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

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Executive Summary

In 1990, Americans received roughly 20 dollars of value in reduced risks of death, illness, and other adverse effects for every one dollar spent to control air pollution.

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 finally 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.

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 passage 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

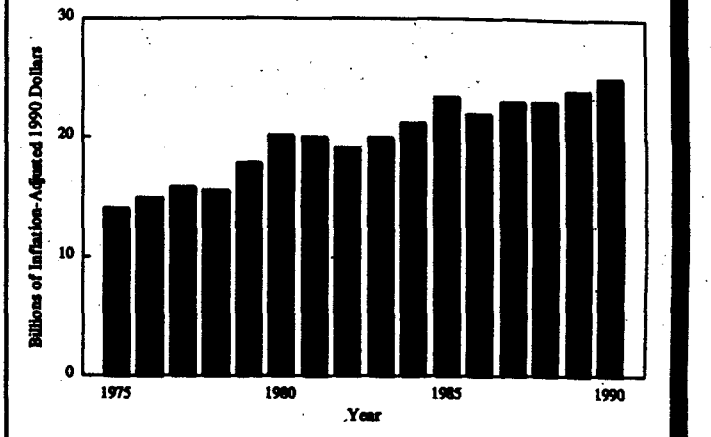
To comply with the Clean Air Act, businesses, consumers, and government entities all incurred higher costs for many goods and services. The costs of providing goods and services to the economy were higher primarily due to requirements to install, operate, and maintain pollution abatement equipment. In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

To tally total compliance costs of 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 1 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.

Figure 1. Total Direct Costs of the CAA (in billions of inflation-adjusted 1990 dollars.)



Indirect Economic Effects

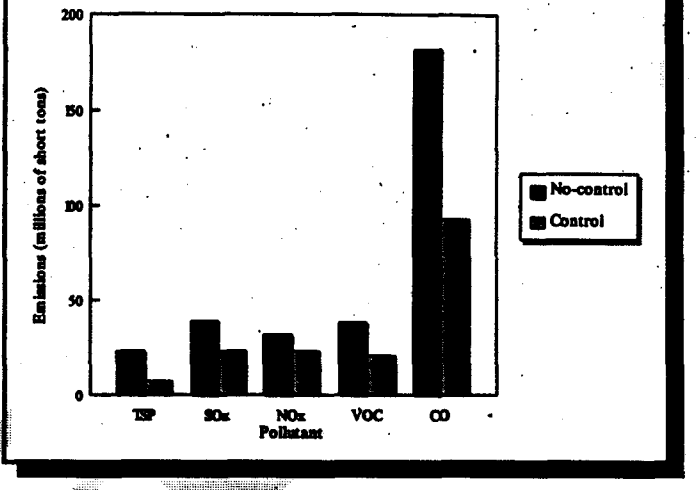
As part of the process of developing input data to model emissions under the control and no-control scenarios, a macroeconomic model was run to estimate changes in key economic variables such as total electricity sales. In addition to providing economic indicators needed for the emissions modeling, the 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 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 model fails to capture the increase in total U.S. production achieved by reductions in air pollution-related worker illness and absenteeism. The macroeconomic model also fails to capture expenditure reductions achieved by the Clean Air Act, such as reductions in medical expenses for air pollution-related illness and disease, and apply those savings to offset the capital displacement effect of compliance expenditures. This imbalanced treatment 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, a conclusion supported by the Science Advisory Board review panel charged with providing scientific and economic review of the study.

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



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 are later 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.

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 thousand of the total 234 thousand ton reduction in total airborne lead emissions under the control scenario. The reduction in total airborne lead emissions from all sources under the control scenario was almost 99 percent.

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 dioxides, 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. 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 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 oxide emissions; and (3) organic aerosols, which derive primarily from 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 oxide 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.

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, valid and reliable quantitative functions 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 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 or millions of cases avoided per year 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 mortality, morbidity, and other outcomes under the two scenarios. The "high" estimates presented in Table 1 reflect the "best estimate" of the concentration-response function with the largest predicted effect. The "low" estimates reflect the "best estimate" of the concentration-response function with the smallest predicted effect. The "mid" value is the average of the "best estimates" of all the concentration-response functions used to estimate each effect.[1] It is important to note that these results do not reflect the uncertainty surrounding the "best estimates," rather, each reported value should be recognized as the average value in a range of possible values.

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

Adverse human health effects of the Clean Air Act "criteria pollutants" sulfur dioxide, nitrogen oxide, 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.

[1] Results for lead (Pb) are handled differently. The lead analysis was conducted using two alternative baselines. The "high" and "low" values reported here reflect the results implied by the two baselines; the "mid" value is the average of the "high" and "low" values.

Reductions in both hazardous air pollutants and criteria pollutants likely yielded widespread improvements in the health 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. Healthier, 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 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 is 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 adverse effects which are amenable to economic valuation permits a summation of monetized benefits and subsequent comparison to the 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, assigning dollar values to outcomes such as loss of human life, pain and suffering, or ecological degradation requires value judgments which should not be left to the economic analyst, or indeed to any one individual or group who would make such judgments on behalf of all Americans. Typically, cost-benefit analysts sidestep such value judgments by adopting the most technically defensible valuation estimates and leave the moral dimensions 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 to perform this cost-benefit analysis. Because of the implied value judgments and substantial uncertainties associated with assigning dollar values to health and environmental effects, EPA strongly encourages readers to look beyond the simple calculus of economic value of the Clean Air Act and consider how they would value the reductions in adverse health and environmental effects estimated in this study.

Even when monetary values can be estimated for a given effect, such estimates are typically fraught with uncertainties. 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.

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, later in the report, as a range of estimates. The mean values are summarized in Table 3 for illustrative purposes, but it must be emphasized that these values only reflect the central tendencies of the ranges of values used for a given effect in this analysis.

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 monetized physical effects calculated 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 benefit category. Using this approach, "restricted activity day" benefits and "lost work day" benefits were not counted in this analysis as separate, duplicative outcomes.

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	$5,350 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 capture the total, aggregate effect of uncertainties in the analysis, a 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 many 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

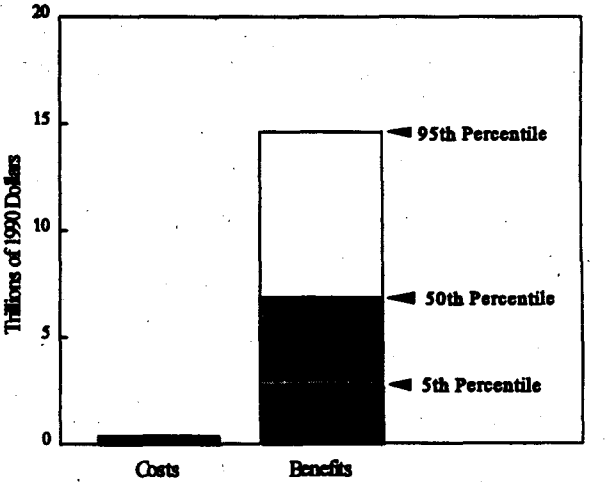
uncertainties associated with earlier steps in the analytical sequence would have required multiple runs of the macroeconomic, emissions, and/or air quality models. This was far beyond the scope of the resources available for this project. However, prior experience in regulatory analysis has demonstrated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

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:

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 2.7 to 14.6 trillion 1990-value dollars, with a central estimate of 6.8 trillion dollars. However, uncertainties in compliance cost estimates, macroeconomic effects estimates, emissions estimates, and air quality modeling results would be expected to broaden the range of possible outcomes. Furthermore, inclusion of unquantified and nonmonetized benefits would likely raise the lower and upper bounds of the range, perhaps by highly significant amounts. More important, the central estimate of 6.8 trillion dollars in benefits may also be significantly underestimated due to the exclusion of large numbers of benefits from the monetized benefit estimate. This range of estimated benefits compares to direct costs of compliance equal to approximately 436 billion dollars in 1990-value dollars over the same 20 year period. Subtracting total costs from total benefits, this analysis indicates that net benefits of 1970 to 1990 Clean Air Act controls probably were within a 90 percent credible interval of 2.3 and 14.2 trillion 1990-value dollars, with a central estimate of 6.4 trillion dollars.

Figure 3 provides a graphical representation of the estimated range of total monetary benefits and compares this range to estimated 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 later in this report, monetized benefits consistently and substantially exceeded costs throughout the 1970 to 1990 period.

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



Dividing the maximum likelihood estimate of 1990 monetized benefits by 1990 estimated direct costs indicates that Americans received approximately 20 dollars of value in reduced risks of death, illness, and other adverse effects for every dollar spent to control air pollution. It is important to remember that even this level of benefit does

not capture the additional value of reductions in hazardous air pollutants, protection of ecosystems, and a variety of other health and environmental effects which could not be quantified or monetized.

Conclusions and Future Directions

First and foremost, the clear implication of 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 eliminating two pollutants: lead and particulate matter. 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. It is inevitable that in any large-scale public policy program, some particular outcomes may yield troubling anecdotes or unfavorable outcomes. These outlier events should clearly be reviewed and mitigated where possible. However, the overriding conclusion to be drawn from this analysis is that the level of investment made under Clean Air Act programs during the 1970 to 1990 period yielded huge dividends.

Because benefits continued to exceed costs by more than an order of magnitude in 1990, it is likely that substantial additional cost-beneficial protection remained to be achieved. In fact, 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 1990 Amendments.[2] Examining this implication of substantial unfinished business to be addressed by the Clean Air Act Amendments of 1990 is the central purpose of the first Section 812 Prospective Study now in progress. As the first in an ongoing series of benefit-cost 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 total 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

[2] U.S. Environmental Protection Agency, *Human Health Benefits from Sulfate Reductions Under Title IV of the 1990 Clean Air Act Amendments*, Office of Air and Radiation / Office of Atmospheric Programs / Acid Rain Division, November 1995.

consistent with the specific control and baseline scenarios to be analyzed. Examples include the Title IV sulfate health study mentioned above and the emissions inventories being developed by the Ozone Transport Assessment Group.

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. To this end, steps have already been taken by EPA to improve coordination between the research and assessment programs of the Agency.

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 adequate resources are not provided, however, or 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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Acronyms and Abbreviations

µeq/L	micro equivalents per liter
µg/m^3	micrograms per cubic meter
µg	micrograms
µM	micrometers, also referred to as microns
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
CAACAC	SAB Clean Air Act Compliance Analysis Council
CAACACPERS	SAB CAACAC Physical Effects Subcommittee
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
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	Department of Agriculture
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

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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 watershed 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 nearly complete picture 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 modern 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 with increased administrative penalties and the role of citizen suits.

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 will assess the costs and benefits to the public health, economy and the environment of legislation before 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.[3] 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 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 include the assumption that states and localities would not have set up air pollution control programs in the absence of the federal CAA. That is, this

[3] While the configuration of these scenarios may appear straightforward, the need to model at least one hypothetical scenario raises a number of issues which are discussed in detail in the main text. For example, the level of state and/or local controls which would have prevailed in the absence of a federal CAA is highly speculative. So while some may argue that states and localities might have provided protections comparable to those required by the federal CAA, adopting that assumption yields a finding of zero costs and zero benefits of the federal Act; a result which is neither informative nor responsive to Congressional and EPA objectives.

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:

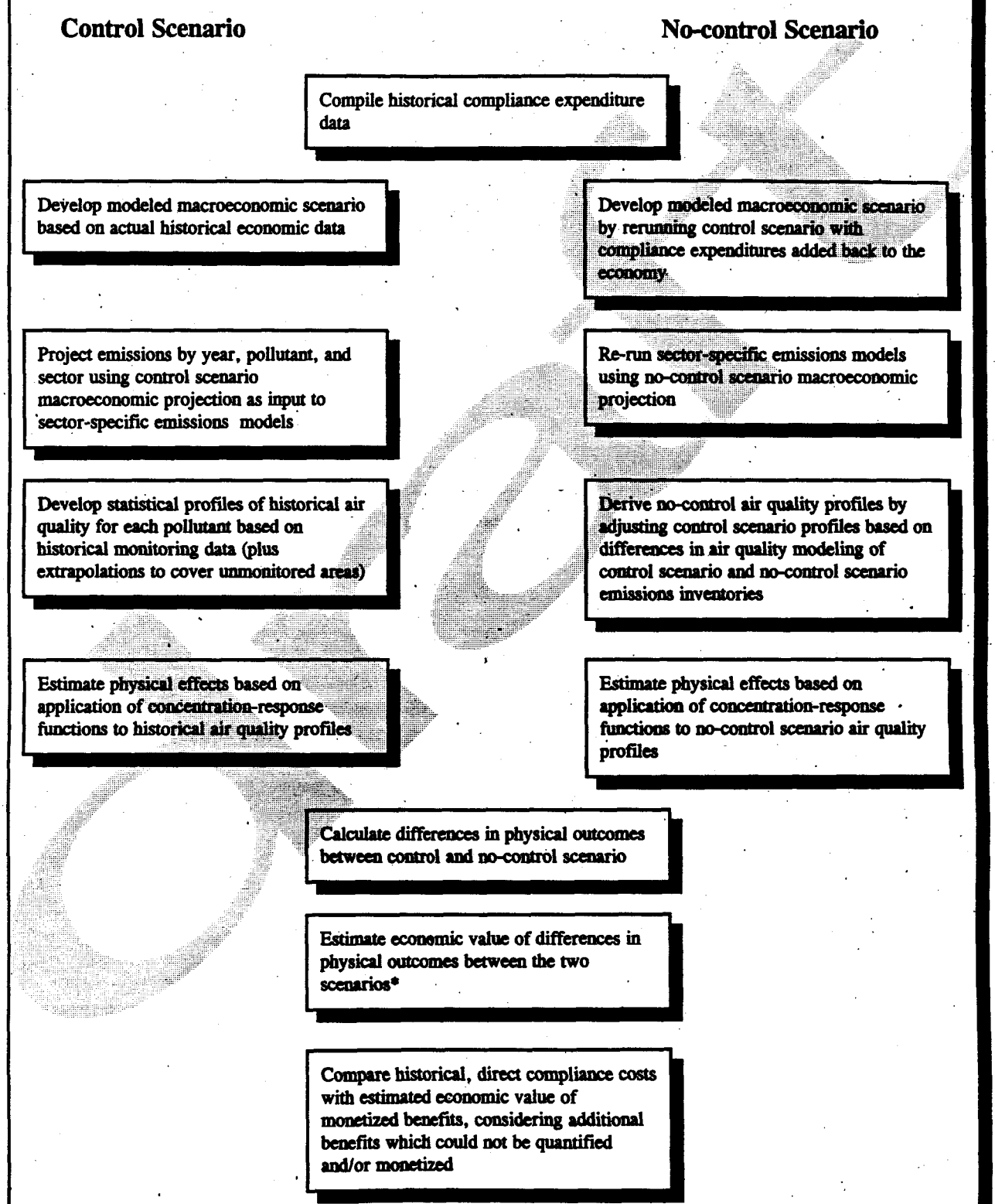
- direct cost estimation
- macroeconomic modeling
- emissions modeling
- air quality modeling
- health and environmental effects estimation
- results aggregation and uncertainty estimation

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 regulatory and budget 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 economicgrowth rates during the 1970 to 1990 period. The model was then re-run for the no-control

[4] It is important to emphasize that 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.

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).

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 were different. 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]

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 staggering 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.[8] 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

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[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.

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.

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

A vastly oversimplified example clarifies the concept: say the actual year-long average 1980 sulfur dioxide concentration at a particular monitor in upstate New York was 0.08 parts per million (ppm). Assume further that the air quality modeling performed for this analysis estimated that the New York state-wide average 1980 concentration were 0.10 ppm under the control scenario and 0.16 ppm under the no-control scenario. In this case, the 0.08 ppm historical figure is adopted as the 1980 control scenario value for that monitor. The value adopted for the 1980 no-control scenario at that monitor would be 0.014 ppm, reflecting the 0.08 ppm control scenario value plus the 0.06 ppm difference between the modeled scenario outcomes.

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.[9]

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.[10]

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, unit values derived from the economic literature were 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.[11] 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.[12] The term "partial" is emphasized because the costs of historical CAA programs were readily obtained but 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. 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]

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[9] 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.

[10] Or, for PM, by county.

[11] 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.

[12] Although considerable uncertainties surround the direct cost, macroeconomic 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.

[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.

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 (CAACAC)– in developing the section 812 analyses. In addition, EPA is required to consult with the Department of Labor and the Department of Commerce.

The CAACAC 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 manner. The SAB CAACAC has met several times during the development of the retrospective study to review methodologies and interim results. While the full CAACAC 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 CAACAC members and members of other SAB committees. The CAACAC Physical Effects Subcommittee has met several times and has provided its own review findings to the full CAACAC. Similarly, the CAACAC Air Quality Subcommittee has held several teleconference meetings to review methodology proposals.

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 CAACAC.

Report Organization

The remainder of the main text of this report summarizes the key methodologies and findings of retrospective study, organized by chapter according to the seven major steps of the analysis. 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 and initial results for the aggregation of monetized benefits of the historical CAA are presented in Chapter 6. Chapter 7 describes the uncertainty analysis and associated re-aggregation of the overall cost and benefit estimates.

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

discusses some of the preliminary research implications of this study and describes plans for future
2 Section 812 analyses.

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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 thee 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.

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.

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

	Expenditures		Annualized Costs
Year	$current	$1990	$1990
1973	7.1	19.5	11.5
1974	8.3	21.3	13.7
1975	10.6	24.4	14.1
1976	11.1	24.1	14.9
1977	11.9	24.0	15.9
1978	12.0	22.6	15.6
1979	14.4	24.8	17.9
1980	16.3	25.6	20.2
1981	17.0	24.4	20.1
1982	16.1	21.8	19.2
1983	16.1	20.1	20.1
1984	18.0	22.4	21.4
1985	19.8	23.7	23.5
1986	18.5	21.6	22.1
1987	18.9	21.4	23.1
1988	18.9	20.6	23.1
1989	19.7	20.6	24.0
1990	20.1	20.1	25.1

Sources: EPA *Cost of Clean* (1990); "Pollution Abatement and Control Expenditures" published annually in the Survey of Current Business by BEA; "Costs and Benefits of Reducing Lead from Gasoline: Final RIA (1985)" (EPA); "Pollution Abatement Costs and Expenditures" published in Current Industrial Reports by Census.

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 over

the 1973 to 1990 period,[14] annual direct CAA compliance expenditures adjusted for inflation were remarkably stable, ranging from $20 billion to $25 billion in 1990 dollars (see Table 4). 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.

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[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.

For this cost/benefit analysis, all capital expenditures were annualized at a 3 percent, inflation-adjusted rate of interest. Therefore, "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 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. Table 11 summarizes costs annualized at 3 percent, as well as annual expenditures.

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. 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

[14] Due to data limitations, the cost analysis for this CAA retrospective starts in 1973, missing costs incurred in 1970-72. *Cost of Clean* (1990), however, which is the source for annualized capital costs, includes capital expenditures for 1972. 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. This limitation is not debilitating in the context of this analysis, 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. It should also be noted that, conversely, only a portion of, for example, the 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 and consistently, 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.

[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.

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.

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 equal to 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 lower than projected under the no-control scenario. This lower level of GNP is equivalent to a reduction of approximately $50 billion in consumer products produced in the United States. However, it is important to emphasize that this analysis addressed only the negative economic impacts of the CAA and did not capture the beneficial economic effects of air pollution control, such as improved worker productivity and reduced medical expenditures.

It is also important to emphasize that changes in GNP 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 "economic 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. That is, changes in production costs and consumer prices should result in decreased consumption by consumers (disregarding the increase in consumption of environmental amenities).

To gauge the magnitude of this effect, the Project Team estimated the monetary compensation which would be required to offset the losses in consumption associated with the control scenario. The resulting estimates indicate that the required monetary compensation is about 120 to 160 percent of CAA compliance cost. That is, if the indirect costs to consumers captured by this analysis are added to direct compliance costs, the resulting total cost would be about 20 to 60 percent greater than direct compliance cost alone. As noted above, however, these additional indirect costs do not reflect the indirect economic benefits associated with improvements in human health and environment. Because of this limitation, the cost-benefit comparisons presented in this report reflect only the differences in direct costs and direct benefits associated with the control and no-control scenarios. If both indirect costs and indirect benefits could be effectively captured, however, the differences between the two scenarios for both total costs and total benefits would likely be higher than presented in this report.

Conclusions

The results of this assessment of the costs and economic effects of the CAA must be carefully interpreted. It is important to remember that the cost and macroeconomic simulation modeling does not incorporate many health, environmental, and economic benefits of the CAA. For example, improvements in worker productivity and reductions in medical expenditures due to the CAA were not incorporated in the macroeconomic model. The direct and indirect benefits of these CAA-related productivity benefits and expenditure reductions are not reflected in the estimates of changes in GNP, consumption, prices, output, and employment presented in this study.

The estimates of compliance cost and economy-wide effects presented in this chapter do not by themselves inform the question of whether the net social and economic benefits of the CAA are positive. This question requires careful and balanced consideration of the benefits of the CAA. The remainder of this report focuses on estimation of these benefits, and the integration and interpretation of the overall costs and benefits of the CAA are presented in subsequent chapters. Additional details about the cost and macroeconomic modeling performed for this assessment are presented in Appendix A.

Differences between control and no-control scenario outcomes were observed at both economy-wide and industry levels. The CAA imposed direct costs of 20 to 25 billion dollars per year on an inflation-adjusted basis. Furthermore, the level of GNP was estimated by the macroeconomic model to be one percent lower in 1990 under the control scenario, and personal consumption was also one percent lower by 1990. The greatest industry-level difference between the two scenarios is realized by the motor vehicle sector, which showed 5.3 percent lower output and 3.8 percent higher price levels under the control scenario. Output effects are also seen in other capital-intensive and energy-intensive industries, including petroleum refining, electric utilities, transportation equipment, primary metals, and metal mining.

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),[16] 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.[17]

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.[18] In other cases, such as the EPA Emissions Trends Report (*Trends)* methodology[19] 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 selection, model configuration, and macroeconomic input data[20] result in unavoidable, but in this case

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

[17] 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.

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

[19] 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.

[20] 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

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.[21]

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.[22]

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.[23] 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 since 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.

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

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estimates and control scenario emission estimates would be further reduced.

[21] 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.

[22] 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)

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

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 5, and are described in more detail in Appendix B.

Table 5. 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.

Summary of Results

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, though 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 CAA 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

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

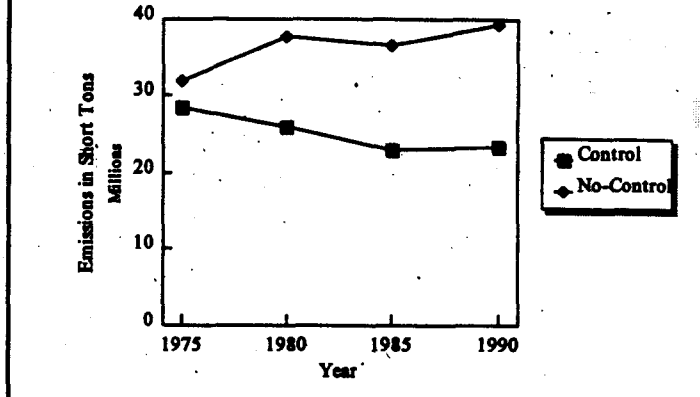


Figure 6. Control and No-control Scenario Total NO_x Emission Estimates.

