienced the same change in emissions as indicated by the national figures.

It should be noted that in addition to the loss in spatial detail, the *Trends* source groups generally spanned several NAPAP source categories. The strength in the *Trends* information is the consistency of emissions estimates over time. It is considered to be the most reliable data for tracking changes in emissions over the time period of the analysis, and was therefore chosen for developing 1975 estimates for TSP and CO.

The 15 source categories reported in *Trends* were matched with those in the 1985 NAPAP Inventory. The ratios of 1975 emissions to 1985 emissions by source category that were applied to the 1985 NAPAP emissions data are shown in Table 56. The 1975 emissions data estimated from the above procedure served as the benchmark and initial value for the CRESS emissions module for both scenarios.

CAA regulation of commercial/residential emissions was limited and largely confined to fuel combustion sources (SO_2, NO_x, TSP), gasoline marketing (VOC), dry cleaning (VOC), and surface coating (VOC). NSPS regulations of small (over 29 MW capacity) fuel combustors were promulgated in 1984 and 1986. For purposes of emissions calculations, the stipulated NSPS for SO_2, NO_x, and TSP were incorporated into the control scenario for 1985 and 1990. Emission rates for source categories subject to VOC regulation were similarly adjusted.

Table 56. *Trends* Source Categories and (1975 to 1985) Scaling Factors for TSP and CO.

Trends Source Category	TSP*	CO*
Commercial/Institutional Fuel Combustion:		
Coal	2.11	0.59
Natural Gas	1.00	0.91
Fuel Oil	2.35	1.43
Other	1.83	0.67
Residential Fuel Combustion:		
Coal	1.33	1.47
Natural Gas	1.17	1.00
Fuel Oil	1.11	1.76
Wood	0.49	0.49
Miscellaneous: Forest Fires	0.67	0.62
Solid Waste Disposal:		
Incineration	3.00	0.64
Open Burning	1.50	1.44
Miscellaneous Other Burning	1.00	1.33
Industrial Processes: Paving	2.71	0.56
Asphalt Paving and Roofing	2.71	0.56
Miscellaneous Other	1.83	0.67

Note: *These values are the ratios of 1985 *Trends* emissions to 1975 *Trends* emissions for each source category. For example, the commercial/ institutional fuel combustion: coal emission ratio of 2.11 is computed as the ratio of the 1975 TSP emissions of 40 gigagrams per year to the corresponding 1985 emissions of 19 gigagrams per year.

Energy Data

Nearly 75 percent of the source categories in CRESS use energy consumption by State and sector as the driver for the emissions calculation. State-level energy consumption statistics are published by EIA in *State Energy Data Report, Consumption Estimates, 1960-1989,* and are electronically available as part of the State Energy Data System (SEDS) (DOE, 1991). The SEDS data base contains annual energy consumption estimates by sector for the various end-use sectors: residential, commercial, industrial and transportation, and electric utilities.

Seven fuel-type categories are used in CRESS: coal, distillate oil, residual oil, natural gas, liquid petroleum gas, wood, and electricity. The model assumes zero consumption of residual fuel oil in the residential sector and zero consumption of wood in the commercial sector. Energy consumption for each fuel-type was expressed in BTUs for purposes of model calculations. With the exception of wood consumption, all of the energy consumption statistics used in CRESS were obtained from SEDS.

Residential wood consumption estimates were derived from two data sources. State-level residential sector wood consumption estimates for 1975 and 1980 were obtained from *Estimates of U.S. Wood Energy Consumption from 1949 to 1981* (EIA, 1982). State-level wood consumption, however, was not available

for 1985 and 1990, therefore, regional information from an alternative publication, *Estimates of U.S. Biofuels Consumption 1990* (EIA, 1990), was used to derive State-level residential wood use figures. Regional 1985 and 1990 wood consumption was distributed among States using 1981 State shares. All wood consumption figures were converted to BTU's using an average value of 17.2 million BTU per short ton.

Economic/Demographic Data

Emissions from slightly more than 25 percent of the CRESS source categories follow State-level economic and demographic activity variables. The demographic variables used by CRESS include State-level population, rural population, and forest acreage. State population is the activity indicator for six emissions source categories for SO_2, NO_x, TSP, and CO, and 13 VOC source categories. State population data were assembled from the SEDS data base. Rural population, which is the indicator of residential open burning activity, is computed as a fraction of total State population. Forest wildfires and managed open burning activity are related to 1977 State-level forest acreage. The demographic information is assumed to be invariant to CAA regulations and thus is the same in the control and no-control scenario scenarios.

Car stock (or vehicle population), the driver variable for the auto body refinishing, is approximated by State motor vehicle registrations. *Highway Statistics*, an annual publication by the FHWA, was the source for data on State motor vehicle registrations. The three source categories connected with gasoline marketing are driven by State-level gasoline sales in gallons. State gasoline consumption was obtained from the SEDS data base. Housing starts and 10 percent of the existing housing stock were combined to form the activity indicator for architectural surface coating emissions. Housing data compiled by the U.S. Bureau of the Census were available in the *Statistical Abstract of the United States* (DOC, 1975; 1977; 1982; 1983; 1987; 1993). Regional-level data for 1975 was allocated to the States based on the 1980 State distribution.

No-control Scenario Emissions

Adjustments to control scenario emissions in each of the target years to reflect conditions under the no-control scenario were achieved through emission factors, energy input data, and economic/demographic data. The adjustments made to each of these variables to generate no-control scenario emissions are discussed individually in the following subsections.

Emissions Data

CAA regulation of the commercial/residential sector was minimal. For regulated source categories, emission factors were revised to reflect pre-regulation emission rates. Six commercial/residential source categories were regulated for VOC emissions: Service Stations Stage I Emissions, Service Stations Stage II Emissions, Dry Cleaning (perchloroethylene), Gasoline Marketed, Dry Cleaning (solvent), and Cutback Asphalt Paving. Commercial-Institutional boilers were regulated for SO_2 and TSP and internal combustion sources were regulated for NO_x emissions. All NSPS were removed for these sources to estimate no-control scenario emissions levels.

Energy Data

State-level energy demand for the residential and commercial sectors for the no-control scenario was estimated from the J/W model forecast. Final energy demand estimates for the household sector were calculated by an EPA contractor for the purposes of the no-control scenario analysis. State allocation of the national-level estimates was based on historic State shares, i.e., this assumes that there is no change in the distribution of energy demand across States as a result of removing regulations. In addition, the J/W model estimates an aggregate refined petroleum category and does not distinguish among liquid petroleum gas, distillate oil, and residual oil. The relative shares among these three categories of petroleum products remained constant between the control and no-control scenarios. The information on percentage change in energy demand by fuel type as provided by the J/W model is listed in Table 57.

Table 57. Percentage Change in Real Energy Demand by Households from Control to No-control Scenario.

Year	Coal	Refined Petroleum	Electric	Natural Gas
1975	1.48	4.76	3.62	2.42
1980	1.50	3.84	4.26	2.12
1985	1.98	3.90	3.88	2.41
1990	2.23	4.33	4.18	2.77

The differential for commercial sector final energy demand was calculated from the combination of four intermediate product flow categories from the J/W forecast. The National Income and Product Accounts (NIPA) for the commercial sector correspond to J/W SIC categories 32 through 35:

(32) Wholesale and Retail Trade;
(33) Finance, Insurance, and Real Estate;
(34) Other Services; and
(35) Government Services.

Percentage change information from the J/W forecast for energy cost shares, value of output, and energy prices was used to calculate the differential in commercial sector energy demand for the no-control scenario. The energy cost share is defined as the cost of energy input divided by the value of the output. In order to calculate the percentage change in commercial sector energy demand, the change in energy price was subtracted from the percentage change in energy cost, and added to the change in the value of output. Each of these variables was available from the J/W model results. This calculation was performed for each of the four energy types, and each of the four NIPA categories. The change in commercial sector energy demand was obtained by taking the weighted average of the four NIPA categories. Since data on relative energy demand for NIPA categories were not readily available, square footage was used as a proxy for calculating the weights. These data were taken from the *Nonresidential Buildings Energy Consumption Survey, Commercial Buildings Consumption and Expenditure 1986* (EIA, 1989). The resulting estimate for commercial sector changes in energy demand is provided in Table 58.

The national-level change in commercial sector energy demand was allocated to the States using historic shares. Implicit is the assumption that removal of CAA regulations does not alter the State distribution of energy use.

Economic/Demographic Data

State population was assumed not to vary as a result of CAA regulations, thus only the economic variables were revised for the no-control scenario. No-control scenario housing starts and car stock were derived from J/W forecast information on construction and motor vehicles. The differential for categories 6 (construction) and 24 (motor vehicles and equipment) was applied to control scenario values to obtain no-control scenario levels. The percentage change from the J/W forecast is given in Table 59.

State-level gasoline sales is one of the activities forecasted by the transportation sector model. The percentage change in gasoline sales calculated by the TEEMS model was used in the no-control scenario as a CRESS model input.

Table 58. Percentage Change in Commercial Energy Demand from Control to No-control Scenario.

Year	Coal	Refined Petroleum	Electric	Natural Gas
1975	-0.13	3.36	1.30	-0.80
1980	0.31	1.90	2.06	-0.82
1985	0.48	1.98	1.72	-0.40
1990	0.39	2.26	1.74	-0.22

Table 59. JW Percent Differential in Economic Variables Used in CRESS.

Year	Construction	Motor Vehicles
1975	0.70	5.04
1980	0.14	4.79
1985	0.41	6.07
1990	0.29	6.25

Table 60. TSP Emissions Under the Control and No-control Scenarios by Target Year (in thousands of short tons).

	With the CAA				Without the CAA				Difference in 1990
Sector	1975	1980	1985	1990	1975	1980	1985	1990	Emissions
Transportation:									
Highway Vehicles	700	760	770	820	770	910	1,030	1,180	(30%)
Off-Highway Vehicles	270	280	270	280	260	270	260	270	5%
Stationary Sources:									
Electric Utilities	1,720	880	450	430	3,460	4,480	5,180	5,860	(93%)
Industrial Processes	5,620	3,650	3,040	3,080	11,120	12,000	11,710	12,960	(76%)
Industrial Boilers	740	480	250	240	780	550	360	400	(41%)
Commercial/Residential	2,020	2,510	2,680	2,550	2,020	2,520	2,700	2,560	(1%)
TOTAL*	11,070	8,550	7,460	7,390	18,410	20,730	21,250	23,230	(68%)

Notes: The estimates of emission levels *with and without the CAA* were developed specifically for this Section 812 analysis using models designed to simulate conditions in the absence of the CAA. These numbers should not be interpreted as actual historical emission estimates.

*Totals may differ slightly from sums due to rounding.

Table 61. SO_x Emissions Under the Control and No-control Scenarios by Target Year (in thousands of short tons).

	With the CAA				Without the CAA				Difference in 1990
Sector	1975	1980	1985	1990	1975	1980	1985	1990	Emissions
Transportation:									
Highway Vehicles	380	450	500	570	380	450	500	560	1%
Off-Highway Vehicles	370	530	410	390	360	530	400	390	1%
Stationary Sources:									
Electric Utilities	18,670	17,480	16,050	16,510	20,690	25,620	25,140	26,730	(38%)
Industrial Processes	4,530	3,420	2,730	2,460	5,560	5,940	5,630	6,130	(60%)
Industrial Boilers	3,440	3,180	2,660	2,820	3,910	4,110	4,020	4,610	(39%)
Commercial/Residential	1,000	800	590	690	1,000	810	610	710	(3%)
TOTAL*	28,380	25,860	22,950	23,440	31,900	37,460	36,310	39,140	(40%)

Notes: The estimates of emission levels *with and without the CAA* were developed specifically for this Section 812 analysis using models designed to simulate conditions in the absence of the CAA. These numbers should not be interpreted as actual historical emission estimates.

*Totals may differ slightly from sums due to rounding.

Table 62. NO_x Emissions Under the Control and No-control Scenarios by Target Year (in thousands of short tons).

	With the CAA				Without the CAA				Difference In 1990
Sector	1975	1980	1985	1990	1975	1980	1985	1990	Emissions
Transportation:									
Highway Vehicles	8,640	9,340	8,610	8,140	9,020	11,060	13,160	15,390	(47%)
Off-Highway Vehicles	1,990	2,180	2,080	2,090	1,980	2,150	2,040	2,060	1%
Stationary Sources:									
Electric Utilities	5,540	6,450	6,660	7,060	5,740	7,150	7,780	8,300	(15%)
Industrial Processes	750	760	690	710	760	830	790	1,090	(35%)
Industrial Boilers	4,090	3,680	3,540	3,710	4,120	3,660	3,680	3,900	(5%)
Commercial/Residential	1,060	960	880	930	1,060	970	890	950	(2%)
TOTAL*	22,060	23,370	22,460	22,640	22,680	25,830	28,350	31,680	(29%)

Notes: The estimates of emission levels *with and without the CAA* were developed specifically for this Section 812 analysis using models designed to simulate conditions in the absence of the CAA. These numbers should not be interpreted as actual historical emission estimates.

*Totals may differ slightly from sums due to rounding.

Table 63. VOC Emissions Under the Control and No-control Scenarios by Target Year (in thousands of short tons).

	With the CAA				Without the CAA				Difference In 1990
Sector	1975	1980	1985	1990	1975	1980	1985	1990	Emissions
Transportation:									
Highway Vehicles	12,220	10,770	9,470	7,740	14,620	16,460	19,800	23,010	(66%)
Off-Highway Vehicles	1,380	1,370	1,340	1,410	1,390	1,420	1,390	1,490	(5%)
Stationary Sources:									
Electric Utilities	20	30	30	40	20	30	30	40	(7%)
Industrial Processes	5,910	6,780	6,230	5,630	6,130	7,930	7,290	6,810	(17%)
Industrial Boilers	150	150	150	150	150	150	140	150	0%
Commercial/Residential	4,980	5,480	5,820	5,870	4,980	5,700	6,080	6,130	(4%)
TOTAL*	24,660	24,580	23,030	20,840	27,290	31,680	34,730	37,630	(45%)

Notes: The estimates of emission levels *with and without the CAA* were developed specifically for this Section 812 analysis using models designed to simulate conditions in the absence of the CAA. These numbers should not be interpreted as actual historical emission estimates.

*Totals may differ slightly from sums due to rounding.

Table 64. CO Emissions Under the Control and No-control Scenarios by Target Year (in thousands of short tons).

	With the CAA				Without the CAA				Difference in 1990
Sector	1975	1980	1985	1990	1975	1980	1985	1990	Emissions
Transportation:									
Highway Vehicles	83,580	79,970	72,490	65,430	90,460	105,530	131,420	149,280	(56%)
Off-Highway Vehicles	8,510	8,100	7,880	8,080	8,510	8,070	7,880	8,080	0%
Stationary Sources:									
Electric Utilities	240	280	290	370	250	290	300	380	(3%)
Industrial Processes	7,580	6,990	4,840	5,140	9,240	9,120	8,860	10,180	(49%)
Industrial Boilers	720	710	670	740	720	710	620	740	0%
Commercial/Residential	10,250	13,130	14,140	13,150	10,250	13,170	14,200	13,210	0%
TOTAL*	110,880	109,170	100,300	92,900	119,430	136,880	163,280	181,860	(49%)

Notes: The estimates of emission levels *with and without the CAA* were developed specifically for this Section 812 analysis using models designed to simulate conditions in the absence of the CAA. These numbers should not be interpreted as actual historical emission estimates.

*Totals may differ slightly from sums due to rounding.

Table 65. Lead (Pb) Emissions Under the Control and No-control Scenarios by Target Year (in thousands of short tons).

	With the CAA				Without the CAA				Difference in 1990
Sector	1975	1980	1985	1990	1975	1980	1985	1990	Emissions
Transportation:									
Highway Vehicles	180	86	22	2	203	207	214	223	(99%)
Stationary Source:									
Industrial Processes	3	1	1	1	7	7	6	5	(87%)
Industrial Combustion	4	2	0	0	5	5	5	5	(96%)
Utilities	1	1	0	0	2	3	4	4	(95%)
TOTAL*	190	90	23	3	217	221	228	237	(99%)

Notes: The estimates of emission levels *with and without the CAA* were developed specifically for this Section 812 analysis using models designed to simulate conditions in the absence of the CAA. These numbers should not be interpreted as actual historical emission estimates.

*Totals may differ slightly from sums due to rounding.

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Appendix C: Air Quality Modeling

Introduction

This appendix describes in greater detail the various methodologies used to translate differences in control and no-control scenario emission estimates into changes in air quality conditions. Summary characterizations of the results of the air quality modeling efforts for 1990 are provided here and in the main text. Further details and discussion of key analytical and modeling issues can be found in a number of supporting documents. These documents, which provide the analytical basis for the results presented herein, are:

- ICF Kaiser/Systems Applications International, *"Retrospective Analysis of Ozone Air Quality in the United States"*, Final Report, May 1995. (Hereafter referred to as "SAI Ozone Report (1995).")
- ICF Kaiser/Systems Applications International, *"Retrospective Analysis of Particulate Matter Air Quality in the United States"*, Draft Report, September 1992. (Hereafter referred to as "SAI PM Report (1992).")
- ICF Kaiser/Systems Applications International, *"Retrospective Analysis of Particulate Matter Air Quality in the United States"*, Final Report, April 1995. (Hereafter referred to as "SAI PM Report (1995).")
- ICF Kaiser/Systems Applications International, *"PM Interpolation Methodology for the §812 Retrospective Analysis"*, Memorandum from J. Langstaff to J. DeMocker, March 1996. (Hereafter referred to as "SAI PM Interpolation Memo (1996).")
- ICF Kaiser/Systems Applications International, *"Retrospective Analysis of SO_2, NO_x and CO Air Quality in the United States"*, Final Report, November 1994. (Hereafter referred to as "SAI SO_2, NO_x and CO Report (1994).")
- ICF Kaiser/Systems Applications International, *"Retrospective Analysis of the Impact of the Clean Air Act on Urban Visibility in the Southwestern United States"*, Final Report, October 1994. (Hereafter referred to as "SAI SW Visibility Report (1994).")
- Dennis, Robin L., US EPA, ORD/NERL, *"Estimation of Regional Air Quality and Deposition Changes Under Alternative 812 Emissions Scenarios Predicted by the Regional Acid Deposition Model, RADM"*, Draft Report, October 1995. (Hereafter referred to as "Dennis, R. RADM Report (1995).")

The remainder of this appendix describes, for each pollutant or air quality effect of concern, (a) the basis for development of the control scenario air quality profiles; (b) the air quality modeling approach used to estimate differences in air quality outcomes for the control and no-control scenario and the application of those results to the derivation of the no-control scenario air quality profiles; © the key assumptions, caveats, analytical issues, and limitations associated with the modeling approach used; and (d) a summary characterization of the differences in estimated air quality outcomes for the control and no-control scenarios.

Carbon Monoxide

Control scenario carbon monoxide profiles

As described in the preceding general methodology section, the starting point for development of control scenario air quality profiles was EPA's AIRS database. Hourly CO air quality monitoring data were compiled for all monitors in the 48 contiguous states for the study target years of 1970, 1975, 1980, 1985, and 1990. Although the CO monitoring network was sparse in 1970, by 1990 506 monitors in 244 counties provided monitoring coverage for 55 percent of the population in the conterminous U.S. Table 66 summarizes the CO monitoring data derived from AIRS. Additional data regarding the EPA Region location, land use category, location-setting category, and objective category of the monitors providing these data are described in the SAI SO_2, NO_x, and CO Report (1994).

Table 66. Summary of CO Monitoring Data.

Year	Number of Monitors	Number of Counties	Percent Population Covered	Number of Samples	Mean Number of Samples per Monitor
1970	82	54	n/a	408,524	4,982
1975	503	246	n/a	2,667,525	5,303
1980	522	250	50 %	3,051,599	5,846
1985	472	232	n/a	3,533,286	7,486
1990	506	244	55 %	3,788,053	7,486

Data Source: SAI SO_2, NO_x and CO Report (1994).

The next step in constructing the control scenario air quality profiles was to calculate moving averages, for a variety of time periods, of the hourly CO data for each monitor. For CO, moving averages of 1, 3, 5, 7, 8, 12, and 24 hours were calculated. Daily maximum concentrations observed at each monitor for each of these averaging periods were then calculated. Finally, profiles were developed to reflect the average and maximum concentrations for each of the seven averaging periods. However, profiles were only developed for a given monitor when at least 10 percent of its theoretically available samples were actually available. The purpose of applying this cutoff was to avoid inclusion of monitors for which available sample sizes were too small to provide a reliable indication of historical air quality.

As discussed in the air quality modeling chapter of the main text, development of representative distributions for these profiles was then necessary to provide a manageable characterization of air quality conditions. Initially, two-parameter lognormal distributions were fitted to the profiles based on substantial evidence that such distributions are appropriate for modeling air quality data. However, given the relative

importance of accurately modeling higher percentile observations (i.e., 90th percentile and higher), a three-parameter modeling approach was used to isolate the effect of observations equal, or very close, to zero. In this approach one parameter defines the proportion of data below a cutoff close to zero and the remaining two parameters describe the distribution of data above the cutoff value. Several other studies have already demonstrated good fit to air quality modeling data with a three-parameter gamma distribution, and both lognormal and gamma distributions using a three-parameter approach were developed for the present study. As documented in the SAI SO_2, NO_x, and CO Report (1994), a cutoff of 0.05 ppm was applied and both the three-parameter lognormal and three-parameter gamma distributions provided a good fit to the empirical data. For CO, the gamma distribution provided the best fit.

The control scenario air quality profiles are available on diskette. The filename for the CO Control Scenario profile database is COCAA.DAT, and adopts the format presented in Table 67.

No-control scenario carbon monoxide profiles

To derive comparably configured profiles representing CO air quality in the no-control scenario, control scenario profile means and variances were adjusted in proportion to the difference in emissions estimated under the two scenarios. Specifically, for all control scenario air quality observations predicted by the three-parameter distributions falling above the "near-zero" cutoff level, comparable no-control estimates were derived by the following equation:

$$X_{NCAA} = \left(\frac{E_{NCAA}}{E_{CAA}}\right)(X_{CAA} - b) + b \qquad (7)$$

where X_{NCAA} = air quality measurement for the non-CAA scenario,
X_{CAA} = air quality measurement for the CAA scenario,
E_{NCAA} = emissions estimated for the non-CAA scenario,
E_{CAA} = emissions estimated for the CAA scenario, and
b = background concentration.

The adjustment for background concentration is made to hold ambient background concentrations of the pollutant constant between the control and no-control scenarios. To the extent background concentrations are affected by transport of anthropogenic pollutants from upwind sites, and to the extent upwind emissions may have been controlled under the control scenario, assuming a fixed background concentration represents a conservative assumption in this analysis. As discussed in the SAI SO_2, NO_x, and CO Report (1994), the CO background concentration used for this analysis was 0.2 ppm, which equals the lowest typical concentration observed in the lower 48 states.

In the SAI SO_2, NO_x, and CO Report (1994) documenting the CO air quality modeling effort, reference is made to using county-level emission estimates as the basis for deriving the no-control profiles. Derivation of these county-level results is described in more detail in the appendix on emissions estimation. It is important to emphasize here, however, that the county-level CO emissions data were derived for both the control and no-control scenarios by simple population-weighted disaggregation of

state-level emission totals. Although CO emission estimates were needed at the county level to support the ozone air quality modeling effort, differences in state-level emissions estimates are

Table 67. Format of Air Quality Profile Databases.

Columns	Format	Description
1 - 2	Integer	Year (70, 75, 80, 85, 90)
4 - 6	Integer	Averaging time (1, 3, 5, 7, 8, 12, 24 hours)
8 - 9	Integer	State FIPS code
11 - 13	Integer	County FIPS code
15 - 19	Integer	Monitor number (digits 6-10 of monitor id)
21 - 30	Real	Latitude
32 - 41	Real	Longitude
43 - 44	Integer	Latitude/longitude flag[a]
46 - 55	Real (F10.3)	Hourly intermittency parameter p^b
56 - 65	Real (F10.3)	Hourly lognormal parameter μ^b
66 - 75	Real (F10.3)	Hourly lognormal parameter σ^b
76 - 85	Real (F10.3)	Hourly gamma parameter α^b
86 - 95	Real (F10.3)	Hourly gamma parameter β^b
96 - 105	Real (F10.3)	Daily max intermittency parameter p^b
106 - 115	Real (F10.3)	Daily max lognormal parameter μ^b
116 - 125	Real (F10.3)	Daily max lognormal parameter σ^b
126 - 135	Real (F10.3)	Daily max gamma parameter α^b
136 - 145	Real (F10.3)	Daily max gamma parameter β^b

[a] Values for flag: 1 = actual latitude/longitude values
2 = latitude/longitude values from collocated monitor or previous monitor location (monitor parameter occurrence code 1)
-9 = latitude/longitude missing (county center substituted)

[b] Units of concentration are ppm for CO and ppb for SO_2, NO_2 and NO.

Source: SAI SO_2, NO_2 and CO Report (1994).

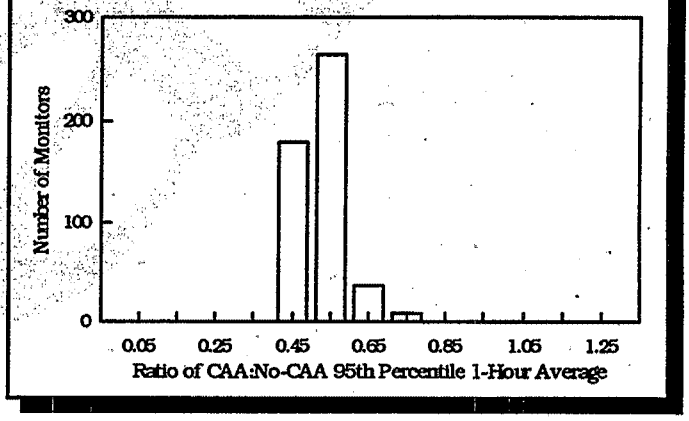
what drive the difference in the control and no-control air quality profiles for CO. In other words, the E_{NCAA} to E_{CAA} ratios used to derive the no-control profiles according the Equation (1) above are essentially based on state-level emissions estimates for CO.

As for the control scenario air quality profiles, the no-control scenario air quality profiles are available on diskette. The filename for the CO No-control Scenario profile database is CONCAA.DAT. The same data format described in Table 67 is adopted.

Summary differences in carbon monoxide air quality

While the control and no-control scenario air quality profiles are too extensive to present in their entirety in this report, a summary indication of the difference in control and no-control scenario CO concentrations is useful. Figure 32 provides this summary characterization. Specifically, the air quality indicator provided is the 95th percentile observation of 1990 CO concentrations averaged over a 1-hour period. The graph shows the number of monitors for which the ratio of 1990 control to no-control scenario 95th percentile 1-hour average concentrations falls within a particular range. The x-axis values in the graph represent the midpoint of each bin. The results indicate that, by 1990, CO concentrations under a no-control scenario would have been dramatically higher than control scenario concentrations.

Figure 32. Frequency Distribution of Estimated Ratios for 1990 Control to No-control Scenario 95th Percentile 1-Hour Average CO Concentrations, by Monitor.



Key caveats and uncertainties for carbon monoxide

A number of important uncertainties should be noted regarding the CO air quality estimates used in this analysis. First and foremost, CO is a highly localized, "hot spot" pollutant. As such, CO monitors are often located near heavily-used highways and intersections to capture the peak concentrations associated with mobile sources. Since this analysis relies on state-level aggregate changes in CO emissions from all sources, the representativeness and accuracy of the predicted CO air quality changes are uncertain. There is no basis, however, for assuming any systematic bias which would lead to over- or under-estimation of air quality conditions due to reliance on state-wide emission estimates.

A second source of uncertainty is the extent to which the three-parameter distributions adequately characterize air quality indicators of concern. Appendix C of the SAI SO_2, NO_x, and CO Report (1994) presents a number of graphs comparing the fitted versus empirical data for one-hour and 12-hour averaging

periods. In the case of CO, the gamma distribution appears to provide a very reasonable fit, though clearly some uncertainty remains.

Finally, as noted in footnote 29, a central premise of this analysis is that changes in CO emissions should be well-correlated with changes in CO air quality. Strong correlation between the state-level emissions estimates used in this analysis and empirical air quality measurements would not be expected due to inconsistencies between the state-level scale of modeled emissions versus the monitor-level scale of the air quality data, and between the modeled control scenario emissions inventories and actual historical air quality measurements. Under these circumstances, it is particularly important to focus on the primary objective of the current analysis, which is to estimate the difference in air quality outcomes between scenarios which assume the absence or presence of historical air pollution controls. In the process of taking differences, some of the uncertainties are expected to cancel out. No attempt is made in the overall analysis to predict historical air quality, or hypothetical air quality in the absence of the Clean Air Act, in absolute terms.

Sulfur Dioxide

Sulfur dioxide (SO_2) emissions lead to several air quality effects, including secondary formation of fine particle sulfates, long range transport and deposition of sulfuric acid, and localized concentrations of gaseous sulfur dioxide. The first two effects are addressed later in this appendix, under the particulate matter and acid deposition sections. The focus of this section is estimation of changes in local concentrations of sulfur dioxide.

The methodology applied to estimation of local sulfur dioxide air quality is essentially identical to the one applied for carbon monoxide. As such, this section does not repeat the "roll-up" modeling methodological description presented in the CO section, but instead simply highlights those elements of the sulfur dioxide modeling which differ from carbon monoxide.

Control scenario sulfur dioxide profiles

Unlike the CO monitoring network, the number of monitors as well as the population coverage of the SO_2 monitoring network shrank during the 1980's. Table 68 summarizes the SO_2 monitoring data used as the basis for development of the control scenario air quality profiles.

As for CO, air quality profiles reflecting average values and daily maxima for 1, 3, 5, 7, 8, 12, and 24 hour averages were compiled from AIRS for monitors in the lower 48 states which had at least 10 percent of their potential samples available. Applying a cutoff of 0.1 ppb to isolate the zero and near-zero observations, three-parameter lognormal and gamma distributions were fitted to these empirical profiles. In the case of SO_2, the three-parameter lognormal distribution was found to provide the best fit.

The control scenario SO_2 air quality profiles are available on diskette, contained in a file named SO2CAA.DAT. The same data format described in Table 67 is adopted.

Table 68. Summary of SO_2 Monitoring Data.

Year	Number of Monitors	Number of Counties	Percent Population Covered	Number of Samples	Mean Number of Samples per Monitor
1970	86	56	n/a	399,717	4,648
1975	847	340	n/a	4,280,303	5,053
1980	1,113	440	60 %	6,565,589	5,899
1985	926	401	n/a	6,602,615	7,130
1990	769	374	50 %	5,810,230	7,556

Data Source: SAI SO_2, NO_x and CO Report (1994).

No-control scenario sulfur dioxide profiles

The no-control air quality profiles for SO_2 are derived using the same equation –equation 7 on page 161– that was applied for CO. For SO_2, the background concentration was assumed to be zero. Although anthropogenic emissions contribute only small amounts to total global atmospheric sulfur, measured background concentrations for the continental U.S. range from only 0.1 to 1.3 ppb. Background SO_2 is discussed in more detail in the supporting document SAI SO_2, NO_x, and CO Report (1994).[84]

The no-control scenario SO_2 air quality profiles are available on diskette, contained in a file named SO2NCAA.DAT. The data format is described in Table 67.

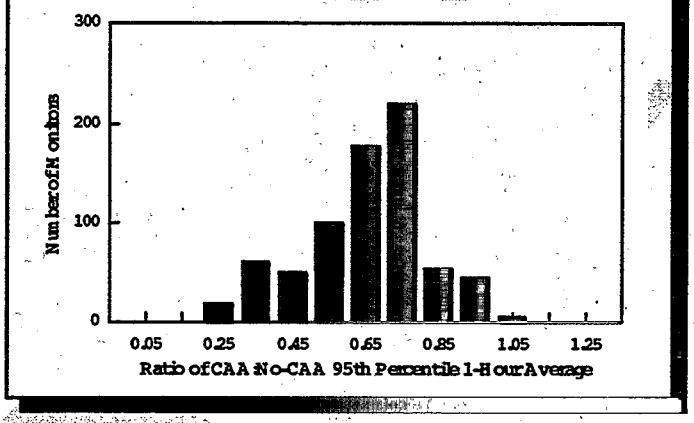
Summary differences in sulfur dioxide air quality

As for CO, reporting differences in control and no-control scenario air quality projections for each monitor covered in the analysis is impractical due to the large amount of data involved. However, Figure 33 provides an illustration of scenario differences similar to the one provided for CO. Specifically, the graph shows the distribution of 1990 control to no-control scenario 95th percentile 1-hour average concentrations ratios at SO_2 monitors. By 1990, SO_2 concentrations under the no-control scenario were substantially higher than those associated with the control scenario.

[84] SAI SO_2, NOx, and CO Report (1994), page 4-9.

Key caveats and uncertainties for sulfur dioxide

Figure 33. Frequency Distribution of Estimated Ratios for 1990 Control to No-control Scenario 95th Percentile 1-Hour Average SO_2 Concentrations, by Monitor.



The height of stacks used to vent flue gases from utility and industrial fossil fuel-fired boilers has a significant effect on the dispersion of sulfur dioxide and on the formation and long-range transport of secondary products such as particulate sulfates. Under a no-control scenario, it is conceivable that some sources might have built taller stacks to allow higher emission rates without creating extremely high ground-level concentrations of flue gases. On the other hand, it is also conceivable that, in the absence of post-1970 air pollution control programs, sources might have built shorter stacks to avoid incurring the higher costs associated with building and maintaining taller stacks. To the extent facilities would have adopted different stack height configurations under a no-control scenario, both local exposures to sulfur dioxides (and other emissions from fossil fuel combustion) and long-range transport, deposition, and exposure associated with secondary formation products may have been different. However, this analysis assumes that both the location of individual facilities and the height and configuration of emission stacks are constant between the two scenarios. If, in fact, stack heights were raised under the historical case due to CAA-related concerns, increases in local SO_2 concentrations under the no-control scenario may be overestimated. However, this same assumption may at the same time lead to underestimation under the no-control scenario of long-range transport and formation of secondary particulates associated with taller stacks. For stacks built lower under a no-control scenario, local SO_2 exposures would have been higher and long-range effects lower. Finally, the comments on uncertainties for carbon monoxide on page 165 apply as well to SO_2.

Nitrogen Oxides

Similarly to sulfur dioxide, emissions of nitrogen oxides (NO_x) –including nitrogen dioxide (NO_2) and nitrous oxide (NO)– lead to several air quality effects. These effects include secondary formation of fine particle nitrates, formation of ground-level ozone, long range transport and deposition of nitric acid, and localized concentrations of both NO_2 and NO. The first three effects are addressed later in this appendix, under the particulate matter, ozone, and acid deposition sections. The focus of this section is estimation of changes in local concentrations of NO_2 and NO.

The methodology applied to estimation of local nitrogen oxides air quality is essentially identical to the one applied for carbon monoxide and sulfur dioxide. As such, this section does not repeat the "roll-up" modeling methodological description presented in the CO section, but instead simply highlights those elements of the nitrogen oxides modeling which differ from carbon monoxide.

Control scenario nitrogen oxides profiles

After peaking around 1980, the number of NO_2 and NO monitors, their county coverage, and their population coverage shrank between 1980 and 1990. Tables 69 and 70 summarize, respectively, the NO_2 and NO monitoring data used as the basis for development of the control scenario air quality profiles.

Table 69. Summary of NO_2 Monitoring Data.

Year	Number of Monitors	Number of Counties	Percent Population Covered	Number of Samples	Mean Number of Samples per Monitor
1970	45	32	n/a	275,534	6,123
1975	308	155	n/a	1,574,444	5,112
1980	379	205	45 %	1,984,128	5,235
1985	305	182	n/a	2,142,606	7,025
1990	346	187	40 %	2,456,922	7,101

Data Source: SAI SO_2, NO_x and CO Report (1994).

Table 70. Summary of NO Monitoring Data.

Year	Number of Monitors	Number of Counties	Percent Population Covered	Number of Samples	Mean Number of Samples per Monitor
1970	39	28	n/a	246,262	6,314
1975	206	94	n/a	1,101,051	5,345
1980	224	124	30 %	1,023,834	4,571
1985	139	86	n/a	956,425	6,881
1990	145	81	15 %	999,808	6,895

Data Source: SAI SO_2, NO_x and CO Report (1994).

As for CO and SO_2, air quality profiles reflecting average values and maxima for 1, 3, 5, 7, 8, 12, and 24 hour NO_2 and NO averages were compiled from AIRS for monitors in the lower 48 states which had at least 10 percent of their potential samples available. Applying a cutoff of 0.5 ppb to both NO_2 and NO to isolate the zero and near-zero observations, three-parameter lognormal and gamma distributions were fitted to these empirical profiles. For NO_2 and NO, the three-parameter gamma distribution was found to provide the best fit.

The control scenario NO_2 and NO air quality profiles are available on diskette, contained in files named NO2CAA.DAT and NOCAA.DAT, respectively. The same data format described in Table 67 is adopted.

No-control scenario nitrogen oxides profiles

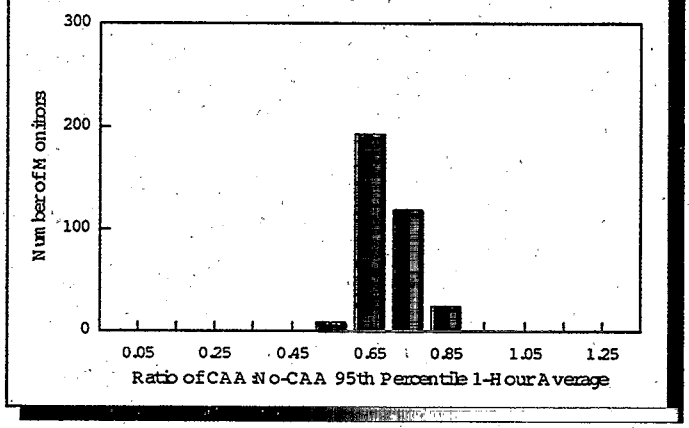
The no-control air quality profiles for NO_2 and NO are derived using the same equation –equation 7 on page 161– that was applied for CO and SO_2. As discussed in detail in the SAI SO_2, NO_x, and CO Report (1994),[85] nitrogen oxides are emitted almost entirely from anthropogenic sources and they do not have long atmospheric residence times. Therefore, global background concentrations are very low, on the order of 0.1 or 0.2 ppb. For the present analysis, background concentrations of NO_2 and NO were assumed to be zero.

The no-control scenario NO_2 and NO air quality profiles are available on diskette, contained in files named NO2NCAA.DAT and NONCAA.DAT, respectively. The data format is described in Table 67.

Summary differences in nitrogen oxides air quality

Figure 34 provides a summary indication of the differences in control and no-control scenario air quality for NO_2. As for CO and SO_2, the graph shows the distribution of 1990 control to no-control scenario 95th percentile 1–hour average concentration ratios at NO_2 monitors. These ratios indicate that, by 1990, no-control scenario NO_2 concentrations were significantly higher than they were under the control scenario. The changes for NO are similar to those for NO_2.

Figure 34. Frequency Distribution of Estimated Ratios for 1990 Control to No-control Scenario 95th Percentile 1-Hour Average NO_2 Concentrations, by Monitor.



[85] SAI SO_2, NO_x, and CO Report (1994), page 4-9.

Key caveats and uncertainties for nitrogen oxides

A number of caveats and uncertainties specific to modeling NO_x should be noted. First, stack height and stack height control strategies likely to have influenced local concentrations of SO_2 may also have influenced local concentrations of NO_2 and NO. (For a fuller discussion of the stack heights issue, refer to the section "Key caveats and uncertainties for SO_2" on page 167.) In addition, the discussion on page 165 of uncertainties resulting from the use of state-level emissions and the cancellation of uncertainties resulting from analyzing only differences or relative changes also applies to NO_x.

Acid Deposition

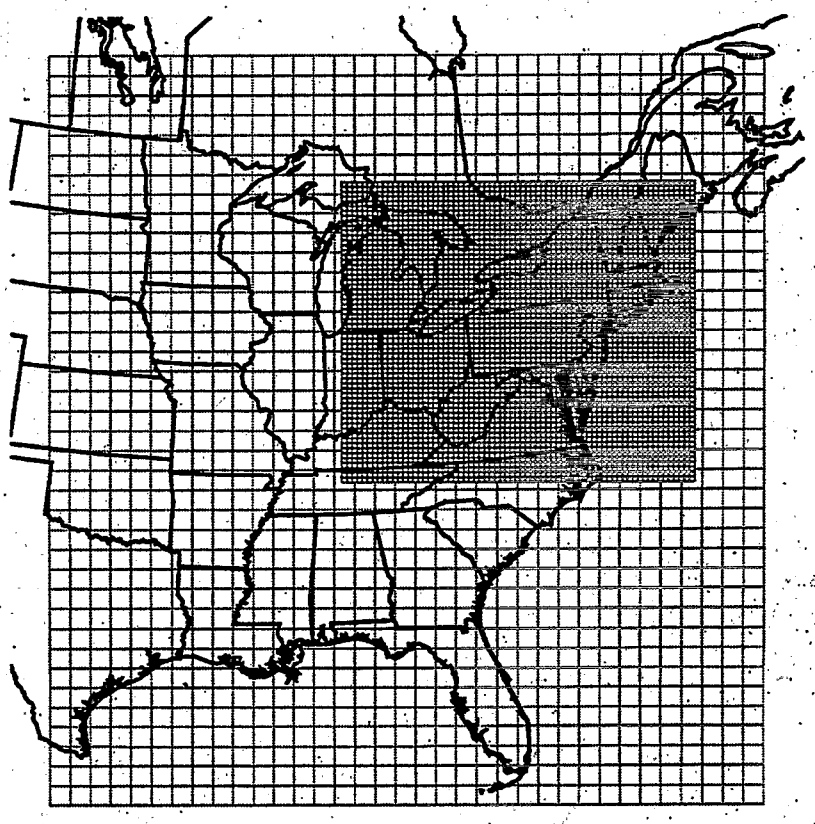
The focus of air quality modeling efforts described above for carbon monoxide, sulfur dioxide, and nitrogen oxides was to estimate the change in ambient concentrations of those pollutants as a result of changes in emissions. Particularly since the emissions modeling was driven by modeled macroeconomic conditions, rather than actual historical economic activity patterns, neither the emissions inventories nor the resultant air quality conditions developed for this analysis would be expected to match historical outcomes. The need to focus on relative changes, rather than absolute predictions, becomes even more acute for estimating air quality outcomes for pollutants subject to long-range transport, chemical transformation, and atmospheric deposition. The complexity of the relationships between emissions, air concentrations, and deposition is well-described in the following paragraph from the RADM report document developed by Robin Dennis of US EPA's National Exposure Research Laboratory in support of the present analysis:[86]

> *"Sulfur, nitrogen, and oxidant species in the atmosphere can be transported hundreds to thousands of kilometers by meteorological forces. During transport the primary emissions, SO_2, NO_x, and volatile organic emissions (VOC) are oxidized in the air or in cloud-water to form new, secondary compounds, which are acidic, particularly sulfate and nitric acid, or which add to or subtract from the ambient levels of oxidants, such as ozone. The oxidizers, such as the hydroxyl radical, hydrogen peroxide and ozone are produced by reactions of VOC and NO_x. The sulfur and nitrogen pollutants are deposited to the earth through either wet or dry deposition creating a load of pollutants to the earth's surface... However, the atmosphere is partly cleansed of oxidants through a number of physical processes including deposition (e.g., ozone is removed by wet and dry deposition). Dry deposition occurs when particles settle out of the air onto the earth or when gaseous or fine particle species directly impact land, plants, or water or when plant stomata take up gaseous species, such as SO_2. In wet deposition, pollutants are removed from the atmosphere by either rain or snow. In addition, fine particles or secondary aerosols formed by the gas- and aqueous-phase transformation processes scatter or absorb visible light and thus contribute to impairment of visibility."*

[86] Dennis, R. RADM Report (1995), p. 1.

The complexity and nonlinearity of the relationships between localized emissions of precursors, such as SO_2 and VOC, and subsequent regional scale air quality and deposition effects are so substantial that the simple "roll-up" modeling methodology used for estimating local ambient concentrations of SO_2, NO_x, and CO is inadequate, even for a broad-scale, aggregate assessment such as the present study. For sulfur deposition, and for a number of other effects addressed in subsequent sections of this appendix, a regional air quality model was required. After careful review of the capabilities, geographic coverage, computing intensity, and resource requirements associated with available regional air quality models, EPA decided to use various forms of the Regional Acid Deposition Model (RADM) to estimate these effects.[87] Figure 35 shows the geographic domain of the RADM.

Figure 35. Location of the High Resolution RADM 20-km Grid Nested Inside the 80-km RADM Domain.



Control scenario acid deposition profiles

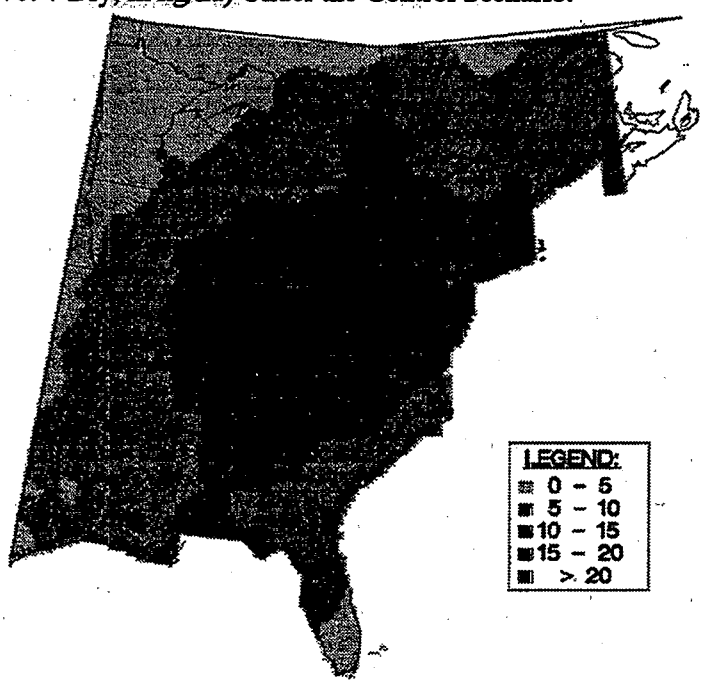
The derivation of control scenario emission inventory inputs to the RADM model is succinctly described in this excerpt from the Dennis, R. RADM Report (1995):

> *The RADM model requires a very detailed emissions inventory in both time and space. The emissions fields are also day-specific to account for the temperature effects on the volatile organics and the wind and temperature effects on the plume rise of the major point sources. At the time of the 812 Retrospective Study RADM runs, these inventories had been developed for 1985, using the 1985 NAPAP (National Acid Precipitation Assessment Program) inventory, and*

[87] For a detailed description of the various forms of the RADM and its evaluation history, see the Dennis, R. RADM Report (1995).

adjusted for point source emissions to 1988 for the Eulerian Model Evaluation Field Study funded by NAPAP. These RADM emissions inventories had county-level and detailed SCC and species-level information incorporated into them to provide the 80- and 20-km detail. The 812 Study emissions are principally computed at the state level. While the 1985 812 Study emissions are close to the NAPAP inventory, they do not exactly match, nor do they have the spatial, nor economic sector, nor species detail within a state needed to run RADM. To connect the 812 Study emissions to the RADM emissions, the following approach was followed: An industry/commercial-level disaggregation (including mobile sources) was developed for the 812 emissions to allow different sectors in a state to change their emissions across time without being in lock step and the detailed NAPAP emissions for every 80- and 20-km RADM grid-cell were grouped by state to the same level of industry/commercial aggregation for an exact correspondence. Then it was assumed that the 812 Study 1985 control emissions were effectively the same as the 1985 NAPAP emissions. Relative changes in emissions between the 812 1985 control and any other scenario (e.g., 1985 no-control, or 1990 control, or 1980 no-control, etc.) were then applied to the 1985 NAPAP state-level industry/commercial groups in the appropriate 80- and 20-km grid cells. Thus, state-level emissions for each group would retain the same state-level geographic pattern in the different scenarios years, but the mix across groups could change with time. In this way, the more detailed emissions required by RADM were modeled for each scenario year using the 812 Study emissions data sets.

Figure 36. RADM-Predicted 1990 Total Sulfur Deposition (Wet + Dry; in kg/ha) Under the Control Scenario.



Although the focus of the present analysis is to estimate the differences between the control and no-control scenarios, it is useful to illustrate the absolute levels of acid deposition associated with the two scenarios. It is particularly important to demonstrate the initial deposition conditions to preclude possible misinterpretations of the maps showing percent change in deposition. A relatively high percentage change in a particular region, for example, may occur when initial deposition is low, even when the change in deposition is also modest. The RADM-modeled 1990 control scenario wet and dry sulfur deposition pattern is shown in Figure 36. A comparable map for nitrogen deposition is presented in Figure 37. Maps of the RADM-predicted 1990 No-control Scenario sulfur and nitrogen deposition are presented in Figures 38 and 39, respectively.

No-control scenario acid deposition profiles

Configuration of the RADM model for the present analysis —including allocation of emission inventories to model grid cells, design of meteorological cases, treatment of biogenic versus anthropogenic emissions, and temporal, spatial, and species allocation of emissions— are described in detail in the Dennis, R. RADM Report (1995). The remainder of this section provides a summary description of the acid deposition modeling effort.

For sulfur deposition, the RADM Engineering Model (RADM/EM), which focuses on sulfur compounds, was used to derive annual average total (wet plus dry) deposition of sulfur in kilograms sulfur per hectare (kg-S/ha) under both the control and no-control scenarios. The relative changes in annual average total sulfur deposition for each of the 80-km RADM/EM grid cells for 1975, 1980, 1985, and 1990 were then compiled.

Nitrogen deposition was calculated in a different manner. Since nitrogen effects are not included in the computationally fast RADM/EM, nitrogen deposition had to be derived from the full-scale, 15-layer RADM runs. Because of the cost and computational intensity of the 15-layer RADM, nitrogen deposition estimates were only developed for 1980 and 1990. As for sulfur deposition, the relative changes in annual average total (wet plus dry) nitrogen deposition, expressed as kg-

Figure 37. RADM-Predicted 1990 Total Nitrogen Deposition (Wet + Dry; in kg/ha) Under the Control Scenario.

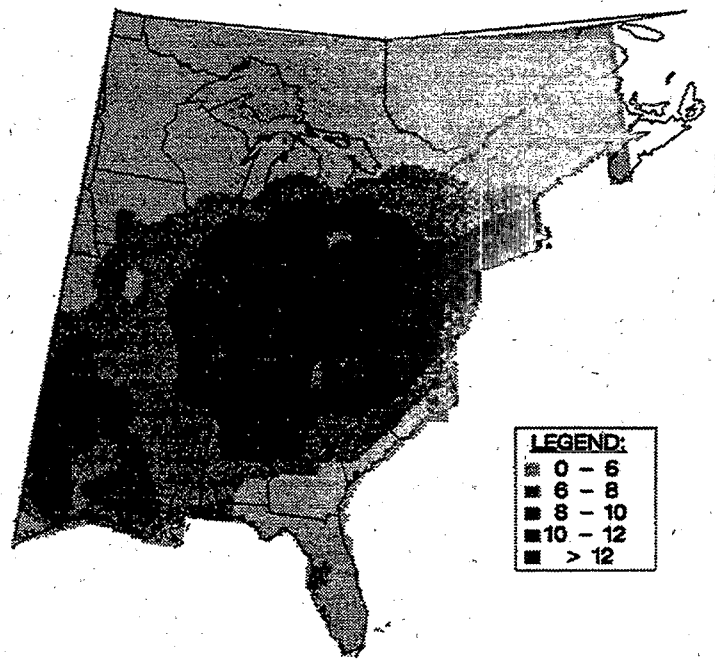
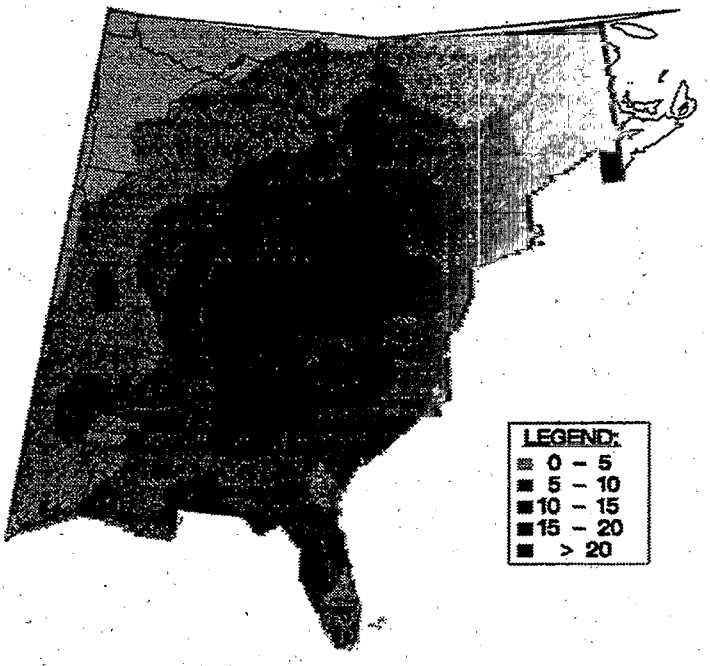


Figure 38. RADM-Predicted 1990 Total Sulfur Deposition (Wet + Dry; in kg/ha) Under the No-control Scenario.



N/ha, were calculated for each 80-km grid cell and for each of the two scenarios. It is important to note that ammonia depositin contributes significantly to total nitrogen deposition. However, the activities of sources associated with formation and deposition of ammonia, such as livestock farming and wildlife, were essentially unaffected by Clean Air Act-related control programs during the 1970 to 1990 period of this analysis. Therefore, ammonia deposition is held constant between the two scenarios.

Summary differences in acid deposition

Figure 40 is a contour map showing the estimated percent increase in sulfur deposition under the no-control scenario relative to the control scenario for 1990. Figure 41 provides comparable information for nitrogen deposition. These maps indicate that by 1990 acid deposition would have been significantly higher across the RADM domain under the no-control scenario.

Examination of the percent change sulfur deposition map indicates relatively large percentage changes in the upper Great Lakes and the Florida-Southeast Atlantic Coast areas. This result may appear somewhat surprising to readers familiar with the historical patterns of acid deposition. However, a review of the emission data and the control scenario sulfur deposition map reveal the reasons for this result.

Figure 39. RADM-Predicted 1990 Total Nitrogen Deposition (Wet + Dry; in kg/ha) Under the No-control Scenario.

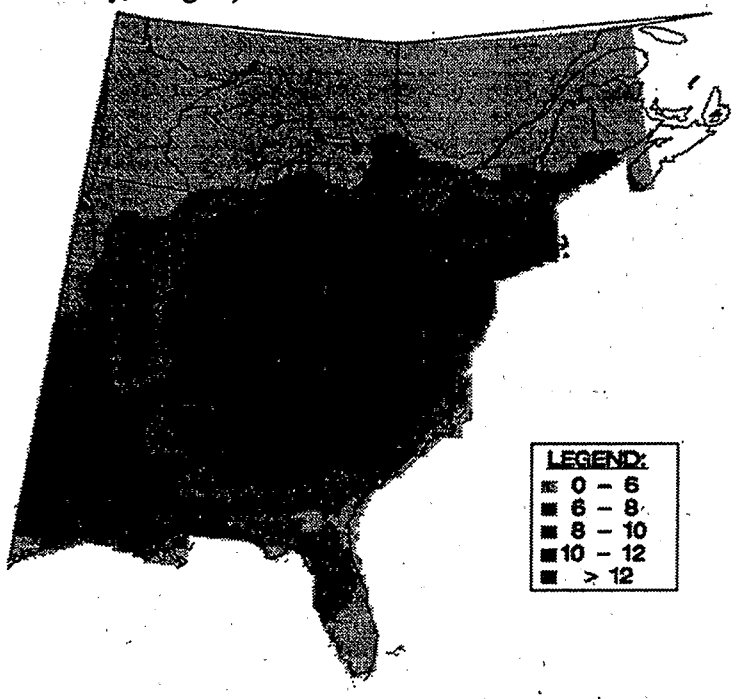
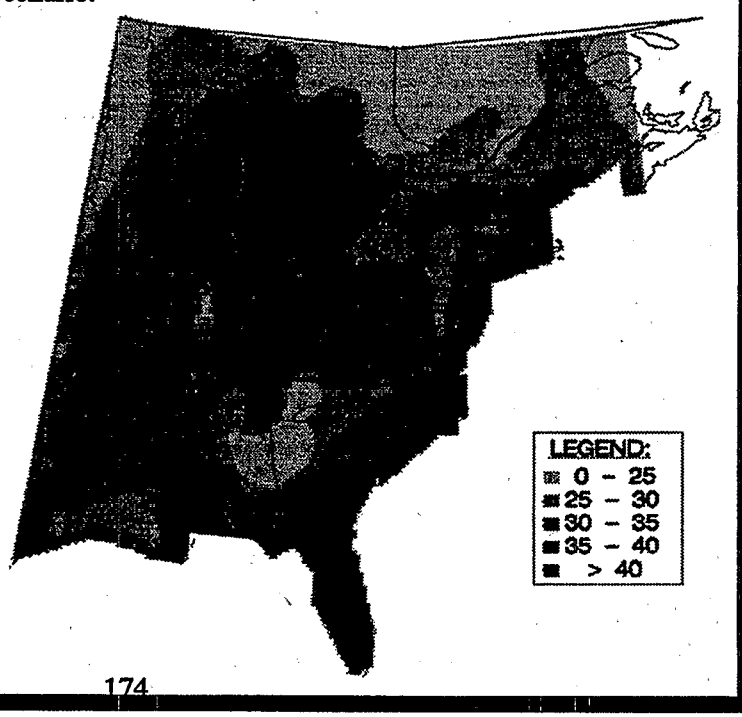


Figure 40. RADM-Predicted Percent Increase in Total Sulfur Deposition (Wet + Dry; in kg/ha) Under the No-control Scenario.



First, Figure 36 shows that control scenario deposition rates are relatively low. As described above, even a small absolute increase in deposition leads to a large percentage increase in areas with low initial rates of deposition. Second, the scenario differences in SO_x emission rates for these areas were substantial. For example, 1990 no-control scenario total SO_x emissions for Michigan were approximately 1.8 million tons but control scenario emissions for the same year were less than 600,000 tons; a reduction of over two-thirds. Similarly, 1990 no-control scenario emissions for Florida were over 2.3 million tons, compared to approximately 800,000 tons under the control scenario; also a reduction of about two-thirds. Almost 1 million tons of the Michigan reduction and approximately 1.3 million tons of the Florida reduction were associated with utilities. Emission reductions of these magnitudes would be expected to yield significant reductions in rates of acid deposition.

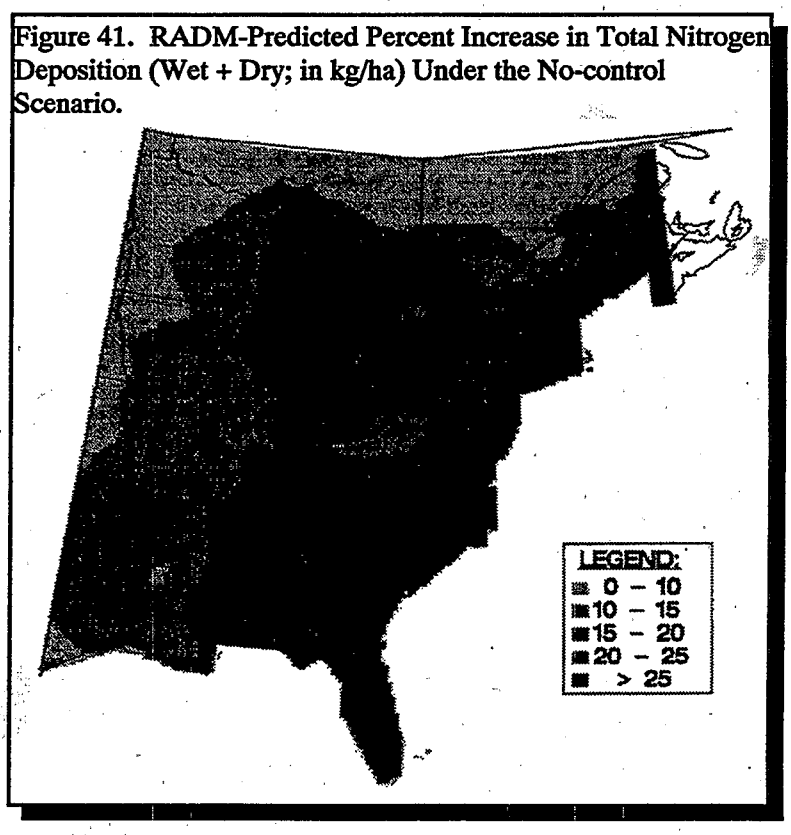
Key caveats and uncertainties for acid deposition

Regional-scale oxidant and deposition modeling involves substantial uncertainty. This uncertainty arises from uncertainties in modeling atmospheric chemistry, incomplete meteorological data, normal seasonal and temporal fluctuations in atmospheric conditions, temporal and spatial variability in emissions, and many other factors. Uncertainties specific to the RADM model, and this particular exercise, are discussed in detail in the Dennis, R. RADM Report (1995). It is important, however, to highlight some of the potential sources of modeling uncertainty unique to this analysis.