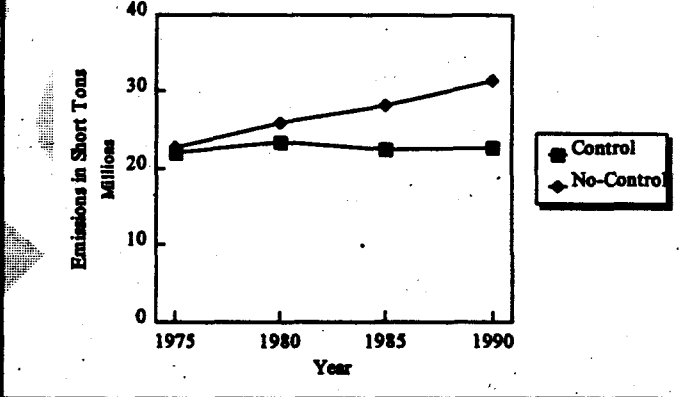
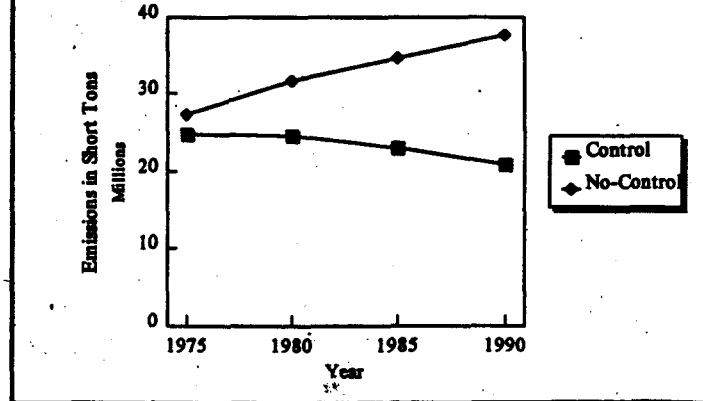


Figure 7. Control and No-control Scenario Total VOC Emission Estimates.



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 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 were 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.

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

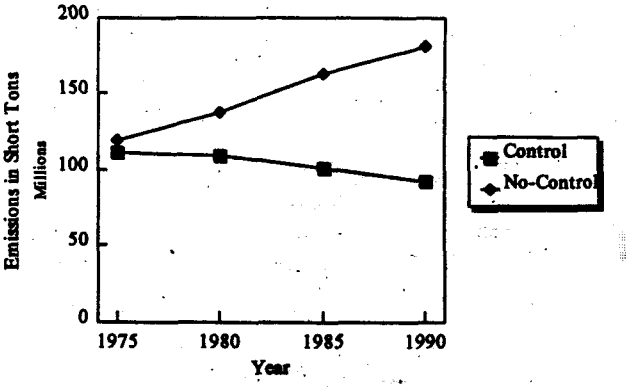


Figure 9. Control and No-control Scenario Total TSP Emission Estimates.

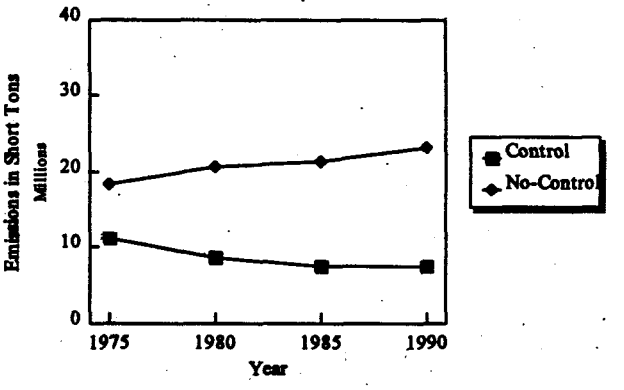
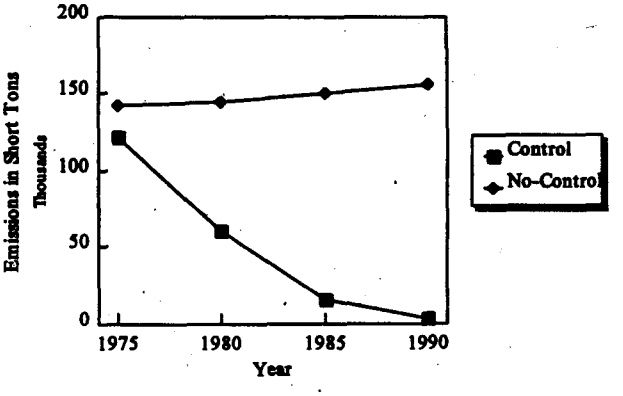


Figure 10. Control and No-control Scenario Total Pb Emission Estimates.



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 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.

Potential errors in the emissions modeling, and their significance relative to overall uncertainty in the monetary benefit estimate, are presented in Table 6.

Table 6. 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 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 fine particle fraction;[24] 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 adverse human health effect estimates require ozone concentration profiles based on a variety of short-term averaging periods.[25]

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 successfully employed for

[24] Particles with an aerometric diameter of less than or equal to 2.5 microns.

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

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.[26] 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[27] 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 7. 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 scenario profiles, pollutant-specific modeling strategies and assumptions, references to key supporting documents, and important caveats and uncertainties are also presented in Appendix C.

[26] 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.

[27] 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.

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[28] 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,[29] both lognormal and gamma distributional forms were tested against actual data to determine the form which provided the best fit to the historical data.[30] 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.

[28] 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.

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

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

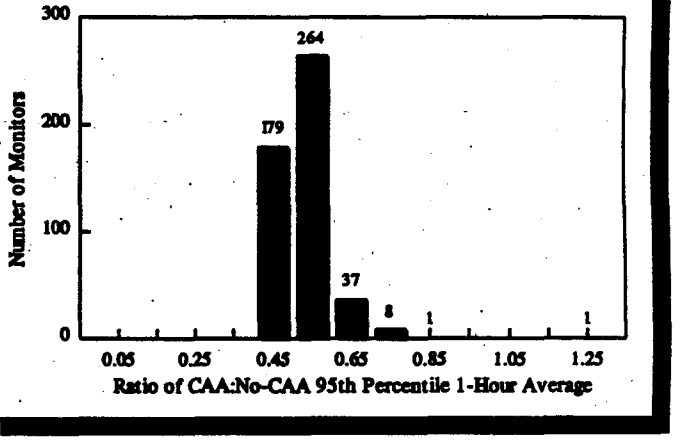
Sample Results

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),[31] 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.[32] 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.[33]

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 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 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

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

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

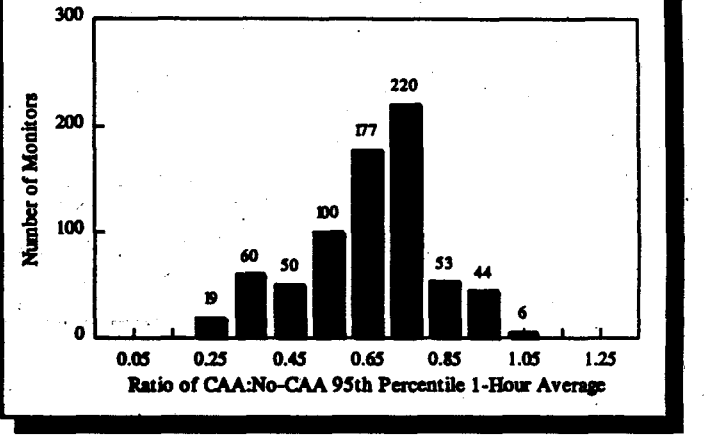
[33] Large percentage changes can result from even modest absolute changes when they occur in areas with good initial (e.g., control scenario) air quality. Considering percentage changes alone might create false impressions regarding absolute changes in air quality in some areas. For example, Appendix 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.

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 for 1990 Control to No-control Scenario 95th Percentile 1-Hour Average SO_2 Concentrations, by Monitor.



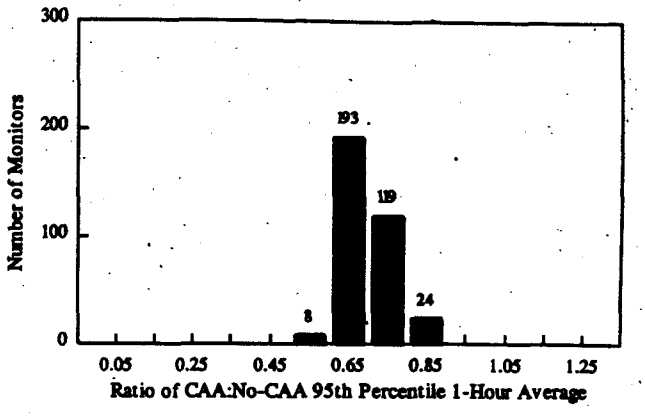
Figures 12 provides a histogram of the predicted control to no-control ratios for SO_2 which is similar to the one presented for CO. The results indicate that, on an overall basis, SO_2 concentrations were reduced by about one-third. The histogram also shows a much wider distribution of control to no-control 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. This source-specific variability was not observed for CO because controls were applied relatively uniformly on highway vehicles.

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.

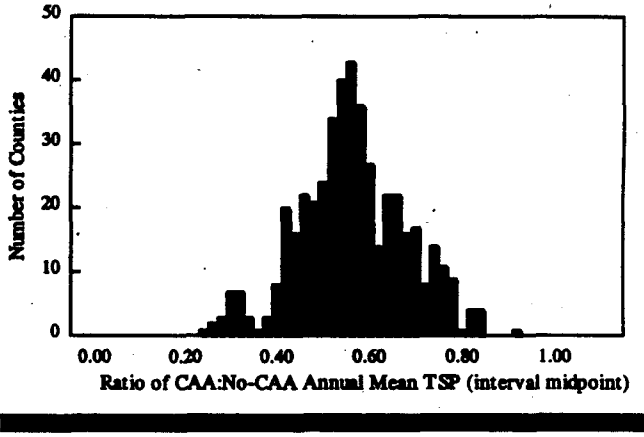
Figure 13. Frequency Distribution for 1990 Control to No-control Scenario 95th Percentile 1-Hour Average NO_2 Concentrations, by Monitor.



Particulate Matter

An indication of the difference in outcomes for TSP 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.[34] 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 1990 County-Level Annual Mean TSP CAA to No-CAA Ratios.



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 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

[34] 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).

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.[35]

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 are projected under the no-control scenario (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

Figure 15. Distribution of 1990 Control to No-control OZIPM4 Simulated Peak Ozone Ratios.

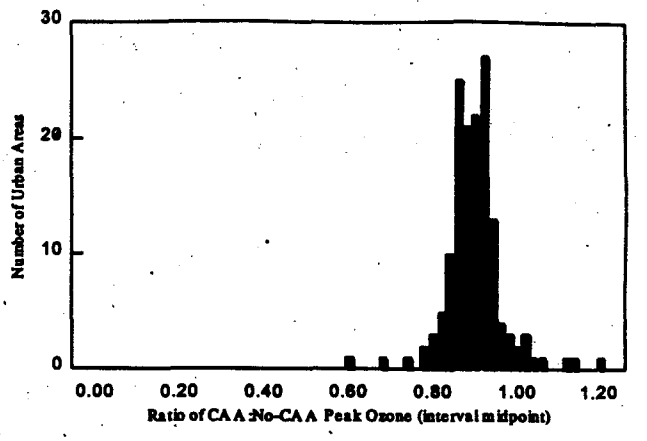


Figure 16. Distribution of 1990 Control to No-control RADM Simulated Ozone Ratios.

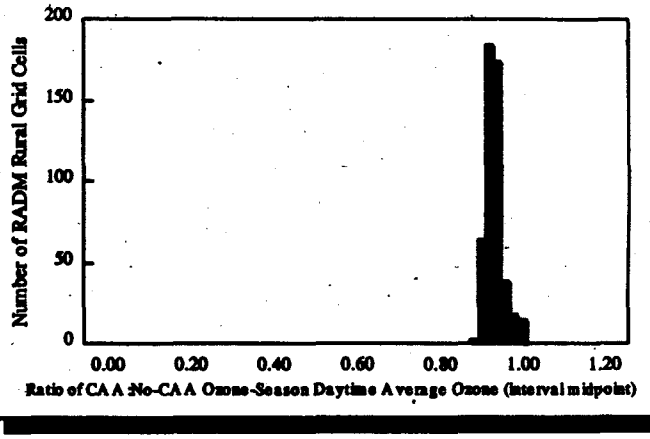
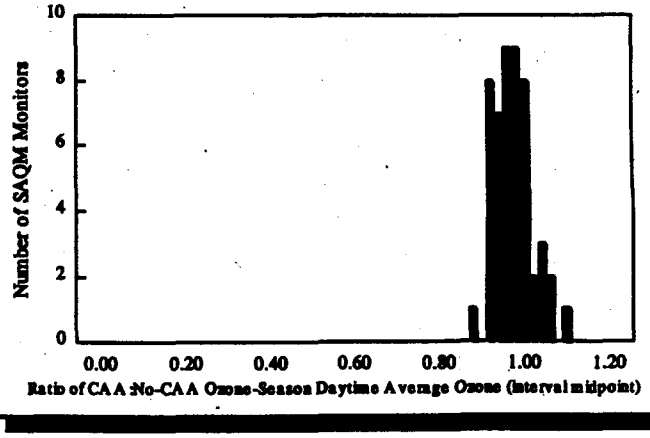


Figure 17. Distribution of 1990 Control to No-control SAQM Simulated Ozone Ratios.



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

precursor emissions in upwind areas and the relatively low deposition rates observed under the control scenario.[36]

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 ma urban corridors of the eastern seaboard.

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 20. This figure shows the increase in modeled annual average visibility degradation, in DeciView[37] 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 18. RADM-Predicted Percent Increase in Total Sulfur Deposition (Wet + Dry) Under the No-control Scenario.

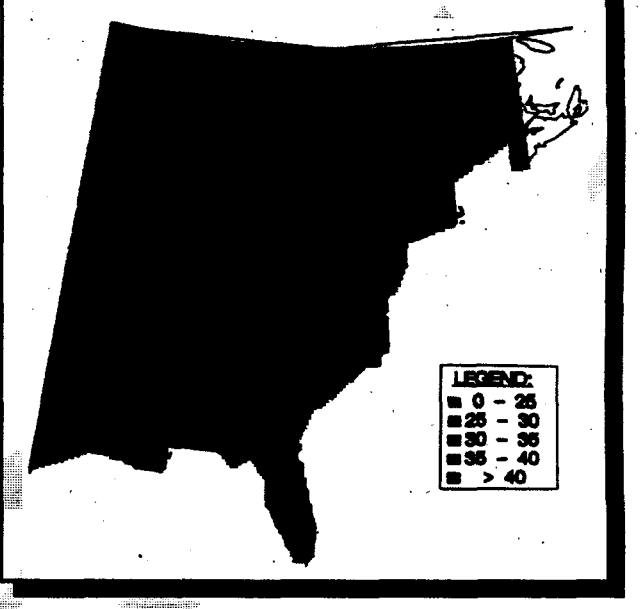
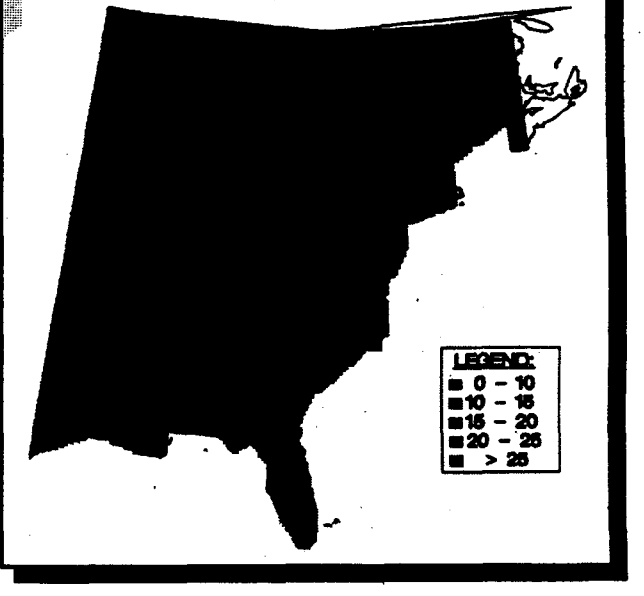


Figure 19. RADM-Predicted Percent Increase in Total Nitrogen Deposition (Wet + Dry) Under the No-control Scenario.



[36] 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.

[37] 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.

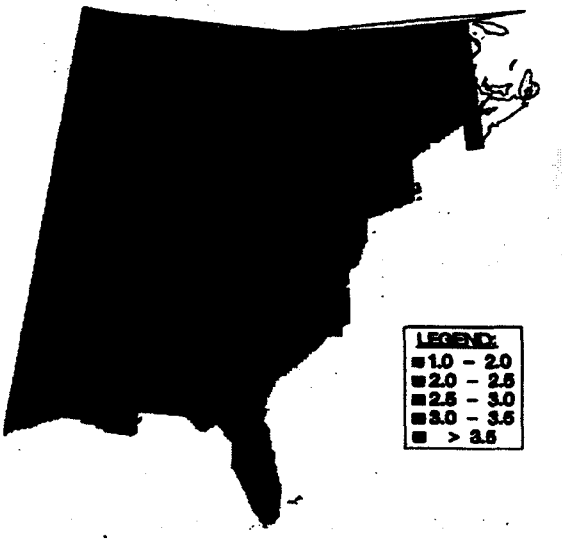


Table 7. Key Uncertainties Associated with Air Quality Modeling.

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 RDM, 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.
U[illegible] early biogenic emission estimates in RADM to e[illegible]ate 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. Control programs were likely 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.
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.
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 were 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 8 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 9 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.

Table 8. 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 Emergency room visits Asthma attacks Decreased income resulting from decreased exercise performance Mortality	Changes in pulmonary function 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 Lower respiratory illness Upper respiratory illness 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 bronchitis	Chronic respiratory diseases other than chronic bronchitis Inflammation in the lung
Carbon Monoxide	Decreased time to onset of angina	Behavioral effects	Other cardiovascular effects Developmental effects
Nitrous Oxides	Respiratory illness	Increased airway responsiveness	Decreased pulmonary function Inflammation in the lung Immunological changes
Sulfur Dioxide	In exercising asthmatics: Changes in pulmonary function Respiratory symptoms 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

Table 9. Selected Welfare Effects of Criteria Pollutants.

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

Welfare effects quantified using this general approach include household soiling damage by PM_{10} and agricultural benefits measured in terms of net economic surplus.

Another important welfare effect assessed herein is the benefits 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.[38] The models provided estimates of changes in light extinction, which were then translated mathematically into various specific measures of perceived visibility change.[39] These visibility change measures were then combined with population data to estimate the economic value of the visibility changes. 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 extensive discussion of many of the important ecological benefits which may have accrued due to historical implementation of the CAA.

[38] These models, and the specific visibility changes estimated by these models, are described in summary fashion in the previous chapter and are discussed in detail in Appendix C.

[39] These visibility measures are described in Appendix C.

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. To the maximum extent practical, 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 fuller discussion 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 13 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 initially assumed that all individuals within a particular air quality modeling grid cell were exposed to air quality changes estimated at the nearest monitor within 50 kilometers of the grid cell. Likewise, for PM_{10} (for which historical air quality data were available at the county level), only the population residing within monitored counties were initially assumed to be exposed to the air quality changes.

Unfortunately, by limiting the quantitative analysis to populations within 50 km of a monitor –or within a PM-monitored county– a significant portion of the U.S. population was initially left out of the analysis (see Table 10). For most pollutants in most years, less than three-quarters of the population lived within 50 km of a monitor (or within a PM-monitored county). Clearly, an analysis that excludes 25 percent of the population from the benefits calculations would drastically understate the benefits of

the CAA. However, a significant portion of the omitted population for a given year or pollutant may live so far from monitors for that pollutant that using those monitors to estimate their exposure may yield inaccurate results for those distant populations. Although the direction of any resulting bias cannot be ascertained definitively, it is likely that simple extrapolation of monitoring data to distant populations would tend to overestimate benefits. This is because monitors are typically located in more populated areas where emissions tend to be higher and air quality poorer.

Table 10. 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	78%	55%	53%	65%	67%
1980	80%	71%	59%	73%	68%
1985	76%	71%	61%	73%	68%
1990	68%	74%	61%	71%	70%

Despite these uncertainties, the Project Team endeavored to develop a more comprehensive benefits assessment by deriving an additional set of physical effects estimates for these previously omitted populations. This supplemental analysis was performed using a different set of assumptions to correlate air quality changes with population. For SO_2, O_3, NO_2, and CO, populations were assigned the air quality changes modeled for the nearest monitor, *without regard to the distance to that monitor*. This approach was tantamount to using the simple extrapolation approach alluded to above. For PM_{10}, however, a more sophisticated approach was used which should provide more reliable extrapolation results for this important indicator pollutant. Rather than extrapolating concentration measurements from the nearest monitored county, the grid cell-specific sulfate particle concentrations predicted by the RADM model were used to provide a scaled interpolation between monitored counties in the eastern 31 states.[40] For counties outside the RADM domain, an alternative method based on state-wide average concentrations was used. While less certain than the results obtained in the 31 eastern states, using state-wide averages provided better estimates than those obtained through simple extrapolation. With this supplemental analysis, estimates were developed of the health effects of the CAA on almost the entire continental U.S. population.[41] It should be noted, however, that 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.

There was a substantial difference between the results of the "50 km" and "all population" analyses. For NO_2, there was a 64 percent increase in the population modeled for 1990 (i.e., 61 percent of the population lives within 50 km of an NO_2 monitor while 100 percent of the continental U.S. population were captured using the extrapolation method). Incidence of increased respiratory illness among adults in 1990 caused by NO_2 exposure increased by 41 percent in the "all population" scenario relative to the "50 km" scenario. A 35 percent increase in 1990 population exposure for ozone (i.e., from 74 percent coverage under the 50 km limit to 100 percent using the extrapolation method) resulted in a predicted 27 percent increase in ozone-related asthma attacks. A 39 percent increase in 1990 population exposure for SO_2 (i.e., from 71 percent under the 50 km case to 98 percent for the all population case) resulted in a predicted 42 percent increase in SO_2-related respiratory symptoms among adults. Finally, the "all-

[40] The specific methodology is described in detail in Appendix C.

[41] While this alternative captures the vast majority of the U.S. population, it does not model exposure for everyone. To improve computational efficiency, those grid cells with populations less than 1,000 were not modeled; the analysis covered more than 95 percent of the population.

population" approach covers 45 percent more persons in 1990 than does the "monitored counties only" approach for PM, resulting in a roughly equivalent increase in PM-related health effects.

Clearly, the "50 km" approach severely understated aggregate health effects because it failed to cover 20 to 40 percent of the population. The "all-population" approach covers more of the population, but projections for persons living great distances from monitors were significantly more uncertain than were those for persons living close to monitors, and may misrepresent the effects on those persons. As noted previously, the results for PM effects in unmonitored eastern U.S. counties were more accurate than the simple extrapolation results used for other pollutants given the use of the regional model results to scale the interpolation to unmonitored counties. In addition, the extrapolated results for ozone should be more accurate than for the other non-PM pollutants because ozone is more of a regional pollutant. Finally, it is important to note that the "all-population" approach still fails to cover up to 5 percent of the population,[42] and the health benefits for that residual portion of the population were not reflected in the quantitative results of this study. Appendix D provides detailed results of both the "50 km" and "all-population" analyses.

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, 14 CR functions were available correlating excess premature mortality to 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, 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. For clarity, the results reported in this chapter and in Chapter 6 reflect only the

[42] Ibid, footnote 41.

"best estimates" of the underlying studies. In considering these "best estimate" results, however, it was important to bear in mind that there is a distribution of possible results around the "best estimates."

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, including results for all quantified health endpoints, are presented in Appendix D.

Table 11 provides a summary of a subset of the health effect modeling results. Results are expressed as cases reduced per year as a result of the CAA, and the pollutants associated with these health effect estimates are listed in the table.

Several studies were used to derive alternative estimates for some of the endpoints: the "high" estimate reflects the "best estimate" of the CR function with the greatest predicted effect, the "low" estimate reflects the "best estimate" of the CR function with the smallest predicted effect, and the "mid" value is the average of the "best estimates" of all the CR functions for mortality.[43] This table does not reflect the uncertainty surrounding the 'best estimates." These "mid" value results should therefore be interpreted as average values in a much wider distribution of possible results.

Table 11. Selected Health Benefits of the CAA, 1970-1990, for population in lower 48 States (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

[43] Results for lead (Pb) were handled differently. The Pb analysis was conducted using two baselines. The "high" and "low" values reported here reflect the results implied by the two baselines; the "mid" value is the average of the "high" and "low" values.

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, and nutrient cycling. 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.[44] 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.

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

[44] In fact, the effect on barley was several orders of magnitude less than 1 percent.

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 12 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).[45]

Table 12. 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 Building Deterioration	Loss of biological diversity

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

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

some pollutants are present in the atmosphere at reference levels that have caused adverse effects in animals.[46]

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.[47]

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.