The first source of uncertainty specific to this analysis is associated with the spatial and geographic disaggregation of emissions data. As discussed in the Dennis, R. RADM report, the RADM model requires emission inventory inputs which are highly disaggregated over both time and space. The ideal emissions inventory fed into the RADM model includes day-specific emissions to account for temperature effects on VOCs and the significance of localized meteorological conditions around major point sources. Given the broad-scale, comprehensive nature of the present study, such detailed emissions inventories were not available. However, the industry/commercial-level disaggregation approach developed for the present analysis would not be expected to introduce any systematic bias, and the contribution of this disaggregation of emissions would not be expected to contribute significantly to the overall uncertainty of the larger analysis.

The acid deposition estimates included in the present analysis are limited in that only the eastern 31 of the 48 coterminous states are covered. Although acid deposition is a problem primarily for the eastern U.S., acid deposition does occur in states west of the RADM domain. The magnitude of the benefits of reducing acid deposition in these western states is likely to be small, however, relative to the overall benefits of the historical Clean Air Act.

Figure 41. RADM-Predicted Percent Increase in Total Nitrogen Deposition (Wet + Dry; in kg/ha) Under the No-control Scenario.



Particulate Matter

Developing air quality profiles for particulate matter is significantly complicated by the fact that "particulate matter" is actually an aggregation of different pollutants with varying chemical and aerodynamic properties. Particulate species include chemically inert substances, such as wind-blown sand, as well as toxic substances such as acid aerosols; and include coarse particles implicated in household soiling as well as fine particles which contribute to human respiratory effects. In addition, emissions of both primary particulate matter and precursors of secondarily-formed particulates are generated by a wide variety of mobile and stationary sources, further complicating specification of particulate air quality models. Finally, particulate air quality models must take account of potentially significant background concentrations of atmospheric particles.

Modeling multiple species and emission sources, however, is not the only major challenge related to particulate matter which is faced in the present study. Over the 1970 to 1990 period being analyzed, understanding of the relative significance of fine versus coarse particles evolved significantly. Up until the mid-1980's, particulate air quality data were collected as Total Suspended Particulates (TSP). However, during the 1980's, health scientists concluded that small, respirable particles, particularly those with an aerodynamic diameter of less than or equal to 10 microns (PM_{10}), were the component of particulate matter primarily responsible for adverse human health effects. As of 1987, federal health-based ambient air quality standards for particulate matter were revised to be expressed in terms of PM_{10} rather than TSP. Starting in the mid-1980's, therefore, the U.S. began shifting away from TSP monitors toward PM_{10} monitors. As a result, neither TSP nor PM_{10} are fully represented by historical air quality data over the 1970 to 1990 period of this analysis. Furthermore, a large number of U.S. counties have no historical PM monitoring data at all, making it difficult to estimate changes in ambient concentrations of this significant pollutant for areas containing roughly 30 percent of the U.S. population.

Given the relative significance of particulate matter to the bottom-line estimate of net benefits of the historical Clean Air Act, it was important to develop methodologies to meet each of these challenges. The methodologies developed and data used are described primarily in the two supporting documents SAI PM Report (1992) and SAI PM Report (1995).[88] To summarize the overall approach, historical TSP data were broken down into principal component species, including primary particulates, sulfates, nitrates, organic particulates, and background particulates. Historical data were used for the control scenario. To derive the no-control profiles, the four non-background components were scaled up based on corresponding no-control to control ratios of emissions and/or modeled atmospheric concentrations. Specifically, the primary particulate component was scaled up by the ratio of no-control to control emissions of PM. Organic constituents were scaled up by the ratio of no-control to control VOC emissions. In the eastern 31 states where RADM sulfate and nitrate data were available, values for SO_4 and NO_3 from an appropriate RADM grid cell were assigned to the relevant county and used to scale these components of PM. For the western states not covered by RADM, sulfates were scaled up by the change in SO_2 emissions and nitrates were scaled up the change in NO_x emissions. No-control profiles were then constructed by adding these scaled components to background concentrations.

To resolve the problem of variable records of TSP and PM_{10} data, both TSP and PM_{10} profiles were generated for the entire 20 year period. Missing early year data for PM_{10} were derived by applying region-specific, land use category-specific PM_{10} to TSP ratios to the historical TSP data. Missing recent year TSP data were derived for those areas where PM_{10} monitors replaced TSP monitors by applying the reciprocal of the relevant PM_{10} to TSP ratio. The methodology is described in detail in the SAI PM Report (1995).

In addition, to increase estimates of air quality on a geographical basis, an interpolation methodology[89] was developed to predict air quality for the control scenario in counties without measured data. PM concentrations were estimated by first estimating the components of PM (i.e., sulfate, nitrate, and organic particulate, and primary particulate). The methodology for developing the concentrations of components within a county differed depending upon whether the county was within or outside the RADM domain.

For those counties within the RADM domain, the RADM modeled concentrations for 1980 and 1990 were used to predict sulfate air quality. Relationships based on linear regressions that related 1980 and 1990 RADM sulfate concentrations to estimated sulfate particulate concentrations were calculated for counties with AIRS data. Sulfate particulate concentrations were then calculated for all counties in the domain by applying the regression results to the RADM grid cell concentration located over the county center. Statewide average nitrate, VOC, and primary particulate concentrations were calculated from measured ambient TSP and PM_{10} to describe these constituents in counties without data. Control PM profiles were developed by adding the RADM-estimated sulfate particulate levels with the statewide average nitrate, VOC, and primary particulate levels, and background.

For counties outside the RADM domain, an alternate procedure was used. Using the primary and secondary particulate estimates for counties with data, statewide average sulfate, nitrate, VOC, and primary particulate concentrations were determined. Control PM_{10} was predicted by adding the statewide averages of all primary and secondary particulate, and background. Using this method, all counties that did not

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[88] In addition, SAI memoranda and reports which supplement the results and methodologies used in this analysis are included in the references.

[89] The interpolation methodology is described in detail in SAI, 1996. Memo from J. Langstaff to J. DeMocker. PM Interpolation Methodology for the §812 Retrospective Analysis. March 1996.

have monitors and are in the same state are assigned the same PM concentration profiles. These interpolated results are clearly less certain than results based on actual historical monitoring data and are therefore presented separately.

Control scenario particulate matter profiles

The number of TSP monitors peaked in 1977 and declined throughout the 1980's. Table 71 summarizes the daily (i.e., 24-hour average) TSP monitoring data used as the basis for development of the control scenario air quality profiles. Most of the TSP and PM_{10} monitors collected samples every six days (i.e., 61 samples per year).

Table 71. Summary of TSP Monitoring Data.

Year	Number of Monitors	Number of Counties	Number of Samples	Mean Number of Samples per Monitor
1970	751	245	56,804	76
1975	3,467	1,146	221,873	64
1980	3,595	1,178	234,503	65
1985	2,932	1,018	189,344	65
1990	923	410	59,184	64

Data Source: SAI PM Report (1995).

Table 72. Summary of PM_{10} Monitoring Data.

Year	Number of Monitors	Number of Counties	Number of Samples	Mean Number of Samples per Monitor
1985	303	194	22,031	73
1990	1,249	556	98,904	79

Data Source: SAI PM Report (1995).

Daily PM_{10} data were also collected for each year between 1983 and 1990. Table 72 summarizes the daily PM_{10} monitoring data used for the control scenario air quality profiles.

Further speciation of TSP and PM_{10} air quality data serves two purposes in the present analysis. First, speciation of TSP into PM_{10} and other fractions allows derivation of PM_{10}:TSP ratios. Such ratios can then be used to estimate historical PM_{10} for those years and monitors which had TSP data but no PM_{10} data. The reciprocal ratio is also applied in this analysis to expand 1985 and 1990 TSP data to cover those areas which monitored PM_{10} but not TSP. The second purpose served by speciation of particulate data is, as described on page 177, to provide a basis for scaling up concentrations of each species to derive no-control scenario TSP and PM_{10} profiles.

To break the TSP and PM_{10} data down into component species, speciation factors were applied to the PM fractions with aerodynamic diameters below 2.5 microns ($PM_{2.5}$) and from 2.5 to 10 microns (PM_{10}).

The $PM_{2.5}$ speciation factors were drawn from a National Acid Precipitation Assessment Program (NAPAP) report on visibility which reviewed and consolidated speciation data from a number of studies.[90] These factors are presented in Table 73. In the table, fine particle concentrations are based on particle mass measured after equilibrating to a relative humidity of 40 to 50 percent; and organics include fine organic carbon.

To develop speciation factors for coarser particles (i.e., in the $PM_{2.5}$ to PM_{10} range), a review was performed by SAI of the available literature, including Conner et al. (1991), Wolff and Korsog (1989), Lewis and Macias (1980), Wolff et al. (1983), Wolff et al. (1991), and Chow et al. (1994).[91] These speciation factors are summarized in Table 74. Data were too limited to allow differentiation between urban and rural locations for coarser particles.

[90] J. Trijonis, "Visibility: Existing and Historical Conditions--Causes and Effects," NAPAP Report 24, 1990.

[91] This literature review, and complete citations of the underlying studies, are presented in the SAI PM Report (1995), pp. 4-2 to 4-6 and pp. R-1 to R-2, respectively.

Table 73. Fine Particle ($PM_{2.5}$) Chemical Composition by U.S. Region.

Component	Units	Number of Data Sets	Arithmetic Mean	Range of Values
RURAL EAST				
Fine particle concentration	$\mu g/m_3$	19	18	6 - 46
Ammonium sulfate	% Fine particles	19	52	41 - 66
Ammonium nitrate	% Fine particles	3	1	1
Organics	% Fine particles	5	24	9 - 34
URBAN EAST				
Fine particle concentration	$\mu g/m_3$	3	36	29 - 43
Ammonium sulfate	% Fine particles	3	55	53 - 57
Ammonium nitrate	% Fine particles	2	1	1
Organics	% Fine particles	2	24	15 - 32
RURAL WEST				
Fine particle concentration	$\mu g/m_3$	25	5	1 - 11
Ammonium sulfate	% Fine particles	25	35	15 - 56
Ammonium nitrate	% Fine particles	17	4	1 - 17
Organics	% Fine particles	25	27	14 - 41
URBAN WEST				
Fine particle concentration	$\mu g/m_3$	16	35	13 - 74
Ammonium sulfate	% Fine particles	16	16	3 - 35
Ammonium nitrate	% Fine particles	14	15	2 - 37
Organics	% Fine particles	16	42	25 - 79

Data Sources: SAI PM Report (1995); and J. Trijonis, *"Visibility: Existing and Historical Conditions--Causes and Effects,"* NAPAP Report 24, 1990.

Table 74. Coarse Particle ($PM_{2.5}$ to PM_{10}) Chemical Composition by U.S. Region.

Component	Units	Number of Data Sets	Arithmetic Mean	Range of Values
EAST				
Coarse particle concentration	$\mu g/m_3$	1	5.5	5.5
Ammonium sulfate	% Coarse particles	3	3	1 - 4
Ammonium nitrate	% Coarse particles	1	4	4
Organics	% Coarse particles	2	10	7 - 13.8
WEST				
Coarse particle concentration	$\mu g/m_3$	18	24	7.7 - 56.7
Ammonium sulfate	% Coarse particles	18	6	2.1 - 10.39
Ammonium nitrate	% Coarse particles	18	18	2.33-28.52
Organics	% Coarse particles	18	14	8.41-25.81

Data Source: SAI PM Report (1995).

The TSP and PM_{10} control scenario profiles developed based on this methodology are available on diskette, under the filenames listed in Table 75.

No-control scenario particulate matter profiles

To derive the no-control TSP and PM_{10} air quality profiles, individual component species were adjusted to reflect the relative change in emissions or, in the case of sulfates and nitrates in the eastern U.S., the relative change in modeled ambient concentration. The following excerpt from the SAI PM Report (1995) describes the specific algorithm used:[92]

"For the retrospective analysis, the no-CAA scenario TSP and PM_{10} air quality was estimated by means of the following algorithm:

- *Apportion CAA scenario TSP and PM_{10} to size categories and species;*
- *Adjust for background concentrations;*

[92] SAI PM Report (1995), p. 5-1.

- *Use a linear scaling to adjust the non-background portions of primary particulates, sulfate, nitrate, and organic components based on emissions ratios of PM, SO_2, NO_x and VOC, and Regional Acid Deposition Model (RADM) annual aggregation results for SO_4 and NO_3;*

- *Add up the scaled components to estimate the no-CAA scenario TSP and PM_{10} concentrations."*

The specific procedures and values used for the linear rollback, speciation, fine to coarse particle ratio, scaling, and background adjustment steps are described in detail in the SAI PM report (1995).[93] Table 76 lists the names of the electronic data files containing the TSP and PM_{10} profiles for the no-control scenario.

Table 75. PM Control Scenario Air Quality Profile Filenames.

Component	Indicator	Filename
TSP	Annual Mean	TSPCMEAN.DAT
TSP	2nd Highest Daily	TSPCHI2.DAT
TSP	(X)th Percentile	TSPC(X).DAT
PM_{10}	Annual Mean	PM10CMEA.DAT
PM_{10}	2nd Highest Daily	PM10CHI2.DAT
PM_{10}	(X)th Percentile	PM10C(X).DAT

Note: "(X)" refers to percentiles from 5 to 95, indicating 19 percentile data files available for TSP and 19 files available for PM_{10}; for example, the filename for the 50th percentile TSP air quality data profile for the control scenario is named TSPC50.DAT.

Table 76. PM No-Control Scenario Air Quality Profile Filenames.

Component	Indicator	Filename
TSP	Annual Mean	TSPCNMEA.DAT
TSP	2nd Highest Daily	TSPNCHI.DAT
TSP	(X)th Percentile	TSPNC(X).DAT
PM_{10}	Annual Mean	PM10NCME.DAT
PM_{10}	2nd Highest Daily	PM10NCHI.DAT
PM_{10}	(X)th Percentile	PM10NC(X).DAT

Note: "(X)" refers to percentiles from 5 to 95, indicating 19 percentile-based data files available for TSP and 19 similar files available for PM_{10}; for example, the filename for the 50th percentile TSP air quality data profile for the no-control scenario is named TSPNC50.DAT.

[93] SAI PM Report (1995), pp. 5-2 to 5-15.

Summary differences in particulate matter air quality

Figure 42. Distribution of Estimated Ratios for 1990 Control to No-Control Annual Mean TSP Concentrations, by Monitored County.

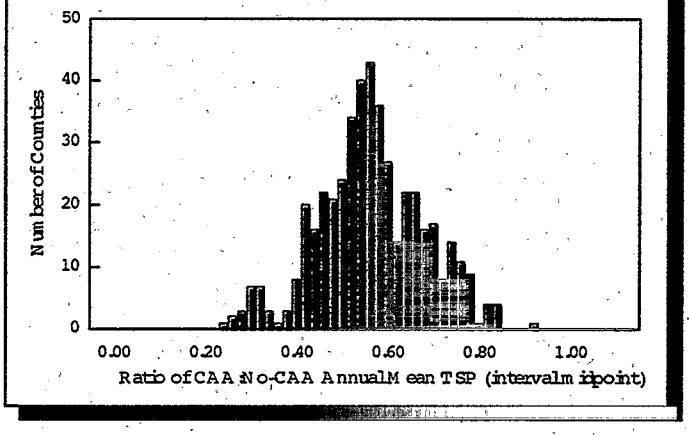


Figure 42 provides one indication of the estimated change in particulate matter air quality between the control and no-control scenarios. Specifically, the graph provides data on the estimated ratios of 1990 control to no-control scenario annual mean TSP concentrations in monitored counties. The X-axis values represent the mid-point of the ratio interval bin, and the Y-axis provides the number of counties falling into each bin. Figure 42 indicates that annual average TSP concentrations would have been substantially higher in monitored counties under the no-control scenario.

Key caveats and uncertainties for particulate matter

There are several important caveats and uncertainties associated with the TSP and PM_{10} air quality profiles developed for this study. Although further reductions in these uncertainties were not possible for this study given time and resource limitations, the relative importance of particulate matter reduction contributions towards total benefits of the Clean Air Act highlights the importance of these uncertainties.

A number of uncertainties were introduced in the process of speciating and rolling up individual components of particulate matter. First, temporal and spatial variability in the size and chemical properties of particulate emissions are substantial. These characteristics change from day to day at any given location. Second, using changes in proxy pollutant emissions, such as using SO_2 as a surrogate for SO_4 in the western states, to roll up individual PM components may introduce significant uncertainty. Third, even assuming a satisfactorily high degree of correlation between target and surrogate pollutants, relying on predicted changes in emissions at the state level further compounds the uncertainty. Finally, and perhaps most important, using PM_{10} to TSP ratios derived from late 1980's monitoring data may lead to significant underestimation of reductions in fine particulates achieved in earlier years. This is because historical Clean Air Act programs focused extensively on controlling combustion sources of fine particulates. As a result, the share of TSP represented by PM_{10} observed in the late 1980's would be lower due to implementation of controls on combustion sources. This would lead, in turn, to underestimation of baseline PM_{10} concentrations, as a share of TSP, in the 1970's and early 1980's. If baseline PM_{10} concentrations in these early years are underestimated, the reductions in PM_{10} estimated by linear scaling would also be underestimated.[94]

[94] See SAI PM Report (1995), p. 5-9.

Ozone

Nonlinear formation processes, long-range atmospheric transport, multiple precursors, complex atmospheric chemistry, and acute sensitivity to meteorological conditions combine to pose substantial difficulties in estimating air quality profiles for ozone. Even in the context of an aggregated, national study such as this, the location-specific factors controlling ozone formation preclude the use of roll-up modeling based on proxy pollutants or application of state-wide or nation-wide average conditions. Such simplifications would yield virtually meaningless results for ozone.

Ideally, large-scale photochemical grid models —such as the Urban Airshed Model (UAM)— would be used to develop control and no-control scenario estimates for ozone concentrations in rural and urban areas. Such models provide better representations of the effects of several important factors influencing air quality projections such as long-range atmospheric transport of ozone. However, the substantial computing time and data input requirements for such models precluded their use for this study.[95] Instead, three separate modeling efforts were conducted to provide urban and rural ozone profiles for those areas of the lower 48 states in which historical ozone changes attributable to the Clean Air Act may be most significant.

First, for urban areas the Ozone Isopleth Plotting with Optional Mechanisms-IV (OZIPM4) model was run for 147 urban areas. Table 77 lists the urban areas modeled with OZIPM4. Although it requires substantially less input data than UAM, the OZIPM4 model provides reasonable evaluations of the relative reactivity of ozone precursors and ozone formation mechanisms associated with urban air masses.[96] Three to five meteorological episodes were modeled for each of the 147 urban areas; and for each of these, four model runs were performed to simulate the 1980 and 1990 control and no-control scenarios. The outputs of these model runs were peak ozone concentrations for each of the target year-scenario combinations. The differentials between the control and no-control scenario outputs were averaged over meteorological episodes and then applied to scale up historical air quality at individual monitors to obtain no-control case profiles. As for the other pollutants, the control scenario profiles were derived by fitting statistical distributions to actual historical data for individual monitors.

[95] For a description of the extensive data inputs required to operate UAM, see SAI Ozone Report (1995), p. 1-1.

[96] See SAI Ozone Report (1995), p. 1-1.

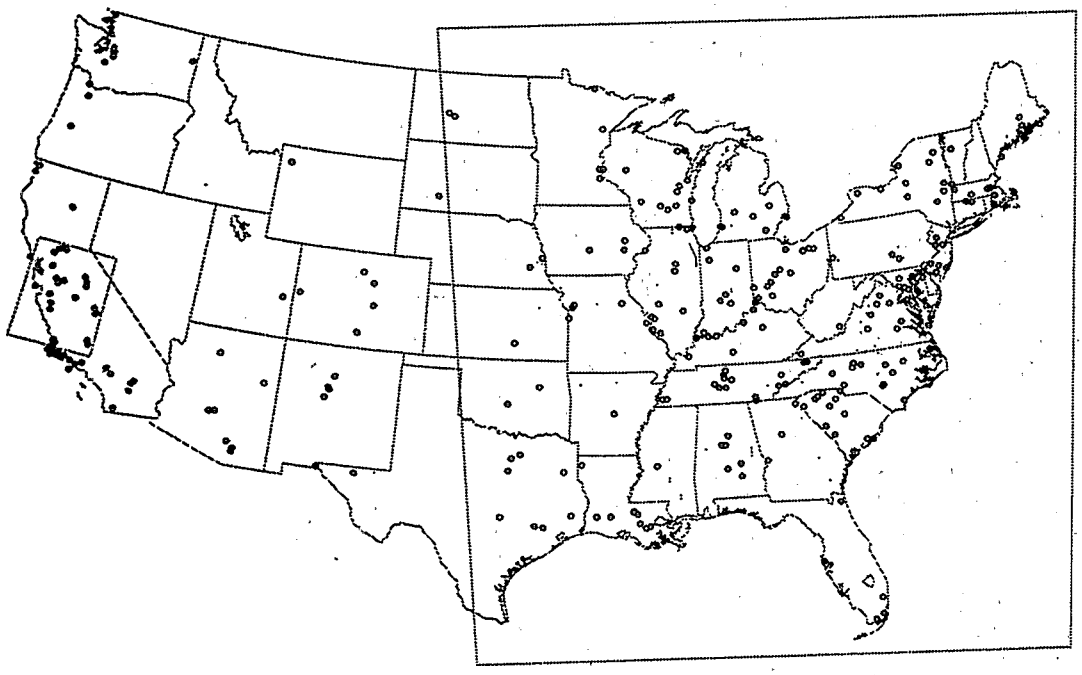
Table 77. Urban Areas Modeled with OZIPM4.

Albany, NY	Fort Wayne, IN	Owensboro, KY
Albuquerque, NM	Grand Rapids, MI	Parkersburg, WV
Allentown, PA-NJ	Greeley, CO	Pascagoula, MS
Altoona, PA	Green Bay, WI	Pensacola, FL
Anderson, IN	Greensboro, NC	Peoria, IL
Appleton, WI	Greenville, SC	Philadelphia, PA
Asheville, NC	Harrisburg, PA	Phoenix, AZ
Atlanta, GA	Hartford, CT	Portland, OR
Atlantic City, NJ	Houston, TX	Portsmouth, NH
Auburn, ME	Huntington, WV-KY	Raleigh, NC
Augusta, GA-SC	Huntsville, AL	Reading, PA
Austin, TX	Indianapolis, IN	Reno, NV
Baltimore, MD	Iowa City, IA	Richmond, VA
Baton Rouge, LA	Jackson, MS	Roanoke, VA
Beaumont, TX	Jacksonville, FL	Rochester, NY
Bellingham, WA	Janesville Rock Co, WI	Rockford, IL
Billings, MT	Johnson City, TN-VA	Sacramento, CA
Birmingham, AL	Johnstown, PA	Salt Lake City, UT
Boston, MA	Kansas City, MO	San Antonio, TX
Boulder, CO	Knoxville, TN	San Diego, CA
Canton, OH	Lafayette, IN	San Francisco, CA
Cedar Rapids, IA	Lafayette, LA	San Joaquin Valley, CA
Champaign, IL	Lake Charles, LA	Santa Barbara, CA
Charleston, SC	Lancaster, PA	Sarasota, FL
Charleston, WV	Lansing, MI	Scranton, PA
Charlotte, NC	Las Cruces, NM	Seattle, WA
Chattanooga, TN-GA	Las Vegas, NV	Sheboygan, WI
Chicago, IL	Lexington, KY	Shreveport, LA
Cincinnati, OH	Lima, OH	South Bend, IN
Cleveland, OH	Little Rock, AR	Springfield, IL
Colorado Springs, CO	Longview, TX	Springfield, MO
Columbia, SC	Los Angeles, CA	Springfield, OH
Columbus, GA-AL	Louisville, KY	St Louis, MO
Columbus, OH	Lynchburg, VA	Steubenville, OH-WV
Corpus Christi, TX	Medford, OR	Syracuse, NY
Cumberland, MD-WV	Memphis, TN	Tallahassee, FL
Dallas, TX	Miami, FL	Tampa, FL
Davenport, IA-IL	Minneapolis, MN-WI	Terre Haute, IN
Decatur, IL	Mobile, AL	Toledo, OH
Denver, CO	Monroe, LA	Tucson, AZ
Detroit, MI	Montgomery, AL	Tulsa, OK
El Paso, TX	Nashville, TN	Utica-Rome, NY
Erie, PA	New Orleans, LA	Ventura County, CA
Eugene, OR	New York, NY	Victoria, TX
Evansville, IN	Norfolk, VA	Washington, DC
Fayetteville, NC	Oklahoma City, OK	Wheeling, WV-OH
Flint, MI	Omaha, NE-IA	Wichita, KS
Fort Collins, CO	Orange Co, CA	York, PA
Fort Smith, AR-OK	Orlando, FL	Youngstown, OH-PA

Second, the 15-layer RADM runs for 1980 and 1990 were used to estimate the relative change in rural ozone distributions for the eastern 31 states. In addition, a limited number of 20-km grid cell high-resolution RADM runs were conducted to benchmark the 15-layer, 80-km RADM median ozone response and to estimate high ozone response. The relative changes in modeled median and 90th percentile rural ozone were then assumed to be proportional to the changes in, respectively, the median and 90th percentile ozone concentrations. The domain of the high-resolution RADM is shown in Figure 35 and the general RADM domain is shown in Figure 43.

Finally, the SARMAP Air Quality Model (SAQM) was run for EPA by the California Air Resources Board (CARB) to gauge the differences in peak ozone concentrations in key California agricultural areas for 1980 and 1990. No-control profiles were developed for ozone monitors in these areas by assuming the relative change in peak ozone concentration also applies to the median of the ozone distribution. The domain of the SAQM is shown in Figure 43.

Figure 43. RADM and SAQM Modeling Domains, with Rural Ozone Monitor Locations.



Control scenario ozone profiles

For ozone, air quality profiles were developed from historical AIRS data and calculated for individual monitors based on 1, 2, 6, 12, and 24 hour averaging times. Profiles based on the daily maximum concentrations for these averaging times were also calculated. Given the significance of seasonal and diurnal ozone formation, twelve separate profiles of hourly ozone distributions were also developed for six 2-month periods and for daytime and nighttime hours. The 2-month periods are January-February, March-April, and so forth. The diurnal/nocturnal profiles are divided at 7 A.M. and 7 P.M. Local Standard Time. All of these profiles are based on constructing 1, 2, 6, 12, and 24-hour moving average profiles from the hourly ozone data from each monitor.[97] A two-parameter gamma distribution is then fitted to characterize each of these air quality profiles.[98] The functional form of the gamma distribution, the basis for deriving the monitor-specific values for mean and variance, and an analysis of the goodness of fit to the data are presented in the SAI Ozone Report (1995).

Table 78 summarizes the ozone monitoring data used as the basis for the control scenario profiles. The distribution of these monitors among urban, suburban, and rural locations is presented in Table C-2 of the SAI Ozone Report (1995).

Given the substantial number of alternative air quality profiles for ozone, approximately 20 high-density disks are required to hold the profiles, even in compressed data format. Resource limitations therefore preclude general distribution of the actual profiles. As discussed in the caveats and uncertainties subsection below, however, the substantial uncertainties associated with model results for any given area preclude application of these profiles in contexts other than broad-scale, aggregated assessments such as the present study. The historical ozone monitoring data used as the basis for this study are, nevertheless, available through EPA's Aerometric Information Retrieval System (AIRS).

Table 78. Summary of Ozone Monitoring Data.

Year	Number of Monitors	Number of Counties
1970	1	1
1975	467	240
1980	791	415
1985	719	415
1990	834	477

Data Source: SAI Ozone Report (1995).

No-control scenario ozone profiles

The specific modeling methodologies for the OZIPM4 runs —including emissions processing, development of initial and boundary conditions, meteorological conditions, simulation start and end times, organic reactivity, and carbon fractions— are described in detail in the SAI Ozone Report (1995).

[97] For the nighttime profiles, only 1, 2, 6, and 12-hour averaged concentrations are derived.

[98] Normal and lognormal distributions were also developed and tested for goodness of fit; however, the gamma distribution provided a better representation of the concentration distribution. See SAI Ozone Report (1995), page 4-2.

Assumptions and modeling procedures not otherwise described in the SAI report were conducted in accordance with standard EPA guidance.[99]

Similarly, the RADM modeling methodology used to estimate changes in day-time rural ozone distributions in the eastern 31 states are described in detail in the Dennis, R. RADM Report (1995). The referenced report also provides complete citations of the literature associated with development, standard application procedures, and evaluation of RADM by the National Acid Precipitation Assessment Program (NAPAP).

To derive the no-control scenario results for key California agricultural areas, the California Air Resources Board and US EPA's Region 9 office agreed to conduct three runs of the SAQM. For the 1990 control scenario, the 1990 SARMAP base case scenario adopted for California State Implementation Plan modeling was adopted.[100] Derivation of 1990 no-control and 1980 control and no-control scenarios was based on adjusting the aggregate mobile, point, and area source VOC and NO_x emissions associated with each of these cases. For example, the 1980 no-control results were derived by, first, multiplying the 1990 SARMAP base case mobile source VOC emissions by the ratio of 1980 no-control scenario to 1990 control scenario mobile source VOC emissions derived for the present study. Similar adjustments were made for point and area sources, and for NO_x. The SAQM was then re-run holding fixed all other conditions associated with the 1990 SARMAP base case, including meteorology, activity patterns, and other conditions. The specific emission ratios used to modify the 1990 SARMAP base case are presented in Table 79. The ratios themselves were derived by adding on-highway and off-highway emissions to represent the mobile source category; adding utility, industrial process, and industrial combustion emissions to represent point sources; and using commercial/residential emissions to represent area sources. The no-control scenarios were then derived by adjusting the peak and median of the control scenario ozone distribution based on the ratio of SARMAP-predicted peak ozone concentrations under the control and no-control scenarios.

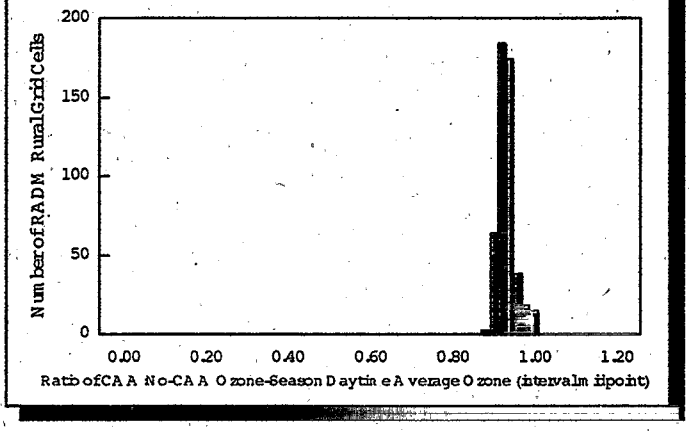
Table 79. Apportionment of Emissions Inventories for SAQM Runs.

	Source Category	1980 Control to 1990 Control Ratio	1980 No-Control to 1990 Control Ratio	1990 No-Control to 1990 Control Ratio
VOC	Mobile	1.344	1.955	3.178
	Area	0.820	0.901	1.106
	Point	1.284	1.439	1.232
NO_x	Mobile	1.042	1.148	1.677
	Area	0.731	0.738	1.058
	Point	0.987	1.339	1.159

[99] US EPA, Office of Air Quality Planning and Standards, *"Procedures for Applying City-Specific EKMA,"* EPA-450/4-89-012, 1989.

[100] Documentation of the SARMAP Air Quality Model and the SARMAP 1990 base case can be found in the SAQM references listed at the end of this appendix.

Figure 44. Distribution of Estimated Ratios for 1990 Control to No-control RADM-Simulated Daytime Average Rural Ozone Concentrations, by RADM Grid Cell.



The relative results of the control and no-control scenario runs of the OZIPM4, RADM, and SAQM models were then used to derive the no-control case air quality profiles. For the urban monitors relying on OZIPM4 results, only ozone-season daytime concentrations could be calculated directly from OZIPM4 results. This is because OZIPM4 provides only the maximum hourly ozone concentration. However, to estimate all the various physical consequences of changes in ambient ozone concentrations, the current study requires estimation of the shift in the entire distribution of ozone concentrations. Since it is daytime ozone season concentrations which are most sensitive to changes in VOC and NO_x emissions, the predicted shifts in the most important component of the ozone concentration distribution are reasonably well-founded. The method adopted for this analysis involved applying the no-control to control peak concentration ratio to all concentrations in the distribution down to a level of 0.04 ppm. The 0.04 ppm level is considered at the high end of hypothetical ambient ozone concentrations in the absence of all anthropogenic ozone precursor emissions. A ratio of 1.0 is used for ozone concentrations at or near zero. The methodology is described in more detail in the SAI Ozone Report (1995) on page 4-6.

Estimating changes in rural ozone concentrations is required primarily for estimating effects on agricultural crops, trees, and other vegetation. For this reason, only the differences in daytime, growing season ozone concentrations are derived for the present study. As described in detail in the SAI Ozone Report (1995) on page 4-7, the no-control rural ozone profiles are calculated by, first, taking the ratio of the average daytime growing season ozone concentrations simulated by RADM or SAQM (whichever is relevant for that monitor). The ratio of no-control to control scenario average ozone concentration is then applied to all the hourly concentrations from that monitor. Profiles based on 1, 2, 6, 12, and 24-hour averages are then calculated for the control case; and averages for daytime hours are calculated for the no-control case.[101] Even though the control and no-control scenario off-season profiles are held constant, profiles for the no-control scenario are developed for all months of the year since the ozone season varies throughout the country.

[101] The no-control scenario nighttime profiles are assumed to be the same as the control scenario profiles.

Summary differences in ozone air quality

Figure 46 presents a summary of the results of the 1990 OZIPM4 results for all 147 of the modeled urban areas. Specifically, the graph depicts a frequency distribution of the ratio of control to no-control scenario peak ozone. While the vast majority of simulated peak ozone concentration ratios fall below 1.00, eight urban areas show lower simulated peak ozone for the no-control scenario than for the control scenario. For these eight urban areas, emissions of precursors were higher under the no-control scenario; however, the high proportion of ambient NO_x compared to ambient non-methane organic compounds (NMOCs) in these areas results in a decrease in net ozone production when NO_x emissions increase. Figures 44 and 45 present frequency distributions for control to no-control ratios of average ozone-season daytime ozone concentrations at rural monitors as simulated by RADM and SAQM, respectively.

These figures indicate that, by 1990, no-control scenario ozone concentrations in the modeled areas would have been generally higher in both urban and rural areas. Rural area concentrations differences are not as great as urban area differences due to (a) the differentially greater effect of CAA emission controls in high population density areas, and (b) potential differences in the models used for urban and rural areas.

Ozone reductions in both rural and urban areas projected in this analysis are not as proportionally large as the estimated reductions in emissions of ozone precursors for at least four reasons. First, current knowledge of atmospheric photochemistry suggests that ozone reductions resulting from emissions changes will be proportionally smaller than the emissions reductions. Second, biogenic emissions of VOCs, an important ozone precursor, are significant and are held constant for the control and no-control scenarios of this analysis. Biogenic emissions are important because they contribute roughly half of the total (manmade plus natural) VOC emissions nationwide. Due to this abundance of VOC loading and the inherent nonlinearity of the ozone-precursor

Figure 45. Distribution of Estimated Ratios for 1990 Control to No-control SAQM-Simulated Daytime Average Ozone Concentrations, by SAQM Monitor.

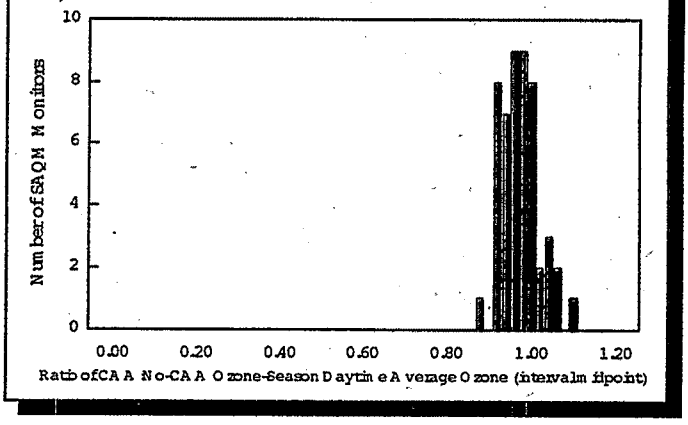
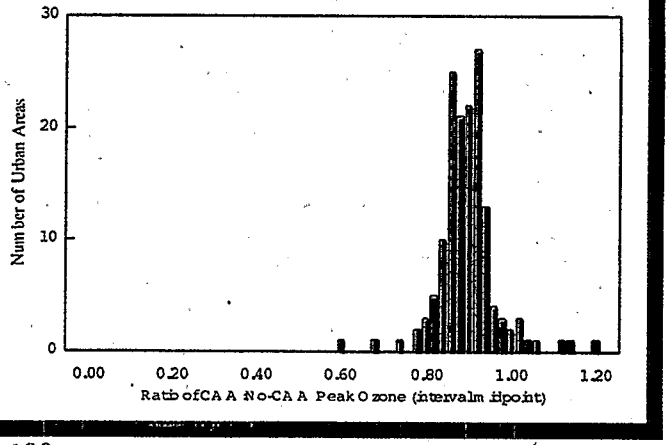


Figure 46. Distribution of Estimated Ratios for 1990 Control to No-control OZIPM4 Simulated 1-Hour Peak Ozone Concentrations, by Urban Area.



response system,[102] historical reductions in anthropogenic VOC emissions can yield minimal reductions in ozone, especially in rural environments. Third, this rural effect also influences urban areas receiving substantial ozone transported in from surrounding areas. Consequently, the effect of emission controls placed in urban areas often is reduced since much of the urban area ozone is imported. Thus, the problem is truly regionalized given the importance of transport, biogenic emissions and associated urban-rural interactions; all contributing toward a relatively non-responsive atmospheric system.[103] Finally, physical process characterizations within OZIPM4 are severely limited and incapable of handling transport, complex flow phenomena, and multi-day pollution events in a physically realistic manner. Consequently, it is possible that the OZIPM4 method used herein produces negative bias tendencies in control estimations. Additional discussion of uncertainties in the ozone air quality modeling is presented in the following section.

Key caveats and uncertainties for ozone

There are a number of uncertainties in the overall analytical results of the present study contributed by the ozone air quality modeling in addition to the potential systematic downward bias discussed above. First, there are substantial uncertainties inherent in any effort to model ozone formation and dispersion. These uncertainties are compounded in the present study by the need to perform city-specific air quality modeling using OZIPM4, which is less sophisticated than an Eulerian model such as the Urban Airshed Model. However, while the absolute ozone predictions for any given urban area provided by OZIPM4 may be quite uncertain, the process of aggregating results for a number of cities and meteorological episodes should significantly reduce this uncertainty.[104] Urban areas for which ozone changes may be overpredicted are offset to some degree by urban areas for which the change in ozone concentrations may be underpredicted. In weighing the significance of this source of uncertainty, it is important to consider the central purpose of the present study, which is to develop a reasonable estimate of the overall costs and benefits of all historical Clean Air Act programs. All analyses are based on relative modeled results, and ratios of the model predictions for the control and no-control scenarios, rather than the absolute predictions. As a result of this, the effect of any bias in the model predictions is greatly reduced due to partial cancellation.

Additional uncertainty is contributed by other limitations of the models, the supporting data, and the scope of the present analysis. Relying on linear interpolation between 1970 and modeled 1980 results to derive results for 1975, and between modeled results for 1980 and 1990 to derive results for 1985, clearly adds to the uncertainty associated with the RADM-based rural ozone estimates. Assuming that changes in peak concentration predicted by OZIPM4 and SAQM can be applied to scale hourly ozone values

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[102] Nonlinear systems are those where a reduction in precursors can result in a wide range of responses in secondary pollutants such as ozone. Ozone response often is "flat" or nonresponsive to reductions of VOCs in many rural areas with significant natural VOC emissions. Also, ozone can increase in response to increases in NO_x emissions in certain localized urban areas.

[103] Both the 1990 CAA and EPA's and the National Academy of Science's Section 185B Report to Congress recognized the consequences of biogenics, transport and the need to conduct regionalized assessments, as reflected in organizational structures such as the Ozone Transport Commission and the North American Research Strategy for Tropospheric Ozone (NARSTO).

[104] Note that aggregating individual urban area results may reduce the effect of uncertainty in individual city projections (i.e., overestimated cities would offset underestimated cities). However, aggregation of individual urban area results would not reduce potential errors caused by systematic biases which arise due to, for example, misestimated emissions inventories.

throughout the concentration distribution also contributes to uncertainty. Resource and model limitations also required that night-time ozone concentrations be held constant between the scenarios. This leads to an underestimation of the night-time component of ozone transport. Finally, changes in rural ozone in areas not covered by RADM or SAQM could not be estimated. As a result, potentially significant changes in ambient ozone in other major agricultural areas, such as in the mid-west, could not be developed for this analysis. The Project Team considered using an emissions scaling (i.e., a roll-back) modeling strategy to develop crude estimates of the potential change in rural ozone concentrations in monitored areas outside the RADM and SAQM domains. However, the Project Team concluded that such estimates would be unreliable due to the nonlinear effect on ozone of precursor emission changes. Furthermore, the Team concluded that baseline levels of ozone and changes in precursor emissions in these areas are relatively low. The decision not to spend scarce project resources on estimating ozone changes in these rural areas is further supported by the relatively modest change in rural ozone concentrations estimated within the RADM and SAQM domains.

Visibility

Two separate modeling approaches were used to estimate changes in visibility degradation in the eastern and southwestern U.S. These are the two regions of the coterminous U.S. for which Clean Air Act programs were expected to have yielded the most significant reductions in visibility degradation. Visibility changes in the eastern 31 states were estimated based on the RADM/EM results for sulfates; and changes in visibility in 30 southwestern U.S. urban areas were calculated using a linear emissions scaling approach. Despite the potential significance of Clean Air Act-related visibility changes in southwestern U.S. Class I areas, such as National Parks, resource limitations precluded implementation of the analysis planned for these areas.

The RADM/EM system includes a post-processor which computes various measures of visibility degradation associated with changes in sulfate aerosols.[105] The basic approach is to allocate the light extinction budget for the eastern U.S. among various aerosols, including particulate sulfates, nitrates, and organics. The change in light extinction from sulfates is provided directly by RADM, thereby reflecting the complex formation and transport mechanisms associated with this most significant contributor to light extinction in the eastern U.S. Nitrates are not estimated directly by RADM. Instead, RADM-estimated concentrations of nitric acid are used as a surrogate to provide the basis for estimating changes in the particulate nitrate contribution to light extinction. The organic fractions were held constant between the two scenarios. Standard outputs include daylight distribution of light extinction, visual range, and DeciViews[106] for each of RADM's 80-km grid cells. For the present study, the RADM visibility post-processor was configured to provide the 90th percentile for light extinction and the 10th percentile for visual range to represent worst cases; and the 50th percentile for both of these to represent average cases. More detailed documentation of the RADM/EM system and the assumptions used to configure the visibility calculations are presented in the Dennis, R. RADM Report (1995).

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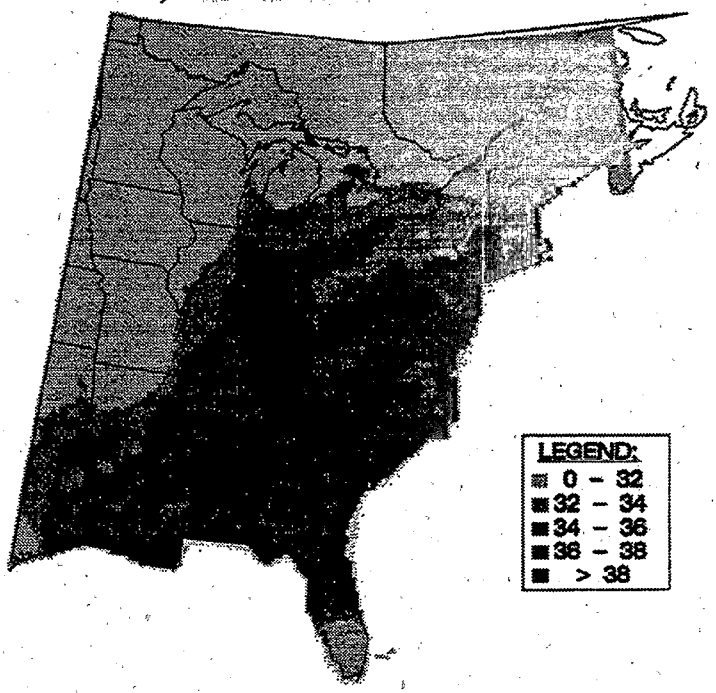
[105] A complete discussion, including appropriate references to other documents, of the RADM and RADM/EM modeling conducted for the present study is presented in the subsection on acid deposition earlier in this appendix.

[106] The DeciView Haze Index (dV) is a relatively new visibility indicator aimed at measuring visibility changes in terms of human perception. It is described in detail in the SAI SW Visibility Report (1994), pp. 4-2 to 4-3.

To estimate differences in control and no-control scenario visibility in southwestern U.S. urban areas, a modified linear rollback approach was developed and applied to 30 major urban areas with population greater than 100,000.[107] For each of the 30 urban centers, seasonal average 1990 air quality data was compiled for key pollutants contributing to visibility degradation in southwestern U.S. coastal and inland cities, including NO_2 and PM_{10}. PM_{10} was then speciated into its key components using city-specific annual average PM_{10} profile data. After adjusting for regional —and for some species city-specific— background levels, concentrations of individual light-attenuating species were scaled linearly based on changes in emissions of that pollutant or a proxy pollutant.[108] Using the same approach used for the 1993 EPA Report to Congress on effects of the 1990 Clean Air Act Amendments on visibility in Class I areas, light extinction coefficients for each of these species were then multiplied by their respective concentrations to derive a city-specific light extinction budget.[109] This process was repeated for pre-1990 control and all no-control scenarios by scaling 1990 results by the relative change in annual county-level emissions of SO_x, NO_x, and PM. Based on the city-specific light extinction budget calculations, measures for total extinction, visual range, and DeciView were calculated for each scenario and target year.

Figure 47. RADM-Predicted Visibility Degradation, Expressed in Annual Average DeciView, for Poor Visibility Conditions (90th Percentile) Under the Control Scenario.



Control scenario visibility

Unlike the other air quality conditions addressed in the present study, modeled visibility conditions are used as the basis for the control scenario rather than actual historical conditions. However, like the other air quality benefits of the historical Clean Air Act, it is the differences between modeled visibility outcomes for the control and no-control scenarios which are used to estimate visibility benefits. Nevertheless, 1990 absolute levels of eastern U.S. visibility predicted by RADM under the control scenario are presented in Figure 47 to provide a sense of initial visibility conditions.

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[107] Complete documentation of the linear scaling modeling, speciation methodologies, spatial allocation of emissions, and other data and assumptions are provided by the SAI SW Visibility Report (1994).

[108] For example, sulfate (SO_4) concentrations were scaled based on changes in sulfur oxide (SO_x) emissions.

[109] The term "light extinction budget" refers to the apportionment of total light attenuation in an area to the relevant pollutant species.

For the southwestern urban areas, 1990 control scenario annual average light extinction budget, visual range, and DeciView conditions are listed in Table 80. These 1990 results are presented to give the reader a sense of the initial visibility conditions in absolute, albeit approximate, terms.

No-control scenario visibility

The no-control scenario visibility results for the eastern U.S. area covered by RADM are presented in Figure 48. No-control scenario 1990 outcomes for the 30 southwestern U.S. urban areas are presented in Table 81.

Table 80. 1990 Control Scenario Visibility Conditions for 30 Southwestern U.S. Cities.

City	Light Extinction Budget (b, Mm^{-1})	Visual Range (km)	DeciView (dV)
Los Angeles, CA	197.6	15.2	29.8
San Bernardino, CA	201.7	14.9	30.0
Riverside, CA	208.3	14.4	30.4
Anaheim, CA	170.1	17.6	28.3
Ventura, CA	113.3	26.5	24.3
San Diego, CA	126.9	23.6	25.4
Santa Barbara, CA	112.8	26.6	24.2
Bakersfield, CA	215.1	13.9	30.7
Fresno, CA	211.7	14.2	30.5
Modesto, CA	148.8	20.2	27.0
Stockton, CA	153.1	19.6	27.3
San Francisco, CA	120.8	24.8	24.9
Oakland, CA	117.5	25.5	24.6
San Jose, CA	154.6	19.4	27.4
Monterey, CA	84.7	35.4	21.4
Sacramento, CA	119.1	25.2	24.8
Redding, CA	83.2	36.1	21.2
Reno, NV	147.4	20.3	26.9
Las Vegas, NV	157.9	19.0	27.6
Salt Lake City, UT	117.5	25.5	24.6
Provo, UT	107.8	27.8	23.8
Fort Collins, CO	80.7	37.2	20.9
Greeley, CO	84.2	35.6	21.3
Denver, CO	153.4	19.6	27.3
Colorado Springs, CO	83.3	36.0	21.2
Pueblo, CO	88.1	34.1	21.8
Albuquerque, NM	91.1	32.9	22.1
El Paso, TX	109.3	27.5	23.9
Tucson, AZ	85.6	35.0	21.5
Phoenix, AZ	125.3	23.9	25.3

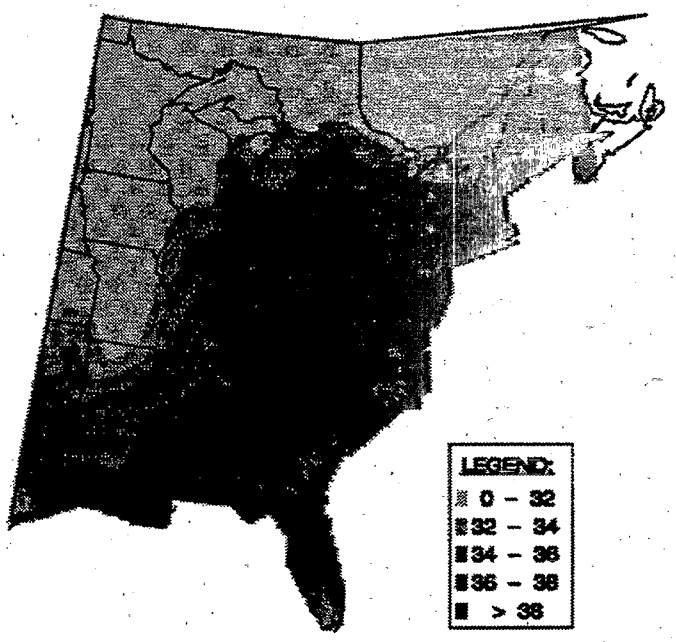
Data Source: SAI SW Visibility Report (1994).

Table 81. 1990 No-control Scenario Visibility Conditions for 30 Southwestern U.S. Cities.

City	Light Extinction Budget (b, Mm^{-1})	Visual Range (km)	DeciView (dV)
Los Angeles, CA	333.4	9.0	35.1
San Bernardino, CA	337.3	8.9	35.2
Riverside, CA	343.2	8.7	35.4
Anaheim, CA	286.3	10.5	33.5
Ventura, CA	194.8	15.4	29.7
San Diego, CA	210.1	14.3	30.4
Santa Barbara, CA	183.2	16.4	29.1
Bakersfield, CA	356.4	8.4	35.7
Fresno, CA	349.0	8.6	35.5
Modesto, CA	240.1	12.5	31.8
Stockton, CA	248.1	12.1	32.1
San Francisco, CA	197.3	15.2	29.8
Oakland, CA	188.6	15.9	29.4
San Jose, CA	253.0	11.9	32.3
Monterey, CA	141.4	21.2	26.5
Sacramento, CA	189.2	15.9	29.4
Redding, CA	128.6	23.3	25.5
Reno, NV	416.6	7.2	37.3
Las Vegas, NV	643.8	4.7	41.6
Salt Lake City, UT	185.8	16.1	29.2
Provo, UT	159.0	18.9	27.7
Fort Collins, CO	191.2	15.7	29.5
Greeley, CO	117.0	25.6	24.6
Denver, CO	284.4	10.5	33.5
Colorado Springs, CO	175.8	17.1	28.7
Pueblo, CO	299.9	10.0	34.0
Albuquerque, NM	175.8	17.1	28.7
El Paso, TX	276.3	10.9	33.2
Tucson, AZ	272.2	11.0	33.0
Phoenix, AZ	429.5	7.0	37.6

Data Source: SAI SW Visibility Report (1994).

Figure 48. RADM-Predicted Visibility Degradation, Expressed in Annual Average DeciView, for Poor Visibility Conditions (90th Percentile) Under the No-control Scenario.



Summary differences in visibility

Deciview Haze Index

The DeciView Haze Index (dV) has recently been proposed as an indicator of the clarity of the atmosphere that is more closely related to human perception than visual range (VR) or total extinction (b_{ext}) (Pitchford and Malm, 1994). It is defined by the equation:

$$dV = 10 \ln_e \left(\frac{b_{ext}}{10}\right) \quad (8)$$

where:

b_{ext} = total extinction in inverse megameters (Mm^{-1})

This index has the value of approximately 0 when the extinction coefficient is equal to the scattering coefficient for particle-free air (Rayleigh scattering) and increases in value by approximately one unit for each 10 percent increase in b_{ext}. Since the apparent change in visibility is related to the percent change in b_{ext} (Pitchford et al., 1990), equal changes in dV correspond to approximately equally perceptible changes

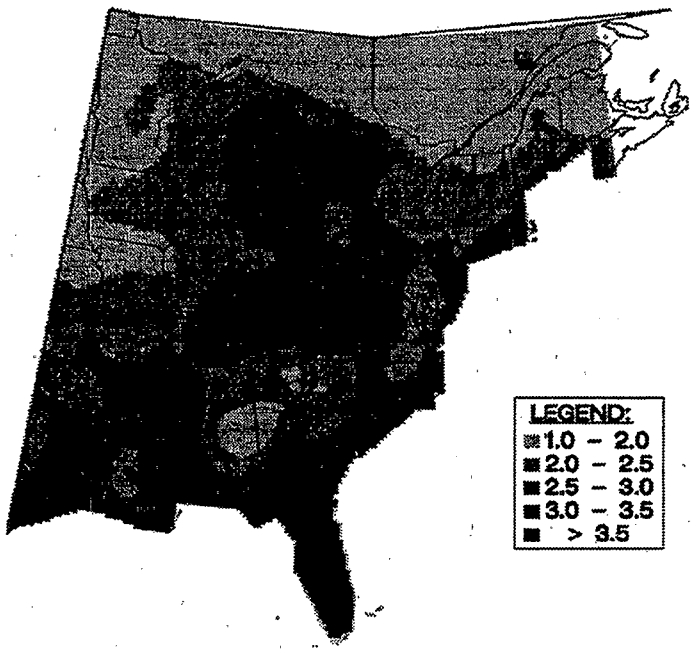
in visibility. Recent research indicates that, for most observers, a "just noticeable change" in visibility corresponds to an increase or decrease of about one to two dV units.

Both VR and dV are measures of the value of b_{ext} at one location in the atmosphere. Both are unaffected by the actual variability of the compositions and illumination of the atmosphere, so neither is closely linked to the human perception of an actual scene. The isolation of these parameters from site-specific variations and temporal fluctuations of the atmospheric illumination increases their usefulness for comparing the effects of air quality on visibility across a range of geographic locations for a range of time periods. Each parameter attempts to scale the b_{ext} data so that changes in air quality can be used to provide an indication of changes in the human perception of a scene.

Modeling Results

The differences in modeled 1990 control and no-control scenario visibility conditions projected by the RADM/EM for the eastern U.S. are presented in Figure 49. The map shows the percent increase in modeled annual average visibility degradation under poor conditions for 1990 when moving from the control to the no-control scenario. The results indicate perceptible differences in visibility between the control and no-control scenario throughout the RADM domain. The relatively large increase in visibility impairment in the Gulf Coast area is a reflection of the significant increases in 1990 sulfate concentrations associated with the no-control scenario. (For related discussion of effects in this region, see page 172.)

Figure 49. RADM-Predicted Increase in Visibility Degradation, Expressed in Annual Average DeciView, for Poor Visibility Conditions (90th Percentile) Under the No-control Scenario.



The differences in modeled 1990 control and no-control scenario visibility conditions in the 30 southwestern U.S. urban areas projected by linear rollback modeling are presented in Table 82. When reviewing these visibility degradation differentials for the 30 southwestern U.S. urban areas, it is important to consider that while estimated differences in visual range were in many cases very large, changes in the DeciView Haze Index (dV) may be relatively small. This is because the perception of visibility degradation measured by dV may be small when baseline visibility is high.[110] Even so, the results indicate that, by 1990, visibility in southwestern U.S. urban areas would be noticeably worse under the no-control scenario.

[110] See SAI SW Visibility Report (1994), page 5-3.

Table 82. Summary of Relative Change in Visual Range and DeciView Between 1990 Control and No-control Scenario Visibility Conditions for 30 Southwestern U.S. Cities.

City	Visual Range (%)	DeciView (dV)
Los Angeles, CA	69	-5
San Bernardino, CA	67	-5
Riverside, CA	65	-5
Anaheim, CA	68	-5
Ventura, CA	72	-5
San Diego, CA	65	-5
Santa Barbara, CA	62	-5
Bakersfield, CA	66	-5
Fresno, CA	65	-5
Modesto, CA	61	-5
Stockton, CA	62	-5
San Francisco, CA	63	-5
Oakland, CA	61	-5
San Jose, CA	64	-5
Monterey, CA	67	-5
Sacramento, CA	59	-5
Redding, CA	55	-4
Reno, NV	183	-10
Las Vegas, NV	308	-14
Salt Lake City, UT	58	-5
Provo, UT	48	-4
Fort Collins, CO	137	-9
Greeley, CO	39	-3
Denver, CO	85	-6
Colorado Springs, CO	111	-7
Pueblo, CO	240	-12
Albuquerque, NM	93	-7
El Paso, TX	153	-9
Tucson, AZ	218	-12
Phoenix, AZ	243	-12

Data Source: SAI SW Visibility Report (1994).

Key caveats and uncertainties for visibility

There are several sources of uncertainty in the RADM and southwestern U.S. linear scaling model analyses. For RADM, the use of nitric acid as a surrogate for estimating changes in light-attenuating nitrate particles ignores the interaction effects of nitrates, sulfates, and ammonia. As a result, increases in nitrates may be overestimated by the model when both sulfates and nitric acid increase. However, the significance of this potential overestimation is mitigated to some extent by the relative insignificance of nitrate-related visibility degradation relative to sulfates which prevails in the eastern U.S.

Several important uncertainties in the southwestern U.S. urban area visibility analysis are described in detail in the SAI SW Visibility Report (1994). First, the need to use seasonal average conditions leads to underestimation of extreme visibility impairment episodes associated with high humidity, since particle growth due to water absorption is highly nonlinear. Second, although the use of city-specific light extinction and PM speciation data is significantly better than reliance on regional averages, uncertainties in city-specific data may contribute to overall uncertainty in the estimates. However, overall uncertainty associated with these factors will be reduced to some extent since overestimation of visibility degradation in some cities will be offset by underestimations in other cities. Finally, the linear scaling used to estimate the pre-1990 control scenarios and the no-control scenarios was based on changes in county-wide or air basin emissions. Uncertainties associated with apportionment of state-wide emission changes to individual counties or air basins may contribute significantly to overall uncertainty in the visibility change estimates. Such apportionment is particularly difficult for SO_x emission changes, since emission reductions achieved by the Clean Air Act tended to be at relatively remote utility and smelter plants. However, sulfates are a relatively minor source of light attenuation in western urban areas.

An important overall limitation of the visibility analysis conducted for the present study is that only southwestern urban areas and the eastern 31 states were included. The Clean Air Act may have contributed toward significant reductions in visibility degradation in other areas. For example, Clean Air Act programs to reduce ambient particulate matter may have motivated reductions in silvicultural burning in some northwestern states. Perhaps the greatest deficiency in geographic coverage by the present study is the omission of visibility changes in Class I areas in the west.

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Since the concentration-response data are particular to specific sex and adult age groups, additional population data were also required to determine the sizes of affected subpopulations. For 1990 age and sex, the U.S. Census, 1992 was used, with age groups tallied as necessary. For 1980 age and sex, the U.S. Census, 1982 was used, with age groups also tallied as necessary. The 1970 age and sex breakdowns were obtained through personal communication with the Census Bureau.[250] The age and sex percentages were interpolated for intervening years.

Pregnant women are often a subpopulation of interest for lead effects. Although pregnant women themselves may be harmed by exposure to lead, this analysis was concerned with pregnant women because of possible effects on their fetuses who will be born and evince effects as young children. To estimate the number of exposed fetuses who were born during the years of interest,[251] birth rates for 1970, 1980 and 1990 were obtained from the Census Bureau.[252] These birth rates were used to interpolate for years between 1970 and 1980, and for the years between 1980 and 1990.

Results

For both the controlled and uncontrolled scenarios, Table 106 shows estimated lead emissions from industrial processes, industrial combustion and electric utilities. Table 107 and Table 108 show the differences in health impacts between the two scenarios (for industrial processes, industrial combustion and electric utilities only) for the "forward-looking" and "backward-looking" analyses.

[250] Karl Kuellmer, Abt Associates, and the Bureau of Census, Population, Age and Sex telephone staff, March, 1994.

[251] Note that we do not record the number of pregnancies, since the valuation only applies if the child is born and lives to exhibit the effect. Neither are we concerned with whether the births are single or multiple births, since each fetus is at risk, whether a pregnant women carries one or more fetuses.

[252] Personal communication, Karl Kuellmer, Abt Associates and the Bureau of Census, Population, Fertility/Births telephone staff.

Table 106. Estimated Lead Emissions from Electric Utilities, Industrial Processes, and Industrial Combustion (in Tons).