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 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.

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

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

Table 13. 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 drastic 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)	None.	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 Section 812 analysis of health 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.[48] 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 values as well as a probability distribution of estimates. This permits an evaluation of some aspects of the uncertainty associated with the point estimates. 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, valuation estimates are reported in dollars per unit of avoided damage. In many cases, the avoided losses were measured directly in monetary terms. For agricultural benefits, however, estimated changes in crop yields were evaluated with an agricultural sector model. 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, 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.

Unit Valuation

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

[48] The literature reviews and valuation estimate development process is described in detail in Appendix I and in the referenced supporting reports and memoranda by Industrial Economics, Incorporated (IEc).

an environmental effect. For example, the value of an avoided respiratory symptom would be a person's WtP to avoid that symptom.[49]

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. However, value may be inferred by 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 used estimates of the WtP among the U.S. population to avoid an environmental effect as the value of avoiding that effect. In some cases, such estimates were not available, and the cost of mitigating or avoiding the effect was used as a rough estimate of the value of avoiding the effect. For example, if an effect results in hospitalization, the avoided hospitalization costs were considered as a possible estimate of the value of avoiding the effect. Finally, where even the "avoided cost" estimate was not available, the analysis relied on other available methods to provide a rough approximation of WtP. As noted above, this analysis used a range of values for most environmental effects, or endpoints. Table 14 provides a summary of the mid-range values used for per unit valuations. The full range of values can be found in Appendix I.

Table 14. Monetized Health and Welfare Effects (1990 dollars).

Endpoint	Pollutant	Valuation (mid-estimate)
Mortality	PM_{10}, O_3, SO_2, & Pb	$4,800,000 per case
Heart Attacks	Pb	$587,000 per case
Strokes	Pb	$587,000 per case
Hospital Admissions		
Respiratory	PM_{10} & O_3	$7,500 per case
Ischemic Heart Disease	PM_{10}	$10,000 per case
Congestive Heart Failure	PM_{10}	$8,000 per case
Respiratory Illness and Symptoms		
Upper Respiratory Illness	PM_{10}	$18 per case
Lower Respiratory Illness	PM_{10}	$10 per case
Acute Bronchitis	PM_{10}	$45 per case
Acute Respiratory Symptoms	PM_{10}, O_3, NO_2, SO_2	$17 per case
Work Loss Days	PM_{10}	$83 per day
Restricted Activity Days	PM_{10} & O_3	$38 per day
Asthma Attacks	PM_{10} & O_3	$32 per case
IQ Changes		
Lost IQ Points	Pb	$5,550 per IQ pt.
Incidence of IQ < 70	Pb	$52,700 per case
Hypertension	Pb	$680 per year per case
Decreased Worker Productivity	O_3	Direct Economic Valuation
Visibility	Extinction	Direct Economic Valuation
Household Soiling	PM_{10}	Direct Economic Valuation
Agriculture (Net Surplus)	O_3	Est. Change in Econ. Surplus

[49] 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.

Willingness-to-Pay Estimates

Mortality

Some forms of air pollution increase the probability that individuals will die prematurely. The concentration-response functions for mortality used in this analysis report 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's 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.[50] For this analysis, the Project Team surveyed the economics literature to determine an appropriate valuation for avoided mortality, and used a range of values with an arithmetic average of $4.8 million per mortality avoided.

The concentration-response functions used to estimate mortality effects of air pollution examined the effects of high-pollution incidents on mortality rates during and immediately following the incidents. This approach is not ideal for the purpose of benefit-cost analysis:

> It may overstate the mortality effect of air pollution insofar as it includes as "excess premature mortalities" some individuals who were very sick and would have died within a few days or weeks even without the air pollution event. The valuation method used in this analysis was based on WtP for reduced mortality risk of individuals who are, generally, healthy and with an expectation of many years of life ahead of them. Applying that valuation to extremely ill individuals may overstate benefits.
>
> -or-
>
> It may understate the mortality effect of air pollution insofar as it fails to address years of life lost due to long-term exposure to air pollution. Air pollution can have both acute and chronic effects. The CR functions used address only the "acute," or short term, effects of air pollution on mortality rates. It is possible that longer term effects exist, causing "premature mortality" that does not necessarily occur during or immediately following a high-pollution event.

Ideally, instead of valuing changes in short-term mortality risk, one would value changes in the risk of incurring a health effect which imposes a specific reduction in life expectancy. Unfortunately, the state of health science is not advanced enough to provide reliable "life-shortening" estimates, so the Project Team estimated and valued the expected number of "excess premature mortalities" caused by air pollution.

[50] For example, if the WtP for a one percent reduction in mortality risk is $20,000, then the "benefit per life saved" would be [$20,000 x 100 =] $2 million.

Survey-Based Values

The use of survey methods (such as the "contingent valuation" method) to determine willingness-to-pay for environmental improvement 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.

For this analysis, unit valuations based on the contingent valuation method have been used for respiratory illnesses and symptoms, restricted activity days, and asthma attacks. The Project Team reviewed the relevant economic literature to define a range of values for those health endpoints, and calculated "mid-range" estimates. Those mid-range estimates are summarized in Table 14, above. Although a large fraction of the monetized health endpoints rely on survey-based valuations, the aggregate benefit calculations are insensitive to these values.[51]

Avoided Cost Estimates

For some health effects, WtP estimates were not available, and the Project Team instead used "costs avoided" as a best estimate of WtP. Presumably, willingness-to-pay to avoid a health effect that would require hospitalization would be equal to the cost of hospitalization plus the value of avoided pain and suffering, lost leisure time, etc. Consequently, the cost of hospitalization would be a lower-bound estimate of one's WtP to avoid a health effect. In a society where the cost of hospitalization is not paid directly by the patient, however, one's WtP to avoid a health effect may be completely independent of the hospitalization cost avoided; thus, it is unclear whether "cost avoided" understates or overstates WtP to avoid the health effect.[52]

This analysis used "avoided costs" to monetize benefits for hospital admissions (the analysis uses three categories of hospital admissions) and "Changes in the Incidence of IQ<70" as a result of diminished cognitive development in young children due to Pb exposure (unit values are summarized in Table 14). In both cases, "avoided costs" must be recognized as a very rough approximation of WtP to avoid those health effects. In the IQ<70 case, the "avoided cost" is the estimated incremental cost of part-time special education for children in regular classrooms as opposed to in special education programs for twelve years (i.e., grades one through twelve).

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[51] That is, changing the unit values derived through the contingent valuation method has little effect on the overall level of benefits found in this analysis. The health endpoints that are monetized through the use of CVM studies account for approximately 1 percent of all monetized benefits in this cost-benefit analysis.

[52] If it could be calculated, one would ideally count as a benefit of environmental improvement WtP to avoid pain and suffering, lost leisure time, etc. Avoided hospitalization costs, whether paid directly by the patient or communally (e.g., through an insurance pool), would be counted as a negative cost (i.e., a reduction in the cost of environmental controls, just as reduced automobile maintenance costs are counted as a "negative cost" of the CAA).

Other Valuation Estimates

Heart Attacks and Strokes

In its review of the economics literature, the Project Team found no studies that estimated WtP to avoid the risk of non-fatal heart attacks and strokes. Although it would be possible to estimate the avoided hospitalization costs due to decreased incidence of heart attack and stroke, such an approach would almost certainly provide a gross underestimate of the WtP to avoid those conditions. Instead, the Project Team used a valuation derived for a different medical condition (chronic bronchitis), assuming that it would provide a rough approximation for WtP to avoid heart attacks and strokes. Based on a review of the literature, the Project Team believes that a reasonable estimate for WtP to avoid the risk of chronic bronchitis is 587,500 dollars. The Project Team has therefore assumed that the WtP to avoid the risk of a case of heart attack or stroke is also 587,500 dollars.

Work Loss Days

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.

Hypertension

As with heart attacks and strokes, the Project Team found no studies which estimate WtP to avoid hypertension (high blood pressure). Studies do exist, however, which estimate the medical costs and work loss days associated with hypertension. Although there are further effects that would affect an individual's WtP to avoid hypertension (e.g., the value of pain and suffering, the value placed on dietary changes), this analysis assumes that the sum of medical costs (per year) and work loss costs per year (valued as discussed above) is a reasonable approximation of the WtP per year to avoid a case of hypertension.

Lost IQ Points

One of the major effects of Pb exposure is permanently impaired cognitive development in children. No ready estimates of society's WtP for improved cognitive ability are currently available. However, estimates of the relationship between IQ and lifetime earnings as well as estimates of the IQ effects of Pb exposure were available. Although this is a very crude estimate (and likely underestimate) of WtP for improved cognitive development in children, this study assumed that changes in lifetime earnings equal to $5,551 per IQ point change were a reasonable approximation of WtP.

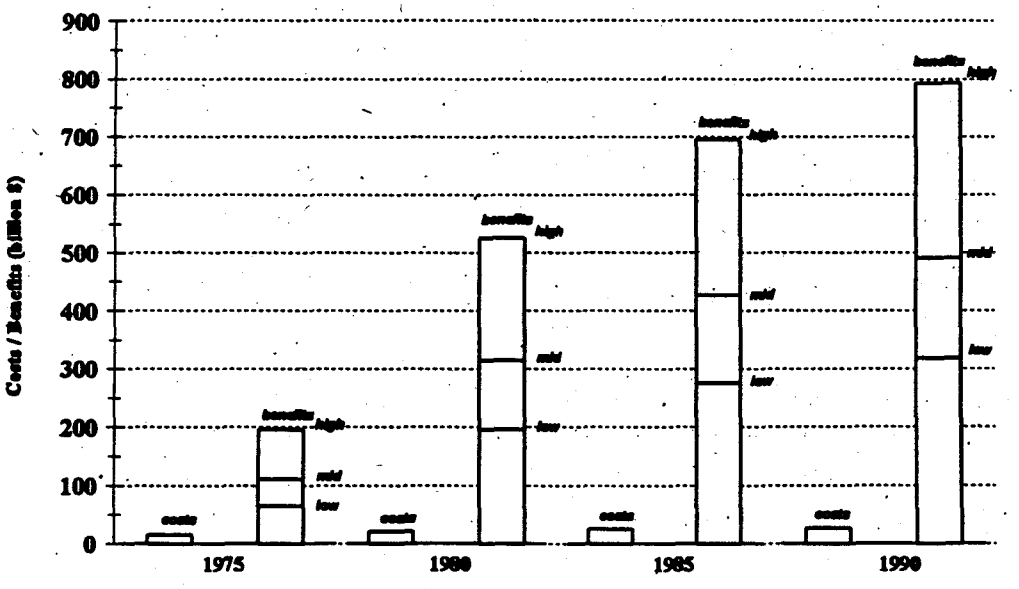
Aggregation of Benefits

The total monetized economic benefit attributable to the CAA was derived by applying the unit valuations discussed above to the stream of physical effects calculated for the 1970 to 1990 period. Some aggregation difficulties arose, however, which required modification to the physical effects and/or

valuation data to avoid double-counting of benefits. For example, some studies that were used as sources for physical effects estimates provided cases of hospital admissions for pneumonia, while other studies provided cases of hospital admissions for all respiratory problems. In this analysis, double-counting of effects was avoided by assuming that the two sets of studies provided alternative estimates of the same health impact (i.e., they were not treated as additive). Similarly, the Project Team had access to several studies estimating the effect of air pollution on "restricted activity days" of various degrees of severity as well as studies that estimated lost work days. To avoid double-counting, the various "restricted activity" health effects were combined into a single benefit category.

Figure 21 summarizes the results of the aggregation of monetized benefits for the entire population of the lower 48 States in each of the target years of this analysis and compares these results with the total direct compliance costs incurred in those years. Each of the bars representing total monetized benefits in the figure include a high, mid, and low estimate. These alternative estimates reflect the fact that, for some health endpoints, several studies were available for use in estimating physical effects. In addition, for some other endpoints more than one method of calculation was used. For those endpoints where more than one estimate was available, the lowest estimate, highest estimate, and average of all available estimates were considered. The "low" lines in Figure 21 reflect the sum of the benefits estimated by using the "low estimate" physical

Figure 21. Aggregate Direct Costs and Monetized Benefits of the Clean Air Act (in billions of 1990 dollars).



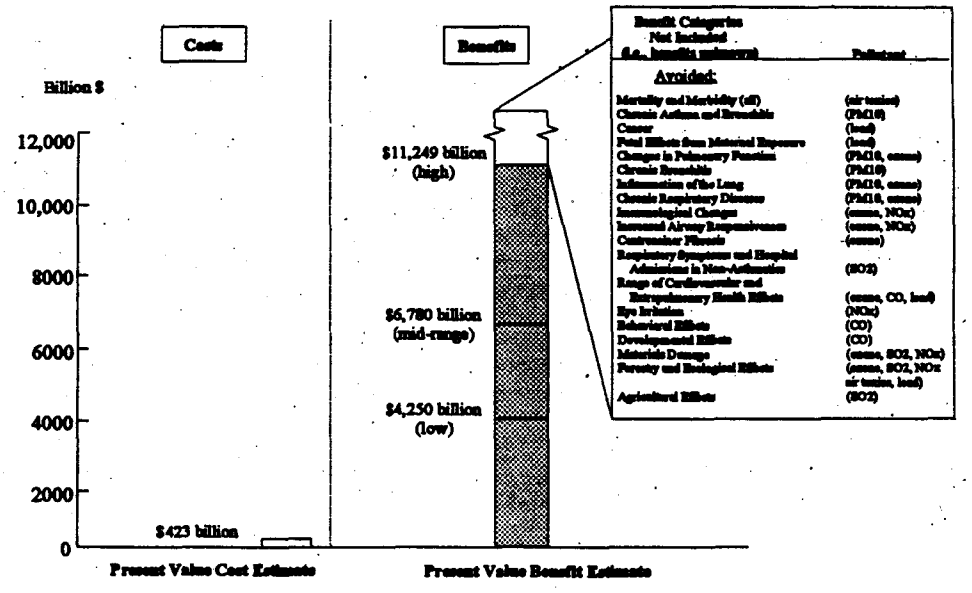
NOTE: Aggregated benefit estimates do not include benefits arising from decreased exposure to pollutants other than the six criteria pollutants. Also omitted are non-quantifiable or non-monetizable benefits arising from control of criteria pollutants (e.g. some avoided adverse ecological effects).

effect studies; the "mid" lines reflect the numerical average of results based on all available effect studies, and the "high" lines reflect the sum of benefits using the "high estimate" studies.

It is important to note that these estimates do not reflect the combined uncertainties in physical effects and economic valuation of those effects. Rather, the results presented in this chapter use mid-range estimates for economic valuation of effects. Therefore the ranges presented reflect only the uncertainties associated with the selection of physical effect studies. Broader ranges of total monetized benefit estimates which also reflect uncertainties in economic valuation are presented in Chapter 7.

The total costs and benefits accumulated over the entire 20 year period of this study are presented in Figure 22. The summary estimates for both costs and benefits are derived by summing the yearly cost and benefit estimates adjusting those yearly values to reflect their equivalent value in 1990.[1] This summary figure also clarifies the substantial number of benefits which could not be included in the total monetized benefit estimate. Including these and other nonmonetized benefits would increase the high, mid-range, and low benefit estimates by unknown degrees. In addition, as noted in the preceding paragraph, these total benefit estimates do not reflect uncertainties in economic valuation of effects. More detailed results, including those which indicate the relative contribution of individual endpoints to the total monetized benefit estimate, are presented in Appendix I.

Figure 22. Comparison of Total 1970 to 1990 Clean Air Act Monetized Benefits and Direct Costs (in billions of 1990 dollars and in 1990 Present Value terms).



[1] Estimated benefits for years between the target years 1975, 1980, 1985, and 1990 were derived by linear interpolation between target years. Consistent with the use of 1970 as the starting point of the analysis, a value of zero benefit for 1970 was used to calculate interpolated results for 1971 to 1974.

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7

Uncertainty in the Benefits Analysis

Some degree of uncertainty was associated with every step of this analysis, from cost estimation through economic valuation of benefits. The results were most sensitive, however, to uncertainties in the physical effects estimation and economic valuation steps. The Project Team evaluated these two key areas of uncertainty through a quantitative analysis of the distribution of likely results, and a discussion of some of the health effects excluded from the analysis.

Modeling Uncertainty of Health Effects and Valuation

Different published results reported in the scientific literature on the adverse health effects associated with elevated levels of air pollution typically do not report identical findings. In some instances the differences are substantial, and considerable controversy exists about the magnitude of virtually all of the suspected effects. Given the level of uncertainty associated with the estimates of any of the concentration-response relationships, it was difficult to identify a single relationship that best represents the nationwide impact of pollution on human health. Rather than try to identify a single "best estimate," two different approaches were used in this assessment to estimate the magnitude of the effects, and to fairly represent the uncertainty in current scientific opinion.

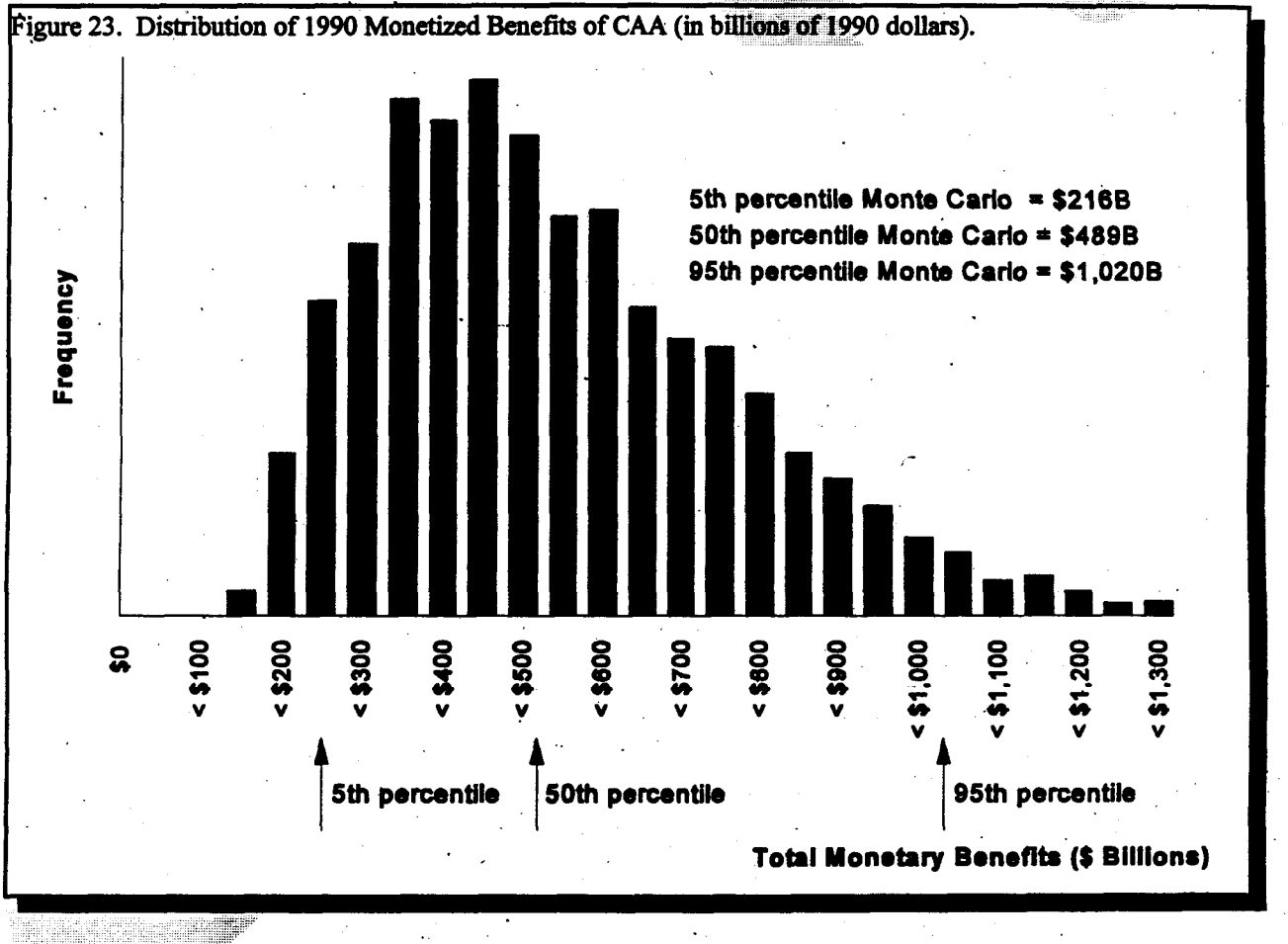
The first approach identified a set of concentration-response relationships drawn from recent scientific literature which included the range of estimated effect magnitude for each known or suspected health endpoint. This approach produced multiple estimates of the avoided incidence of health effects. The set of point estimates which make up this range of estimates, both in terms of the incidence of the health effect and in monetary terms, indicate not only the range of credible estimates, but the extent to which there is a clustering of estimates in some portion of the range. This analytical approach was used to derive the ranges of results presented in Chapters 5 and 6.

The second approach used in this assessment relied on a Monte Carlo method to evaluate the effect of combining the uncertainties in the concentration-response function and the economic value for each endpoint. The first step in the Monte Carlo technique used here was to randomly select a study from the set of available studies for a particular health effect. Each study provided, when selected by the Monte Carlo model, an estimate of the concentration-response function and its standard deviation. In the second step, a normal statistical distribution was derived based on the mean and variance 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. 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 under the control and no-control scenarios.

Concurrently, this two-stage Monte Carlo technique was used to estimate the range of monetary benefits associated with the avoided health effects. The monetary values assigned to each health or welfare endpoint were also measured with considerable uncertainty, and a distribution was used to reflect the uncertainty of monetary values. Each iteration of the Monte Carlo assessment picked a specific level for the monetary value of the endpoint from the assigned distribution of values. The estimate of the monetary benefits for changes in the health effect incidence for that Monte Carlo iteration was thus composed of a random draw of the estimated change in incidence multiplied times an independent random draw of the monetary value of the health effect. Figure 23 illustrates the results of the Monte Carlo analysis of benefits for the year 1990.

Figure 23. Distribution of 1990 Monetized Benefits of CAA (in billions of 1990 dollars).



The vertical bars in Figure 23 represent the relative frequency of a given result in the Monte Carlo analysis. The largest bar, located above the "<$400", indicates that more Monte Carlo iterations generated monetized benefits of $350 billion to $400 billion than in any other $50 billion range bin. This tallest bar, however, does not represent the true central, or expected, value of the estimate for total monetized benefit; rather the 50th percentile estimate provides this value. The 50th percentile estimate was derived by determining the median outcome of all of the Monte Carlo iterations. In other words, half of the iteration results fell above and half fell below this value. In this analysis, the 50th percentile result for the year 1990 was 489 billion dollars. A 90 percent credible interval was also derived from the Monte Carlo analysis. This credible interval was defined by the 5th and 95th percentile outcomes. Only 10 percent of the iteration results fell outside of this range, implying a probability of 90 percent that this

range captures the true value for total 1990 monetized benefits. The 90 percent credible range for total 1990 monetized benefits estimated in this analysis was from 216 billion to 1.02 trillion dollars.

Similar 50th percentile and 90 percent credible interval estimates were also derived for the other target years of the study: 1975, 1980, and 1985. To estimate the 50th percentile and 90 percent credible interval for total monetized benefits accruing over the entire 1970 to 1990 period, the Project Team developed estimates for intervening years through linear interpolation between target years. For example, results for 1971 through 1974 were derived by linearly interpolating between the 1975 results and the 1970 result (which equaled zero in this case since the control and no-control scenarios were assumed to be equal at the start of the study period). Finally, the results for each year were adjusted to their equivalent value in 1990 and summed. The 50th percentile estimate for total 1970 to 1990 monetized benefit of the Clean Air Act derived by this method was 6.8 trillion 1990-value dollars. The 90 percent credible interval was 2.7 to 14.6 trillion 1990-value dollars.

Studies and Health Effects Excluded from the Analysis

This assessment was designed to be as comprehensive as possible and to include "best estimates" of air pollution effects even when there were significant uncertainties surrounding those estimates. Generally, the assessment quantified all physical effects where concentration-response relationships were found in the published literature. Two categories of effects for which concentration-response relationships were found, however, were excluded because of inconsistencies in the characteristics of the effects studied in the literature and the effects estimated in the context of the present analysis. These two effects were (1) premature mortality predicted through "cohort" studies and (2) chronic bronchitis.

Avoided Premature Mortality -- Cohort Studies

As noted in Chapter 6, the preferred basis for valuing avoided premature mortality in the context of cost-benefit analysis would be the willingness-to-pay for some probability of an extension of life years, which would then be applied to a measure of "life-years lost" due to long-term, rather than episodic, exposure to air pollution. Some research has been published which addresses the mortality impacts of long-term exposure to air pollution, in particular due to exposure to ambient particulate matter. Use of the concentration-response relationships found in these studies (referred to herein as the "cohort studies") would allow estimation of average life-years lost across the population as a whole.

The Project Team, however, was not satisfied by any of the methods proposed for converting the concentration-response relationships defined by the cohort studies to forms which could be used for the present study. To evaluate the sensitivity of the estimates of total monetized benefit to this exclusion, an attempt was made to derive some estimate of the value of estimated reductions in mortality consistent with the inferences of the cohort studies. Specifically, concentration response relationships were inferred from the cohort studies and the "life years lost" results were converted into "premature mortalities avoided." The resulting estimate of premature mortalities avoided were then valued using the same valuation methods used for the primary results presented herein.

The sensitivity analysis was conducted by, first, estimating total 1970 to 1990 monetized benefits in 1990-value dollars absent all particulate matter-related mortality and reflecting numerical averages of alternative outcomes based on alternative studies of a particular effect. (Note that this estimate differs from the 50th percentile Monte Carlo result presented as the expected benefit estimated by this study, although they are very close.) The second estimate was identical except that particulate matter mortality was added by applying equal weights to the alternative mortality outcomes predicted by the fourteen "episodic" particulate matter mortality studies. The third estimate was identical to the second estimate

except that the two cohort mortality studies for particulate matter were included with the episodic studies. The two cohort studies were given weights equal to the fourteen episodic studies in this third estimate, which significantly dampened the effect of including the cohort studies. The fourth estimate derived was identical to the second estimate except that the episodic studies were dropped and only the two cohort studies were used to estimate particulate matter mortality. These four estimates are presented in Table 15.

Table 15. Sensitivity of 1970 to 1990 All 48 State Population Monetized Benefits Mid-Point Estimate to Inclusion of Cohort Mortality Studies (in billions of 1990-value dollars).

Estimate 1: Without PM Mortality	2,245
Estimate 2: With PM Mortality Based on Episodic Studies Only	6,780
Estimate 3: With PM Mortality Based on Weighted Average of 14 Episodic Studies and 2 Cohort Studies	10,518
Estimate 4: With PM Mortality Based on Cohort Studies Only	35,680

The clear implication of the sensitivity analysis was that the inclusion of the cohort studies would significantly increase the estimated total monetized benefits of the Clean Air Act. However, further research is needed to improve the potential interpretation and incorporation of the results of cohort studies for benefit-cost analyses. Based on currently available information, the Project Team was not confident enough in its interpretation method to include the results of the cohort studies in the primary results of this assessment.

Chronic Bronchitis

Evidence indicates there may be a relationship between exposure to PM_{10} and development of chronic bronchitis. As with other health endpoints studied for this study, the Project Team identified the published scientific literature addressing PM_{10} and chronic bronchitis, and inferred concentration-response relationships from those studies. As with the other health endpoints studied, a range of estimated incidence outcomes associated with the control and no-control scenarios were developed. The differences between these scenarios in the estimated outcomes for chronic bronchitis, reflecting low, high, and mid estimates similar to those derived for other endpoints in Chapter 5, are summarized in Table 16.

Table 16. Estimated Reductions in Incidence of Chronic Bronchitis under the Control Scenario (in thousands of new cases per year).

	1975	1980	1985	1990
Low estimate	173	454	565	603
Mid estimate	185	493	620	656
High estimate	196	532	675	709

The Project Team also developed a unit value for avoidance of a case of chronic bronchitis. Based on a review of the economics literature and recommendations from the Science Advisory Board Advisory Council on Clean Air Compliance Analysis, the Project Team used a value of $587,500 per case avoided to translate avoided incidences into estimated monetized benefit. Using these values, a sensitivity analysis was performed to evaluate the potential significance of including chronic bronchitis reduction benefits in the total 1970 to 1990 monetized benefit estimate. The first estimate is identical to Estimate 2 presented in Table 15. The second estimate is identical to this estimate with the total 1970 to 1990 estimated monetized value of chronic bronchitis reductions added. The results of this sensitivity analysis are summarized in Table 17.

Upon further review, it became apparent that the "chronic bronchitis" valued in the economics literature is far more severe than the "chronic bronchitis" predicted in the health effects literature used by in the present study.[54] There is almost certainly a chronic bronchitis health effect which could, theoretically, be valued. At this time, however, the Project Team considers the valuation procedure so uncertain that it borders on speculative. Consequently, the Project Team did not include benefits from avoided cases of chronic bronchitis in the primary estimate of benefits presented in this study. As for the particulate matter-related cohort mortality studies, additional research is needed to improve monetary characterizations of these benefits and facilitate their incorporation in benefit-cost studies.

Table 17. Sensitivity of 1970 to 1990 All 48 State Population Monetized Benefits Mid-Point Estimate to Inclusion of Chronic Bronchitis (in billions of 1990-value dollars).

Estimate 1: Without Chronic Bronchitis	6,780
Estimate 2: With Chronic Bronchitis	12,735

draft

[54] With an average of one-half million new cases of chronic bronchitis each year for over a decade, the total number of persons with chronic bronchitis would approach two to three percent of the U.S. population by 1990. It was considered unlikely that two percent of the population could develop a health condition of the severity of the chronic bronchitis valued in the economics literature.

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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"[55] (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 estimated by the macroeconomic model. The effects reported by the model are indirect costs and sectoral impacts; the model has not estimated the indirect benefits attributable to improvements in environmental quality, or avoidance of environmental or natural resource degradation. That is, the benefits that result from a cleaner environment are not captured and allowed to propagate through the economy as are compliance costs. For example, air quality improvements leading to improvements in human health translate into reduced medical expenditures. These savings in medical expenditure would then flow into alternative economic activities. A fully integrated model would contain environmental-economic relationships such as this so there would be a simultaneous accounting for changes in costs and benefits. However, no existing macroeconomic model integrates all market and non-market costs and benefits of environmental programs.

[55] 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.

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 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.[56] 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.

One significant limitation of current macroeconomic modeling tools is that they only capture those effects that manifest themselves in the market. Furthermore, they may not even capture all potentially significant market effects.[57] For example, although the prices and quantities of residential housing stocks are included in the J/W model, changes in prices caused by changes in air pollution-related visibility in residential areas are not captured. Also, some of the economic *benefits* of air pollution control are not fully characterized by the J/W model. For example, since there is no distinct sector representing the pollution control industry, increases in production, export sales, and employment in this particular

[56] 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.

[57] See March 24, 1993 letter from the SAB CAACAC review panel to the EPA Administrator, pp 2-3, which states: *"In principle, effects such as decreases in worker productivity and increases in medical care expenditure that would occur in the no control scenario should affect Gross Domestic Product (GDP). We do not believe, however, that it is likely to be productive to attempt formally to 'close the loop' by incorporating such effects in the general equilibrium modeling. As we understand the Jorgenson/Wilcoxen (J/W) model, it may be also impossible for it to show impacts on the pollution control industry, narrowly defined, since such impacts importantly reflect changes in the detailed composition of investment goods produced."*

industry created by air pollution control programs are not reported by the J/W model.[58] The limited capabilities of macroeconomic models are even more acute for non-market effects, some of which may have highly significant consequences for social and economic welfare. For example, the J/W model does not capture pain and suffering caused by air pollution-related respiratory disease, the effects of increased deposition in lakes of airborne heavy metals and subsequent uptake by fish caught by individuals for personal consumption, adverse effects of air pollution on wildlife and sensitive ecosystems, and many other health and environmental effects.

In general, the value of the investment in using a general equilibrium model in any given analysis must be weighed against alternative investments which could reduce other sources of uncertainty and bias in the overall analysis. Although the project team chose to perform macroeconomic modeling for the present study, significant questions regarding the benefits of investing scarce project time and resources in macroeconomic modeling have led to the conclusion that a sector-level approach will be more appropriate for the first §812 Prospective Study of the costs and benefits of the CAA. This conclusion is reinforced by the recommendation of the CAACAC review panel.[59]

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..."[60]

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.

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[58] *Ibid*, footnote 57.

[59] See May 14, 1993 letter from SAB CAACAC to the EPA Administrator, p. 6, which states: "... *the Agency should learn from the retrospective study that as a rule, resources are better spent on developing sound, comprehensive data relevant to key issues than on elaborate modeling or literature review efforts.*" (authors' emphasis)

[60] 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.

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 18). 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 18. 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 19. Definitions of Industries Within the J/W Model.

Industry Number	Description
1	Agriculture, forestry, and
2	fisheries
3	Metal mining
4	Coal mining
5	Crude petroleum and natural gas
6	Nonmetallic mineral mining
7	Construction
8	Food and kindred products
9	Tobacco manufacturers
10	Textile mill products
11	Apparel and other textile
12	products
13	Lumber and wood products
14	Furniture and fixtures
15	Paper and allied products
16	Printing and publishing
17	Chemicals and allied products
18	Petroleum refining
19	Rubber and plastic products
20	Leather and leather products
21	Stone, clay, and glass products
22	Primary metals
23	Fabricated metal products
24	Machinery, except electrical
25	Electrical machinery
26	Motor vehicles
27	Other transportation equipment
28	Instruments
29	Miscellaneous manufacturing
30	Transportation and warehousing
31	Communication
32	Electric utilities
33	Gas utilities
34	Trade
35	Finance, insurance, and real estate
	Other services
	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 19).[61] 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.

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,

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

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.[62]

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

[62] 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.

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.[63]

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.

[63] 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).[64] Clean air regulations have typically led 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

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

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 motors 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.[65] Estimates of capital and operating costs resulting from new and forthcoming regulations were derived from EPA's Regulatory 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 1988.[66]

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.

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

[66] The most recent BEA article used in the Cost of Clean was "Pollution Abatement and Control Expenditures, 1985-88" in *Survey of Current Business*, November 1990.

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.

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.[67] Table 20 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.[68]

Table 20. Estimated Capital and O&M Expenditures for Stationary Source Air Pollution Control (millions of current dollars).

	Nonfarm Business[1]		Government Enterprise	
Year	Cap.	O&M	Cap.	O&M
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				

Sources: "Pollution Abatement and Control Expenditures" published annually in the *Survey of Current Business* by BEA

1. Consists of new plant and equipment expenditures for pollution abatement according to results from the Pollution Abatement and Costs Expenditures and the Structures and Equipment Survey-Supplement for Pollution Abatement by the Bureau of Census, as well as indirect estimation by BEA for selected nonmanufacturing industries.

Operation and Maintenance Expenditures Data

Stationary source O&M expenditures are made by manufacturing establishments, private and public electric utilities, and other nonmanufacturing

[67] 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.

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

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 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.[69] O&M expenditures are net of depreciation and payments to governmental units, and are summarized in Table 20. O&M data were disaggregated to the two digit SIC level for use in the macroeconomic model.

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 21. 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 20) less recovered costs. Recovered cost data were disaggregated to the two digit SIC level for use in the macroeconomic model.

Table 21. 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

* Air cost recovered as reported in PACE

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

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.[70]

[69] 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.

[70] 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.

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 22 summarizes mobile source capital costs.

Table 22. Estimated Capital and Operation and Maintenance Expenditures for Mobile Source Air Pollution Control (millions of current dollars).

Year	Capital	O&M
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: *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).

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 23 summarizes mobile source O&M expenditures and cost savings by categories, with net O&M costs summarized above in Table 22. 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 annually 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.[71] 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.

Table 23. 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	-3839	-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.

Source: *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).

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.[72] 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.[73] Under this assumption, the decrease in total fuel cost to consumers is considered a benefit of the program.

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

[71] 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.

[72] 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.

[73] 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.

for 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.[74] 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,[75] 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 24.

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

[75] 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 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 24. Other Air Pollution Control Expenditures.

Year	Abatement		Regulations and Monitoring		Research and Development			Total
	Federal	State	Federal	State & Local	Private	Federal	State & Local	
1973	47	0	50	115	451	126	6	795
1974	56	0	52	131	492	100	7	838
1975	88	1	66	139	466	108	8	876
1976	105	1	69	135	543	131	6	990
1977	106	1	80	161	654	144	7	1,153
1978	90	0	93	183	789	146	8	1,309
1979	103	0	100	200	924	105	7	1,439
1980	95	0	122	207	869	130	5	1,428
1981	85	0	108	226	852	131	0	1,402
1982	87	0	93	230	912	126	2	1,450
1983	136	4	88	239	1,315	133	6	1,921
1984	115	14	101	250	1,359	165	4	2,008
1985	98	12	103	250	1,427	247	3	2,140
1986	67	14	106	307	1,499	217	4	2,214
1987	80	15	110	300	1,574	200	2	2,281
1988	65	10	120	320	1,652	220	1	2,388
1989	70	12	130	360	1,718	230	2	2,522
1990	71	13	133	343	1,820	231	2	2,613

Source: *Cost of Clean* (1990)

Uncertainties in the Direct Cost Data

Potential Sources of Error

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.[76] Table 25 contains a summary of the results of the analysis.

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. Both BEA and Census instruct respondents to exclude research and development costs from their estimates of pollution abatement

[76] 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 25. 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[1] 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 I/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.

expenditures. Private R&D averages about 18 percent of all private, non-capital pollution control outlays over the period 1972-89. The range is from 12 to 25 percent with the vast majority (12 of 18) of the annual observations clustered in the 17 to 18 percent range. For example, it has been estimated that air pollution abatement R&D spending by private parties in 1987 totaled $1.574 billion, or 17 percent of all current account air pollution abatement spending by private business.[77] 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.

[77] Bratton, David M. and Gary L. Rutledge. "Pollution Abatement and Control Expenditures, 1982-1988," *Survey of Current Business*, November 1990.

It should be noted that 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 26. 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.

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.

Table 26. 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.

Source:

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. For the 1970 to mid 1980s period, the BEA's estimated fuel price penalty exceeded EPA's by a factor of three to four. In the late 1980s, the two series converge, and there is no substantial difference between the fuel price penalties estimated by the two methods.

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 27, 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 ⅛ 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 27. Estimated Annual Direct CAA Compliance Expenditures ($millions).

	Expenditures (current dollars)				
Year	Capital	O&M and "Other"	Total	GDP Price Defl.	Total Exp. ($1990)
1973	3,326	3,797	7,123	41.3	19,524
1974	3,674	4,788	8,462	44.9	21,334
1975	5,586	5,009	10,595	49.2	24,377
1976	5,915	5,219	11,134	52.3	24,099
1977	6,256	5,605	11,861	55.9	24,019
1978	6,695	5,324	12,019	60.3	22,563
1979	7,839	6,523	14,362	65.5	24,821
1980	8,398	7,924	16,322	71.7	25,769
1981	9,120	7,917	17,037	78.9	24,443
1982	9,145	6,963	16,108	83.8	21,759
1983	8,908	7,236	16,144	87.2	20,958
1984	10,377	7,649	18,026	91.0	22,424
1985	10,856	8,895	19,751	94.4	23,684
1986	11,288	7,209	18,497	96.9	21,608
1987	11,307	7,591	18,898	100.0	21,393
1988	11,716	7,180	18,896	103.9	20,587
1989	12,048	7,680	19,728	108.5	20,583
1990	11,707	8,383	20,090	113.2	20,090
average expenditure, 1973-1990 ($1990) =					22,446

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 a 3 percent (real) rate of interest (and, for illustrative purposes, a 7 percent rate). 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 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 28 summarizes annual expenditures and costs annualized at both 3 percent and 7 percent.

Due to data limitations, the cost analysis for this CAA retrospective starts in 1973, missing costs incurred in 1970-72. *Cost of Clean* (1990), however, which is is the source for annualized capital costs, includes capital expenditures for 1972. 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, 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 28. Annualized Costs and Annual Compliance Expenditures, 1973-1990 ($1990 millions).

		Annualized Costs	
Year	Expend.	at 3%	at 7%
1973	19,524	11,462	11,866
1974	21,334	13,734	14,373
1975	24,377	14,140	15,092
1976	24,099	14,891	16,154
1977	24,019	15,939	17,512
1978	22,563	15,582	17,463
1979	24,821	17,939	20,152
1980	25,769	20,222	22,760
1981	24,443	20,147	23,012
1982	21,759	19,190	22,359
1983	20,958	20,146	23,597
1984	22,424	21,426	25,197
1985	23,684	23,469	27,500
1986	21,608	22,070	26,349
1987	21,393	23,052	27,579
1988	20,587	23,092	27,863
1989	20,583	23,996	28,989
1990	20,090	25,094	30,305

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.

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 29). During the period 1973-1990, 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.[78] 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 30). 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).

Table 29. 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	1.09
1989	-0.21	0.99
1990	-0.18	1.00

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 1973-1990. 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.

Prices

A principal direct 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 31). 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

[78] 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.

substitution toward fossil fuel energy sources and toward energy in general. Total Btu consumption also increases.

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.

Table 30. Difference in Personal Consumption Between the Control and No-Control Scenarios.

Year	Nominal % Change	Real % Chang e
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

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.

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 31. 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 which 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.

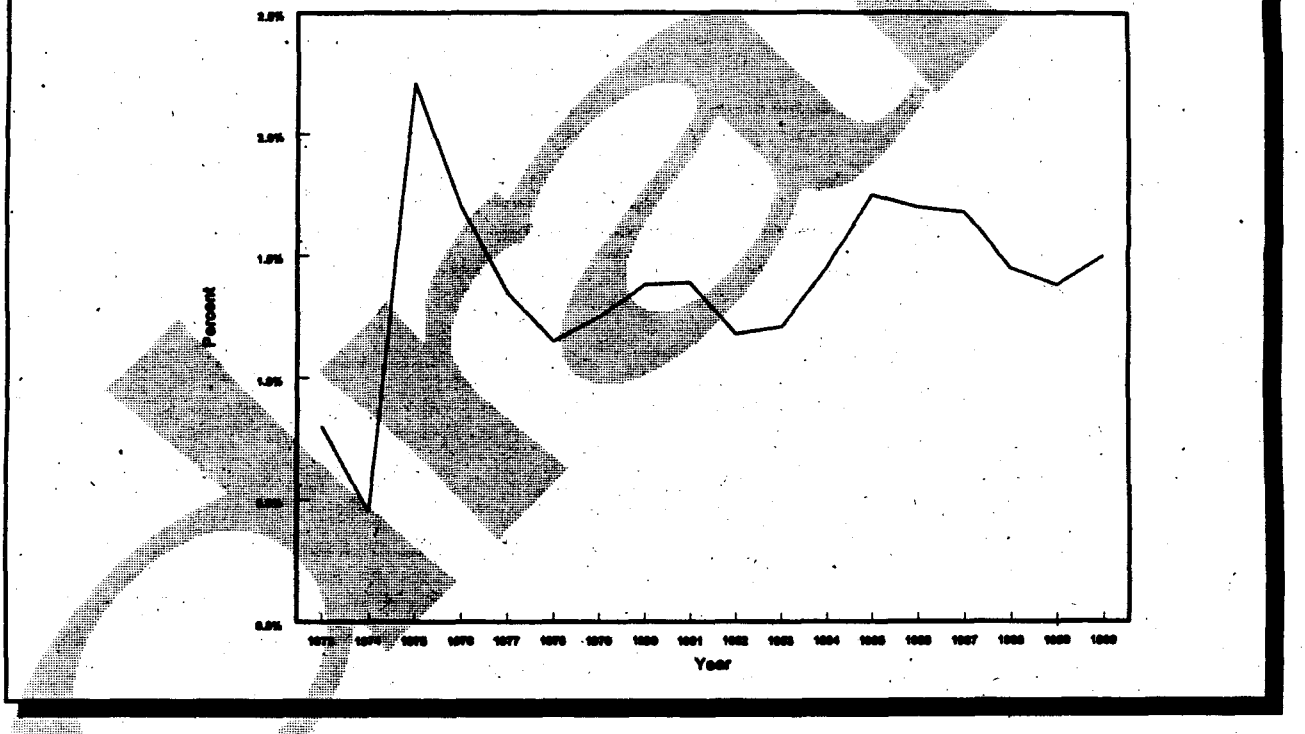


Figure 25. Percent Difference in Price of Output by Sector Between Control and No-control Scenario for 1990.

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

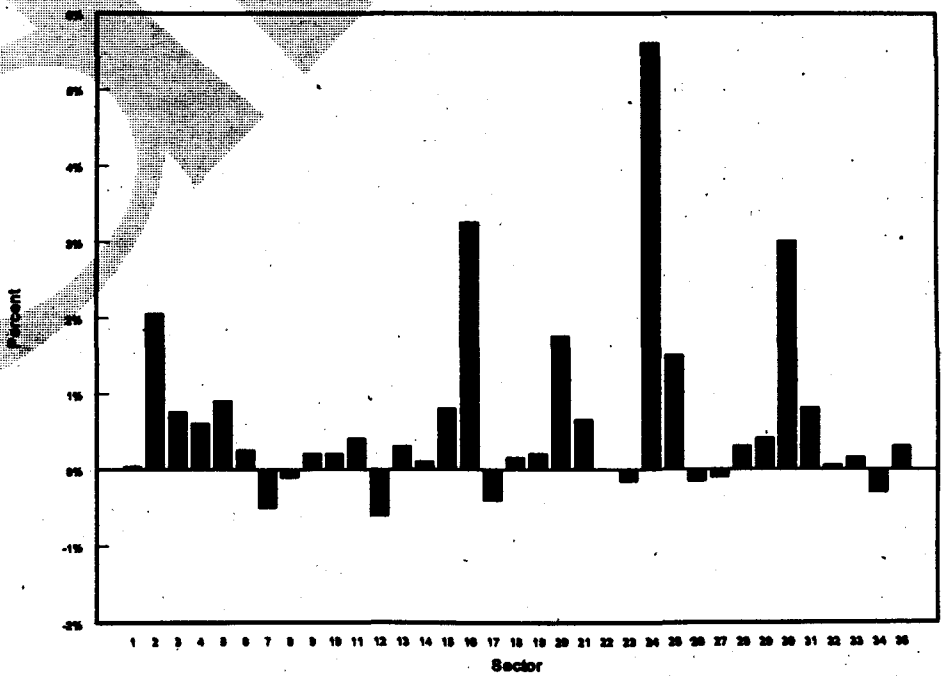
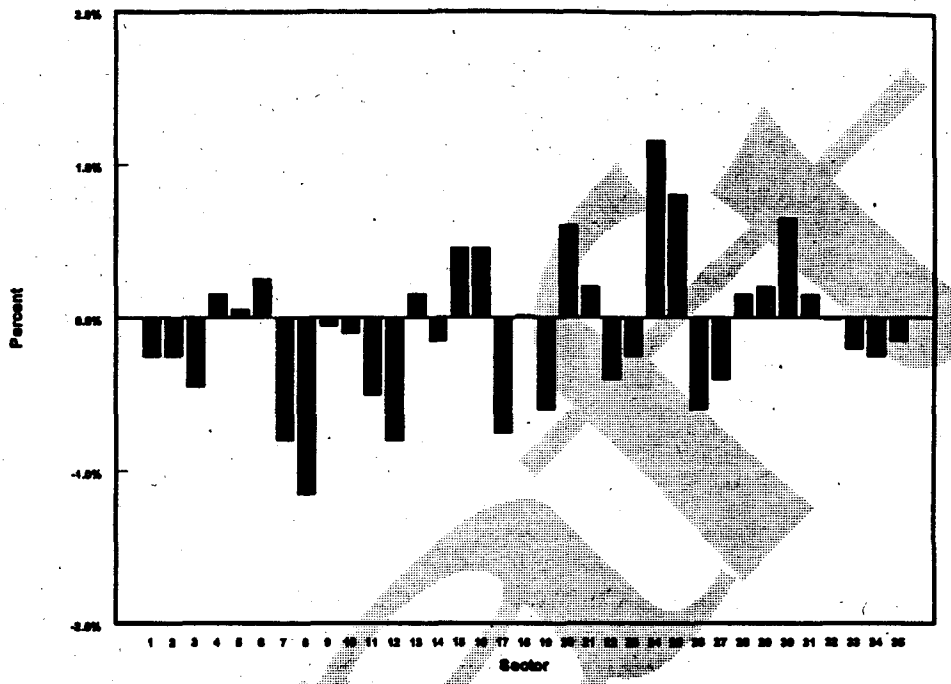


Figure 27. Percent Difference in Employment by Sector Between Control and No-control Scenario for 1990.