	1970	1975	1980	1985	1990
Electric Utilities* Controlled Scenario		1,351	636	175	190
Electric Utilities Uncontrolled Scenario		2,309	3,143	3,670	3,864
Industrial Processes Controlled Scenario	7,789	3,317	1,032	670	658
Industrial Processes Uncontrolled Scenario	7,789	7,124	6,550	5,696	5,305
Industrial Combustion Controlled Scenario	4,329	4,354	1,880	190	187
Industrial Combustion Uncontrolled Scenario	4,329	4,457	4,653	4,584	4,596

*Appropriate data on electric utilities do not exist for years prior to 1975.

Table 107. Yearly Differences in Number of Health Effects Between the Controlled and Uncontrolled Scenarios: Industrial Processes, Boilers, and Electric Utilities (Holding Other Lead Sources at Constant 1970 Levels).

Year	Population Exposed (millions)	Male Hypertension	Male Coronary Heart Disease	Male Stroke	Male Mortality	Female Coronary Heart Disease	Female Stroke	Female Mortality	Neonatal Mortality	Lost IQ Points	IQ<70 (cases)
1970	179	0	0	0	0	0	0	0	0	0	0
1975	188	149	0.1	0.1	0.1	0.1	0	0	0	716	4
1980	197	3,785	3.5	1.8	3.1	1.3	0.9	0.8	.001	16,242	92
1985	207	6,351	5.9	2.9	5.0	2.1	1.5	1.3	.002	25,819	145
1990	217	6,672	6.2	2.9	5.3	2.1	1.5	1.4	.002	27,183	153

Table 108. Yearly Differences in Number of Health Effects Between the Controlled and Uncontrolled Scenarios: Industrial Processes, Boilers, and Electric Utilities (Holding Other Lead Sources at Constant 1990 Levels).

Year	Population Exposed (millions)	Male Hypertension	Male Coronary Heart Disease	Male Stroke	Male Mortality	Female Coronary Heart Disease	Female Stroke	Female Mortality	Neonatal Mortality	Lost IQ Points	IQ<70 (cases)
1970	179	0	0	0	0	0	0	0	0	0	0
1975	188	422	0.7	0.3	0.6	0.2	0.2	0.2	0	716	2
1980	197	10,768	16.2	7.8	14.0	5.9	4.2	3.9	0.001	16,250	43
1985	207	18,113	27.0	12.7	23.3	9.6	6.8	6.4	0.002	25,837	67
1990	217	19,025	28.6	12.9	24.6	9.7	6.9	6.4	0.002	27,197	70

Reduction in Health Effects Attributable to Gasoline Lead Reductions

Estimating Changes in Amount of Lead in Gasoline from 1970 to 1990

The relationship between the national mean blood lead level and lead in gasoline is calculated as a function of the amount of lead in gasoline consumed. Thus, to calculate the health benefits from gasoline lead reductions, necessary inputs are estimates of lead in gasoline consumed over the period 1970 to 1990 and the amount of lead in gasoline that would have been consumed in the absence of the Clean Air Act. These values are calculated using the quantity of both leaded and unleaded gasoline sold each year and the concentration of lead in leaded and unleaded gasoline for each year in the period of interest. For each year, the relationship is expressed as:

$$LEAD = \left(\frac{SOLD}{365\,days} \right) \times [FRAC_{Pb} \times PB_{leaded} + (1 - FRAC_{Pb}) \times PB_{unleaded}] \qquad (63)$$

where:

$LEAD$	=	average lead per day in gasoline sold in a given year (metric tons/day),
$SOLD$	=	total quantity of gasoline sold (million gal/yr),
$FRAC_{Pb}$	=	fraction of total gasoline sales represented by leaded gasoline (dimensionless),
Pb_{leaded}	=	lead content of leaded gasoline (g/gal), and
$Pb_{unleaded}$	=	lead content of unleaded gasoline (g/gal).

Gasoline Sales (SOLD): Data on annual gasoline sales were taken from a report by Argonne National Laboratories (1993) which presented gasoline sales for each state in five year intervals over the period 1970-1990. This analysis used linear interpolation to estimate the gasoline sales for years between the reported years. These data were summed to obtain national sales figures.

Fraction of Sales that were Leaded and Gasoline ($FRAC_{Pb}$): For the controlled scenario, this analysis used information reported by Kolb and Longo (1991) for the fraction of the gasoline sales represented by leaded gasoline for the years 1970 through 1988. For 1989 and 1990, data were taken from DOE (1990 and 1991, respectively). For the uncontrolled scenario, all of the gasoline sold was assumed to be leaded for all years.

Lead Content of Gasoline (Pb_{leaded} and $Pb_{unleaded}$): Argonne National Laboratory in Argonne, Illinois was the source for the data on the lead content of leaded and unleaded gasoline for the period 1974-1990. Argonne compiled these data from historical sales data submitted to EPA, from Clean Air Act regulations on lead content, and from recent MVMA surveys. For 1970 through 1973, this analysis assumed the lead content of gasoline to be at the 1974 level. For the uncontrolled scenario, this analysis used the 1974 lead content in leaded gasoline as the lead content in all gasoline for each year.

Estimating the Change in Blood Lead Levels from the Change in the Amount of Lead in Gasoline

Several studies have found positive correlations between gasoline lead content and blood lead levels.[253] Data from the National Health and Nutrition Examination Survey (NHANES II) has been used by other researchers who determined similar positive correlations between gasoline lead and blood lead levels.[254]

The current analysis used a direct relationship between consumption of lead in gasoline and blood lead levels to estimate changes in blood lead levels resulting from Clean Air Act regulation of the lead content of gasoline. This relationship was based on regression analyses of the reduction of leaded gasoline presented in the 1985 Regulatory Impact Analysis (RIA).[255] Several multiple regressions were performed in the RIA to relate gasoline usage with individuals' blood lead levels, which were taken from NHANES II. These regressions of blood lead on gasoline usage controlled for such variables as age, sex, degree of urbanization, alcohol consumption, smoking, occupational exposure, dietary factors, region of the country, educational attainment, and income. The regressions suggested that a decrease of 100 metric tons per day (MTD) of lead used in gasoline is associated with a decrease in mean blood lead concentration of 2.14 μg/dL for whites and 2.04 μg/dL for blacks. In both of these regressions, gasoline use was found to be a highly significant predictor of blood lead ($p < 0.0001$).[256]

To determine a single gasoline usage-blood lead slope for the entire population of the U.S., this analysis used the average of the slopes for blacks and for whites, weighted by the percentage of blacks and whites in the U.S. during the time period of the analysis.[257] The resulting relationship is 2.13 μg/dl blood lead per 100 metric tons of lead in gasoline consumed per day. The same relationship was used to model changes in both children's and adults' blood lead levels. The U.S. EPA (1985) analyzed data from a study of black children in Chicago during the time period 1976 to 1980 and determined a slope of 2.08 μg/dl per 100 MTD. This slope for children is very similar to the one used in this analysis.

1970-Forward and 1990-Backward Approaches

As with the industrial processes and boilers analysis, this analysis used two different approaches to determine mean blood lead levels based on changes in lead concentrations in gasoline. In the 1970-forward approach, the calculations began with the estimated blood lead level for 1970. The change in blood lead level from one year to the next was based upon the change in the amount of lead in gasoline sold, as discussed above, for both the controlled and uncontrolled scenarios. For example, to calculate the blood lead level for 1971, the calculated change in blood lead from 1970 to 1971 was added to the 1970 value. This process was repeated for each succeeding year up to 1990.

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[253] U.S. EPA, 1985; Billick et al., 1979; Billick et al., 1982.

[254] Janney, 1982; Annest et al., 1983; Center for Disease Control, 1993; National Center for Health Statistics, 1993.

[255] U.S. EPA, 1985.

[256] U.S. EPA, 1985.

[257] U.S. Department of Commerce, 1992. Although the percentages of blacks and whites changed slightly over this time period (1970-1990), the change did not affect the value of the weighted slope.

The 1990-backward approach began with a mean blood lead level in 1990 for the controlled scenario. For the uncontrolled scenario, the starting blood lead was estimated from the 1990 level used in the controlled scenario, plus an additional blood lead increment resulting from the difference between the 1990 consumption of lead in gasoline in the controlled and uncontrolled scenarios. Again, the difference in mean blood lead levels from one year to the next was based on the change in gasoline lead for the corresponding years. For example, the difference in blood lead levels between 1990 and 1989 was subtracted from the 1990 level to determine the 1989 level. The process was continued for each year back to 1970.

Relating Blood Lead Levels to Population Health Effects

The mean blood lead levels calculated using the methods described above were used in the dose-response functions for various health effects (e.g., hypertension, chronic heart disease, mortality). This information was then combined with data on the resident population of the 48 conterminous states in each year to determine the total incidence of these health effects attributable to lead in gasoline in the controlled and uncontrolled scenarios. A Department of Commerce Publication (1991) was used to obtain the total population in 1970, 1980, and 1983-1990, while a different publication was the source of the 1975 population values.[258] Linear interpolation was used to estimate the populations in years for which specific data were not available.

For certain health effects, it was necessary to know the size of various age groups within the population. Two different sources were used to estimate the proportions of the population in the age groups of interest. A U.S. Census summary (Dept. of Commerce, 1990) was used for information for 1990 for children and adults and for 1980 for adults, and Census Telephone Staff (Dept. of Commerce, 1994) provided information for 1980 for children and 1970 for children and adults. The populations for the intervening years were estimated by linear interpolation.

[258] Dept. of Commerce, 1976.

Changes in Leaded Gasoline Emissions and Resulting Decreased Blood Lead Levels and Health Effects

Table 109 shows the estimated quantity of lead burned in gasoline in the five year intervals from 1970 to 1990. Tables 110 and 111 show the difference in health impacts between the two scenarios (for lead in gasoline only) for the "forward-looking" and "backward-looking" analyses.

Table 109. Lead Burned in Gasoline (in tons).

	1970	1975	1980	1985	1990
Controlled Scenario	176,100	179,200	86,400	22,000	2,300
Uncontrolled Scenario	176,100	202,600	206,900	214,400	222,900

Table 110. Yearly Differences in Number of Health Effects Between the Controlled and Uncontrolled Scenarios: Lead in Gasoline only (Holding Other Lead Sources at Constant 1970 Levels).

Year	Population Exposed (millions)	Male Hypertension	Male Coronary Heart Disease	Male Stroke	Male Mortality	Female Coronary Heart Disease	Female Stroke	Female Mortality	Neonatal Mortality	Lost IQ Points	IQ<70 (cases)
1970	202	0	0	0	0	0	0	0	0	0	0
1975	214	677,000	653	331	610	237	172	155	456	1,170,000	6,690
1980	225	4,200,000	3,970	1,990	3,680	1,430	1,040	939	2,340	5,720,000	32,500
1985	237	7,840,000	7,440	3,630	6,720	2,620	1,890	1,710	3,930	9,760,000	52,800
1990	247	9,740,000	9,380	4,410	8,110	3,140	2,270	2,060	4,940	11,800,000	63,900

Table 111. Yearly Differences in Number of Health Effects Between the Controlled and Uncontrolled Scenarios: Lead in Gasoline only (Holding Other Lead Sources at Constant 1990 Levels).

Year	Population Exposed (millions)	Male Hypertension	Male Coronary Heart Disease	Male Stroke	Male Mortality	Female Coronary Heart Disease	Female Stroke	Female Mortality	Neonatal Mortality	Lost IQ Points	IQ<70 (cases)
1970	202	0	0	0	0	0	0	0	0	0	0
1975	214	984	1,010	507	946	368	266	242	456	1,170,000	5,960
1980	225	6,350	6,670	3,270	6,180	2,420	1,740	1,590	2,340	5,720,000	27,400
1985	237	12,300	13,800	6,590	12,400	4,860	3,490	3,210	3,930	9,760,000	41,900
1990	247	15,600	18,100	8,340	15,700	6,090	4,360	4,030	4,940	11,800,000	49,700

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Appendix H: Air Toxics

Introduction

Air toxics are defined as air pollutants other than those six criteria pollutants for which EPA sets acceptable concentrations in ambient air. The SARA 313 Toxic Release Inventory (TRI), covering 328 of the approximately 3000 potentially hazardous compounds detected in air, estimated that approximately 1.2 millions tons of air toxics were released to the atmosphere in 1987 from U.S. stationary sources alone. While the TRI estimate tends to understate emissions of toxics for a number of reasons, it does show that large quantities of toxics are emitted into the atmosphere annually.

Effects of air toxic emissions are divided into three categories for study and assessment: cancer; "noncancer" effects, e.g. a wide variety of serious health effects such as abnormal development, birth defects, neurological impairment, or reproductive impairment, etc.; and ecological effects. Each year, these air toxic emissions contribute to significant adverse effects on human health, human welfare, and ecosystems. In EPA's 1987 *Unfinished Business Report*[259] cancer and noncancer air toxic risk estimates were considered sufficiently high, relative to risks addressed by other EPA programs, that the air toxics program area was among the few rated "high risk".

Limited Scope of this Assessment

The effects of air toxic emissions are difficult to quantify. Adverse health effects of toxics are often irreversible, not mitigated or eliminated by reduction in ongoing exposure, and involve particularly painful and/or protracted disease. Therefore these effects are not readily studied and quantified in human clinical studies, as contrasted with, for example, ambient ozone. In addition, epidemiological studies of effects in exposed populations are often confounded by simultaneous exposure of subjects to a variety of pollutants. Therefore, effects of these pollutants are often quantified by extrapolating data from animal studies to human exposure and expressed as risk per unit of exposure. Incidence of noncancer effects, for example, often are difficult to translate into monetized benefits.

Similarly, the quantification of ecological effects due to emissions of air toxics is hampered by lack of sufficient information regarding contribution of sources to exposure, associations between exposure to mixtures of toxics and various ecological endpoints, and economic valuation for ecological endpoints.

[259] U.S. EPA. Office of Policy Planning and Evaluation. *Unfinished Business: A Comparative Assessment of Environmental Problems.* February 1987.

Table 112. Health and Welfare Effects of Hazardous Air Pollutants.

Effect Category	Quantified Effects	Unquantified Effects	Other Possible Effects
Human Health	Cancer Mortality - nonutility stationary source - mobile source	Cancer Mortality - utility source - area source Noncancer effects - neurological - respiratory - reproductive - hematopoetic - developmental - immunological - organ toxicity	
Human Welfare		Decreased income and recreation opportunities due to fish advisories Odors	Decreased income resulting from decreased physical performance
Ecological		Effects on wildlife Effects on plants Ecosystem effects Loss of biological diversity	Effects on global climate
Other Welfare		Visibility Materials Damage	

The air toxics portion of this study is, of necessity, separate and more qualitative in nature than the benefit analysis conducted for the criteria air pollutants. Limitations in the quantitative analyses of air toxic effects led the Project Team to decide to exclude the available quantitative results from the primary analysis of CAA costs and benefits. Table 112 presents the range of potential human health and ecological effects that can occur due to air toxics exposure. As indicated, this appendix presents alternative quantitative estimates of benefits of CAA air toxics control for the cancer mortality endpoint for only nonutility stationary source and mobile source categories. Noncancer effects and ecological effects are described qualitatively.

History of Air Toxics Standards under the Clean Air Act of 1970

The 1970 Clean Air Act required the EPA to list a chemical as a hazardous air pollutant if it met the legislative definition provided:

> *"The term 'hazardous air pollutant' means an air pollutant to which no ambient air quality standard is applicable and which in the judgment of the Administrator may cause, or contribute to, an increase in mortality or an increase in serious irreversible, or incapacitating reversible, illness."*[260]

Once a HAP was listed, the EPA was required to:

> *"establish any such standard at the level which in his judgment provides an ample margin of safety to protect the public health from such hazardous air pollutant."*[261]

In other words the EPA had to first determine that a chemical was a HAP, and then regulate the emissions of each HAP based solely on human health effects and with an ample margin of safety. This regulatory mandate proved extremely difficult for EPA to fulfill, for reasons discussed below, and the result was that only seven HAPs were regulated over a period of 20 years.

Listing chemicals became a difficult task because of debates within and outside of the EPA surrounding issues of how much data are needed and which methodologies should be used to list a chemical as a HAP. An even more difficult issue was how to define the Congressional mandate to provide an "ample margin of safety." For carcinogens, there is generally no threshold of exposure considered to be without risk. What level of risk, then, is acceptable, and how should it be calculated? The EPA struggled to provide answers to these questions, and was challenged in court. The end result was a 1987 ruling by the D.C. Circuit Court that provided the EPA with a legal framework with which to determine an "ample margin of safety." This framework was interpreted and used by the EPA in its 1989 benzene regulations.

Quantifiable Stationary Source Air Toxics Benefits

One might be tempted to presume that the few federal HAP standards set would have achieved relatively substantial reductions in quantifiable risk. While some standards set under section 112 of the Clean Air Act appear to have achieved significant reductions in cancer incidence, the coverage, quantification, and monetization of the full range of potential adverse effects remains severely limited. This fact serves to highlight the inadequacy of current methods of evaluating HAP control benefits. This limited ability to estimate the total human health and ecological benefits of HAP reductions is an important

[260] 42 U.S.C. §1857(a)(1).

[261] 42 U.S.C. §1857(b).

area for future research. Thus the quantifiable benefits for CAA air toxics control presented here are limited in scope.

There are three sources of information that provide a picture of potential stationary source air toxics benefits of the CAA. EPA's Cancer Risk studies attempted to broadly assess the magnitude and nature of the air toxics problem by developing quantitative estimates of cancer risks posed by selected air toxics and their sources. Secondly, risk assessments conducted in conjunction with the promulgation of National Emissions Standards for Hazardous Air Pollutants (NESHAPs) offer a snapshot of potential monetized cancer mortality benefits. Finally, the Project Team attempted to estimate historical non-utility stationary source HAP-related direct inhalation cancer incidence reductions. Results from each of these studies are presented below.

EPA Analyses of Cancer Risks from Selected Air Toxic Pollutants

The Agency conducted two efforts to broadly assess the magnitude and nature of the air toxics problem. The 1985 report entitled, "The Air Toxics Problem in the United States: An Analysis of Cancer Risks for Selected Pollutants"[262] otherwise known as the "Six Month Study," was intended to serve as a "scoping" study to provide a quick assessment of the air toxics problem utilizing only readily available data on compound potencies, emissions, and ambient pollutant concentrations. The Agency updated this analysis of cancer risks in the 1990 report entitled "Cancer Risk from Outdoor Exposure to Air Toxics" referred to here as the "1990 Cancer Risk study."[263]

For the pollutant and source categories examined, the 1990 Cancer Risk study estimated the total nationwide cancer incidence due to outdoor concentrations of air toxics to range from as many as 1,700 - 2,700 excess cancer cases per year with 14 compounds accounting for approximately 95 percent of the annual cancer cases. Additionally, point sources contribute 25 percent of annual cases and area sources contribute 75 percent of annual cases, with mobile sources accounting for 56 percent of the nationwide total.[264]

The Six Month study indicates that the criteria air pollutant programs appear to have done more to reduce air toxics levels during the 1970 to 1990 period than have regulatory actions aimed at specific toxic compounds promulgated during the same period. Metals and polynuclear compounds usually are emitted as particulate matter and most of the volatile organic compounds are ozone precursors. As such, they are regulated under State Implementation Plan (SIP) and New Source Performance Standard (NSPS) programs and Title II motor vehicle regulations. A number of reports cited indicate significant reductions in air toxic emissions attributable to actions taken under SIP, NSPS and mobile source programs. Additionally, EPA conducted a comparison of air quality and emissions data for 1970 with the estimates of cancer incidence

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[262] U.S. EPA. Office of Air Quality Planning and Standards. *The Air Toxics Problem in the United States: An Analysis of Cancer Risks for Selected Pollutants*. May 1985. EPA-450/1-85-001.

[263] U.S. EPA. Office of Air Quality Planning and Standards. *Cancer Risk from Outdoor Exposure to Air Toxics*. September 1990. EPA-450/1-90-004a.

[264] The 1990 Cancer Risk study reported approximately 500 - 900 more cancer cases per year than the Six Month Study due primarily to the inclusion of more pollutants, better accounting of emissions sources, and, in some cases, increases in unit risk estimates.

for 1980.[265] Methods, assumptions and pollutants included were held constant over the period. The analysis showed a significant decrease in incidence during the decade due to improvements in air quality, presumably related to general regulatory programs. For the 16 pollutants studied, estimated nationwide cancer incidence decreased from 3600 in 1970 to 1600 in 1980. The 1990 Cancer Risk Study did not attempt to update this analysis.

Although it is difficult to make quantitative conclusions from these two studies regarding the benefits of CAA air toxics control, it is apparent that the pollutant-specific and source category-specific NESHAPs were not structured to reduce significant air toxic emissions from area and mobile sources. In fact, the 1990 Cancer Risk Study indicates that considerable cancer risk remained prior to passage of the 1990 CAA Amendments: as many as 2,700 excess cancer cases annually. However, some studies indicate that the criteria air pollutant program played a critical role during the 1970 to 1990 period in achieving air toxic emission reductions and therefore decreasing cancer risk.

Cancer Risk Estimates from NESHAP Risk Assessments

In looking back at the estimated effects of the HAP standards, EPA found that the effects of the NESHAPs were not quantified completely. These estimates occurred at a time when emission estimation and risk assessment methodologies for HAPs were first being developed. One consequence is that because emissions were not fully characterized, air toxic exposures could not be completely assessed. Additionally, most assessments only focused on the specific HAP being listed under the CAA and did not assess the reduction of other pollutants, which are currently considered HAPs. For example, while the vinyl chloride standard reduces emissions of ethylene dichloride, these emission reductions were not assessed in the risk assessment. In a different context, reductions of HAP may also achieve reductions of VOC and PM. The benefits of such reductions generally were also not evaluated. In addition, EPA generally did not assess the potential exposure to high, short-term concentrations of HAP and therefore did not know whether toxics effects from acute exposures would have been predicted and possibly addressed by the HAP standards.

In addition, people living near emission sources of concern are often exposed to a mix of pollutants at once. Some pollutants have been shown to act synergistically together to create a health risk greater than the risk that would be expected by simply adding the two exposure levels together. More research is needed to understand the effects of multiple-pollutant exposures. Finally, HAP risks tend to be distributed unevenly across exposed populations, with particularly high exposures occurring closest to emission sources. It should be noted that HAP exposure to specific populations may tend to fall disproportionately among the poor and minorities, who are more likely to live in close proximity to emitting facilities.

With the above caveats in mind, Table 113 provides information about maximum individual risk taken from the Federal Register notices for the NESHAPs promulgated before the 1990 amendments to the Clean Air Act. The benefits are calculated by multiplying the estimated annual incidence reduction by $4.8 million valuation per statistical life (1990 dollars). These benefit estimates provide a snapshot of potential monetized benefits for the year in which each NESHAP was promulgated. Of course these

[265] Hunt, W.F., Faoro, R.B. and Curran, T.C., "Estimation of Cancer Incidence Cases and Rates for Selected Toxic Air Pollutants Using Ambient Air Pollution Data, 1970 vs. 1980". U.S. EPA. April 1985.

estimates do not include air toxics benefits for other health and ecological benefit categories or air toxics benefits from co-control of criteria air pollutants. All uncertainties associated with the original estimates remain.

Table 113. Cancer Incidence Reductions and Monetized Benefits for NESHAPs.

Pollutant	Source Category	Year Promulgated	Pre-Reg Maximum Individual Risk	Post-Reg Maximum Individual Risk	Reduction in Cancer Incidence (per year)	Benefits in $million per year (1990$)
benzene		1985	$1.5x10^{-3}$	$4.5x10^{-4}$	31	1.5
benzene	coke by-product	1984	$7x10^{-3}$	$2x10^{-4}$	1.95	9.4
benzene	storage vessels	1982	$4.5x10^{-4}$	$3x10^{-5}$	0.01 to 0.06	0.05 to 0.3
benzene	waste operations	1986	$2x10^{-3}$	$5x10^{-5}$	0.55	2.6
benzene	transfer operations	1987	$6x10^{-3}$	$4x10^{-5}$	0.98	4.7
arsenic	primary copper	1986	$1.3x10^{-3}$ to $5x10^{-6}$	$1.2x10^{-3}$ to $3x10^{-6}$	0.09	0.4
arsenic	glass manuf.	1986	$7x10^{-4}$ to $3x10^{-5}$	$1.7x10^{-4}$ to $6x10^{-6}$	0.117 to 0.0034	0.02 to 0.6
asbestos	demolition	1973			100	480
vinyl chloride	PVC production	1975			10.5	50.4

Non-utility Stationary Source Cancer Incidence Reductions

The Project Team commissioned two studies to estimate reductions in cancer incidence due to pre-1990 NESHAPs: the PES Study and the ICF Re-analysis. The methodology used for most air pollutant evaluations involved a "back calculation" for the estimation of incidence reductions. However, the EPA has elected not to rely on the results of this analysis given critical methodological flaws. Despite the Project Team's concerns, the methodology and results of the two studies are presented below in the interest

of full disclosure and to guide efforts to develop a more valid and reliable analysis of the health-related benefits of HAP reductions in the upcoming section 812 Prospective studies.

PES Study

Methodology

The first attempt to estimate, for this study, historical non-utility stationary source HAP-related direct inhalation cancer incidence reductions was conducted by Pacific Environmental Services (PES). The basic approach used in the PES study was to adjust the cancer incidence estimates developed for EPA's 1990 Cancer Risk study to reflect the changes in emissions of, and exposures to, 14 key HAPs: arsenic, asbestos, benzene, 1,3-butadiene, carbon tetrachloride, chloroform, hexavalent chromium, dioxin, ethylene dichloride, ethylene dibromide, formaldehyde, gasoline vapors, products of incomplete combustion (PICs), and vinyl chloride.

The first step was to compile baseline incidence levels, defined as cancer cases per million population, for each of the 14 pollutants. The point estimates of incidence from the 1990 Cancer Risk study were used for this purpose. For some source categories, the "best point estimate" from the 1990 Cancer Risk study was used, for others a mid-point was selected.[266] These baseline incidence levels were based on measured ambient concentrations of the pollutant, modeled concentrations, or both.

The second step involved allocating baseline incidence levels to the individual source categories known to emit the relevant pollutant. In some cases, adjustments were made to reflect differences among the vintages of source category-specific data.[267] All baseline incidence estimates were ultimately expressed relative to a 1985 base year.[268] The assumption was then made that source-category incidence rates were proportional to the level of emissions from that source category.

Next, levels of control for each source category-specific incidence rate were estimated for each of the target years of the present analysis (i.e., 1970, 1975, 1980, 1985, and 1990).[269] Source category-specific activity level indicators were then established and linked to changes in corresponding activity indicators provided by the J/W macroeconomic modeling results. Activity levels were estimated for each source category, for each of the target years, and for each of the two scenarios.

Finally, source category/pollutant combination incidence levels for both the control and no-control scenarios were developed. These incidence levels were developed based on the baseline incidence levels, the activity indicators, and the control levels for each year. Both of these latter two factors varied between

[266] For some of the source categories, the original NESHAP/Air Toxic Exposure and Risk Information System (NESHAP/ATERIS) estimates of incidence were not available, in which case the baseline incidence was obtained from the 1989 National Air Toxics Information Clearinghouse(NATICH) Database Report. (See PES, "Draft Summary of Methodology Used for Cancer from Stationary Sources," memorandum from Ken Meardon, PES to Vasu Kilaru, US EPA, March 22, 1993, p. 2.)

[267] For example, six discrete sources for vinyl chloride were identified in the Six-Month Study Update. Point estimate incidences for each of these source categories came from separate references with databases corresponding to different years. (See PES, "Retrospective Analysis for Section 812(a) Benefits Study," September 30, 1992, p. 8.)

[268] See PES, March 22, 1993 memorandum, p, 3.

[269] Control level estimates were based on one of the following: control efficiencies for related criteria pollutants defined in the criteria pollutant analysis, reference documents such as Control Technology Guidelines (CTGs) or Background Information Documents (BIDs), preambles for related regulations, or EPA experts. (See PES, March 22, 1993 memorandum, p. 3.)

the control and no-control scenarios. The activity levels differed based on the specific levels of related sector economic activity predicted by the J/W model for the control and no-control scenario. The control levels prevailing in each of the target years were used for the control scenario, and the 1970 control level was applied throughout the 1970 to 1990 period for the no-control scenario.[270] The formula used for these calculations was as follows:[271]

$$I_{ty} = I_{by} \times \left[\frac{A_{ty}}{A_{by}}\right] \times \left[\frac{P_{ty}}{P_{by}}\right] \times \left[\frac{(1-C_{ty})}{(1-C_{by})}\right] \quad (64)$$

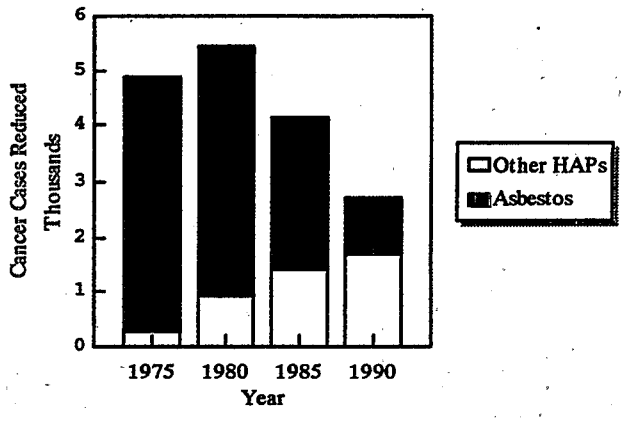
where:

I	=	cancer incidence for a source category-pollutant combination
A	=	activity level for a source category
P	=	population
C	=	control level for a source category-pollutant combination
ty	=	target year (1970 ... 1990)
by	=	base year

Findings

The PES analysis concluded that substantial reductions in HAP-related cancer cases were achieved during the reference period of the present study. The vast majority of these estimated reductions were attributable to reduced exposures to asbestos, particularly from manufacturing and fabricating sources.[272] In fact, roughly 75 percent of the total reduction in cancer cases averaged over the 1970 to 1990 period were attributed to asbestos control.[273] Figure 50 summarizes the PES study overall cancer incidence reductions and the relative contribution of asbestos-related reductions over the study period.

Figure 50. PES Estimated Reductions in HAP-Related Cancer Cases.



The Project Team had several concerns about the PES results. First and foremost, the reductions in asbestos-related cancer cases appeared to be substantially higher than expected, particularly in the earlier target years. Second, the control scenario activity level indicators for several sources with which Project Team members were familiar did not appear to be even remotely consistent

[270] More detailed descriptions of the methodology and associated uncertainties are provided in "Retrospective Analysis for Section 812(a) Benefits Study," a September 30, 1992 memorandum from Ken Meardon, PES to Vasu Kilaru, US EPA.

[271] See PES, March 22, 1993 memorandum, p. 4.

[272] PES, "Cancer Risk Estimates from Stationary Sources," memorandum from Ken Meardon, PES to Vasu Kilaru, US EPA, March 5, 1993.