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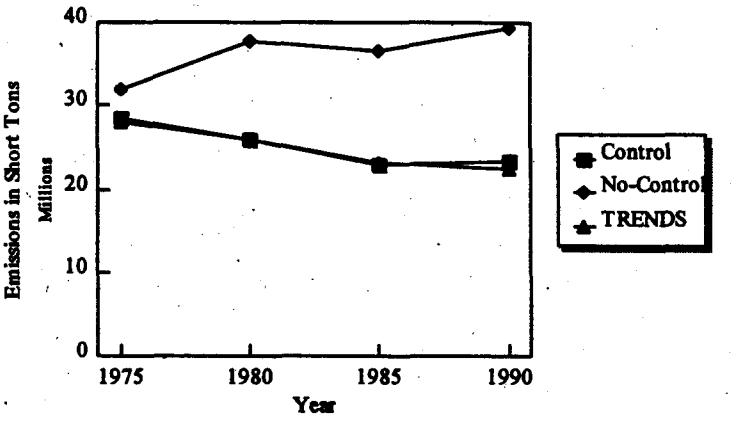
Appendix B: Emissions Modeling

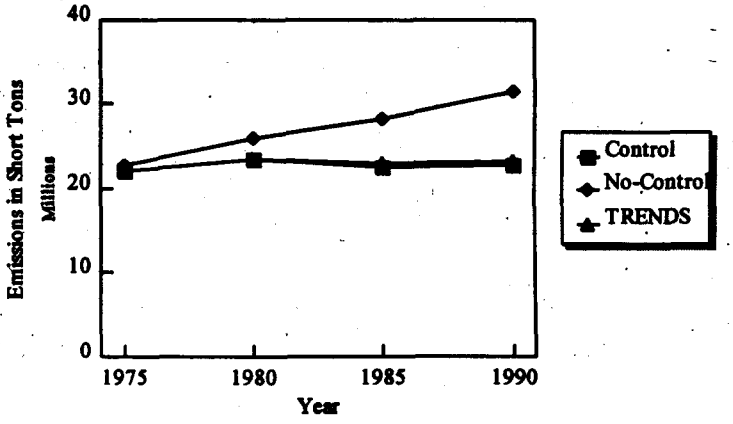
This appendix provides additional detail regarding the methodologies used to estimate control and no-control scenario emissions from each of the six principal emission sectors: industrial combustion, industrial processes, electric utilities, on-highway vehicles, off-highway vehicles, and commercial/residential 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).

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



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.[79] Comparisons between the current estimates and the *Trends* data for SO_2, NO_x, VOC, and CO are presented in Figures 28, 29, 30, and 31, 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 47, 48, 49, 50, 51, and 52, respectively, at the end of this appendix.

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

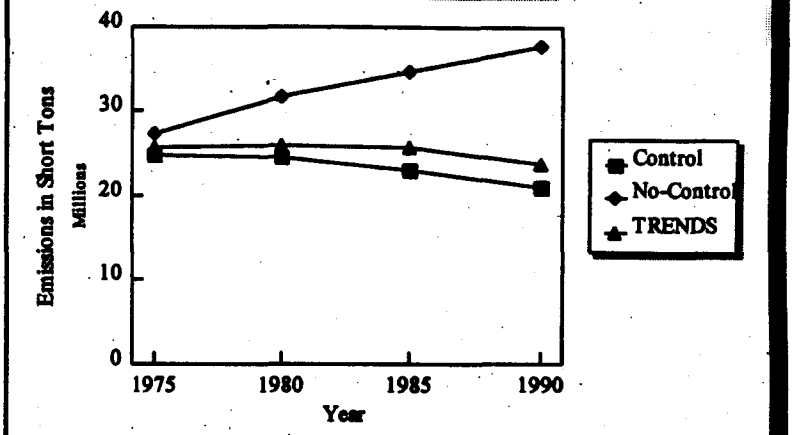


Though the EPA *Trends* and the present study emission profiles are similar to each other, they should not be expected to match precisely. This is because the emission estimates developed for the present study are based on modeled macroeconomic and emission sector conditions. Even though the

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

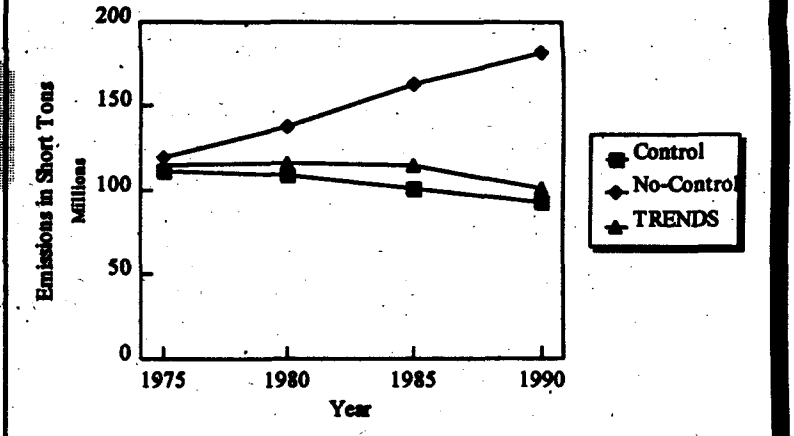
macroeconomic and sector models themselves are constructed and calibrated using historical data, modeled replications of historical trends would not be expected to precisely capture actual historical events and conditions which affect emissions. Relying on modeled historical scenarios is considered reasonable 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 32 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 32. 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 32 (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.[80]

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.

[80] 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 32). 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:

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

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

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.[81]

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 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

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

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 33. These estimates represent an average of several sectors, which were developed by ANL as part of the modeling process for ICE.

Table 33. 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

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 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 34 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

Table 34. 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.

control scenario and it is possible that these technological changes may also have occurred under a no-control scenario.

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 89. 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.[82]

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 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.[83]

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-

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

[83] 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.

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 35 lists data sources for the control scenario run.

Table 36 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. 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 37 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 38. TEEMS' activity and energy

Table 35. 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

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 36. 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 37. 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 38. 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	2,533	83	19.6
Compact (4)	3,625	115	14.6	3,338	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 39 shows the revised shares. Comparing Table 39 no-control scenario shares with those in Table 36 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 40. Changes in fleet characteristics are summarized in Table 41.

Table 39. 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 40. 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 mean of the vehicle choice model (DVSAM). Vehicle weight and seating capacity were kept unchanged from the *with C44* run. Table IV-7 shows the changes in various vehicle attributes.

Table 41. 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 steam 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.

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 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 116, 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 121.)

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.[84]

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.

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

[84] 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.

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 42 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.

Table 42. 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%

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:

$$Q_{t,b} = \left(\frac{Q_0}{E_0}\right)_{t,b} \times \left(\frac{D_t}{D_0}\right) \times \sum (f_{tj} \times E_{tj}) \qquad (3)$$

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,

$$Q_t = Q_0 \times \left(\frac{D_t}{D_0}\right) \qquad (4)$$

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:

$$Q_{t,b} = \left(\frac{Q_0}{E_0}, b \right) \times (E_{t,b} + g_{t,n} \times E_{t,n}) \quad (6)$$

In equation 6, 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 43. 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 43. *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 44.

Table 44. 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 45.

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 46.

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 45. 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 46. 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 47. 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	890	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 48. SO_2 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	430	500	570	380	460	500	560	1%
Off-Highway Vehicles	370	330	410	390	360	330	400	390	1%
Stationary Sources:									
Electric Utilities	19,670	17,480	16,050	16,310	20,690	25,020	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	22,930	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 49. 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	880	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 50. 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	1,380	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	7,200	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	6,090	6,130	(4%)
TOTAL*	24,660	24,580	23,030	20,840	27,290	31,680	34,730	37,650	(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 51. 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	Reductions
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 52. 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	Reductions
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).")
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- 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 53 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 53. 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 54.

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 54. 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, 4, 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 ρ^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 ρ^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).

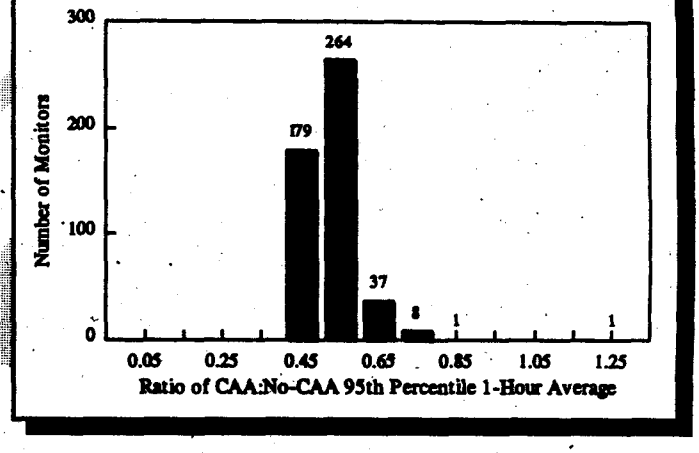
what drive the difference in the control and no-control air quality profiles for CO. In other words, the E_{NCAA} to E_{CAA} ratios used to derive the no-control profiles according the Equation (1) above are essentially based on state-level emissions estimates for CO.

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 54 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 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 26, 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 55 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 54 is adopted.

Table 55. 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_x, 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 139– 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).[85]

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 54.

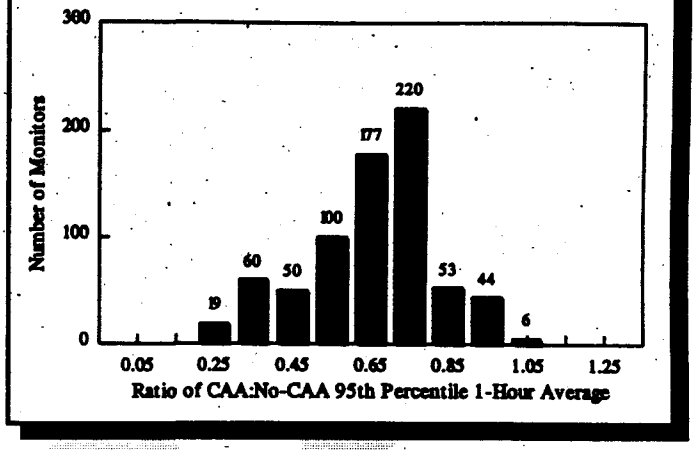
Summary differences in sulfur dioxide air quality

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.

[85] SAI SO_2, NOx, and CO Report (1994), page 4-9.

Key caveats and uncertainties for sulfur dioxide

Figure 33. Frequency Distribution 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 142 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 56 and Table 57 summarize, respectively, the NO_2 and NO monitoring data used as the basis for development of the control scenario air quality profiles.

Table 56. 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_x, NO_x and CO Report (1994).

Table 57. 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_x, 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 54 is adopted.

No-control scenario nitrogen oxides profiles

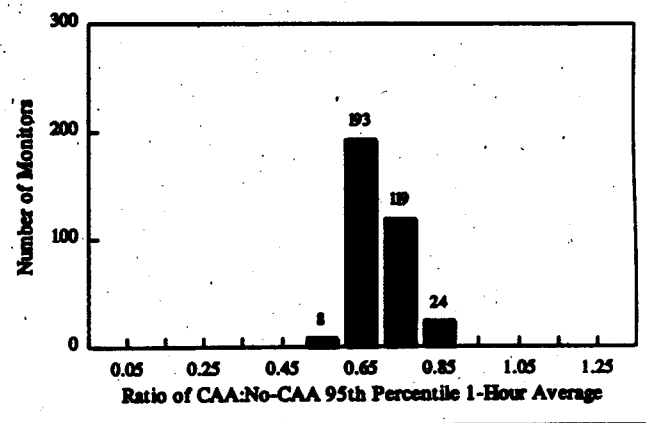
The no-control air quality profiles for NO_2 and NO are derived using the same equation –equation 7 on page 139– that was applied for CO and SO_2. As discussed in detail in the SAI SO_2, NO_x, and CO Report (1994),[86] nitrogen oxides are emitted almost entirely from anthropogenic sources and they do 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 54.

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 for 1990 Control to No-control Scenario 95th Percentile 1-Hour Average NO_2 Concentrations, by Monitor.



[86] 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 144.) In addition, the discussion on page 142 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:[87]

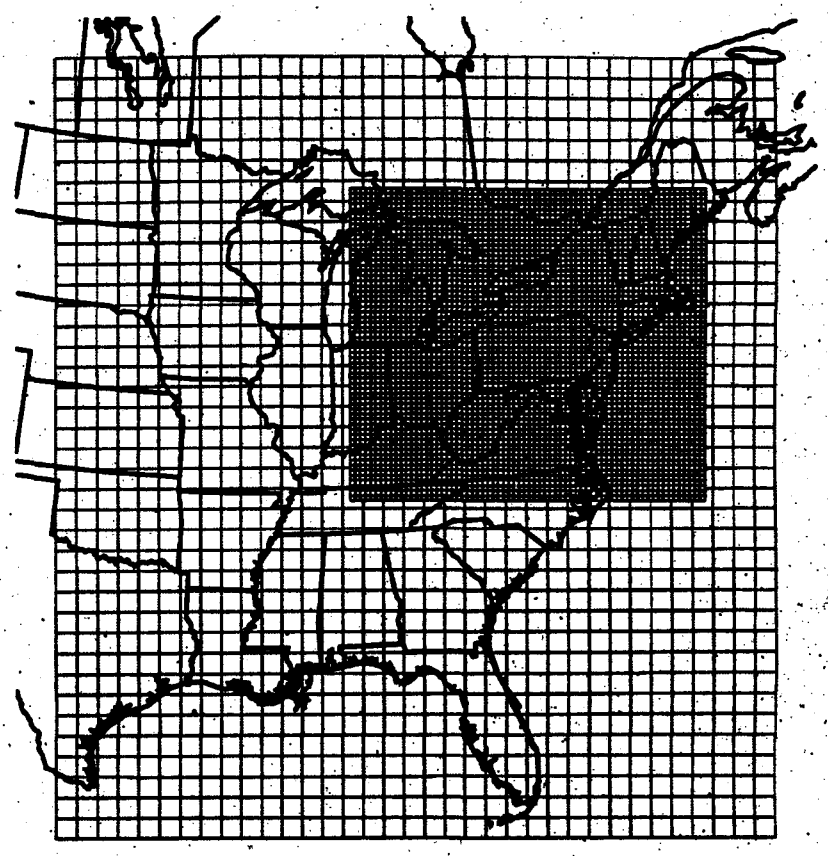
> *"Sulfur, nitrogen, and oxidant species in the atmosphere can be transported hundreds to thousands of kilometers by meteorological forces. During transport the primary emissions, SO_2, NO_x, and volatile organic emissions (VOC) are oxidized in the air or in cloud-water to form new, secondary compounds, 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."*

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,

[87] Dennis, R. RADM Report (1995), p. 1.

NO_y 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.[88] 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):

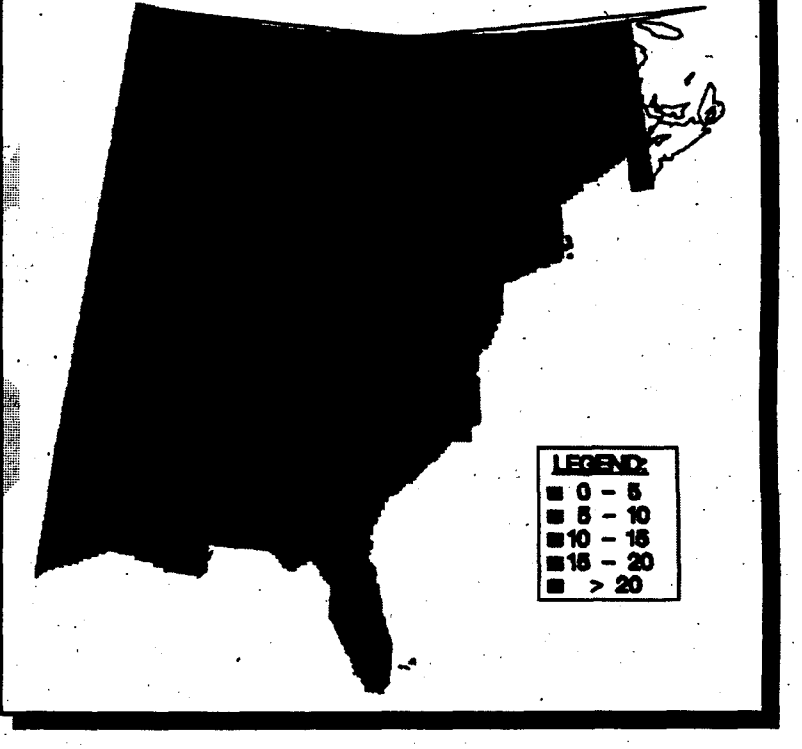
The RADM model requires a very detailed emissions inventory in both time and space. The emissions fields are also day-specific to account for the temperature effects on the volatile organics and the wind and temperature effects on the plume rise of the major point sources. At the time of the 812 Retrospective Study RADM runs, these inventories had been developed for 1985, using the 1985 NAPAP (National Acid Precipitation Assessment Program) inventory, and 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

[88] For a detailed description of the various forms of the RADM and its evaluation history, see the Dennis, R. RADM Report (1995).

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.

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.

Figure 36. RADM-Predicted 1990 Total Sulfur Deposition (Wet + Dry; in kg/ha) Under the Control Scenario.



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-N/ha, were calculated for each 80-km grid cell and for each of the two scenarios. It is important to note that ammonia deposition 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.

Figure 37. RADM-Predicted 1990 Total Nitrogen Deposition (Wet + Dry; in kg/ha) Under the Control Scenario.

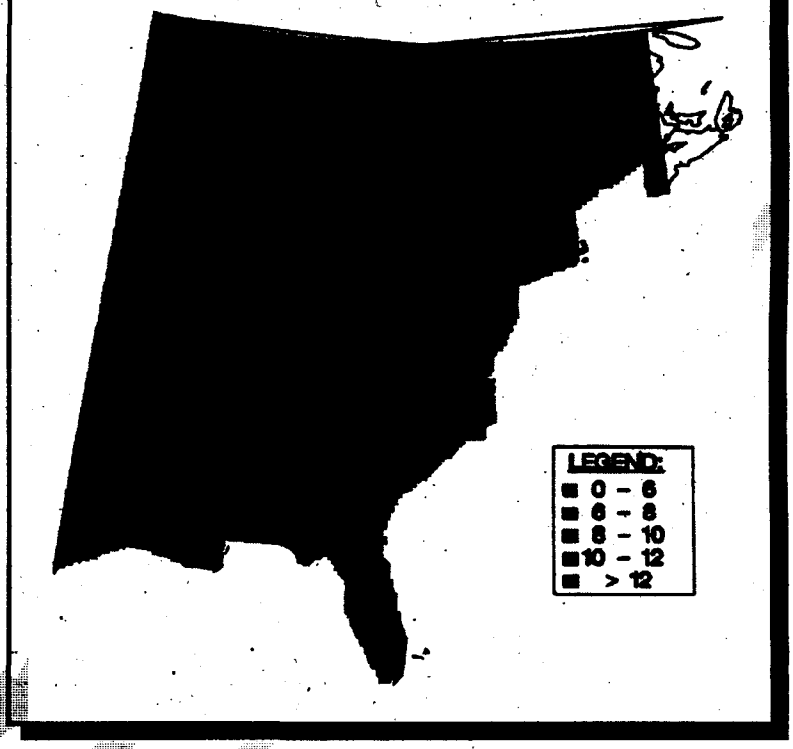
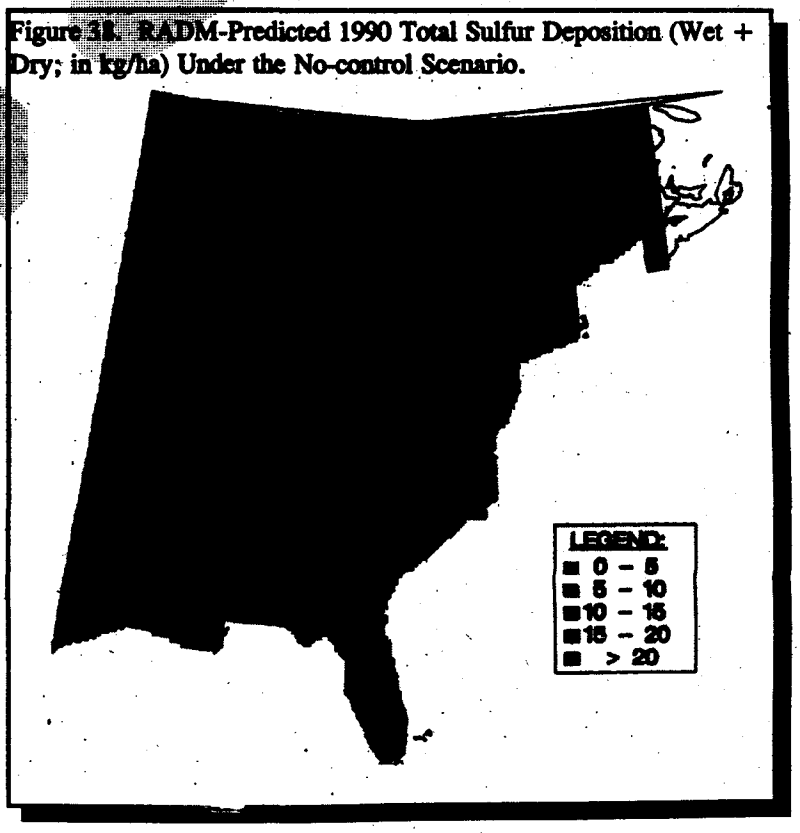


Figure 38. RADM-Predicted 1990 Total Sulfur Deposition (Wet + Dry; in kg/ha) Under the No-control Scenario.



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. 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

Figure 39. RADM-Predicted 1990 Total Nitrogen Deposition (Wet + Dry; in kg/ha) Under the No-control Scenario.