[273] ICF, "Direct Inhalation Incidence Benefits," Draft Report, November 11, 1994, p. 10.

with actual historical activity patterns.[274] Finally, the level of documentation of the analytical methodologies, assumptions, and results was insufficient to ascertain the validity and reliability of the results. Ultimately, the Project Team determined that it was necessary to conduct a formal review and re-analysis of the cancer incidence reductions associated with non-utility stationary source HAP controls. The results of the PES analysis remain a relevant part of the record of the present study, however, since they provided a substantial basis for the subsequent re-analysis by ICF Incorporated.

ICF Re-analysis

Methodology

The purposes of the ICF Re-analysis were to examine the methodology and results of the PES study, particularly to address the aforementioned concerns of the Project Team, and to develop a revised set of estimates. Due to significant constraints on the resources remaining for HAP analysis in the section 812 study, however, only a few key HAPs could be investigated in depth and many important issues could not be addressed.[275] Furthermore, the effects of two early and potentially important HAP standards –the Beryllium and Mercury NESHAPs– could not be evaluated. Nevertheless, the ICF Re-analysis clarified some potential sources of uncertainty in the PES results and provided revised cancer incidence reduction estimates for several HAPs.

A key uncertainty in the PES results was associated with the use of a "back-calculation" technique to estimate incidence reductions for some HAPs. The back-calculation technique estimates uncontrolled incidence by dividing residual incidence by the assumed control efficiency. This approach means uncontrolled incidence, and therefore incidence reductions, are highly sensitive to small changes in assumed control efficiency.[276] In some cases, the PES analysis may have used control efficiencies which were too high, resulting in overestimation of uncontrolled incidence and therefore incidence reductions attributable to the CAA.[277] The vinyl chloride incidence reduction estimates appear to be significantly influenced by the use of this back-calculation technique. Another important source of uncertainty identified by ICF involved the potential overestimation of incidence totals when source apportionment is based on measured ambient concentrations.[278] ICF was unable, however, to perform an extensive evaluation of the activity level indicators used in the PES study.[279]

The first step undertaken in the re-analysis was to conduct a screening test to identify the HAPs which accounted for the most significant estimated incidence reductions. Based on this screening analysis,

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[274] For example, the activity indicators for Municipal Waste Combustors (MWCs) incorporated in the PES analysis decline dramatically throughout the 1975 to 1990 period. (See PES, March 5, 1993 memorandum to Vasu Kilaru, p. 10). In reality, overall MWC capacity and throughput increased significantly over this period.

[275] For example, the Project Team sought to develop and apply a methodology for estimating a central tendency estimate for the total carcinogenic risk imposed by all the HAPs examined. The intent was to address concerns about potential overestimation of aggregate risk measures when combining upper bound risk estimates of multiple HAPs. Unfortunately, resources were insufficient to continue development of this methodology.

[276] An example of this back-calculation technique illustrating the sensitivity to the assumed control efficiency is presented on page 12 of the draft ICF report.

[277] See ICF Draft Report, p. 12.

[278] See ICF Draft Report, p. 9.

[279] See ICF Draft Report, p. 13.

ICF eliminated 1,3-butadiene, carbon tetrachloride, chloroform, gasoline vapors, chromium, formaldehyde, and PICs from the detailed re-analysis effort.

Detailed reviews were then conducted for the remaining HAPs: vinyl chloride, dioxins, ethylene dibromide (EDB), ethylene dichloride (EDC), benzene, asbestos, and arsenic. In the re-analysis of these HAPs, ICF determined whether a forward- or back-calculation technique was used for the relevant source categories of a given HAP, reviewed the regulatory history of the relevant source categories to re-evaluate the assumed control efficiencies, and reviewed the upper-bound unit risk factor for each HAP. Revised total incidence reduction estimates for each HAP and for each target year were then calculated using the same basic calculation procedure used by PES. Finally, ICF identified a number of residual deficiencies in the analysis which could only be addressed through additional research and analysis.[280]

Findings

The ICF Re-analysis largely affirmed the original results obtained by PES; primarily because the PES analysis itself served as the basis for the re-analysis and only minor adjustments were adopted for many critical variables. In particular, most Project Team concerns regarding the PES methodology could not be resolved, including uncertainties associated with activity levels, assumed control efficiencies, and the unexpectedly high estimated incidence reductions associated with asbestos. In fact, the ICF Re-analysis produced a revised upper bound estimate for vinyl chloride-related incidence reductions which were even higher than the asbestos benefits.

Several sets of results were developed by ICF and presented in either the November 1994 draft report or in briefing materials prepared for the Science Advisory Board Clean Air Act Compliance Analysis Council Physical Effects Subcommittee (SAB/CAACACPERS) in May 1995. The first set of results is based on the assumption of 100 percent source compliance with HAP control requirements. An alternative set of results was developed assuming an 80 percent compliance rate with applicable standards. Given the linear effect of changes in compliance rates, these results were precisely 20 percent lower than the first set of estimates. At the May 1995 CAACACPERS briefing, estimates based on the 100 percent compliance estimates were presented. For asbestos, the revised incidence reductions were presented and characterized as upper bound. The asbestos estimates were then combined with upper and lower bound estimates for vinyl chloride and for "all other compounds." Figure 51 presents the total cancer incidence reductions derived from the ICF Re-analysis, using the asbestos estimates combined with the lower bound estimates for non-asbestos HAPs. Figure 52 presents a comparable compilation reflecting the upper bound estimates for all HAPs.

[280] Additional details of the ICF Re-analysis methodology can be found in ICF, "Direct Inhalation Incidence Benefits," Draft Report, November 11, 1994.

The Project Team remains concerned about these incidence reduction estimates, particularly given the doubts raised by the SAB/CAACACPERS at the May 1995 presentation of these results. For instance, several critical assumptions are needed to make this analysis valid when applied to EPA's NESHAPs. The flaws in these assumptions are described below.

(1) The risk estimates described in the 1990 Cancer Risk study, which served as the baseline for determining risk reductions, were accepted without question. There are myriad uncertainties in these estimates that must be recognized, as the study was designed only to generate rough order-of-magnitude estimates of the extent of the air toxics cancer problem.

(2) The percent control efficiency for emission reductions, which is calculated in each standard, would have to apply across every facility. Typically, the emission reductions vary between facilities; using a single average reduction could skew the results.

(3) There is a direct correlation between the number of tons of emissions reduced and incidence reduced by a specific regulation. Given the assumption of a linear, non-threshold dose-response curve (as is typically done for cancer), this is theoretically correct.

(4) Finally, the back calculation approach assumes that there is 100 percent compliance with the regulation.

Figure 51. ICF Estimated Reductions in Total HAP-Related Cancer Cases Using Upper Bound Asbestos Incidence and Lower Bound Non-Asbestos HAP Incidence.

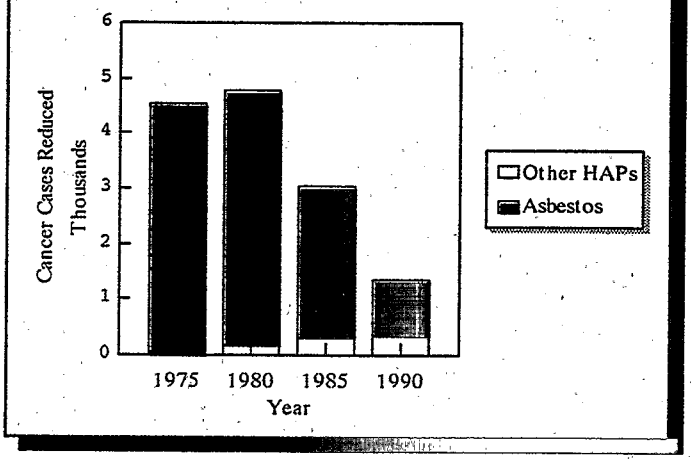
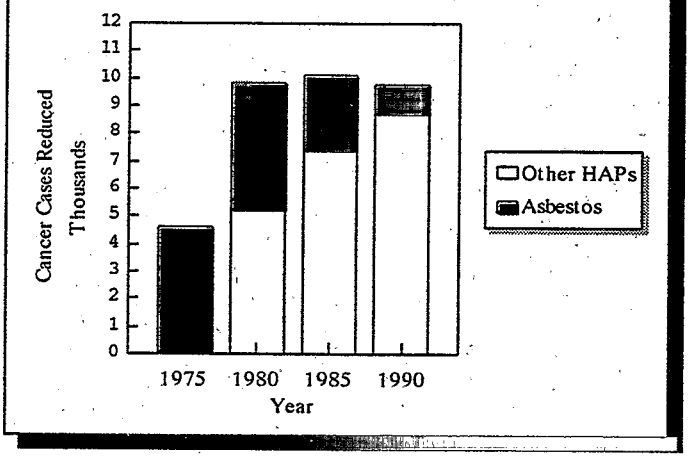


Figure 52. ICF Estimated Reduction in Total HAP-Related Cancer Cases Using Upper Bound Incidence for All HAPs.



EPA staff reviewed the "back calculation" approach for one of the more controversial aspects of the vinyl chloride (VC) NESHAP. The PES study estimates benefits at 426 cases reduced in 1990. The ICF Re-analysis resulted in an even higher estimate, between 1,000 and 7,000 cases annually. An analysis by EPA staff indicated that these vinyl chloride risk estimates are highly suspect given historical cancer incidence data for hepatic angiocarcinoma, a specific cancer that has been linked to vinyl chloride (Koppikar and Fegley, 1995). The following analysis demonstrates the inadequacies of the assumptions in the 1993 study.

(1) In the actual standard, no control technology was required for emissions from oxychlorination vents at ethylene dichloride (EDC)/VC plants. Applying "back calculation" for these emissions is inappropriate.

(2) In 1985, there were an estimated 8,000 fabrication plants which processed resins produced by PVC plants, thus resulting in VC emissions, which were exempt from the VC NESHAP. They emit very small quantities of VC and back calculation is not appropriate.

(3) The 1993 study uses a baseline estimate of 18 residual cases from the NESHAP/ATERIS data base. There is no evidence that these cases resulted only from emissions from PVC and EDC/VC plants.

(4) The risk analysis performed for the October 21, 1976 final VC regulation projected an incidence reduction of 11 cases per year.

In contrast, the PES study, using the "back calculation" method derived the following annual incidence reductions:

1980 - 250 cases
1985 - 360 cases
1990 - 430 cases

The subsequent back calculation conducted in the ICF Re-analysis resulted in incidence reductions as much as an order of magnitude higher than these.

Even considering the slightly different industrial output assumptions imposed by macroeconomic modeling, such a stark contrast is difficult to explain except for a critically flawed approach. Growth in activity and population nor other factors explain the difference in these two estimates. Given that the same general methodology was used for all of the air toxic pollutant assessments as was used for the VC NESHAP evaluation, there is reason to believe that cancer incidence results for the other air toxic pollutants are also flawed.

Mobile Source HAP Exposure Reductions

EPA's Cancer Risk report estimated that approximately 60 percent of the total carcinogenic risk posed by HAPs was attributed to mobile sources, with stationary sources contributing 15 percent and area sources contributing the remaining 25 percent.[281] The relative importance of mobile sources to total HAP exposure was a significant motivation behind EPA's subsequent effort to examine exposures and risks from mobile source HAPs.[282] Although available analytical resources were severely limited, the Project Team nevertheless decided it was necessary to perform at least an initial screening analysis to estimate the differences in mobile source HAP exposures between the control and no-control scenarios configured for the present study.

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[281] Cancer Risk report, Page ES-12.

[282] See US EPA/OAR/OMS, "Motor Vehicle-Related Air Toxics Study," EPA 420-R-93-005. April 1993.

Methodology

The approach used by ICF/SAI in conducting the mobile source HAP analysis closely followed the approach used in the EPA Motor Vehicle-Related Air Toxics Study (MVATS).[283] Recognizing the dearth of HAP ambient concentration and exposure data, both studies use carbon monoxide (CO) concentrations as the basis for estimating mobile source HAP concentrations and exposures. An important difference between the two studies, however, is that the ICF/SAI study adjusted the estimated change in ambient CO concentrations to take account of background[284] and non-mobile source[285] CO emissions. The HAP exposure function used in the ICF/SAI analysis is summarized by the following equation:

$$E = ((C \times A) - B) \times S \times M \times \frac{(VOC \times HAP)}{CO} \quad (65)$$

where :

E	=	exposure to motor vehicle-emitted HAP
C	=	annual ambient CO concentration to annual CO exposure concentration conversion factor
A	=	county-level annual average ambient CO concentration
B	=	background CO concentration
S	=	no-control to control scenario CO concentration adjustment factor (equals 1 for the control scenario)
M	=	total CO exposure to mobile source CO exposure conversion factor
VOC	=	VOC emissions by year, county, and scenario
HAP	=	VOC speciation factor by mobile source HAP
CO	=	CO emissions by year, county, and scenario

Details of the derivation of each of the variables applied in the above equation are provided in the ICF/SAI report. However, in essence, the calculation involves the following basic steps.

First, annual average, county-level CO ambient monitoring data are compiled from the EPA Aerometric Information Retrieval System (AIRS) database. After adjusting for background and non-mobile source contributions, these annual average ambient CO concentrations are converted to annual average CO exposure concentrations. As in the EPA MVATS, this conversion is made based on the Hazardous Air Pollutant Exposure Model - Mobile Sources (HAPEM-MS) population exposure model, which takes account of time spent in five indoor and outdoor microenvironments: indoors at home, other indoor, in-vehicle, outdoors near roadway, and other outdoor.[286] After adjusting for CO exposures attributable to non-mobile sources of CO, the CO exposures are converted to exposures for each of the mobile source HAPs based on available VOC speciation data and the ratio of co-located VOC and CO

[283] ICF/SAI, "Retrospective Analysis of Inhalation Exposure to Hazardous Air Pollutants from Motor Vehicles," October 1995, p. 4.

[284] Background CO is produced by the oxidation of biogenic hydrocarbons. See ICF/SAI, p. 7.

[285] The EPA MVATS attributed all measured CO to motor vehicles, resulting in an overestimation of motor-vehicle contributions to CO concentration changes. See ICF/SAI, p. 8. The MVATS assumption would also lead to a subsequent overestimation of changes in HAP exposures.

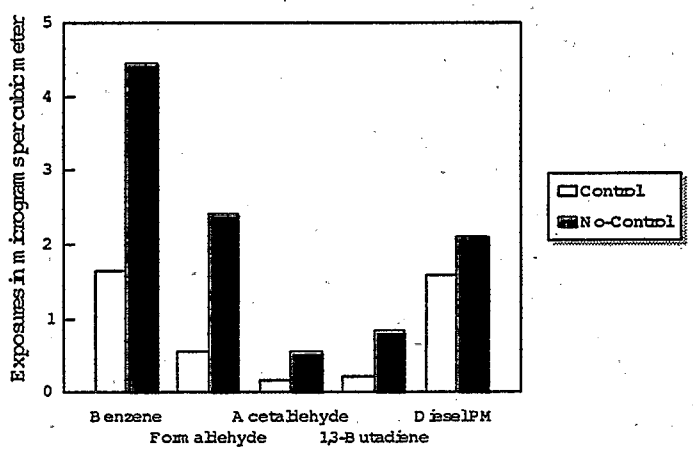
[286] See ICF/SAI, p. 3.

emissions.[287] These calculations are repeated for the no-control scenario after adjusting for differences in CO ambient concentrations for each target year and for differences in fuel composition.

Results

By 1990, CAA controls resulted in significant reductions in exposure to motor vehicle HAPs. Figure 53 summarizes the nationwide annual average exposure levels, in micrograms per cubic meter, for each of the five HAPs analyzed under the control and no-control scenarios. Additional detailed results, including breakdown by urban versus rural environments and comparisons with the EPA MVATS estimates, are provided in the ICF/SAI report.

Figure 53. National Annual Average Motor Vehicle HAP Exposures (μg/m3).



Analytical resources to carry forward these exposure estimates to derive estimates of the changes in motor vehicle HAP-related adverse effects attributable to historical CAA programs were not available.

Non-Cancer Health Effects

Broad gaps exist in the current state of knowledge about the quantifiable effects of air toxics exposure. This is particularly true for a wide range of health effects such as tumors, abnormal development, birth defects, neurological impairment, or reproductive impairment, etc. For example, the EPA's Non-Cancer Study[288] found that ambient concentrations for a substantial number of monitored and modeled HAPs exceeded one or more health benchmarks.[289] However no accepted methodology exists to quantify the effects of such exceedences. More data on health effects is needed for a broad range of chemicals.

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[287] The same HAP emission fractions used in the EPA MVATS were used herein, except for diesel PM which is not proportional to VOC emissions. Instead, diesel PM emission factors were developed using year-specific PART5 diesel PM emission factors and VMT estimates for diesel-powered vehicles.

[288] U.S. Environmental Protection Agency, "Toxic Air Pollutants and Noncancer Risks: Screening Studies," External Review Draft, September, 1990.

[289] Relevant benchmarks include Acceptable Daily Intake (ADI), the estimate of daily exposure at which adverse health effects are unlikely; and Lowest Observed Actual Effect Level (LOAEL), which is the lowest exposure level at which significant adverse health effects are observed.

Ecological Effects

Through the 1970s and 1980s, the adverse effects of toxic pollution on the Great Lakes became clear and undeniable. Over the same time period, scientists began collecting a convincing body of evidence that toxic chemicals released to the air can travel long distances and be deposited on land or water far from the original sources. An example of this evidence is the presence of such contaminants as PCBs, toxaphene, and other pesticides in fish in Lake Siskiwit, a lake on an island on upper Lake Superior, which has no waterborne sources of pollution. Toxaphene, a pesticide used primarily in the southeastern U.S. cotton belt, has been found as far away as the Arctic, with a decreasing air concentration gradient from the southeast toward the Great Lakes and the north Atlantic regions.

Similarly, a growing body of evidence showed that pollutants that were persistent (do not easily break down) and bioaccumulating (not significantly eliminated from the body) were magnifying up the food chain, such that levels in top predator fish contained levels up to millions of times greater than the harmless levels in the water. As such, those who ate those large fish, such as humans, eagles, mink, and beluga whales could receive very high exposures to the pollutants. Wildlife were beginning to show adverse effects in the wild, that could be duplicated in the lab. In the Great Lakes, such chemicals as PCBs, mercury, dieldrin, hexachlorobenzene, Lindane, lead compounds, cadmium compounds, DDT/DDE, and others are of significant concern. In other places in the country, similar effects are being experienced, especially with mercury, which is transported primarily by air, but exposure to which is primarily through contaminated fish. It was this kind of information about DDT and toxaphene that led to their being banned in the U.S. under FIFRA.

While ecological and economical sciences are not yet sufficiently advanced to support the kind of comprehensive, quantitative evaluation of benefits needed for the present study, selected local and regional scale adverse ecological effects of HAPs, and their adverse consequences for human health and welfare, can and have been surveyed. In May 1994, the EPA issued its first "Report to Congress on Deposition of Air Pollutants to the Great Waters."[290] The Great Waters Report examined the pollutants contributing to adverse ecological effects, the potential significance of the contribution to pollutant loadings from deposition of airborne pollutants, and the potential adverse effects associated with these pollutant loadings. Key HAPs identified in the Great Waters Report include PCBs, mercury, dioxins, and other heavy metals and toxic organics.

Of particular relevance to the present assessment, the Great Waters Report demonstrated the significance of transport and transformation of HAPs through food webs, leading to increased toxicity and biomagnification. A prime example of adverse transport and transformation is mercury. Transformation from inorganic to methylated forms significantly increases the toxic effects of mercury in ecosystems. A prime example of biomagnification is PCBs. As noted in the Great Waters Report:

> *"Pollutants of concern [such as PCBs] accumulate in body tissues and magnify up the food web, with each level accumulating the toxics from its diet and passing the burden along to the animal in the next level of the food web. Top consumers in the food web, usually consumers of large fish, may accumulate chemical concentrations many millions of times greater than the concentrations present in the water...High risk groups...include*

[290] USEPA/OAR/OAQPS, "Deposition of Air Pollutants to the Great Waters, First Report to Congress," EPA-453/R-93-055, May 1994.

breast-feeding mothers because breast-fed babies continue to accumulate [pollutants] from their mothers after birth. For example, they can have PCB levels four times higher than their mothers after six to nine months of breast feeding."[291]

Because of the risk of significant exposure to infants and other high-risk groups, such as "sport anglers, Native Americans, and the urban poor,"[292] a substantial number of fish consumption advisories have been issued in recent years. Current fish advisories for the Great Lakes alone include widespread advisories for PCB's, chlordane, mercury and others, cautioning that nursing mothers, pregnant women, women who anticipate bearing children, female children of any age and male children age 15 and under not eat certain high-food chain fish species. It should be noted as well that 40 states have issued mercury advisories in some freshwater bodies, and nine states have issued mercury advisories for every freshwater waterbody in the state (these states are Maine, New Hampshire, Vermont, Massachusetts, New York, New Jersey, Missouri, Michigan, and Florida).

There is little evidence indicating that the CAA had much beneficial effect on air toxic deposition to water bodies. Since the early NESHAPs were based on direct inhalation, primarily cancer effects close to a plant, they did not address the issue of cumulative effects of persistent pollutants far from the source. It was for this reason that Section 112(m) was included in the 1990 CAA Amendments, with requirements to study and document the atmospheric contribution of water pollutants, the adverse human health and environmental effects resulting and the sources that should be controlled to prevent adverse effects, and additionally, to promulgate regulations to prevent adverse effects.

Conclusions -- Research Needs

As has been demonstrated, there are broad gaps in the current state of knowledge about the quantifiable effects of air toxics exposure for a wide range of both human health and environmental effects. The following discussion outlines areas in which further research is needed in order to adequately quantify the benefits of air toxics control.

Health Effects

- Develop health effects data on pollutants for which limited or no data currently exists. Such studies should be focused on pollutants with a relatively high probability of exposure and/or potential adverse health effects.
- Understand mechanism of action of pollutants, for example through pharmacokinetic modeling. This will allow for a more accurate assessment of the effects of these pollutants on humans.
- Conduct research on factors that affect variations in susceptibility of human populations and determine the distribution of these factors in the U.S.
- Conduct research to better understand interactive effects of multiple pollutant exposures.

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[291] EPA-453/R-93-055, May 1994, p. ix.

[292] EPA-453/R-93-055, May 1994, p. x.

Develop methodologies to derive alternative estimates of human cancer risk from existing upper-bound methods.

- Acquire data and develop dose-response relationships for critical noncancer effects such as developmental, neurotoxic, mutagenic, respiratory and other effects. In particular, design methodology to quantify effects of exposures above health benchmarks.
- Acquire data and develop methods to estimate effects from acute exposure.

Exposure Assessment

- Expand data collection efforts: pre- and post-control emissions; HAP speciation; facilities location; facility parameters (stack heights, distances from stacks to fencelines, etc.)
- Develop more comprehensive exposure models which incorporate activity patterns, indirect exposures, total body burden, ratios of time spent indoors to outdoors.
- Continue to refine uncertainty analysis methods.

Ecosystem Effects

- Reliable estimates/measures of the levels of persistent bioaccumulating toxics in different media (air, water column, soils and sediments)
- Work to correlate levels of persistent bioaccumulating toxics with exposures, biota concentrations/accumulation, and adverse effects, especially subtle effects such as wasting, behavioral effects, and developmental effects.
- Criteria for effects, such as a wildlife correlate to a RfD or dose-response curve. This work should be done to complement the mass balance efforts now being completed, which will model source emissions to water column concentrations, then design research to predict effects on living resources given those predicted levels.
- Work to determine the effects of mixtures of persistent bioaccumulating toxic pollutants, and to determine cause-effect relationships of exposures over long periods of time.
- Studies to evaluate toxic effects in less well understood terrestrial systems such as: soil organisms/invertebrates, food web effects, amphibian effects, effects on endangered species and phytotoxic effects.

Economic Valuation

- Develop valuation estimates for endpoints for which inadequate estimates currently exist. These valuation estimates must be consistent with the kinds of damages expected.
- Initiate broad-scope economic valuation of air toxics program using survey techniques.

Appendix I: Valuation of Human Health and Welfare Effects of Criteria Pollutants

For the Section 812 analysis of health benefits, valuation estimates were obtained from the literature and reported in dollars per case reduced for health effects, and dollars per unit of avoided damage for welfare effects. Similar to estimates of physical effects provided by health studies, the monetary values of benefits were reported both in terms of mean values as well as a probability distribution of estimates. This permitted an evaluation of the uncertainty associated with the point estimates. It is interesting to note that the distributions of benefit values varied by endpoint. For example, while the estimate of the dollar value of a reduced mortality was lognormally distributed, the estimate for the value of a reduced case of acute bronchitis was uniformly distributed between a minimum and maximum value.

For the welfare benefits analysis, the avoided losses were in many cases measured in monetary terms directly. For agricultural benefits, however, the benefits associated with estimated changes in crop yields were evaluated with an agricultural sector model, AGSIM. This model incorporates agricultural price, farm policy, and other data for each year. Based on expected yields, the model estimates the production levels for each crop, the economic benefits to consumers, and the economic benefits to producers associated with these production levels. To the extent that alternative exposure-response relationships were available, a range of potential benefits was calculated.

Methods Used to Value Health Effects

This section describes the derivations of the economic valuations for health and welfare endpoints considered in the benefits analysis. An introduction to the method for monetizing improvements in health and welfare is followed by a summary of dollar estimates used to value benefits and detailed descriptions of the derivation of each estimate. Economic valuations are given both in terms of a central (point) estimate as well as a probability distribution which characterizes the uncertainty about the central estimate. All dollar values are in 1990 dollars.

Willingness to pay (WTP) is the measure used for the value an individual places on something, whether it is something that can be purchased in a market or not. WTP is the maximum amount of money such that the individual would be indifferent between having the good (or service) and having the money.

For both market and non-market goods, WTP reflects individuals' preferences. Because preferences are likely to vary from one individual to another, WTP for both market and non-market goods (e.g., health-related improvements in environmental quality) is likely to vary from one individual to another. In contrast to market goods, however, non-market goods such as environmental quality improvements are public goods, whose benefits are shared by many individuals. The individuals who "consume" the environmental quality improvement may have different WTPs for this non-market good. The total social value of the good is the sum of the WTPs of all individuals who consume the good.

In the case of health improvements related to pollution reduction, it is not certain specifically who will receive particular benefits of reduced pollution. For example, the analysis may predict 100 days of cough avoided resulting from a reduction of pollutant concentrations to achieve a given standard, but the analysis does not estimate which individuals will be spared those days of coughing. The health benefits conferred on individuals by a reduction in pollution concentrations are, then, actually *reductions in the probabilities* of having to endure certain health problems. These benefits (reductions in probabilities) may not be the same for all individuals (and could be zero for some individuals). Likewise, the WTP for a given benefit is likely to vary from one individual to another. In theory, the total social value associated with the decrease in incidence of a given health problem resulting from a given reduction in pollution concentrations is

$$\sum_{i=1}^{N} B_i \; WTP_i(B_i) \qquad (66)$$

where B_i is the benefit (i.e., the reduction in probability of having to endure the health problem) conferred on the ith individual (out of a total of N) by the reduction in pollution concentrations, and $WTP_i(B_i)$ is the ith individual's WTP for that benefit. If a reduction in pollution concentrations affects the incidences of several health endpoints, the total health-related social value of the reduction in pollution concentrations is

$$\sum_{i=1}^{N} \sum_{j=1}^{J} B_{ij} \; WTP_i(B_{ij}) \qquad (67)$$

where B_{ij} is the benefit related to the jth health endpoint (i.e., the reduction in probability of having to endure the jth health problem) conferred on the ith individual by the reduction in pollution concentrations, and $WTP_i(B_{ij})$ is the ith individual's WTP for that benefit.

The reduction in probability of each health problem for each individual is not known, nor is each individual's WTP for each possible benefit he or she might receive known. Therefore, in practice, benefits analysis estimates the value of a *statistical* health problem avoided. For example, although a reduction in pollutant concentrations may save actual lives (i.e., avoid premature mortality), whose lives will be saved cannot be known *ex ante*. What is known is that the reduction in air pollutant concentrations results in a reduction in mortality risk. It is this reduction in mortality risk that is valued in a monetized benefit analysis. Individual WTPs for small reductions in mortality risk are summed over enough individuals to infer the value of a *statistical* life saved. This is different from the value of a particular, identified life saved. Rather than "WTP to avoid a death," then, it is more accurate to use the term "WTP to avoid a statistical death," or, equivalently, "the value of a statistical life."

Suppose, for example, that a given reduction in PM concentrations results in a decrease in mortality risk of 1/10,000. Then for every 10,000 individuals, one individual would be expected to die in the absence of the reduction in PM concentrations (who would not die in the presence of the reduction in PM concentrations). If WTP for this 1/10,000 decrease in mortality risk is $500 (assuming, for now, that all individuals' WTPs are the same), then the value of a statistical life is 10,000 x $500, or $5 million.

WTP for a particular health benefit is, of course, unlikely to be the same for all individuals. If the individuals receiving health improvements, however, are a random sample from the population (i.e., if all individuals have the same chance of receiving these benefits), then the mean WTP is the population parameter of interest. Predicted benefits are, in this case, just the mean WTP for the benefit times the number of individuals predicted to receive the benefit.

The individuals actually receiving health improvements, however, may not be a random sample of the population. Suppose, for example, that most of those receiving a particular benefit (e.g., a reduction in the probability of dying in the current year) as a result of a reduction in pollution are the elderly. If WTP for this particular health risk improvement among the elderly is substantially different from WTP for the same health risk improvement among younger individuals, then economic theory would indicate that using the population mean WTP will give a biased result.

While the estimation of WTP for a market good (i.e., the estimation of a demand schedule) is not a simple matter, the estimation of WTP for a non-market good, such as a decrease in the risk of having a particular health problem, is substantially more difficult. Estimation of WTP for decreases in very specific health risks (e.g., WTP to avoid one day of coughing or WTP to avoid admission to the hospital for respiratory illness) is further limited by a paucity of information. Derivation of the dollar value estimates discussed below was often limited by available information. In the case of hospital admissions, for example, cost-of-illness estimates combined with opportunity cost estimates were used in lieu of WTP because of the lack of other information regarding willingness to pay to avoid such hospital admissions. These estimates are likely to understate total WTP to avoid a hospital admission because they do not include the value of avoiding the pain and suffering associated with the illness for which the individual entered the hospital. WTP to avoid a day of specific morbidity endpoints, such as coughing or shortness of breath, has been estimated by only a small number of studies (two or three studies, for some endpoints; only one study for other endpoints). The estimates for health endpoints involving these morbidity endpoints are therefore similarly based on only a few studies. However, it is worth noting that the total benefit associated with any reduction in pollutant concentrations is determined largely by the benefit associated with the corresponding reduction in mortality risk because the dollar value associated with mortality is significantly greater than any other valuation estimate.

Estimates of WTP may be understated for a couple of reasons. First, if exposure to pollution has any cumulative or lagged effects, then a given reduction in pollution concentrations in one year may confer benefits not only in that year but in future years as well. Benefits achieved in later years are not included. Second, the possible effects of altruism are not considered in any of the economic value derivations. Individuals' WTP for reductions in health risks for others are implicitly assumed to be zero.