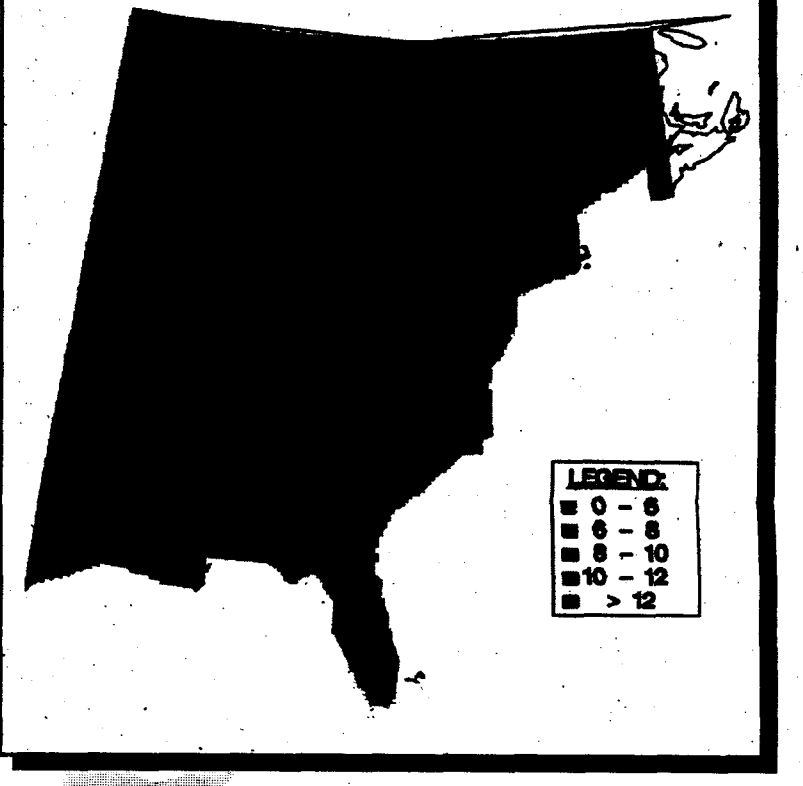
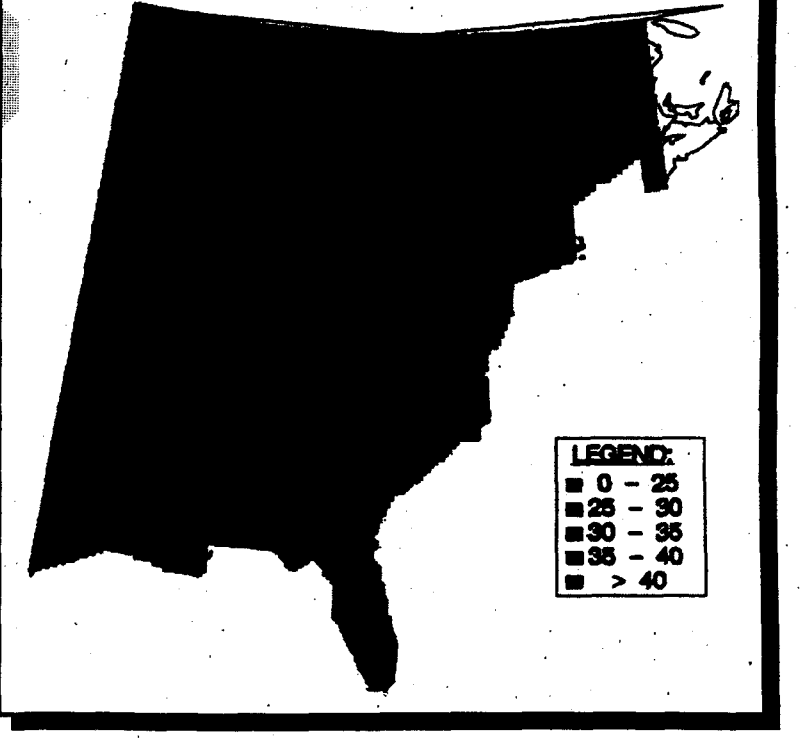


Figure 40. RADM-Predicted Percent Increase in Total Sulfur Deposition (Wet + Dry; in kg/ha) Under the No-control Scenario.



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.

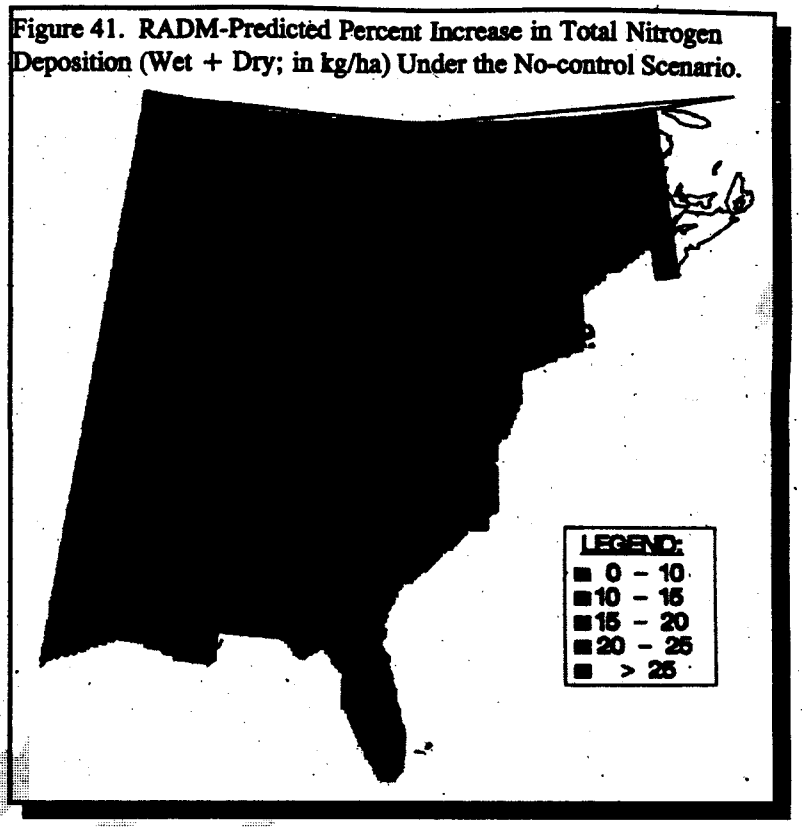


Figure 41. RADM-Predicted Percent Increase in Total Nitrogen Deposition (Wet + Dry; in kg/ha) Under the No-control Scenario.

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.

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 fine particles, especially 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).[89] 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.

[89] In addition, SAI memoranda and reports which supplement the results and methodologies used in this analysis are included in the references.

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[90] 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 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 58 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).

Daily PM_{10} data were also collected for each year between 1983 and 1990. Table 59 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

[90] 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.

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 153, to provide a basis for scaling up concentrations of each species to derive no-control scenario TSP and PM_{10} profiles.

Table 58. 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 59. 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).

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.[91] These factors are presented in Table 60. 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).[92] These speciation factors are summarized in Table 61. Data were too limited to allow differentiation between urban and rural locations for coarser particles.

[91] J. Trijonis, "Visibility: Existing and Historical Conditions--Causes and Effects," NAPAP Report 24, 1990.

[92] 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 60. 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 61. 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 62.

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:[93]

"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;*

[93] 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."*

Table 62. 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.

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).[94] Table 63 lists the names of the electronic data files containing the TSP and PM_{10} profiles for the no-control scenario.

Table 63. 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.

Summary differences in particulate matter air quality

Figure 42 provides one indication of the estimated change in TSP air quality between the control and no-control scenarios. Specifically, the graph provides data on the percentage of counties having TSP monitors with estimated annual mean TSP concentrations above 120 micrograms per cubic meter ($\mu g/m^3$). Figure 43 provides similar county data for estimated 2nd High 24-Hour TSP concentrations

[94] SAI PM Report (1995), pp. 5-2 to 5-15.

above 260 $\mu g/m^3$. Another indication of the difference in outcomes between the two scenarios is provided by Figure 44. This graph shows the distribution of annual mean TSP control to no-control ratios for 1990. 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.

Figures 42 and 43 indicate that by 1990 a significantly greater proportion of monitored counties would have observed high PM concentrations under the no-control scenario than under the control scenario. Figure 44 further indicates that annual average PM concentrations would also 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

Figure 42. Counties with Annual Mean TSP Concentrations >120 $\mu g/m^3$, Expressed as a Percent of the Number of Counties with TSP Monitors.

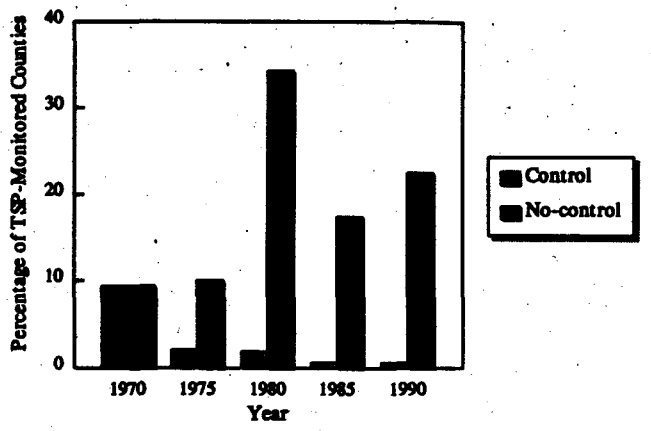


Figure 43. Counties with 2nd High 24-Hour TSP Concentrations >260 $\mu g/m^3$, Expressed as a Percent of the Number of Counties with TSP Monitors.

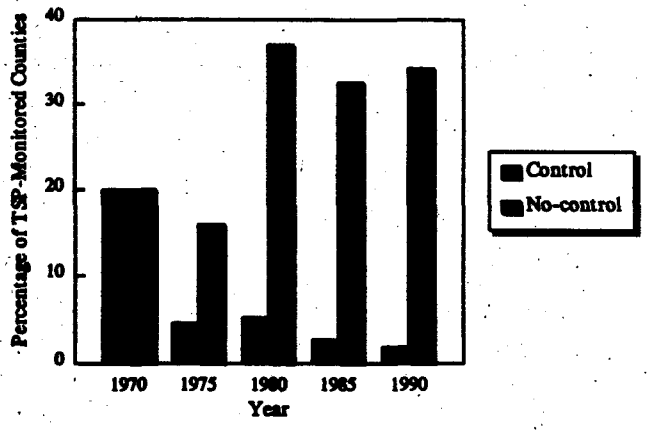
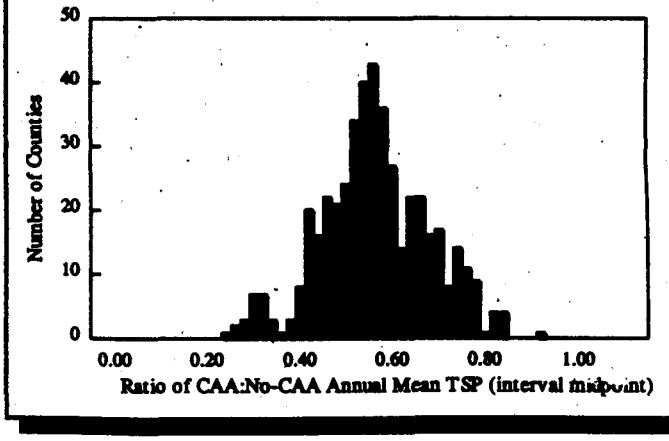


Figure 44. Distribution of County-Level Annual Mean TSP CAA to No-CAA Ratios.



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.[95]

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 oxidant models —such as the Regional Oxidant Model (ROM)— would be combined with Eulerian photochemical grid models —such as the Urban Airshed Model (UAM)— to develop control and no-control scenario estimates for ozone concentrations in rural and urban areas. However, the substantial computing time and data input requirements for both ROM and UAM precluded their use for this study.[96] Instead, three separate modeling efforts were conducted to provide urban and rural ozone profiles for those areas of the lower 48 states for which ozone effects may be most important.

First, for urban areas the Ozone Isopleth Plotting with Optional Mechanisms-IV (OZIPM4) model was run for 147 urban areas. Table 64 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.[97] 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] See SAI PM Report (1995), p. 5-9.

[96] For a description of the extensive data inputs required to operate UAM, see SAI Ozone Report (1995), p. 1-1.

[97] See SAI Ozone Report (1995), p. 1-1.

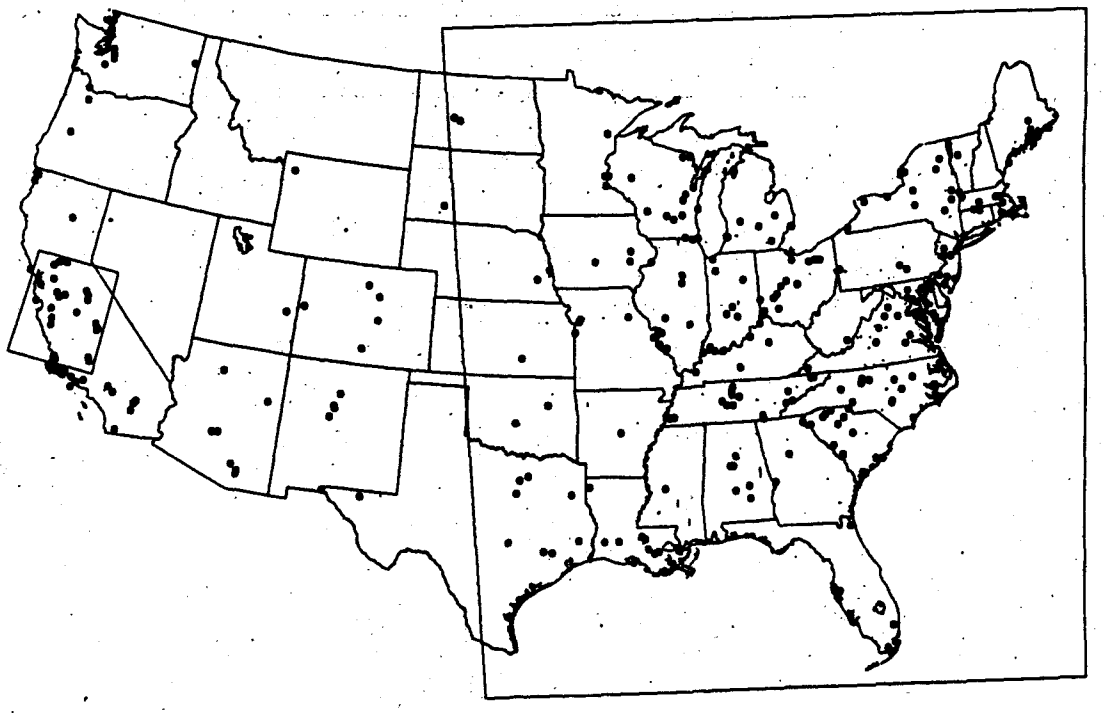
Table 64. Urban Areas Modeled with OZIPM4.

Albany, NY
Albuquerque, NM
Allentown, PA-NJ
Altoona, PA
Anderson, IN
Appleton, WI
Asheville, NC
Atlanta, GA
Atlantic City, NJ
Auburn, ME
Augusta, GA-SC
Austin, TX
Baltimore, MD
Baton Rouge, LA
Beaumont, TX
Bellingham, WA
Billings, MT
Birmingham, AL
Boston, MA
Boulder, CO
Canton, OH
Cedar Rapids, IA
Champaign, IL
Charleston, SC
Charleston, WV
Charlotte, NC
Chattanooga, TN-GA
Chicago, IL
Cincinnati, OH
Cleveland, OH
Colorado Springs, CO
Columbia, SC
Columbus, GA-AL
Columbus, OH
Corpus Christi, TX
Cumberland, MD-WV
Dallas, TX
Davenport, IA-IL
Decatur, IL
Denver, CO
Detroit, MI
El Paso, TX
Erie, PA
Eugene, OR
Evansville, IN
Fayetteville, NC
Flint, MI
Fort Collins, CO
Fort Wayne, IN
Grand Rapids, MI
Greeley, CO
Green Bay, WI
Greensboro, NC
Greenville, SC
Harrisburg, PA
Hartford, CT
Houston, TX
Huntington, WV-KY
Huntsville, AL
Indianapolis, IN
Iowa City, IA
Jackson, MS
Jacksonville, FL
Janesville Rock Co, WI
Johnson City, TN-VA
Johnstown, PA
Kansas City, MO
Knoxville, TN
Lafayette, IN
Lafayette, LA
Lake Charles, LA
Lancaster, PA
Lansing, MI
Las Cruces, NM
Las Vegas, NV
Lexington, KY
Lima, OH
Little Rock, AR
Longview, TX
Los Angeles, CA
Louisville, KY
Lynchburg, VA
Medford, OR
Memphis, TN
Miami, FL
Minneapolis, MN-WI
Mobile, AL
Monroe, LA
Montgomery, AL
Nashville, TN
New Orleans, LA
New York, NY
Norfolk, VA
Oklahoma City, OK
Omaha, NE-IA
Orange Co, CA
Owensboro, KY
Parkersburg, WV
Pascagoula, MS
Pensacola, FL
Peoria, IL
Philadelphia, PA
Phoenix, AZ
Portland, OR
Portsmouth, NH
Raleigh, NC
Reading, PA
Reno, NV
Richmond, VA
Roanoke, VA
Rochester, NY
Rockford, IL
Sacramento, CA
Salt Lake City, UT
San Antonio, TX
San Diego, CA
San Francisco, CA
San Joaquin Valley, CA
Santa Barbara, CA
Sarasota, FL
Scranton, PA
Seattle, WA
Sheboygan, WI
Shreveport, LA
South Bend, IN
Springfield, IL
Springfield, MO
Springfield, OH
St Louis, MO
Steubenville, OH-WV
Syracuse, NY
Tallahassee, FL
Tampa, FL
Terre Haute, IN
Toledo, OH
Tucson, AZ
Tulsa, OK
Utica-Rome, NY
Ventura County, CA
Victoria, TX
Washington, DC
Wheeling, WV-OH
Wichita, KS
York, 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 45.

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 45.

Figure 45. 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.[98] A two-parameter gamma distribution is then fitted to characterize each of these air quality profiles.[99] 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 65 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 65. 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).

[98] For the nighttime profiles, only 1, 2, 6, and 12-hour averaged concentrations are derived.

[99] 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.[100]

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.[101] 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 66. 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 66. 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

[100] US EPA, Office of Air Quality Planning and Standards, *"Procedures for Applying City-Specific EKMA,"* EPA-450/4-89-012, 1989.

[101] 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.

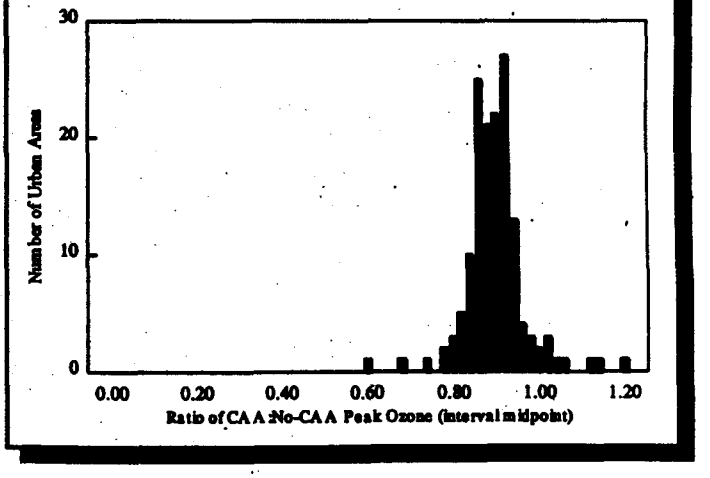
The relative results of the control and no-control scenario runs of the OZIPM4, RADM, and SAQM models were then used to derive the no-control case air quality profiles. For the urban monitors relying on OZIPM4 results, only ozone-season daytime concentrations could be calculated directly from OZIPM4 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.[102] 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.

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

Figure 46. Distribution of 1990 Control to No-control OZIPM4 Simulated Peak Ozone Ratios.



[102] The no-control scenario nighttime profiles are assumed to be the same as the control scenario profiles.

(NMOCs) in these areas results in a decrease in net ozone production when NO_x emissions increase. Figures 47 and 48 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.

Figure 47. Distribution of 1990 Control to No-control RADM-Simulated Rural Ozone Ratios.

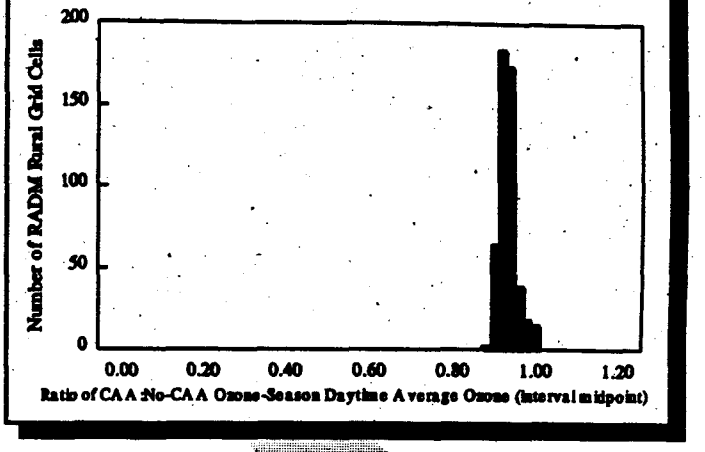
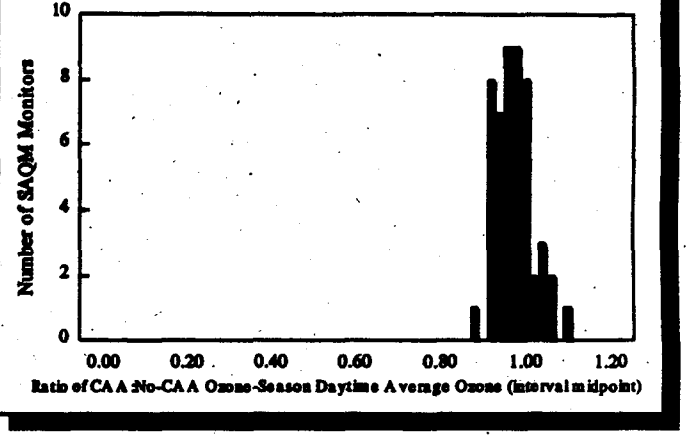


Figure 48. Distribution of 1990 Control to No-control SAQM-Simulated Ozone Ratios.



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 response system,[103] 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.[104] Finally, physical process characterizations

[103] 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.

[104] 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).

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. 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 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).

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

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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.

[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).

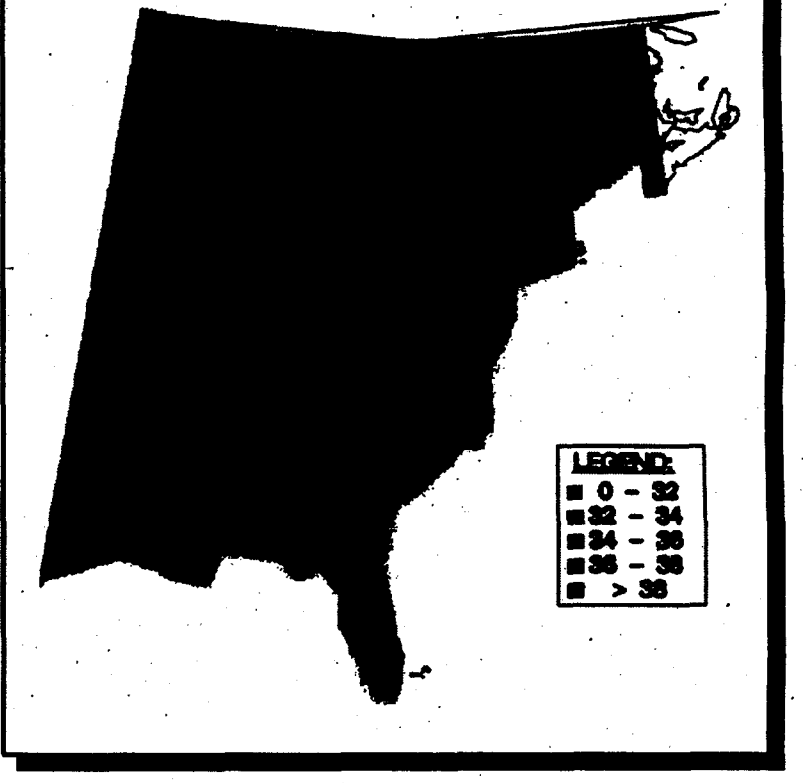
[108] For example, sulfate (SO_4) concentrations were scaled based on changes in sulfur oxide (SO_x) emissions.

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.

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 49 to provide a sense of initial visibility conditions.

Figure 49. RADM-Predicted Visibility Degradation, Expressed in Annual Average DeciView, for Poor Visibility Conditions (90th Percentile) Under the Control Scenario.



For the southwestern urban areas, 1990 control scenario annual average light extinction budget, visual range, and DeciView conditions are listed in Table 67. 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 50. No-control scenario 1990 outcomes for the 30 southwestern U.S. urban areas are presented in Table 68.

[109] The term "light extinction budget" refers to the apportionment of total light attenuation in an area to the relevant pollutant species.

Table 67. 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	27.6
Stockton, CA	153.1	19.6	27.3
San Francisco, CA	120.8	24.8	24.9
Oakland, CA	117.3	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	117.3	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

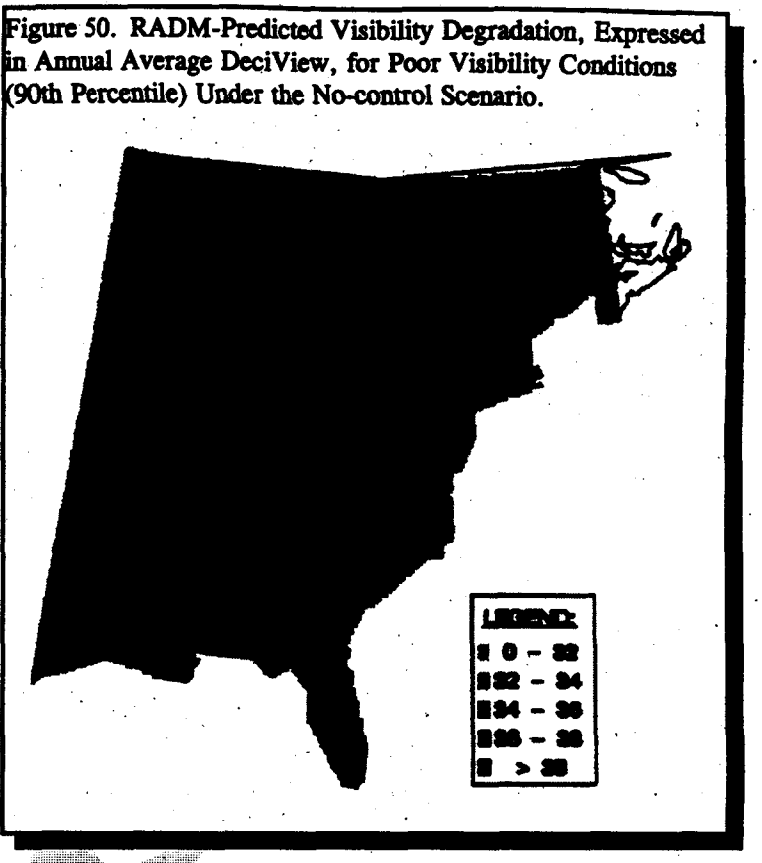
Data Source: SAI SW Visibility Report (1994).

Table 68. 1990 No-control Scenario Visibility Conditions for 30 Southwestern U.S. Cities.

City	Light Extinction Budget (b_{ext}, 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 50. 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})

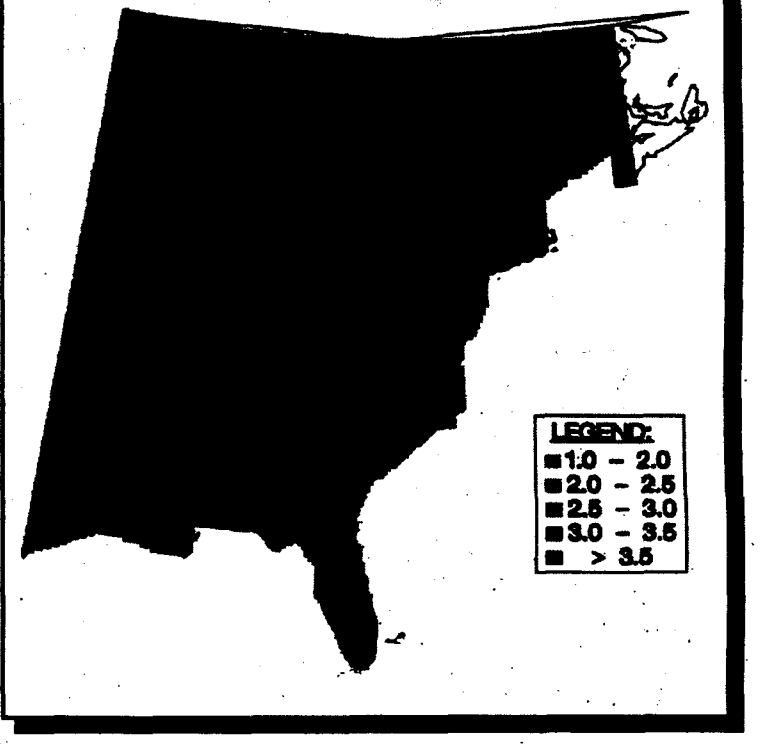
This index has the value of approximately 0 when the extinction coefficient is equal to the scattering coefficient for particle-free air (Rayleigh scattering) and increases in value by approximately one unit for each 10 percent increase in b_{ext}. Since the apparent change in visibility is related to 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 51. 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 149.)

Figure 51. 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 69. 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 69. 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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Appendix D: Human Health and Visibility Effects of Criteria Pollutants

Introduction and Overview

The language of §812 provides guidance on the range and scope of the benefits analysis to be considered in preparing the assessment for Congress. An overview of the approach EPA will use for selecting information on human health effects and several critical conceptual and implementation issues for using the health effect information to conduct a comprehensive benefits analysis are discussed.

Principles for the §812 Benefits

Estimating the effects (in both physical and economic terms) of even modest shifts in environmental releases involves complex chemical, environmental, biological, psychological and economic processes. The task of estimating the broad changes associated with adoption and implementation of the Clean Air Act challenges the limits of scientific knowledge and modeling capability to pragmatically synthesize available information and techniques into a practical framework. A pragmatic plan for a comprehensive assessment must fairly reflect the complexities and uncertainties, but still produce a policy-relevant analysis in a timely fashion. In order to achieve this ambitious goal, the following principles have been used to guide the §812 benefits assessment.[111]

Comprehensiveness: The assessment should include as many benefit categories reasonably believed affected by implementation of the Clean Air Act. Comprehensiveness requires assessing effects we have greater levels of scientific confidence as well as less well understood effects. The degree of relative certainty among effects must be carefully described in order to fairly present a broad portrayal of the physical and social benefits accruing to the nation from implementing the Act. In addition, the legislation explicitly directs a comprehensive benefits coverage that avoids zero values.

Quantification Where Feasible: The basic purpose of a benefit assessment is to indicate the magnitude of change in each effect category, and to reflect the relative importance (in monetary terms) of changes in each effect category and the overall magnitude of total benefits. Measuring changes in economic terms is the most direct way to reflect relative importance of each category, and is the fundamental goal of an economic benefit analysis. This requires quantifying and presenting estimated effects in both physical and monetized economic terms. Because the goal of the assessment is to produce an economic benefit analysis, emphasis will be largely on categories having direct and perceptible effects on human health. Thus, the emphasis of the analysis has been on categories such as symptoms and diseases rather than on physical changes (such as cell level changes) that do not directly result in a decreased health status noticeable to the individual.

[111] These principles are adopted in part from the "Acid Rain Benefit Assessment: Draft Plan for the [§812] 1992 Assessment," Acid Rain Division, U.S. EPA, 1991.

Efficient Use of Previous Research Results: Significant research effort has been spent to understand and quantify the complex relationships between air pollution and human health. The §812 assessment has relied as much as possible on available research results, making adjustments as necessary to apply the existing results to the current analysis.

Incorporate Uncertainty: To properly convey the results of any benefits assessment, it is important to include an evaluation of how much confidence the analysts have in the estimates. Ideally this would include a formal quantitative assessment of the potential for error, and what are the sources and directions of the likely biases and omissions. A method for considering and reporting uncertainty must be built into the fundamental structure of the research design for the assessment.

Health Effects Studies

Scientific research about air pollution's adverse impacts uses a broad array of methods and procedures. The research methods used to investigate the health effects of air pollution have become considerably more sophisticated over time, and will continue to evolve in the future. This progress is the result of better available research techniques and data, and the ability to focus further research more sharply on key remaining issues based on the contributions of earlier work.

The available health effects studies that could potentially be used as the basis of the §812 assessment are divided into Epidemiology Studies and Human Clinical Studies. Epidemiological research in air pollution investigates the association between exposure to air pollution and observed health effects in the study population. Human clinical studies involve examination of human responses to controlled conditions in a laboratory setting. EPA has conducted research on health effects from exposure to pollution using each approach, and studies using these techniques have been considered in various formal regulatory proceedings. Each type of study (as it is used for air pollution research) is described below, and the relative strengths and weaknesses for the purposes of the §812 assessment are examined.

Epidemiological Studies

Epidemiological studies evaluate the relationship between ambient exposures to and health effects in the human population, typically in a "natural" setting. Statistical techniques (typically variants of multivariate regression analysis) are used to estimate quantitative concentration-response (or exposure-response) relationships between pollution levels and health effects.

Epidemiology studies can examine many of the types of health effects that are difficult to study using a clinical approach. Epidemiological results are well-suited for quantitative benefit analyses because they provide a means to estimate the incidence of health effects related to varying levels of ambient air pollution without extensive further modeling effort. These estimated relationships implicitly take into account at least some of the complex real-world human activity patterns, spatial and temporal distributions of air pollution, synergistic effects of multiple pollutants and other risk factors, and compensating or mitigating behavior by the subject population. Suspected relationships between air pollution and the effects of both long-term and short-term exposure can be investigated using an epidemiological approach. In addition, observable health endpoints are measured, unlike clinical studies which often monitor endpoints that do not result in observable health effects (e.g. forced expiratory volume). Thus, from the point of view of conducting a benefits analysis, the results of epidemiological

studies, combined with measures of ambient pollution levels and the size of the relevant population, provide all of the essential components.

Two types of epidemiological studies are considered for dose-response modeling: cohort and longitudinal studies. Cohort-based studies are population based studies where initially disease free individuals are followed over a certain period of time, with periodic reporting of the health status from the individuals. Studies about relatively rare events such as cancer incidence or mortality can require tracking the individuals over a long period of time, while more common events (e.g., respiratory symptoms) occur with sufficient frequency to evaluate the relationship over a much shorter time period. An important feature of cohort studies is information is known about each individual, including other potential variables related to the disease being studied. These variables, called confounders, are important to identify because if they are not accounted for in the study and they are associated with air pollution levels, a false association may be associated with air pollution. The exposure information can range from data from a personal exposure monitor carried by the participants for a few weeks in a study concerning minor symptoms, to monitoring data of ambient air concentrations.

A second type of study used in this analysis is longitudinal or time-series studies. The relationship between population-wide health information such as counts for daily mortality, hospital admissions, or emergency room visits (e.g. heart attack, mortality, acute asthma attack or chronic bronchitis) and ambient levels of air pollution are evaluated. One advantage of this study design is it allows "the population to serve as its own control". Changes in such factors as tobacco, alcohol and illicit drug use, access to health care, employment, and nutrition may have a pronounced impact on the health of an individual, and, if they vary with the pollution variable of interest, need to be included in a cohort study. However, such potential confounding factors are unlikely to vary with pollution levels and these variables do not need to be evaluated in the analysis.

Some of the drawbacks of epidemiological research are it is difficult to adequately characterize exposure, measurement errors in the explanatory variables, the influence of unmeasured variables, and correlations between the pollution variables of concern and both the included and omitted variables. These can potentially lead to spurious conclusions. However, epidemiological studies involve a large number of people and do not suffer extrapolation problems like clinical studies.

Human Clinical Studies

Clinical studies of air pollution involve exposing human subjects to various levels of air pollution in a carefully controlled and monitored laboratory situation. The physical condition of the subjects are measured before, during and after the pollution exposure. The measured physical condition can include general biomedical information (e.g., pulse rate and blood pressure), physiological effects specifically affected by the pollutant (e.g., lung function), the onset of symptoms (e.g., wheezing or chest pain), or the ability of the individual to perform specific physical or cognitive tasks (e.g., maximum sustainable speed on a treadmill). These studies often involve exposing the individuals to pollutants while exercising, increasing the amount of pollutants that are actually introduced into the lungs.

Clinical studies can isolate cause-effect relationships between pollutants and certain human health effects. Repeated experiments altering the pollutant level, exercise regime duration and types of participants can potentially identify effect thresholds, the impact of recovery (rest) periods, and the differences in response among population groups. While cost considerations tend to limit the number of participants and experimental variants examined in a single study, clinical studies can follow rigorous laboratory scientific protocols, such as the use of placebos (clean air) to establish a baseline level of effects and precise measurement of certain health effects of concern.

There are drawbacks to using clinical studies as the basis for a comprehensive benefits analysis. Clinical studies are appropriate for examining acute symptoms caused by short-term exposure to a pollutant. While this permits examination of some important health effects from air pollution, such as bronchoconstriction in asthmatic individuals caused by sulfur dioxide, it excludes studying more severe effects or effects caused by long term exposure. Another drawback is health effects measured in some well-designed clinical studies are selected on the basis of the ability to measure precisely the effect rather than a larger symptom, for example forced expiratory volume. The impact of some clinically measurable health effects such as lung function on future medical condition or lifestyle changes are not well understood.

Ethical limits on experiments involving humans also impose important limits to the potential scope of clinical research. Chronic effects cannot be investigated because people cannot be kept in controlled conditions for an extended period of time, and because these effects are generally irreversible. Participation is generally restricted to healthy subjects, or at least to exclude people with substantial health conditions that compromise their safe inclusion in the study. This can cause clinical studies to avoid providing direct evidence about populations of most concern, such as people who already have serious respiratory diseases. Ethical considerations also limit the exposures to relatively modest exposure levels, and to examining only mild health effects that do no permanent damage. Obviously for ethical reasons human clinical evidence cannot be obtained on the possible relationship between pollution and mortality, heart attack or stroke, or cancer.

The very precision about exposure conditions possible in clinical studies also creates an obstacle to using clinical dose-response functions for benefit analysis. It is difficult to extrapolate results from clinical settings to daily exposures faced by the whole population. For example, many clinical studies evaluate effects on exercising individual. Only a small portion of the population engages in strenuous activity (manual labor or exercise) at any time. Reflecting these fundamental differences between the laboratory setting and the "real world" imposes a formidable burden on researchers to provide information about human activity patterns, exercise levels, and pollution levels.

Issues In Selecting Studies To Estimate Health Effects

A number of issues arise when selecting and linking the individual components of a comprehensive benefits analysis. The appropriate procedure for handling each issue must be decided within the context of the current analytical needs, considering the broader analytical framework. While more sophisticated or robust studies may be available in some circumstances, the potential impact on the overall analysis may make using a simpler, more tractable approach the pragmatic choice. In considering the overall impact of selecting a study for use in the §812 assessment, important factors to consider include the likely magnitude the decision will have on the overall analysis, the balance between the overall level of analytical rigor and comprehensiveness in separate pieces of the analysis, the effect on the scientific defensibility of the overall project, and the potential impact on the schedule and EPA's available resources.

This section discusses seven critical issues in selecting health information for use in the §812 assessment: types of studies considered, confounding variables, concentration-response function thresholds, exposure modeling, meta-analyses, uncertainty and the links to other portions of the §812 assessment. The previous discussion about the types of research methods available for the health information alluded to some of these issues, as they are potentially important factors in selecting between

studies using different methods. Other issues address how scientific research will be used in the overall analytical framework, and have not been previously discussed.

Types of Studies Considered

Given the major advantages of epidemiological studies - exposures do not need to be modeled and health effects are observed in a large, more heterogeneous population, epidemiological studies are used as the basis for determining both health effects and dose response curves. Clinical studies are used if there are health effects observed in clinical studies not observed in epidemiological studies.

Confounding Factors

Confounding can occur when the real cause of disease is associated with the disease and the variable being evaluated. If only the variable being evaluated is used in the analysis, a false association can occur. For example, in longitudinal epidemiology studies of air pollution, it is important to take into account weather conditions, because weather is associated with air pollution and health outcomes. If only air pollution is evaluated, a false associated could occur. Potential confounders include weather-related variables, age and gender mix of the subject population, and the type and distribution of the pollution emissions. Studies that include a broad range of likely confounders can offer a more robust conclusion about an individual pollutant, even if the statistical confidence interval is larger due to including more variables in the analysis.

Thresholds

Exposure-response relationships are conceptualized as either exhibiting a threshold of exposure below which adverse effects are not expected to occur, or as having no response threshold, where any exposure level theoretically poses a non-zero risk of response. The methods employed by health researchers to characterize exposure-response relationships may or may not explicitly analyze the data for the existence of a threshold. Studies may analyze relationships between health and air pollution without considering a threshold. However, extrapolating the risk relationship beyond levels in the study may overestimate benefits. If a threshold for population risk exists but is not identified by researchers, then Clean Air Act benefits would be underestimated, because non-existent risks would be estimated at low levels of air pollution and subtracted from the risks associated with high exposure levels. On the contrary, if a threshold is artificially imposed where one does not exist, the relative benefits of the Clean Air Act may be overestimated, because the population risk associated with levels of pollution below the threshold would be zero. In general, those studies that explicitly consider the question of a threshold (whether a threshold is identified or not) provide stronger evidence and will be considered a positive feature when selecting studies for this analysis.

Exposure Modeling

One potential obstacle to using dose-response information from clinical research methods in a benefits assessment is the need for an exposure model. This requirement adds an additional step in the analytical process, introducing another source of uncertainty and possible error. The dose-response functions developed from clinical research are specific to the population participating in the study and the exposure conditions used in the laboratory setting. For air pollution research, important factors affecting exposures include concentration, duration of exposure, recovery times in cleaner conditions, and exercise (i.e., ventilation) levels.

To apply the clinical results to model the general population, two decisions must be made. First, how far to expand the conditions in the clinical setting? For example, if the subjects in the clinical study were healthy male college students, should the results be applied to the entire population, including children? Second, how many people in the general population are exposed to conditions similar to those used in the clinical setting Frequently clinical studies are conducted at relatively high exercise levels (increasing the dose, or the quantity of pollutants actually delivered to the lungs). In the general population few people experience these conditions very often, and people do not reach these exercise levels with equal frequencies during the day and night. In addition to the exercise level question, the analyst must also determine the number of people that are exposed to the levels of ambient conditions seen in the laboratory. Air quality varies throughout a city and is typically reported by data from monitors located at various places throughout the city. However, people are not exposed to the conditions at any one monitor all day. As people move around in the city, they are exposed to ambient air quality conditions represented by different monitors at different times during the day. To further compound the problem, air quality also varies between indoors and outdoors, within a car or garage, and by such factors as proximity to a roadway or major pollution source (or sink). The exposure model must account for the ambient conditions in the "microenvironments" that the population actually experiences.

The issues of study subjects, exercise and microenvironments can influence the choice of clinical studies selected for the §812 assessment. Clinical studies that use exposure regimes and exercise levels more similar to what larger groups of the population see are easier to apply in a benefits model than are more narrow studies. Similarly, studies that use a diverse group of subjects are easier to apply to the general population than are more narrow studies.

As noted above in the discussion of epidemiological research, exposure models are not necessary to apply epidemiology results to a benefits model. The diverse activity patterns, microenvironments, and pollution levels are already considered in the aggregate through the concentration-response functions. This simplifies the assessment task by reducing the analytical burden.

Uncertainty

The stated goal of the §812 assessment of providing a comprehensive estimate of benefits deriving from the Clean Air Act means information with very different levels of confidence will have to be used. Benefit categories will not be omitted simply because they are highly uncertain or controversial, but the ultimate audience needs to understand which of the benefit categories are reasonably well understood, and which are more tentative. Documentation of suspected and identified biases, omissions and uncertainties must be included throughout technical documentation of the analysis.

The ideal approach to characterizing uncertainty is to conduct a formal quantitative uncertainty analysis. A common approach develops an estimated probability distribution for each component of the analysis. The information on distributions may be a continuous distribution (such as would typically come from a statistical confidence interval about an estimated result), or it may involve a selection of two or three equally likely alternatives. A Monte Carlo procedure draws simultaneous combinations of random draws from these distributions and estimates the entire model. The model is estimated for many such random combinations, creating a distribution of results that reflects the impact of the joint uncertainties in each piece of the analysis.

The most serious obstacle to preparing a formal quantitative uncertainty analysis is identifying all the necessary distributions for each component of the analysis. Monte Carlo approach requires all components of the model be repeatedly rerun many times. However, the broad plan for the §812 project is to link the outputs from independent modeling activities. It is be impractical to simultaneously rerun

the macroeconomic, emissions, dispersion, and exposure models because of the diverse origins of the models. Therefore, instead of a complete formal uncertainty analysis, the §812 assessment includes a less rigorous analysis of the inherent uncertainties in the modeling effort. The uncertainty analysis combines quantitative and qualitative elements designed to sufficiently describe the implications of the uncertainties. A primary goal of the sensitivity/uncertainty analysis is to identify the health effects that make a sizable contribution to the overall assessment of the monetary benefits. There may be situations where there are significant differences in the available information used to predict the incidence of a particular health effect (i.e., the uncertainty bounds are large). It is important to alert the reader to situations where using the lower incidence estimates may portray the health effect as only modestly contributing to the overall total benefits, but using reasonable alternative higher estimated incidence figures (or higher monetized values) would substantially impact not only the monetized value of the individual health effect, but actually make a noticeable difference in the total benefits assessment.

Consideration of the overall uncertainties inherent in the §812 assessment has several important implications on health study selection. It is important to carefully examine the pragmatic balance between the level of uncertainties in the analysis and the need for comprehensive coverage of all benefit categories. There are frequent situations where there is a direct tradeoff between more comprehensive coverage and the restriction of the analysis to more certain information. The relationship between the overall uncertainty in other parts of the analysis and the uncertainty choices concerning each particular health effect will be carefully considered. In addition, a balance must also be maintained between comprehensiveness of an assessment and the allocation of limited analytical resources.

Links to Air Quality Modeling

Selection of health studies for the §812 assessment must consider the need to match the health information to the air quality modeling conducted for assessment. For example, information on the health effects from short term (five minute) exposure to sulfur dioxide cannot be readily combined with information on average daily sulfur dioxide levels. There are two key components of matching the air quality information, which are actually intimately related: averaging times and spatial resolution. Matching the health studies, the exposure modeling (if needed) and the air quality modeling is a case of searching for a "lowest common denominator" between the components of the assessment.

Target Population

Many of the studies relevant to quantifying the benefits of air pollution reductions have focused on specific sensitive subpopulations suspected to be most susceptible to the effects of the pollutant. Some of these effects may be relevant only for the studied subpopulation; effects on other individuals are either unknown, or not expected to occur. For such studies, the challenge of analysis is to identify the size and characteristics of the subpopulation and match its occurrence to exposure. Other studies have examined specific cohorts who may be less susceptible than the general population to health effects from air pollution (e.g., healthy workers), or who differ in age, gender, race, ethnicity or other relevant characteristics from the target population of the benefits analysis. Extrapolating results from studies on nonrepresentative subpopulations to the general population introduces uncertainties to the analysis, but the magnitude of the uncertainty and its direction is unknown. Because of these uncertainties, analyses often limit the application of the dose-response functions only to those subpopulations with the characteristics of the study population. While this approach has merit in minimizing uncertainty in the analysis, it can also severely underestimate benefits if in fact similar effects are likely to occur in other populations. For these reasons, studies that examine broad, representative populations are preferable to studies with narrower scope because they allow application of the functions to larger numbers of persons without introducing additional uncertainty.

Exposure Ranges

One component of the §812 analysis will estimate the air pollution levels that would have occurred in the absence of the Clean Air Act. These estimates will certainly be much larger than currently observed levels of US air pollution, and perhaps even levels currently observed elsewhere in the world. This aspect of the analysis poses difficulties with the application of concentration-response functions that have been based on exposures at much lower pollution levels. The shape of the concentration-response function much above observed exposures levels is unknown. It is possible that biological mechanisms affecting response that are unimportant at low levels of exposure may dominate the form of response at higher levels, introducing nonlinearity to the mathematical relationship. In general, studies that include exposure levels spanning the range of interest in the §812 assessment are preferable to studies at levels outside of the range, or that only include a narrow part of the range. A possible drawback to this approach is that studies which fit this criterion have often been conducted outside the US The application of foreign studies to US populations introduces additional uncertainties regarding the representativeness of the exposed population and the relative composition of the air pollution mix for which the single pollutant is an indicator. These difficult issues must be balanced by the analyst when selecting studies for the benefits analysis.

Peer Reviewed Research

Whenever possible, peer reviewed research rather than unpublished information has been relied upon. Research that has been reviewed by the EPA's own peer review processes, such as review by the Clean Air Science Advisory Committee (CASAC) of the Science Advisory Board (SAB) has been used whenever possible. Research reviewed by other public scientific peer review processes such as the National Academy of Science or the National Acidic Precipitation Assessment Program is also be included in this category.