The following Exhibit summarizes the derivations of the economic values used in the analysis.

Table 114. Unit Values Used for Economically Valuing Endpoints

Health or Welfare Endpoint	Estimated Value Per Incidence (1990$)		Derivation of Estimates
	Central Estimate	Uncertainty Distribution	
Mortality			
Short- & Long-term Exposure	$4.8 million	Weibull distribution, mean = $4.8 million std. dev. = $3.24 million	Central Est: $ value is the mean of value-of-life estimates from 26 studies (5 contingent valuation and 21 labor market studies). Uncertainty: Best-fit distribution to the 26 sample means. The Weibull distribution prevents selection of negative WTP values.
Chronic Bronchitis	$587,500	Equal probabilities of: (1) $883,000 (2) $457,000 (3) $210,000 (4) $800,000	Central Est: $ value is the mean of estimates from 4 studies of the benefit of avoiding a case of chronic bronchitis. Uncertainty: Distribution based on the four discrete estimates from which the mean value (used as the central estimate) was derived. Society's willingness to pay to avoid either endpoint is best represented by all four discrete values, with equal likelihood of occurrence.

Health or Welfare Endpoint	Estimated Value Per Incidence (1990$)		Derivation of Estimates
	Central Estimate	Uncertainty Distribution	
Hospital Admissions			
1. for coronary heart disease (ICD codes 410-414)	$10,279	Normal distribution, with mean = $10,279 std. dev. = $120	Central Est: $ value combines a cost-of-illness estimate, including the hospital charge, based on patients of all ages, and the cost of associated physician care, with the opportunity cost of time spent in the hospital. Source of hospital charge estimate: AHCPR, 1993. Source of physician charge estimates: Abt Associates Inc., 1992. Uncertainty: variation about the central estimate based on the standard error reported for the hospital charge component (greater than the other two components by an order of magnitude).
2. for initial cerebrovascular accidents (ICD code 436)	$7,554	Normal distribution, with mean = $7,554 std. dev. = $209	Central Est: $ value combines a cost-of-illness estimate, including the hospital charge, based on patients 65 and older, and the cost of associated physician care, with the opportunity cost of time spent in the hospital. Source of hospital charge estimate: AHCPR, 1993. Source of physician charge estimates: Abt Associates Inc., 1992. Uncertainty: variation about the central estimate based on the standard error reported for the hospital charge component (greater than the other two components by an order of magnitude).
3. f or initial atherothrombotic brain infarctions (ICD code 434)	$10,056	Normal distribution, with mean = $10,056 std. dev. = $229	Central Est: $ value combines a cost-of-illness estimate, including the hospital charge, based on patients 65 and older, and the cost of associated physician care, with the opportunity cost of time spent in the hospital. Source of hospital charge estimate: AHCPR, 1993. Source of physician charge estimates: Abt Associates Inc., 1992. Uncertainty: variation about the central estimate based on the standard error reported for the hospital charge component (greater than the other two components by an order of magnitude).

Health or Welfare Endpoint	Estimated Value Per Incidence (1990$)		Derivation of Estimates
	Central Estimate	Uncertainty Distribution	
4. for "respiratory illness"	$6,131	Normal distribution, mean = $6,131 std. dev. = $55	Central Est: $ value combines a cost-of-illness estimate, including the hospital charge, based on patients of all ages, and the cost of associated physician care, with the opportunity cost of time spent in the hospital. Source of hospital charge estimate: AHCPR, 1993. Source of physician charge estimates: Abt Associates Inc., 1992. Uncertainty: variation about the central estimate based on the standard error reported for the hospital charge component (greater than the other two components by an order of magnitude).
5. for COPD (ICD codes 490-496)	$8,075	Normal distribution, with mean = $8,075 std. dev. = $189	Central Est: $ value combines a cost-of-illness estimate, including the hospital charge, based on patients 65 and older, and the cost of associated physician care, with the opportunity cost of time spent in the hospital. Source of cost-of-illness estimates: Abt Associates Inc., 1992. Uncertainty: variation about the central estimate derived from a standard error estimated for the hospital charge component measured by another study (AHCPR). The reported standard error for hospital charge was applied to the combined cost-of-illness and opportunity cost estimate by assuming that relative variabilities surrounding the respective means were similar (i.e., coefficients of variation are equal). The hospital charge represents the vast majority of the total value to avoid a hospital admission for COPD.
6. for pneumonia (ICD codes 480-487)	$7,929	Normal distribution, with mean = $7,929 std. dev. = $112.50	Central Est: $ value combines a cost-of-illness estimate, including the hospital charge, based on patients of all ages, and the cost of associated physician care, with the opportunity cost of time spent in the hospital. Source of hospital charge estimate: AHCPR, 1993. Source of physician charge estimates: Abt Associates Inc., 1992. Uncertainty: Applied the standard error associated with the hospital charge component to the central estimate of $7,929. The hospital charge represents the vast majority of the total value to avoid a hospital admission for pneumonia.

Health or Welfare Endpoint	Estimated Value Per Incidence (1990$)		Derivation of Estimates
	Central Estimate	Uncertainty Distribution	
7. congestive heart failure (ICD code 428)	$8,280	Normal distribution, with mean = $8,280 std. dev. = $116.70	Central Est: $ value combines a cost-of-illness estimate, including the hospital charge, based on patients of all ages, and the cost of associated physician care, with the opportunity cost of time spent in the hospital. Source of hospital charge estimate: AHCPR, 1993. Source of physician charge estimates: Abt Associates Inc., 1992. Uncertainty: Applied the standard error associated with the hospital charge component to the central estimate of $8,280. The hospital charge represents the vast majority of the total value to avoid a hospital admission for congestive heart failure.
8. ischemic heart disease (ICD codes 410-414)	$10,308	Normal distribution, with mean = $10,308 std. dev. = $88.10	Central Est: $ value combines a cost-of-illness estimate, including the hospital charge, based on patients of all ages, and the cost of associated physician care, with the opportunity cost of time spent in the hospital. Source of hospital charge estimate: AHCPR, 1993. Source of physician charge estimates: Abt Associates Inc., 1992. Uncertainty: Applied the standard error associated with the hospital charge component to the central estimate of $10,308. The hospital charge represents the vast majority of the total value to avoid a hospital admission for ischemic heart disease.
Respiratory Ailments Not Requiring Hospitalization			
1. Upper Resp. Illness (URI) (defined as one or more of the following: runny or stuffy nose, wet cough, burning, aching, or red eyes)	$18.70	Continuous uniform distribution over the interval [$7.00, $32.72]	Central Est: Combinations of the 3 symptoms for which WTP estimates are available that closely match those listed by Pope et al. result in 7 different "symptom clusters," each describing a "type" of URI. A $ value was derived for each type of URI, using IEc mid-range estimates of WTP to avoid each symptom in the cluster and assuming additivity of WTPs. The $ value for URI is the average of the $ values for the 7 different types of URI. Uncertainty: taken to be a continuous uniform distribution across the range of values described by the 7 URI types.

Health or Welfare Endpoint	Estimated Value Per Incidence (1990$)		Derivation of Estimates
	Central Estimate	Uncertainty Distribution	
2. Lower Resp. Illness (LRI) (defined in the study as two or more of the following: cough, chest pain, phlegm, and wheeze.)	$11.82	Continuous uniform distribution over the interval [$5.27, $18.57]	Central Est: Combinations of the 4 symptoms for which WTP estimates are available that closely match those listed by Schwartz et al. result in 11 different "symptom clusters," each describing a "type" of LRI. A $ value was derived for each type of LRI, using IEc mid-range estimates of WTP to avoid each symptom in the cluster and assuming additivity of WTPs. The $ value for LRI is the average of the $ values for the 11 different types of LRI. Uncertainty: taken to be a continuous uniform distribution across the range of values described by the 11 LRI types.
3. Acute Bronchitis	$45.02	Continuous uniform distribution over the interval [$13.29, $76.74]	Central Est: Average of low and high values recommended by IEC for use in §812 analysis (Neumann et al., 1994). Uncertainty: continuous distribution between low and high values (Neumann et al., 1994) assigns equal likelihood of occurrence of any value within the range.
4. Presence of any of 19 acute respiratory symptoms 5. Any Symptom 6. Increase in Resp. Illness	$18.31	1. URI, probability = 40% LRI, probability = 40% URI+LRI, prob. = 20% 2. If URI, use URI $ dist. If LRI, use LRI $ dist. If URI+LRI, randomly select one value each from URI and LRI $ distributions; sum the two	Central Est: Assuming that respiratory illness and symptoms can be characterized as some combination of URI and LRI, namely: URI with 40% probability, LRI with 40% probability, and both URI and LRI with 20% probability. The $ value for these endpoints is the weighted average (using the weights 0.40, 0.40, and 0.20) of the $ values derived for URI, LRI, and URI + LRI. Uncertainty: based on variability assumed for central estimate, and URI and LRI uncertainty distributions presented previously.

Health or Welfare Endpoint	Estimated Value Per Incidence (1990$)		Derivation of Estimates
	Central Estimate	Uncertainty Distribution	
7. Asthma - Acute	$32.48	Continuous uniform distribution over the interval [$11.81, $53.80]	Central Est: Mean of average WTP estimates for the four severity definitions of a "bad asthma day." Source: Rowe and Chestnut (1986), a study which surveyed asthmatics to estimate WTP for avoidance of a "bad asthma day," as defined by the subjects. Uncertainty: based on the range of values estimated for each of the four severity definitions.
8. Shortness of breath	$5.29	Continuous uniform distribution over the interval [$0, $10.57]	Central Est: From Ostro et al., 1995. This is the mean of the median estimates from two studies of WTP to avoid a day of shortness of breath: Dickie et al., ($0.00), and Loehman et al., 1979 ($10.57). Uncertainty: taken to be a continuous uniform distribution across the range of values obtained from the two studies.
Restricted Activity and Work Loss Days			
1. WLDs	$83.00		Central Est: Median weekly wage for 1990 divided by 5 (USDC, 1992) Uncertainty: Insufficient information to derive an uncertainty estimate.
2. RADs	not monetized[a]	--	--
3. MRADs	$38.37	triangular distribution centered at $38.37 on the interval [$15.72, $61.02]	Central Est: Median WTP estimate to avoid 1 MRRAD -- minor respiratory restricted activity day -- from Tolley et al. (1986) (recommended by IEc as the mid-range estimate). Uncertainty: range is based on assumption that value should exceed WTP for a single mild symptom (the highest estimate for a single symptom--for eye irritation--is $15.72) and be less than that for a WLD. The triangular distribution acknowledges that the actual value is likely to be closer to the point estimate than either extreme.
4. RRADs	not monetized[a]	--	--
IQ Changes			

Health or Welfare Endpoint	Estimated Value Per Incidence (1990$)		Derivation of Estimates
	Central Estimate	Uncertainty Distribution	
1. Lost IQ Points	$3,531 per lost IQ point	none available	Central Est: $ value is the mean of estimates based on results of 2 studies. With an assumed 5% discount rate, the results in Schwartz (1994) yield an estimate of $2,995 per IQ point; the results of Salkever (1995) yield an estimate of $4,067. These estimates include the combined effects on lifetime earnings: (1) *directly* based on IQ decrement, and (2) *indirectly* based on lower educational attainment and reduced labor force participation.
2. Incidence of IQ < 70	$41,787	none available	Central Est: $ value measures reduction in education costs in terms of special needs for lower IQ students (in mainstream schools).
Hypertension	$681 per case per year	none available	Central Est: $ value quantifies costs associated with physician care, medications, and hospital charges, in addition to opportunity cost of lost work time due to the disability.
Soiling and Materials Damage	$2.52	Beta distribution with mean=$2.52, standard deviation=$1.00 on the interval [$1.26, $10.08]. The shape parameters of this distribution are α=1.2 and β=7.3.	Central Est: Source: RCG/Hagler Bailly. RCG uses $1.26 as its low estimate of annual cost of soiling and materials damage per household (assuming 2.63 persons per household), taken from Manuel et al. (1982). The Manuel study measured particulate matter as TSP rather than PM-10. Hypothesizing that at least half of the costs of household cleaning are for the time value of do-it-yourselfers, which was not included in the Manuel estimate, RCG multiplied the Manuel estimate by 2 to get a point estimate of $2.52, in 1990 $ (reported by RCG as $2.70 in 1992 dollars). Uncertainty: The Beta distribution selected is a smooth, continuous function with its probability mass near the mean and it covers the range of reported estimates.

NOTES:

[a] This endpoint was not monetized because including it in the aggregation of economic benefits would result in double-counting (overlap with WLDs and MRADs).

[b] annual cost of soiling and materials damage per household per $\mu g/m^3$ PM.

Results of Valuation of Health and Welfare Effects

Table 115 presents the results of combining the economic valuations described in this Appendix with the health and welfare effects results presented in Appendix D. As noted in Appendix D, there are alternative estimates for some health and welfare impacts, which form the basis of several alternative benefit estimates. Each of the health effects estimates also has quantified statistical uncertainty. The range of estimated health and welfare effects, along with the uncertain economic unit valuations, were combined to estimate a range of possible results. The combining of the health and economic information used the Monte Carlo method presented in Chapter 7. Table 115 shows the mean estimate results, as well as the measured credible range (upper and lower five percentiles of the results distribution), of economic benefits for each of the quantified health and welfare categories.

The results for aggregate monetized benefits were also calculated using a Monte Carlo method. The results of the Monte Carlo simulations for the economic values for each of the major endpoint categories are presented in Table 116. Note that for the upper and lower fifth percentiles the sum of the estimated benefits from the individual endpoints does not equal the estimated total. The Monte Carlo method used in the analysis assumes that each health and welfare endpoint is independent of the others. There is a very low probability that the aggregate benefits will equal the sum of the fifth percentile benefits from each of the ten endpoints.

Table 117 shows the estimated total benefits ranges for the four modeled target years of this study: 1975, 1980, 1985, and 1990. The results of the Monte Carlo simulations of the aggregate economic benefits for these four target years are depicted in Figure 21.

Table 118 examines the impact of limiting the scope of the analysis to locations with more certain air quality estimates. The main analysis (as shown in Tables 115 through 117) covers almost the entire population of the 48 States.[293] However, the air quality information is less certain for locations far from a monitor. Table 118 presents the results of limiting the analysis to people living within 50 km of an ozone, NO_2, SO_2, or CO monitor, or in counties with a PM monitor. The availability of monitors changes over time. Hence the proportion of the population included in this analysis changes over time as well. Table 118 indicates that approximately a quarter of the total benefits estimated in the main analysis comes from areas with less certain air quality information.

The results of the "all U.S. population" analysis provides a more accurate depiction of the *pattern* of economic benefits across years. The accuracy of the *scale* of incidence is less certain. These results provide a better characterization of the total direct benefits of the Clean Air Act in the lower 48 states than do the "monitored area only" results because the latter completely omits historical air quality improvements for about 25 percent of the population. However, the "all U.S. population" results rely on uncertain extrapolations of pollution concentrations, and subsequent exposures, from distant monitoring sites to provide coverage for the 25 percent or so of the population living far from air quality monitors. Thus, the main results presented in Tables 115 through 117 include important uncertainties.

[293] Except for lead, two to five percent (depending on pollutant) of the population who live in sparsely populated areas are excluded from the main analysis to maximize computer efficiency. All of the population of the 48 states is included in the lead analysis.

Table 115. Criteria Pollutants Health and Welfare Benefits -- Extrapolated to Entire 48 State Population
Present Value (in 1990 using 5% discount rate) of Benefits from 1970 - 1990 (in billions of 1990 dollars).

Endpoint	Pollutant(s)	Present Value (billions of 1990$) 5th %ile	Mean	95th %ile
Mortality				
*Mortality (long-term exposure)	PM-10	$3,040	$21,933	$51,500
*Mortality (short-term exposure)	PM-10	$820	$5,340	$11,863
Mortality (Lead exposure)	Lead	$125	$1,550	$4,096
Chronic Bronchitis	PM-10	$2,145	$7,146	$12,643
Other Lead-induced Ailments				
Lost IQ Points	Lead	$275	$438	$626
IQ < 70	Lead	$22	$26	$31
Hypertension	Lead	$77	$99	$120
Coronary Heart Disease	Lead	$1	$4	$11
Atherothrombotic brain infarction	Lead	$0	$1	$2
Initial cerebrovascular accident	Lead	$0	$1	$2
Hospital Admissions				
**All Respiratory	PM-10 & O3	$8	$9	$11
**COPD + Pneumonia	PM-10 & O3	$8	$9	$10
Ischemic Heart Disease	PM-10	$1	$4	$6
Congestive Heart Failure	PM-10 & CO	$3	$5	$7
Other Respiratory-Related Ailments				
Children				
Shortness of breath, days	PM-10	$0	$7	$17
***Acute Bronchitis	PM-10	$0	$7	$18
***Upper & Lower Respiratory Symptoms	PM-10	$1	$2	$4
Adults				
Any of 19 Acute Symptoms	PM-10 & O3	$7	$49	$119
All				
Asthma Attacks	PM-10 & O3	$0	$1	$2
Increase in Respiratory Illness	NO2	$1	$2	$4
Any Symptom	SO2	$0	$0	$0
Restricted Activity and Work Loss Days				
MRAD	PM-10 & O3	$4	$6	$9
Work Loss Days (WLD)	PM-10	$2	$2	$3
Human Welfare				
Household Soiling Damage	PM-10	$6	$75	$189
Visibility - Eastern U.S.	particulates	$55	$72	$91
Decreased Worker Productivity	O3	$5	$5	$5
Agriculture (Net Surplus)	O3	$11	$23	$34

To avoid double-counting of benefits, the following endpoints were treated as alternatives:

*Mortality benefits estimated by the 19 short-term exposure studies were pooled with the single long-term mortality study, assigning equal weight to each study *type*.

**Hospital admissions for COPD combined with those for pneumonia are treated as an equally-weighted alternative to hospital admissions for all respiratory illnesses.

***The definitions of acute bronchitis and upper and lower respiratory illness overlap; both studies count trouble breathing, dry cough, and wheezing in their estimates. These two studies are treated as alternatives, which reflects the variability of pollution-induced respiratory effects in children.

Table 116. Present Value of 1970 to 1990 Monetized Benefits by Endpoint Category for 48 State Population and Total Monetized Benefits (discount rate of 5 percent)

Endpoint	Pollutant(s)	Present Value		
		5th %ile	Mean	95th %ile
Mortality	PM-10	$1,991	$13,542	$30,968
Mortality	Pb	$125	$1,550	$4,096
Chronic Bronchitis	PM-10	$2,143	$7,156	$12,613
IQ (Lost IQ Pts. + Children w/ IQ<70)	Pb	$299	$466	$656
Hypertension	Pb	$77	$99	$120
Hospital Admissions	PM-10, O3, Pb, & CO	$18	$24	$33
Respiratory-Related Symptoms, Restricted Activity, & Decreased Productivity	PM-10, O3, NO2, & SO2	$31	$76	$149
Soiling Damage	PM-10	$7	$76	$196
Visibility	particulates	$55	$72	$91
Agriculture (Net Surplus)	O3	$11	$23	$35
	TOTAL ($Billions)	$10,500	$23,000	$40,600

Table 117. Monte Carlo Simulation Model Results for Target Years, Plus Present Value in 1990 Terms of Total Monetized Benefits for Entire 1970 to 1990 Period (in billions of 1990-value dollars).

Total Benefits By Year ($Billions)	1975	1980	1985	1990	Present Value (5%)
5th percentile	$161	$431	$558	$623	**$10,500**
Mean	$364	$957	$1,204	$1,318	**$23,000**
95th percentile	$655	$1,710	$2,075	$2,317	**$40,600**

Notes:

Present value reflects compounding of benefits from 1971 to 1990.

"Uncertainty Estimates" are results of Monte Carlo analysis combining economic and physical effects uncertainty (i.e., using both between- and within-study variability).

Full uncertainty analysis done only for years shown. Uncertainty estimates for intermediate years computed based on ratios of 5th to 50th percentile and 95th to 50th Ratios interpolated between years shown and applied to point estimates for intermediate years.

Figure 54. Monte Carlo Simulation Model Results for Target Years (in billions of 1990 dollars)

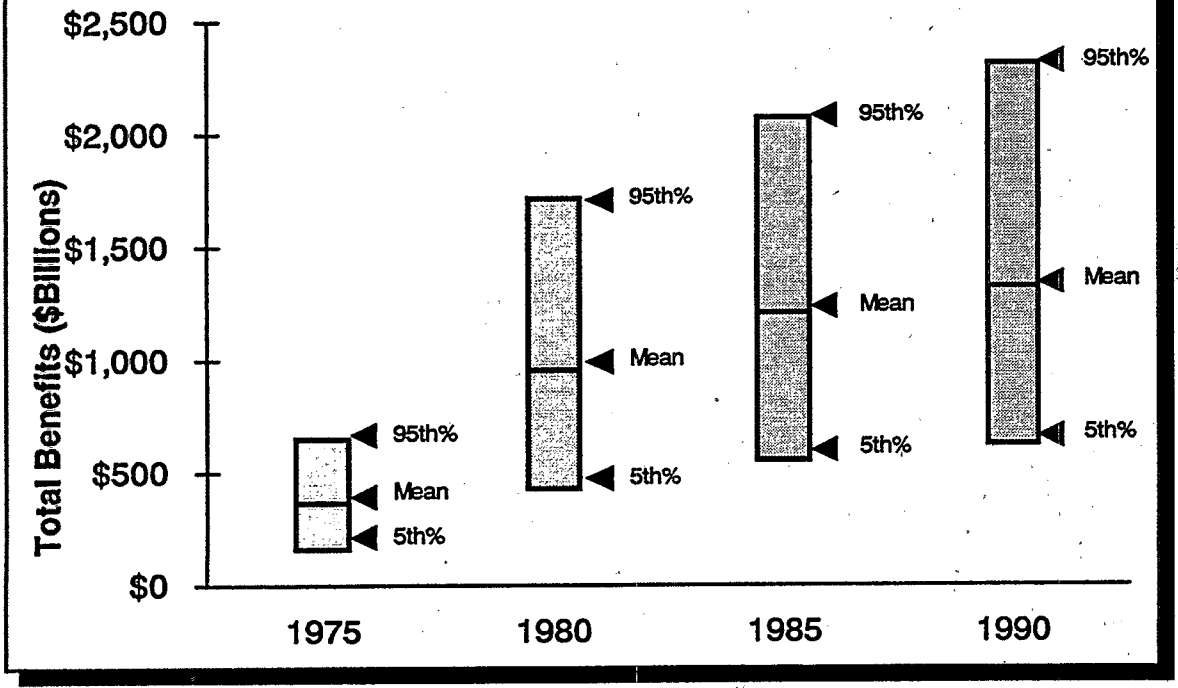


Table 118. Comparison of 1990 (Single Year) Monetized Benefits by Endpoint for 48 State Population and Monitored Areas (in millions of 1990 dollars).

Endpoint	Pollutant(s)	Mean Estimate of Monetized Benefits (millions of 1990-value dollars)	
		48 State Pop.	Monitored Areas*
Mortality	PM-10	$737,408	$465,905
Mortality	Pb	$131,752	$131,752
Chronic Bronchitis	PM-10	$398,042	$259,870
IQ (Lost IQ Points + Children with IQ < 70)	Pb	$37,639	$37,639
Hypertension	Pb	$8,695	$8,695
Hospital Admissions	PM-10, O3, Pb, & CO	$1,566	$1,154
Respiratory-Related Symptoms, Restricted Activity, & Decreased Productivity	PM-10, O3, NO2, & SO2	$4,500	$3,029
Soiling Damage	PM-10	$4,137	$2,718
Visibility	particulates	$4,479	$4,479
Agriculture (Net Surplus)	O3	$991	$991
	TOTAL ($Millions)	$1,317,993	$905,257

* Monitored areas are those within 50 km of an O3, NO2, SO2, or CO monitor or a PM-monitored county. The "48 State Population" modeling estimate captures benefits for populations in unmonitored areas. Air pollution concentrations in these areas are assigned based on concentrations measured at the closest monitor, for O3, NO2, SO2, and CO. PM-10 concentrations in unmonitored counties are derived from those in monitored counties.

Sensitivity Analyses

The uncertainty ranges for the results on the present value of the aggregate measured monetary benefits reported in Table 115 reflect two important sources of measured uncertainty:

- uncertainty about the avoided incidence of health and welfare effects deriving from the concentration-response functions, including both selection of scientific studies and statistical uncertainty from the original studies; and
- uncertainty about the economic value of each quantified health and welfare effect.

These aggregate uncertainty results incorporate many decisions about analytical procedures and specific assumptions discussed in the Appendices to this report.

In order to provide a more complete understanding of the economic benefit results in Table 115, sensitivity analyses examines several additional important aspects of the analysis. First, this section explores the effect of selecting alternative discount rates on the aggregate present value benefits estimation. Second, this section examines the sources of the measured aggregate uncertainty, identifying which of the measured uncertainty components of incidence and valuation for individual health effects categories drive the overall uncertainty results. Third, this section examines several issues involving the estimated economic benefits of mortality.

The main analysis reflected in present value results shown in Table 115 uses a five percent discount rate. The discount rate primarily enters the calculations when compounding the economic benefits estimates from individual years between 1970 and 1990 to estimate the present value of the benefits in 1990. The discount rate also directly enters in the calculation of the economic value of an IQ point.[294] There is considerable controversy in the economics and policy literature about the appropriate discount rate to use in different settings. Major alternatives recommended by various authors include a discount rate based on the social discount rate (typical estimates are in the 2 to 3 percent range), and a discount rate based on the risk-free rate of return on capital (typically in the 7 to 10 percent range). Table 119 presents the aggregate uncertainty results using three different discount rates: 3%, 5% and 7%. While the aggregate benefits estimates are sensitive to the discount rate, selecting one of these alternative discount rates affects the aggregate benefits estimates by only about 15 percent.

Table 119. Effect of Alternative Discount Rates on Present Value of Total Monetized Benefits for 1970 to 1990 (in trillions of 1990 dollars).

Present Value in 1990 of Total Benefits ($Trillions of 1990 Dollars)	3%	5%	7%
5th percentile	$9.1	$10.5	$12.2
Mean	$19.9	$23.0	$26.7
95th percentile	$35.1	$40.6	$47.3

Notes:

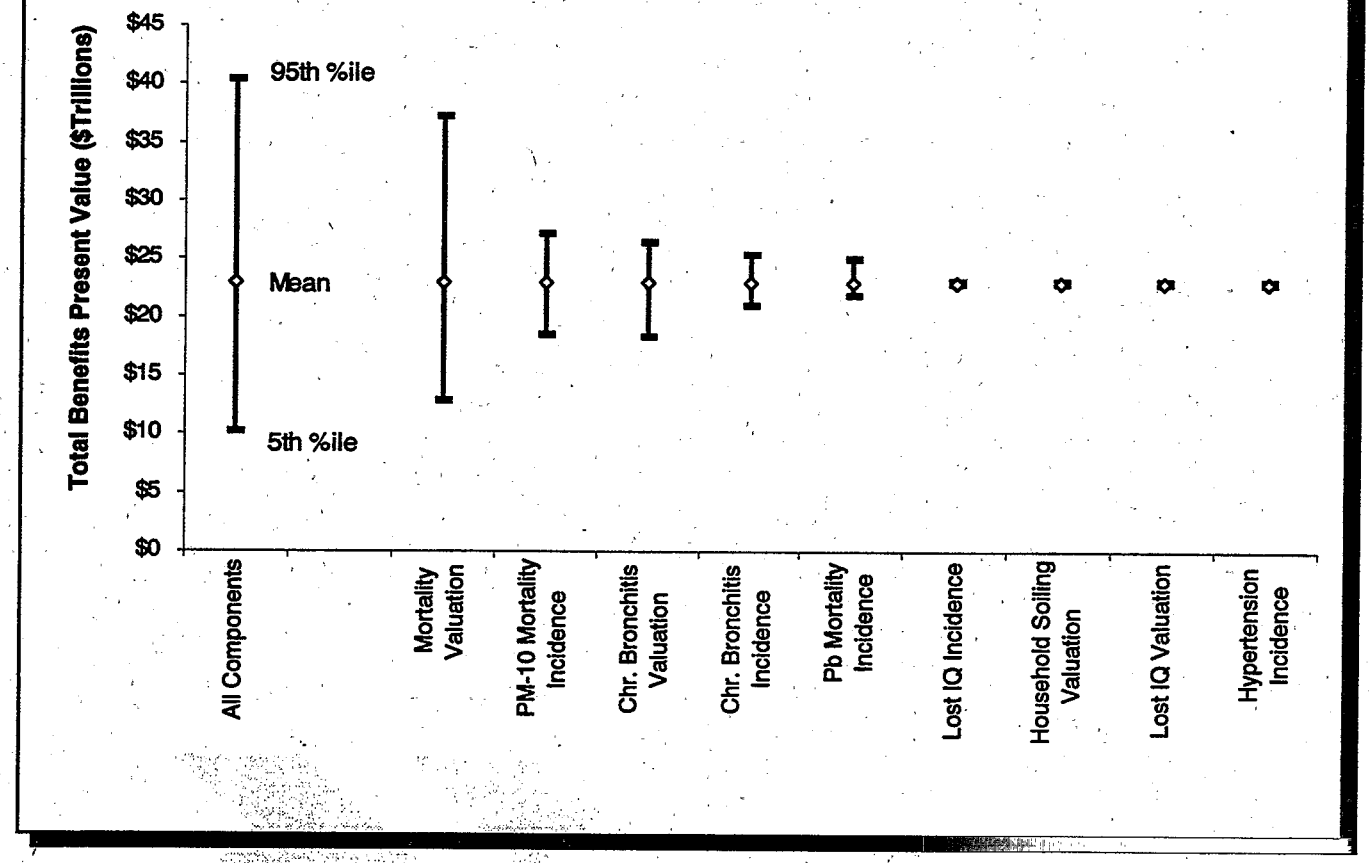
Present value reflects compounding of benefits from 1971 to 1990.

The estimated uncertainty ranges in Table 115 reflect the measured uncertainty associated with both avoided incidence and economic valuation. A better understanding of the relative influence of individual uncertain variables on the overall uncertainty in the analysis can be gained by isolating the individual effects of important variables on the range of estimated benefits. This can be accomplished by holding all the inputs to the Monte Carlo uncertainty analysis constant (at their mean values), and allowing only one variable -- for example, the economic valuation of mortality -- to vary across the range of that variable's uncertainty. The sensitivity analysis then isolates how this single source of variability contributes to the variation in estimated total benefits. The results are summarized in Figure 56. The nine individual uncertainty factors that contribute the most to the overall uncertainty are shown in Figure 56, ordered by the relative significance of their contribution to overall

[294] The estimated economic value of lost IQ points due to lead exposure is based on the present value of the impact on lifetime earnings. A discount rate is required to calculate that present value. The impact on income primarily occurs during adulthood, which is 20 to 70 years after the initial lead exposure. This significant lag results in the discount rate having a significant impact on the estimated economic benefits of the IQ loss.

uncertainty. Each of the additional sources of quantified uncertainty in the overall analysis not shown contribute a smaller amount of uncertainty to the estimates of monetized benefits than the sources that are shown.

Figure 55. Uncertainty Ranges Deriving From Individual Uncertainty Factors



Because of the multiple uncertainties in the benefits estimation, the total estimated present value of the monetary benefits of the 1970 to 1990 Clean Air Act range from a low of about $10 trillion to a high of about $40 trillion (in 1990 dollars, discounted at five percent). Most of the uncertainty in the total estimated benefit levels comes from uncertainty in the estimate of the economic valuation of mortality, followed by the uncertainty in the incidence of mortality from PM (as a surrogate for all non-lead air pollution). The incidence of lead-induced mortality also has a significant influence on the overall uncertainty. The importance of mortality is not surprising, because the benefits associated with reduced mortality are such a large share of overall monetized benefits.

The uncertainty in both the incidence and valuation of chronic bronchitis are the two other significant factors driving the overall uncertainty range. The modeled uncertainty in the other remaining health and welfare endpoints in the analysis contribute relatively small amounts to the overall uncertainty in the estimate of total monetary benefits of the Clean Air Act. Most of these other endpoints account for a relatively small proportion of the overall benefits estimates, making it unlikely that they could contribute significantly to the overall uncertainty. Estimates of either the mean values or standard errors of these variables are generally very small relative to estimated total monetary benefits.

Because the economic benefits associated with premature mortality are the largest source of monetized benefits in the analysis, and because the uncertainties in both the incidence and value of premature mortality are the most important sources of uncertainty in the overall analysis, it is useful to examine the mortality benefits estimation in greater detail. The analytical procedure used in the main analysis to estimate the monetary benefits of avoided mortality assumes that the appropriate economic value for each incidence is a value from the currently accepted range of the value of a statistical life. As documented previously, the estimate value per predicted incidence of excess premature mortality is modeled as a Weibull distribution, with a mean value of $4.8 million and a standard deviation of $3.2 million. This estimate is based on 26 studies of the value of mortal risks. Although there is considerable variation in the analytical designs and data used in the 26 underlying studies, the majority of the studies involve the value of risks to a middle-aged population. Most of the studies examine differences in wages of risky occupations, using a wage-hedonic approach. Certain characteristics of both the population affected and the mortality risk facing that population are believed to affect the average willingness to pay (WTP) to reduce the risk. The appropriateness of a distribution of WTP estimates from 26 studies for valuing the mortality-related benefits of reductions in air pollution concentrations therefore depends not only on the quality of the studies (i.e., how well they measure what they are trying to measure), but also on (1) the extent to which the subjects in the studies are similar to the population affected by changes in pollution concentrations, and (2) the extent to which the risks being valued are similar.

If the individuals who die prematurely from air pollution are consistently older than the population in the valuation studies, the mortality valuations based on middle-aged people may provide a biased estimate of the willingness to pay of older individuals to reduce mortal risk. There is some evidence to suggest that the people who die prematurely from exposure to ambient particulate matter tend to be older than the populations in the valuation studies. In the general U.S. population far more older people die than younger people; 88 percent of the deaths are among people over 64 years old. It is difficult to establish the proportion of the pollution-related deaths that are among the older population because it is impossible to isolate individual cases where one can say with even reasonable certainty that a specific individual died because of air pollution.

There is considerable uncertainty whether older people will have a greater willingness to pay to avoid risks than younger people. There is reason to believe that those over 65 are, in general, more risk averse than the general population, while workers in wage-risk studies are likely to be less risk averse than the general population. More risk averse people would have a greater willingness to pay to avoid risk than less risk averse people. Although the list of recommended studies excludes studies that consider only much-higher-than-average occupational risks, there is nevertheless likely to be some selection bias in the remaining studies -- that is, these studies are likely to be based on samples of workers who are, on average, more risk-loving than the general population. In contrast, older people as a group exhibit more risk averse behavior.

In addition, it might be argued that because the elderly have greater average wealth than those younger, the affected population is also wealthier, on average, than wage-risk study subjects, who tend to be blue collar workers. It is possible, however, that among the elderly it is largely the poor elderly who are most vulnerable to pollution-related mortality risk (e.g., because of generally poorer health care). If this is the case, the average wealth of those affected by a pollution reduction relative to that of subjects in wage-risk studies is uncertain.

Although there may be several ways in which job-related mortality risks differ from air pollution-related mortality risks, the most important difference may be that job-related risks are incurred voluntarily whereas air pollution-related risks are incurred involuntarily. There is some evidence (see, for example, Violette and Chestnut, 1983) that people will pay more to reduce involuntarily incurred risks than risks incurred

voluntarily. If this is the case, WTP estimates based on wage-risk studies may be downward biased estimates of WTP to reduce involuntarily incurred air pollution-related mortality risks.

Finally, another important difference related to the nature of the risk may be that workplace mortality risks tend to involve sudden, catastrophic events, whereas air pollution-related risks tend to involve longer periods of disease and suffering prior to death. Some evidence suggests that WTP to avoid a risk of a protracted death involving prolonged suffering and loss of dignity and personal control is greater than the WTP to avoid a risk (of identical magnitude) of sudden death. To the extent that the mortality risks addressed in this assessment are associated with longer periods of illness or greater pain and suffering than are the risks addressed in the valuation literature, the WTP measurements employed in the present analysis would reflect a downward bias.

The direction of bias resulting from the age difference is unclear, particularly because age is confounded by risk aversion (relative to the general population). It could be argued that, because an older person has fewer expected years left to lose, his WTP to reduce mortality risk would be less than that of a younger person. This hypothesis is supported by one empirical study, Jones-Lee et al. (1985), that found the value of a statistical life at age 65 to be about 90 percent of what it is at age 40. Citing the evidence provided by Jones-Lee et al. (1985), a recent sulfate-related health benefits study conducted for EPA (Hagler Bailly Consulting, 1995) assumes that the value of a statistical life for those 65 and over is 75 percent of what it is for those under 65.

There is substantial evidence that the income elasticity of WTP for health risk reductions is positive (see, for example, Alberini et al., 1994; Mitchell and Carson, 1986; Loehman and Vo Hu De, 1982; Gerking et al., 1988; and Jones-Lee et al., 1985), although there is uncertainty about the exact value of this elasticity). Individuals with higher incomes (or greater wealth) should, then, be willing to pay more to reduce risk, all else equal, than individuals with lower incomes or wealth. Whether the average income or level of wealth of the population affected by PM reductions is likely to be significantly different from that of subjects in wage-risk studies, however, is unclear, as discussed above.

Finally, there is some evidence (see, for example, Violette and Chestnut, 1983) that people will pay more to reduce involuntarily incurred risks than risks incurred voluntarily. If this is the case, WTP estimates based on wage-risk studies may be downward biased estimates of WTP to reduce involuntarily incurred pollution-related mortality risks.

The need to adjust wage-risk-based WTP estimates downward because of the likely upward bias introduced by the age discrepancy has received significant attention (see Chestnut, 1995; IEc, 1992). If the age difference were the only difference between the population affected by pollution changes and the subjects in the wage-risk studies, there might be some justification for trying to adjust the point estimate of $4.8 million downward. Even in this case, however, the degree of the adjustment would be unclear. There is good reason to suspect, however, that there are biases in both directions. Because in each case the extent of the bias is unknown, the overall direction of bias in the mortality values is similarly unknown. Adjusting the estimate upward or downward to compensate for any one source of bias could therefore increase the degree of bias. Therefore, the range of values from the 26 studies is used in the primary analysis without adjustment.

Examining the sensitivity of the overall results to the mortality values can help illuminate the potential impacts of alternative mortality valuations. As mentioned above, a contractor study performed for EPA used one approach to evaluate the economic value of sulfate-related human health improvements resulting from CAAA90 Title IV acid rain controls. That study assumed that 85 percent of the people dying from sulfates (an important component of particulate matter) were over 65, and that people over 65 have a willingness to pay to avoid a

mortal risk that is 75 percent of the values that middle-aged people have. Using this approach, the value of an average statistical life (using a weighted averages) is reduced to 79 percent of the previous value.

An alternative mortality valuation approach emphasizes differences in the amount of remaining life foregone, measured as life-years-lost (LYL), by individuals dying at different ages. Under this approach mortality in older people, with shorter life expectancies than younger people, will have a smaller number of LYL than younger people. Implementing this approach requires three separate estimates: (1) the value of each life-year-lost; (2) the number of people dying at each specific and age, and (3) the number of life-years-lost by people of specific ages dying of air pollution. At this time, the state of the epidemiological science and economic valuation science are insufficiently advanced to allow confident use of the results of this method. However, the following section briefly discusses each of these issues, and suggests possible approaches for each factor. In spite of the substantial uncertainties and lack of available information, this section presents an example of a preliminary estimate of the present value of avoided premature mortality using this approach.

Moore and Viscusi (1988) suggest one approach for determining the value of a life-year-lost. Assume that the reported willingness to pay to avoid a mortal risk is the sum of the value of a single year times the number of years of expected life remaining for an individual. They suggest that a typical respondent in a mortal risk study may have a life expectancy of an additional 35 years. Using a mean estimate of $4.8 million, their approach would yield an estimate of $137,000 per life-year-lost. A variation on this approach assumes that the mean value of a mortal risk is the present value of the value for each additional life year. If an individual discounts future additional years using a standard discounting procedure, the value of each life year lost must be greater than the value assuming no discounting. Using a 35 year life expectancy, a $4.8 million value of a statistical life, and a 5 percent discount rate, the implied value of each life year lost is $293,000. The Moore and Viscusi procedure is identical to this approach, but uses a zero discount rate.

Using the LYL approach, the estimates of the value of a statistical life vary by age. For example, if the mean value of avoiding a statistical death is $4.8 million for a person with a life expectancy of 35 additional years, the resulting value using a five percent discount rate for a person between 65 and 74, who has a life expectancy[295] of 14 years, is 60% ($2.9 million) as much. Using a zero percent discount rate, the value is 40% ($1.9 million) as large. Older people, who have shorter life expectancies, have smaller values. For comparison, the approach used in the acid rain benefits study is 63% as large for all people greater than 65.

The second component required to implement a life-years-lost approach is an estimate of the number of deaths in each age cohort. In order to do this, it is preferable to have age-specific relative risks. Many of the epidemiological studies do not provide any estimate of such age-specific risks. In this case, the age-specific relative risks must be assumed to be identical.

Some of the available epidemiology studies on PM do provide some estimates about age specific relative risks. The information that is available suggests that the increase in relative risk is greater for older people. Most of the available information comes from short-term exposure studies. There is considerable uncertainty in applying the evidence from short-term exposure studies to results from long-term (chronic exposure) studies. Moreover, the available studies offer different estimates of the relationship between the relative risks at various ages, and some of the information (such as risks decreasing for people ages 65 to 74) is counter-intuitive. However, using the available information on the relative magnitudes of the relative risks, it is possible to form a preliminary assessment of the relative risks by different age classes.

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[295] All life expectancies and death rates used in this analysis come from the 1992 U.S. Vital Statistics.

The analysis presented below uses two alternative assumptions about age specific risks; (1) there is a constant relative risk (obtained directly from the health literature) that is applicable to all age cohorts, and (2) the relative risks differ by age, as estimated from the available literature. The age-specific relative risks used in the example below assume that the relative risks for people under 65 are only 16% of the population-wide average relative risk, the risks for people from 65 to 74 are 83% of the population-wide risk, and people 75 and older have a relative risk 55 percent greater than the population average. Details of this approach is provided in Post and Deck, (1996).

The third component of the life-years-lost approach is an estimate of the number of life years lost by a person dying prematurely at each given age. The approach developed for this analysis[296] assumes that exposure to elevated levels of PM increases the probability of dying at a specific age. Increasing the probability of dying at each age (using the age-specific relative risks) lowers the life expectancy for each age cohort. The average number of life-years-lost will depend on the distribution of ages in the population in a location. Using identical age specific relative risks and data for Los Angeles County, each premature mortality is a loss of 15.6 years of life (on average across all age cohorts). Using the age-specific relative risk estimates developed for this analysis, each premature mortality is a loss of 9.8 years.

The present value benefits estimates for PM-related mortality using these alternative approaches are shown in Table 120. Table 120 is based on a single health study: Pope et al., 1996. Alternative studies, or the uncertainty approach used in the primary analysis, would result in a similar pattern of the relationship between valuation approaches.

Table 120. Alternative Estimates of the Present Value of Mortality Associated With PM (based on Pope et al., 1996, in trillions of 1990 dollars).

Valuation Procedure	Present Value of PM Mortality Benefits
Primary Analysis Method ($4.8 million per statistical life saved)	$21.0
Assuming 85% of people are >64, with a value of a statistical life = 75% of average value	$16.5
Life Years Lost approaches	
Single relative risk, valuation using 5% discounting	$12.4
Approximate age-specific relative risk, valuation using 5% discounting	$9.5
Approximate age-specific relative risk, valuation using 0% discounting	$6.1

Notes:

Present value reflects compounding of benefits from 1971 to 1990, using a 5 percent discount rate..

The life-years-lost approach develops two important aspects of uncertainty in the mortality estimates used in the primary analysis. There is another important uncertainty about these estimates. The life-years-lost

[296] The basic approach developed here is identical to the approach used in an EPA study of the Phase I NO_x emission provisions of the acid rain bill (EPA, 1994). That approach, based on unpublished work by John Evans, estimated 11.7 life years lost per death associated with pollution.

approach used here assumes that people who die from air pollution are typical of people in their age group. The estimated value of the quantity of life lost assumes that the people who die would have had an average life expectancy. However, it is possible that the people who die from air pollution are already in ill health, and that their life expectancy is less than a typical person of their age. If this is true, than the number of life years lost per PM-related death would be lower than calculated here, and the economic value would be smaller.

The extent to which adverse effects of particulate matter exposure are differentially imposed on people of advanced age and/or poor health is one of the most important current uncertainties in air pollution-related health studies. There is limited information, primarily from the short-term exposure studies, which suggests that at least some of the estimated premature mortality is imposed disproportionately on people who are elderly and/or of poor health. The Criteria Document (EPA, 1996) identifies only two studies which attempt to evaluate this disproportionality. Spix et al. (1993) suggests that a small portion of the PM-associated mortality occurs in individuals who would have died in a short time anyway. Cifuentes and Lave (1996) found that 37 to 87 percent of the deaths from short-term exposure could have been premature by only a few days, although their evidence is inconclusive.

Long-term studies provide evidence that a portion of the loss of life associated with long-term exposure is independent of the death from short-term exposures, and that the loss of life-years measured in the long-term studies could be on the order of years. Estimates have been made that the chronic exposure studies suggest a loss of life on the order of 1 to 2 years (Lippmann, 1995), but there are no published results derived from the data used in the chronic exposure studies.

If much of the premature mortality associated with PM represents short term prematurity of death imposed on people who are elderly and/or of ill health, the estimates of the monetary benefits of avoided mortality may substantially overestimate society's total willingness to pay to avoid particulate matter-related premature mortality. On the other hand, if the premature mortality measured in the chronic exposure studies is detecting excess premature deaths which are largely independent of the deaths predicted from the short term studies, and the disproportionate effect on the elderly and/or sick is modest, the benefits measured in this report could be underestimates of the total value. At this time there is insufficient information from both the medical and economic sciences to satisfactorily resolve these issues from a theoretical/analytical standpoint. Until there is evidence from the physical and social sciences which is sufficiently compelling to encourage broad support of age-specific values for reducing premature mortality, EPA will continue to use for its primary analyses a range of values for mortaltiy risk reduction which assumes society values reductions in pollution-related premature mortality equally regardless of who receives the benefit of such protection.

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Appendix J: Future Directions

Research Implications

In virtually any benefit analysis of environmental issues, the state of scientific information limits the degree of coverage possible and the confidence in benefit estimation. For most benefit categories, further scientific research would allow for a better quantification of benefits. One of the major outcomes of the retrospective analysis is a clear delineation of the major limitations in the scientific and economics literature in carrying out an analysis of this scope. Often, a list of research needs is generated in studies such as this, but there is no clear internal mechanism to address these needs. With this study (and the ongoing Section 812 program), a process has been initiated where identified research needs are to be integrated into EPA's overall extramural research grants program, administered by the Office of Research and Development, particularly in the areas of air toxics and ecological effects. It is hoped that the research projects that flow from this process will enable future analyses to be more comprehensive.

Certain of the limitations in the retrospective analysis are directly related to the historical nature of the analysis, such as sparse information about air quality in the early 1970's in many areas in the country. Other important limitations are related to the effects of elevated airborne lead concentrations, which has been virtually eliminated by the removal of lead from gasoline. A better understanding of these relationships would improve our understanding of the historical impact of the Clean Air Act, but would only indirectly contribute to developing future air pollution policy. However, most of research that will reduce the major gaps and uncertainties needed to improve the Section 812 analyses will be directly relevant to EPA's primary ongoing mission of developing and implementing sound environmental policies to meet the national goals established in the Clean Air Act and other legislation.

There are a number of areas of biological, physical and economic research areas that the EPA Project Team identified as particularly important for improving future Section 812 analyses. The following discussion is not an evaluation of the current state of scientific knowledge designed to identify specific areas where a modest amount of targeted research efforts would likely yield important and useable information in the near future. Instead, the research topics discussed below are likely to have a significant impact on the overall estimates of the costs and benefits of the Clean Air Act. Significant impacts may arise from research involving two basic types of information gaps: cost and benefit categories that are not currently quantified in the analysis, and reducing the uncertainty in the costs and benefit categories that are included.

One of the challenges in preparing the retrospective analysis was modeling the integrated relationships between emissions of many different chemicals, the subsequent mixture of pollutants in the ambient air, and the resulting health and welfare effects of simultaneous exposure to multiple pollutants. A major portion of the uncertainty in the analysis derives from the limited current understanding of any interactive (synergistic or antagonistic) effects of multiple pollutants. The need to better understand these complex issues is not a limited scientific question to improve Section 812 analyses, but is the primary focus of EPA's current activities (organized under the Federal Advisory Council Act (FACA) process) to develop an integrated set of attainment policies dealing with ozone, particulate matter, sulfur and nitrogen

oxides, and visibility. Further research on multi-pollutant issues is likely to substantially reduce an important source of unmeasured uncertainty in the Section 812 analyses.

Effect categories that are not included in the quantified results of the retrospective analysis are perhaps the research subjects with the greatest potential for significantly affecting future Section 812 analyses. It is very difficult to get a balanced overall view of the impact of the Clean Air Act when there are effects (both costs and benefits) that are not reflected in the aggregate results. Whether it is a case of known (or highly likely) effects that cannot be adequately quantified at this time because of certain knowledge gaps, or the potential of widespread or devastating effects we are only beginning to understand, omitted effects greatly expand the subjective uncertainty range. Research may also lead to a better understanding that certain potential effects are not as likely to occur, or are not as damaging, as the current state of knowledge suggests. Physical, biological and economic science research that helps fill these gaps and narrow the uncertainty ranges will improve the EPA's ability to better depict all the effects of the Clean Air Act.

Examples of presently omitted or underrepresented effect categories which could have a substantial effect on the aggregate results include health effects of air toxics, health effects of chronic exposure to criteria pollutants, ecosystem effects, any long-term impact of displaced capital on productivity slowdown and redirected technological innovation, and potential social costs of transitional unemployment.

Although much of the emissions reductions that occurred between 1970 and 1990 as a result of the CAA were aimed at attaining the criteria pollutant standards, the emissions of air toxics were substantially reduced as well. Some air toxics were deliberately controlled because of their known or suspected carcinogenicity, while other toxic emissions were reduced indirectly due to control procedures aimed at other pollutants, particularly ozone and particulate matter. The current Section 812 analyses was able to present only limited information on the changes in air toxic emissions. These knowledge gaps will be even more serious for future Section 812 analyses, which will be examining the effects of an expanded air toxic program under the CAA Title III, as well as toxic emissions reduced by other sections of the Act. Existing knowledge gaps that prevented a more complete consideration of toxics in the retrospective analysis include methods to estimate changes in acute and chronic ambient exposure conditions nationwide, concentration-response relationships linking exposure and health or ecological outcomes, and economic valuation methods for a broad array of potential serious health effects such as renal damage, reproductive effects and non-fatal cancers, and potential ecological effects of air toxics.

Scientific information about the effects of long-term exposure to air pollutants is just beginning to emerge, but continues to be the subject of intense scientific inquiry. The relationship between chronic PM exposure and excess premature mortality (Pope et al., 1995) included in the quantified results of the retrospective analysis is one example of such research. However, many other potential chronic effects that are both biologically plausible and suggested by existing research are not included. Research to identify the relationship linking certain known or hypothesized physical effects (e.g., ozone's effects on lung function or fibrosis) with the development of serious health effects (e.g., cardio-pulmonary diseases or premature mortality), and the appropriate economic valuation of the willingness to pay to avoid the risks of such diseases, would reduce the uncertainty caused by a major category of omitted health effects that could have a significant impact on the aggregate benefits estimates.

Future research concerning certain effects that are included in the retrospective analysis will also help to substantially reduce the range of uncertainty that is reflected in the aggregate results. As described in Chapter 7 and Appendix I, premature mortality is both the largest source of benefits and the major source of quantified uncertainty in the retrospective analysis. In addition to the quantified uncertainty, there is considerable additional unquantified uncertainty about premature mortality associated with air pollution. Much of the information base about these relationships is relatively new, more is coming out virtually daily, and there is substantial disagreement in the scientific community about many of the key issues. EPA's Research Strategy and Research Needs document for particulate matter, currently under development, will address many of these scientific issues as they relate to PM. The following selection of highly uncertain issues could have a significant impact on both the aggregate mortality benefits estimates and the measured uncertainty range:

- The relationship of specific pollutants in the overall premature mortality effect, including the individual or interactive relationships between specific chemicals (e.g., ozone, sulfates, nitrates, and acid aerosols), and particle sizes (i.e., coarse, fine and ultra-fine particles).

- The degree of overlap (if any) between the measured relationships between effects associated with short term exposures and effects from long term exposure.

- The confounding effect of changes in historic air pollution, including changes over time in both pollution levels and the composition of the pollutant mix.

- The extent to which life spans are shortened by exposure to the pollutants, and the distribution of ages at the time of death.

- The willingness to pay to avoid the risks of shortened life spans.

After premature mortality, chronic bronchitis is the next largest health effect benefit category included in the retrospective analysis. There is considerable measured uncertainty about both the incidence estimation and the economic valuation. Additional research could reduce uncertainties about the level of the pollutants associated with the observed effects, the baseline incidence used to model the changes in the number of new cases, and the correspondence between the definition of chronic bronchitis used in the health effects studies and the economic valuation studies.

Further research on the willingness to pay to avoid the risk of hospital admissions for specific conditions would reduce a potentially significant source of non-measured uncertainty. The Project Team used "avoided costs" for the value of an avoided hospital admission, based on the avoided direct medical cost of hospitalization (including lost wages for the employed portion of the hospitalized population). Avoided costs are likely to be a substantial underestimate of the appropriate willingness to pay, especially for such serious health effects as hospitalization for strokes and congestive heart failure, particularly because they omit the value of avoided pain, suffering, and inconvenience. Furthermore, in addition to hospitalization, there is evidence that some people seek medical assistance as outpatients. It is also likely that there are additional people adversely affected by short-term air pollution levels who seek physician services (but stop short of hospital admissions). Revised estimates of the appropriate economic value of avoided hospitalization and other primary care medical services could increase the total economic benefits of this cluster of health effects sufficiently that it could be a much larger portion of the aggregate benefit total.

In addition to research to improve the understanding of the impacts of changes in air pollution on human health and well-being, further research on the non-health impacts could result in a better depiction of the adverse impacts on a wide array of effect categories. Improvements in air quality have likely resulted in improvements in the health of aquatic and terrestrial ecosystems and the myriad of ecological services they provide, but knowledge gaps prevent them from being included in the Section 812 analysis at this time. Additional research in both scientific understanding and appropriate modeling procedures could lead to including such benefits as improvements in water quality stemming from a reduction in acid deposition-related air pollutants. Such improvements would benefit human welfare through enhancements in certain consumptive services such as commercial and recreational fishing, in addition to non-consumptive services such as wildlife viewing, maintenance of biodiversity, and nutrient cycling. Similarly, increased growth, productivity and overall health of U.S. forests could occur from reducing ozone, resulting in benefits from increased timber production, greater opportunities for recreational services such as hunting, camping, wildlife observation, and nonuse benefits such as nutrient cycling, temporary CO_2 sequestration, and existence value. While there is insufficient information to adequately model the short-run ecological and ecosystem impacts, even less is known about the long-run effects of prolonged exposure. Permanent species displacement or altered forest composition are examples of potential ecosystem effects that are not reflected in the current monetized benefit analysis, and could be a substantial source of additional benefits.

Future Section 812 Analyses

This Retrospective Study of the benefits and costs of the Clean Air Act was developed pursuant to Section 812 of the 1990 Clean Air Act Amendments. Section 812 also requires EPA to generate an ongoing series of prospective studies of the benefits and costs of the Act, to be delivered as Reports to Congress every two years.

Design of the first Section 812 Prospective Study commenced in 1993. The EPA Project Team developed a list of key analytical design issues and a "strawman" analytical design reflecting notional decisions with respect to each of these design issues.[297] The analytical issues list and strawman design was presented to the Science Advisory Board Advisory Council on Clean Air Compliance Analysis (ACCACA), the same SAB review group which has provided review of the Retrospective Study. Subsequently, the EPA Project Team developed a preliminary design for the first Prospective Study. Due to resource limitations, however, full-scale efforts to implement the first Prospective Study did not begin until 1995 when expenditures for Retrospective Study work began to decline as major components of that study were completed.

As for the Retrospective, the first Prospective Study is designed to contrast two alternative scenarios; however, in the Prospective Study the comparison will be between a scenario which reflects full implementation of the CAAA90 and a scenario which reflects continued implementation only of those air pollution control programs and standards which were in place as of passage of the CAAA90. This means that the first Prospective Study will provide an estimate of the incremental benefits and costs of the CAAA90.

[297] Copies of the Prospective Study planning briefing materials are available from EPA.

The first Prospective Study is being implemented in two phases. The first phase involves development of a screening study, and the second phase will involve a more detailed and refined analysis which will culminate in the first Prospective Study Report to Congress. The screening study compiles currently available information on the costs and benefits of the implementation of CAAA90 programs, and is intended to assist the Project Team in the design of the more detailed analysis by providing insights regarding the quality of available data sources and analytical models, and the relative importance of specific program areas; emitting sectors; pollutants; health, welfare, and ecological endpoints; and other important factors and variables.

In developing and implementing the Retrospective Study, the Project Team developed a number of important modeling systems, analytical resources, and techniques which will be directly applicable and useful for the ongoing series of section 812 Prospective Studies. Principal among these are the Criteria Air Pollutant Modeling System (CAPMS) model developed to translate air quality profile data into quantitative measures of physical outcomes; and the economic valuation models, coefficients, and approaches developed to translate those physical outcomes to economic terms.

The Project Team also learned valuable lessons regarding analytical approaches or methods which were not as productive or useful. In particular, the Project Team plans not to perform macroeconomic modeling as an integral part of the first Prospective Analysis. In fact, there are currently no plans to conduct a macroeconomic analysis at all. Essentially, the Project Team concluded, with confirmation by the SAB ACCACA, that the substantial investment of time and resources necessary to perform macroeconomic modeling would be better invested in developing high quality data on the likely effects of the CAA on key emitting sectors, such as utilities, on-highway vehicles, refineries, etc. While the intended products of a macroeconomic modeling exercise –such as overall effects on overall productivity, aggregate employment effects, indirect economic effects– are of theoretical interest, the practical results of such exercises in the context of evaluating environmental programs may be disappointing.

First, the CAA has certainly had a significant affect on several industrial sectors. However, the coarse structure of a model geared toward simulating effects across the entire economy requires crude and potentially inaccurate matching of these polluting sectors to macroeconomic model sectors. For example, the J/W model used for the Retrospective Study has only 35 sectors, with Electric Utilities comprising a single sector. In reality, a well-structured analysis of the broader economic effects of the CAA would provide for separate and distinct treatment of coal-fired utility plants, oil-fired plants, and so on. Furthermore, the outputs of the macroeconomic model are too aggregated to provide useful and accurate input information for the sector-specific emission models used to project the emissions consequences of CAA programs. Again, the critical flaw is the inability to project important details about differential effects on utilities burning alternative fuels.

The second critical problem with organizing a comprehensive analysis of the CAA around a macroeconomic modeling approach is that the effect information produced by the macroeconomic model is relatively unimportant with respect to answering the fundamental, target variable: *"How do the overall health, welfare, ecological, and economic benefits of Clean Air Act programs compare to the costs of these programs?"* The Project Team believes that any adverse effect, no matter how small in a global context, should not be deemed "insignificant" if even one individual is seriously harmed. However, the Retrospective Study results themselves have shown that, when analytical resources are limited, the resources invested in the macroeconomic modeling would have been better spent to provide a more complete and less uncertain assessment of the benefit side of the equation. Even on the cost side of the

equation, it is far more important to invest in developing accurate and reliable estimates of sector-specific compliance strategies and the direct cost implications of those strategies. This will be even more true in the Prospective Study context when the Project Team will be faced with forecasting compliance strategies and costs rather than simply compiling survey data on actual, retrospective compliance expenditures.

The third and most important limitation of macroeconomic modeling analysis of environmental programs is that, unlike the economic costs of protection programs, the economic benefits are not allowed to propagate through the economy. For example, while productivity losses associated with reduced capital investment due to environmental regulation are counted, the productivity gains resulting from reduced pollution-related illness and absenteeism of workers are not counted. The resulting imbalance in the treatment of regulatory consequences raises serious concerns about the value of the macroeconomic modeling evaluation of environmental programs. In the future, macroeconomic models which address this and other concerns may be developed; however, until such time EPA is likely to have limited confidence in the value of macroeconomic analysis of even broad-scale environmental protection programs.

Based on these findings and other factors, the design of the first Prospective Study differs in important ways from the Retrospective Study design. First, rather than relying on broad-scale, hypothetical, macroeconomic model-based scenario development and analysis, the first Prospective Study will make greater use of existing information from EPA and other analyses which assess compliance strategies and costs, and the emission and air quality effects of those strategies. After developing as comprehensive a data set as possible of regulatory requirements, compliance strategies, compliance costs, and emissions consequences, the data set will be reviewed, refined, and extended as feasible and appropriate. In particular, a number of in-depth sector studies will be conducted to develop up-to-date, detailed projections of the effects of new CAA requirements on key emitting sectors. Potential sectors include, among others, utilities, refineries, and on-highway vehicles.

The first Prospective Study will also differ from the Retrospective Study in that analytical resources will be directed toward development of a more complete assessment of benefits. The deficiencies which prevailed in the Retrospective Study relating to assessment of the benefits of air toxics control will be addressed. In addition, the Project Team will endeavor to provide a more complete and effective assessment of the ecological effects of air pollution control.