Research published in peer reviewed journals but not reviewed by CASAC has also been considered for use in the §812 assessment, and has been used if it is determined to be the most appropriate available study. Research accepted for publication by peer reviewed journals ("in press") has been considered to have been published. Indications that EPA intends to submit research to the CASAC (such as inclusion in a draft Criteria Document or Staff Paper) provide further evidence that the journal-published research should be used.

Air pollution health research is a very active field of scientific inquiry, and new results are being produced constantly. Many research findings are first released in University Working Papers, dissertations, government reports, non-reviewed journals and conference proceedings. Some research is published in abstract form in journals, which does not constitute peer review. In order to use the most recent research findings and be as comprehensive as possible, unpublished research has been examined for possible use in the §812 assessment. Any unpublished research used will be carefully identified in the report, and treated as having a higher degree of uncertainty than published results. The peer review of the §812 assessment by the Advisory Council on Clean Air Compliance Analysis will provide one review process for not only all components of the assessment, but also the way in which the components have been used.

Application Of Health Science Research And Air Quality Results To Estimate Health Effects

The general modeling approach used in the §812 assessment is a "reduced form" or "embedded model" approach.[112] The concept of a reduced form is to use simplified versions of previously constructed complex models to characterize the impact of a series of linked physical and socioeconomic processes. The results of three major types of models have been used in the integrated reduced form model, but the models themselves remain exogenous to the reduced form model. Those three models are the macroeconomic model, the emissions model, and the air quality dispersion models for the criteria pollutants.

The reduced form model differs primarily from the previous health effects benefits models prepared by EPA for criteria pollutant analysis in two ways. First, a single integrated model is used for all criteria pollutants. This increases the consistency in the integrated assessment, and also facilitates sensitivity analysis of vital components.

The second major difference is that the level of detail used in the exposure model component is limited to a level of detail that matches the air quality modeling results prepared for the assessment. For example, EPA's previous models for conducting ozone health analysis require spatially dispersed ozone data on an hourly basis. Because that level of time- disaggregated data is not available for the §812 assessment (particularly for the "no-control" scenario), the §812 reduced form model has not been run at that level of disaggregation. This creates difficulties in estimating health effects with dose-response functions based on short-term exposures. To mitigate this problem, a procedure was developed for estimating the appropriate daily distribution of pollutant levels (e.g., one, two or seven hour data) from available data from the §812 air quality modeling. A simplified version of the human activity pattern portion of the exposure model was also developed. While these simplifications increase the uncertainty levels, the overall effect is small. Considering the uncertainties in the other components of the analysis (especially the steps leading to the "no-control" air quality modeling), the pragmatic advantages of a simplified integrated model are a valuable tradeoff for less detailed modeling.

Some reanalysis and manipulation has been done on information used in various portions of the analytical structure to link all the diverse pieces appropriately. In general, the approach favored adjusting the exposure components of the analysis rather than the dose-response (or concentration-response) information.

One difficult issue that confronts any health benefits estimation project of air pollution is aggregating results. An important objective of any integrated analytical approach is to avoid double counting of effects while also avoiding inadvertently omitting some effects altogether. Two main facets of this objective are aggregating effects across symptoms (or diseases) and across pollutants. Many of the criteria air pollutants are closely related, and considerable effort has gone into health effects research to isolate the effects of each pollutant. This is especially difficult in the case of particulate matter, because that category includes all of another criteria pollutant (lead), and a large and important portion of two others (sulfur oxides). In addition, all the criteria pollutants can and do occur simultaneously, making it difficult to isolate impacts.

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[112] This approach draws heavily from the proposed modeling approach in "Acid Rain Benefit Assessment: Draft Plan for the [§812] 1992 Assessment," Acid Rain Division, U.S. EPA, 1991.

Concentration-Response Functions

Issues Common to Several Pollutants

The concentration-response functions describe the relationships between pollutant concentrations and a variety of health effects. The following discussion includes general issues common to several or all pollutants regarding the selection of health endpoints and studies, the form of the concentration-response functions, choice of values used for baseline incidence, and the choice of affected populations modeled. Additional pollutant-specific information is included in sections pertaining to individual pollutants. Each pollutant section also includes a table that presents the dose-response functions, specific information needed to apply these functions for the current analysis, and references for the information.

For each pollutant, the choice of the health effects modeled was as inclusive as possible. For those pollutants associated with a variety of endpoints, the health effects may overlap with each other, as described in sections below. Despite this overlap, the current analysis presents separate estimates of incidence for all health effects.

The analysis includes as many studies related to a given health effect as possible, except for studies inapplicable to the current analysis. For some endpoints, the group of adequate studies yielded mixed results, with some showing statistically significant responses to pollutant concentrations and others with insignificant associations. Unless study methods have been judged inadequate, dose-response functions with both statistically significant and insignificant coefficients have been included to characterize the possible range of risk estimates. However, it should be noted that some studies that found insignificant effects for a pollutant could not be used because they did not report the insignificant coefficient values.

The analysis includes modeling of the health effects that have been associated with exposure to a number of pollutants. Thus, in cases where the health effect is being modeled for the several correlated pollutants of interest, regression models that included multiple pollutant exposures were chosen in preference to single pollutant models to avoid over counting. Using separate regressions for each pollutant (if the pollutants are correlated) may overstate the number of effects caused by each pollutant alone. As an example, this analysis models incidence of premature mortality for both particulate matter and ozone. Thus, coefficients for both these pollutants come from the regression model that considers the effects of both pollutants simultaneously.

A few studies have identified pollutant concentration thresholds below which health effects have been shown not to occur. However, for the current analysis, all concentration-response functions assume no threshold effects. Thus, in certain cases, the incidence of health effects may be overstated at low concentrations.

Many studies reported only a relative risk value (defined as the ratio of the incidence of disease in two groups exposed to two different exposure levels). The analysis required conversion of these values to their corresponding regression coefficients when the coefficients were not reported. When converting the relative risk to a coefficient value, the analysis used the functional form of the regression equation reported by the authors of the study.

The coefficients from a number of studies measured the change in the number of health effects for the study population rather than a change per individual. These coefficients were divided by the size of the study population to obtain an estimate of change per individual. The coefficient could then be multiplied by the size of the population modeled in the current analysis to determine total incidence of health effects.

Certain dose-response functions (those expressed as a change relative to baseline conditions) required baseline incidence data associated with ambient levels of pollutants. Incidence data necessary for the calculation of risk and benefits were obtained from national sources whenever possible, because these data are most applicable to a national assessment of benefits. The National Center for Health Statistics provided much of the information on national incidence rates. However, for some studies, the only available incidence information come from the studies themselves; in these cases, incidence in the study population is assumed to represent typical incidence nationally.

Many studies focus on a particular age cohort of the population for the identification of health effects. The choice of age group is often a matter of convenience (e.g., extensive Medicare data may be available for the elderly population) and not because the effects are, in reality, restricted to the specific age group (even though their incidence may vary considerably over the life span). Thus, the application of the concentration-response functions was, in most cases, carried out for the entire population using incidence values appropriate to the population at large. Additional calculations were also carried out for the subset of the population specifically studied in the report which served as the source of a concentration-response function. In other cases, studies were performed on individuals with specific occupations, activity patterns, or medical conditions because these traits relate to the likelihood of effect. In these cases, application of the dose-response function has been restricted to populations of individuals with these same characteristics.

Pollutants Using Dose-response Information from Epidemiological Studies

Ozone

Ozone has been associated with a variety of health effects. Table 70 includes the concentration-response functions used in this analysis as well as information needed to apply these functions. This section addresses several specific issues concerning selection and use of ozone dose-response functions.

Choice of concentration-response functions for each health effect. For each health effect, the analysis includes concentration-response functions identified in the peer-reviewed literature. The current analysis excludes some studies based on the quality of the study or inapplicability of the study to this analysis (e.g., use of exposures adjusted for indoor and outdoor areas near individuals rather than monitored ambient exposures).

Ozone concentration thresholds. Several studies have found statistically significant relationships between ozone and health effects at ozone concentrations below the current hourly standard of 0.12 ppm. Although all functions in the current analysis assume no thresholds of ozone concentration below which health effects are assumed not to occur, one study listed in Table 70 does suggest a threshold of 0.025 ppm.

Conversion of coefficients dependent on symptom status during the previous day. Krupnick et al. (1990) employed a Markov process to determine the probability of symptoms that were dependent on symptom status of the previous day. The current analysis adjusts the regression coefficients produced by the model in order to eliminate this dependence on previous day's symptom status.

Within-study meta-analysis. Ostro and Rothschild (1989) report six separate regression coefficients that correspond to regression models run for six separate years. To estimate a single coefficient, this analysis performed a fixed coefficient meta-analysis on the six years of data.

Normalization of β coefficients by population. For several studies of mortality and hospital admissions, the current analysis required normalizing the β coefficient by the population in the geographical area of study. For example, Thurston et al. (1994) reported the dependent variable as the number of admissions/day in the Toronto, Canada study area. Thus, the β coefficient was divided by the population of the study area to determine admissions/person/day.

Issues related to the applied populations. In general, the analyses apply to the entire population, with the exception that some studies applied only to the particular cohort studied in the research papers. For example, Crocker and Horst (1981) has been applied only to individuals who work in occupations requiring heavy outdoor physical labor. Whittemore and Korn (1980) has been used only for the population of asthmatics. In addition, Ostro and Rothschild (1989) measured effects on working age adults, thus their findings were applied only to those 18 to 65 years of age.

Issues related to the overlap of health effects. Several health endpoints for ozone overlap with each other. These relate primarily to studies with differing definitions of symptom occurrence, such as studies of the presence of any of 19 symptoms (Krupnick et al., 1990) and studies of restrictions in daily activity (Ostro and Rothschild, 1989; Portney and Mullahy, 1986). All endpoints are included in the current analysis. The possibility of double-counting overlapping effects was considered in aggregating results across endpoints.

Table 70. Summary of Dose Response Functions for Ozone

Except where noted otherwise, the functional form is

$$\Delta cases = cases * (e^{\beta * \Delta O_3} - 1);$$

where "cases" refers to incidence at the first pollution level.

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas from original study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
mortality	803/year[b] (of U.S. pop.)	24-hr average O_3 (in ppb)	all in the St. Louis SMSA, nonaccidental	all	$\beta = 0.00029$ (The model includes O_3 as the only pollutant)	s.e. = 0.00076	Dockery et al., 1992 St. Louis
mortality	803/year[b] (of U.S. pop.)	24-hr average O_3 (in ppb)	all in 11 counties in eastern TN, nonaccidental	all	$\beta = -0.00065$ (The model includes O_3 as the only pollutant)	s.e. = 0.0018	Dockery et al., 1992 Eastern Tennessee
mortality	803/year[b] (of U.S. pop.)	1 hour daily max O_3 (in ppb)	all in Los Angeles, nonaccidental	all	$\beta = 0.00$[c] (Model includes O_3 and PM_{10})	s.e. = 0.00021[c]	Kinney et al., 1995
mortality	803/year[b] (of U.S. pop.)	24-hr average O_3 (in ppb)	all in Philadelphia County, nonaccidental	all	$\beta = 0.000611$ (The model includes ozone, TSP, and SO_2)	s.e.= 0.000216	Moolgavkar, et al., 1995 Philadelphia

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas from original study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
hospital admissions – all resp. illnesses (ICD 460-519)	504/year[d] (incidence in pop. > 65 years of total U.S. pop.)	24 hr avg ($\mu g/m^3$)	65 and older in New Haven, CT, Tacoma, WA	over 65 only	β = New Haven: 0.0027 Tacoma: 0.007 where 1 $\mu g/m3$= 0.510 ppb (two pollutant model with PM_{10} and O_3)	New Haven: s.e. = 0.0014 Tacoma: s.e. = 0.0025 where $1 \mu g/m^3 = 0.51$ ppb	Schwartz, 1995 New Haven and Tacoma
daily respiratory admissions-includes 466, 480, 481, 482, 485, 490, 491, 492, 493	n/a	1 hour daily max ozone (ppb)	all	all	for Toronto: β =0.0388/2.4 million =1.62×10^{-8} delta cases/day = $\beta * \Delta O_3 *$ pop (ozone and PM_{10} model used)	se = 0.0241/2.4 million =1.0×10^{-8}	Thurston et al., 1994 Toronto
hospital admissions pneumonia (480-487)	229/year[d] (incidence in pop. > 65 years of total U.S. pop.)	24-hr avg ppb	over 65, Birmingham AL	over 65	β = 0.00262 for O_3 alone (single pollutant model only avail.)	s.e. = 0.00196	Schwartz, 1994a Birmingham
hospital admissions COPD (490-496)	103/year[d] (incidence in pop. > 65 years of total U.S. pop.)	24-hr avg ppb	over 65, Birmingham AL	over 65	β = 0.00314 for O_3 only (only single pollutant model avail.)	s.e. = 0.00316	Schwartz, 1994a Birmingham
hospital admissions pneumonia (480-487)	229/year[d] (incidence in pop. > 65 years of total U.S. pop.)	24-hr avg ppb	over 65, Detroit	over 65	β = 0.00521 (two pollutant model with O_3 and PM_{10}) note: authors suggest a threshold of 25 ppb	s.e. = 0.0013	Schwartz, 1994b Detroit

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas from original study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
hospital admissions COPD (490-496)	103/year[d] (incidence in pop. > 65 years of total U.S. pop.)	24-hr avg ppb	over 65, Detroit	over 65	$\beta = 0.00549$ (two pollutant model with O_3 and PM_{10}) note: authors suggest a threshold of 25 ppb	s.e. = 0.00205	Schwartz, 1994b Detroit
hospital admissions pneumonia (ICD 480-487)	229/year[d] (incidence in pop. > 65 years of total U.S. pop.)	24 hr avg ppb	65 and over in Mpls	over 65	$\beta = 0.002795$ (two pollutant model with O_3 and PM_{10})	s.e. = 0.00172	Schwartz 1994c Mpls, St. Paul
presence of any of 19 acute respiratory symptoms	n/a	daily one-hour max. O_3 (pphm)	adult members of families of elementary school-aged children in Glendora-Covina-Azusa, CA	all	$\Delta\ Sympt_{day}/day = (p_1 - p_0) * Pop$ where $p_1 = \frac{1}{1 + e^{-(\ln \frac{p_0}{1 - p_0} + \beta * \Delta O_3)}}$ and p_0 = the probability of having $Sympt_{day}$ per individual for a 24-hour period in the base case = 0.19 $\beta = 1.4 \times 10^{-4}$[k] (Model includes O_3, COH, SO_2)	s.e. 6.7×10^{-5}[k]	Krupnick et al., 1990

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas from original study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
self-reported asthma attacks	n/a	1 hour daily max. oxidants (ppm)	asthmatics in Los Angeles	all asthmatics (4%[f] of the total population)	Δ asthma attacks/day = $(p_1 - p_0) * Pop$ where $p_1 = \frac{1}{1 + e^{-(\ln \frac{p_0}{1 - p_0} + \beta * \Delta O_3)}}$ and p_0 = the probability of attacks per asthmatic for a 24-hour period in the base case, = 0.027[g] $\beta = 1.9 \times 10^{-3}$ [h] 1.11 = factor to convert measured O_3 levels to oxidants (only model includes oxidants and TSP)	s.e. = 7.2×10^{-6} [h,i]	Whittemore and Korn, 1980 and U.S. EPA, 1993
respiratory and nonrespiratory conditions resulting in a minor restricted activity day (MRAD)	780,000/ year[l] (of study pop.)	1 hour daily max. O_3 (ppm) averaged over 2 weeks	employed adults across the U.S. between the ages of 18-65 (urban residents)	all adults aged 18-65	equation predicts Δ MRAD over 2 weeks $\beta = 2.2 \times 10^{-3}$ [k] (converted from $\mu g/m^3$ using the relationship 1 $\mu g/m^3$ = 0.51 ppb at 25 °C) (Model includes O_3 and fine particulates)	s.e. = 6.6×10^{-4} [l]	Ostro and Rothschild, 1989
respiratory restricted activity days (RRADs)	310,000/ year[m] (of study pop.)	1 hour daily max O_3 (ppm) averaged over 2 weeks	employed adults across the U.S. between the ages of 18-65 (urban residents)	all adults aged 18-65	equation predicts Δ RRAD over 2 weeks β = -0.0054[k] (Model includes O_3 and fine particulates)	s.e. = 0.0017[l]	Ostro and Rothschild, 1989

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas from original study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
sinusitis and hay fever	n/a	hourly O_3 averaged over six years (1974-1979) in ppm	adults in urban areas surveyed in the National Health Interview Survey	all	$\Delta cases = \frac{[\Phi(\alpha + \beta x_1) - \Phi(\alpha + \beta x_0)]}{6} * Pop$ where: Φ = standard normal distribution function x_1 = average hourly O_3 concentration over six years in the no-CAA scenario x_0 = average hourly O_3 concentration over six years in the CAA scenario a = -1.13[n] β = 0.017 maximum likelihood probit model	s.e. = 0.0070[o]	Portney and Mullahy, 1990
worker productivity (resulting in changes in daily wages)	n/a	hourly O_3 concentration averaged over a workday or 24-hours (ppm)	citrus workers in Southern California	individuals in occupations that require heavy outdoor physical labor	$\Delta I = I*\eta*(X_1 - X_0)/X_0 * Pop * W$ ΔI = change in total daily income, η = income elasticity with respect to O_3 conc., η = -0.14 for 24-hour period, I = total daily income per worker engaged in strenuous outdoor labor = $73[p] W = proportion of outdoor workers in the U.S. population = 0.012[q] X_0 = average hourly O_3 concentrations with CAA, X_1 = average hourly O_3 concentrations without CAA (NOTE: Average number of days worked per year for workers engaged in strenuous outdoor labor = 213)[r] (model includes O_3 only)	not available	Estimated using data from Crocker and Horst (1981) and U.S. EPA, 1994c

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas from original study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
development of "definite" asthma (not present 10 years earlier)	n/a	mean hourly ozone in pphm (Data averaged over 15 years)	adult men	adult males	$p_1 = \frac{1}{1 + e^{-(\ln \frac{P_0}{1 - P_0} + B * delta O_3)}}$ $\Delta cases = \frac{(p_1 - p_0)}{10} * Pop$ where: p_1 = probability of having asthma in the no-CAA scenario p_0 = probability of having asthma in the CAA scenario = 0.022[s] B = 1.18×10^{-4} [t]	s.e. = 3.5×10^{-5} [y]	Abbey et al., 1995a; 1993

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas from original study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
The following two rows should be combined, e.g., cases of $DFEV_1 \geq 15\%$ for heavy exercisers (using equation based on Avol et al., 1984) should be added to cases of $DFEV_1 \geq 15\%$ for moderate exercisers (using equation based on data from Seal et al., 1993)							
Decrements in lung function as measured by forced expiratory volume in one second (FEV_1)	n/a	Exposure to ozone for 1.33 hours during which individuals were exercising continuously for one hour (controlled setting)	Heavily exercising male and female bicyclists (mean age = 26.4 yrs)	all under age 50[a]	$\Delta cases = \alpha * \beta * \Delta O_3 * Pop.$ where, β = 0.00297 for $DFEV_1 \geq 15\%$ = 0.00268 for $DFEV_1 \geq 20\%$ α = 0.06656[v]	--	Avol, et al., 1984
Decrements in lung function as measured by FEV_1	n/a	Exposure to ozone for 2.33 hours during which individuals were exercising intermittently (total exercise time = 1 hour) (controlled setting)	Moderately exercising male and female college students (ages 18-35)	all under age 50[a]	$= \sum_{i=1}^{3} \left\{ \Phi\left[\frac{\ln\left(\frac{X_0 \cdot d_i}{1 - X_0 \cdot d_i}\right) - a}{b} \right]_{e_i} - \Phi\left[\frac{\ln\left(\frac{X_1 \cdot d_i}{1 - X_1 \cdot d_i}\right) - a}{b} \right]_{e_i} \right\} \cdot Pop$ where, a = -0.664 for $DFEV_1 \geq 15\%$ = -0.326 for $DFEV_1 \geq 20\%$ b = 0.000840 for $DFEV_1 \geq 15\%$ = 0.000919 for $DFEV_1 \geq 20\%$ d_1 = 1.06[w] d_2 = 1.00 d_3 = 0.70 e_1 = 0.288[x] e_2 = 0.224 e_3 = 0.640 X_0 and X_1 are ozone concentrations in the CAA and No-CAA scenarios	--	Seal, et al., 1993

NOTES:

[a] Pollutant coefficients expressed as a change in health effects per change in ppb O_3.

[b] Mortality data for general U.S. population taken from Vital Statistics of the United States, Vol. II - Mortality, Part B, pgs. 198-200 (U.S. Dept. of Health and Human Services). Incidence rates were generated for total mortality and mortality excluding accidental deaths and adverse effects, suicide, homicide, and other external causes (ICD E800-E999). Rates calculated based on 1990 population (City and County Databook, 12th edition, 1994, U.S. Dept. of Commerce, Bureau of the Census, Washington, D.C.).

[c] β and s.e. determined from the relative risk of 1.00 and 95% confidence intervals on the relative risks of 0.94 and 1.06. (Relative risks were determined for a 143 ppb change in O_3).

[d] Centers for Disease Control, 1992. Vital and Health Statistics, Detailed Diagnoses and Procedures, National Hospital Discharge Survey, 1990. Number of 1990 discharges divided by 1990 U.S. population (248,709,873) from City and County Databook, 12th edition, 1994, U.S. Dept. of Commerce, Bureau of the Census, Washington, D.C.

[e] Determined the incremental effect/unit O_3 by incorporating stationary probabilities from transitional probabilities. ESEERCO (1994) obtained transitional probabilities for adults from original study authors.

[f] U.S. EPA, 1994a.

[g] Calculated as baseline asthma attack rate (number of attacks per person per year) divided by 365 days per year. Number of attacks per person per year = 9.9 from National Center for Health Statistics, National Health Interview Survey, 1979 (as cited by Krupnick, 1988).

[h] β coefficient and s.e. converted to Δ in cases/ppb O_3 based on the following relationship: 1 ppb O_3 = 1.11 ppb oxidants.

[i] Study did not report a s.e. Thus, the analysis assumed the largest s.e. possible (at $p = 0.01$, using a two-tailed test of significance)

[j] Ostro and Rothschild (1989) report average annual MRADs as 7.8 per person, using data from 6 years.

[k] β is a weighted mean using separate coefficients for six years. Each year's coefficient was weighted by the inverse of the variance for that coefficient.

[l] Standard error is the square root of the sum of the weights (sqrt[sum($1/var_I$)], where I indicates the individual year).

[m] Ostro and Rothschild (1989) report average annual RRADs as 3.1 per person, using data from 6 years.

[n] Obtained by determining the products of beta coefficients for other independent variables and their mean values and summing these and the constant value.

[o] Calculated by dividing β by asymptotic t-ratio.

[p] Average daily wage, assuming an 8-hour day, by workers in the job categories listed above, taken from U.S. Bureau of the Census, Earnings by Occupation and Education, 1990.

[q] Full- and part-time workers (total of 3,100,000) taken from U.S. Bureau of the Census, Earnings by Occupation and Education, 1990. Includes the following job categories: farm workers; groundskeepers and gardeners, except farm; forestry workers, except logging; timber cutting and logging occupations; brickmasons and stonemasons; brickmason and stonemason apprentices; roofers; structural metal workers; construction trades, n.e.c.; construction laborers; garbage collectors; and stevedores. Value is divided by total U.S. population.

[r] Average number of days worked per year, assuming an 8-hour day, by workers in the job categories listed above, taken from U.S. Bureau of the Census, Earnings by Occupation and Education, 1990.

[s] Abbey et al., 1993.

[t] Obtained by personal communication with Floyd Petersen, Loma Linda University.

[u] From Table 12 in 1992 Statistical Abstracts, the percent of individuals in the U.S. population under age 50 = 75%.

[v] Factor to adjust for differences in concentration among microenvironments and amount of time spent in different microenvironments at heavy exercise rates.

[w] The values, d_i, adjust ozone concentrations for various microenvironments (outdoor -- near road, outdoor -- other, and indoor) using values reported in U.S. EPA, 1993.

[x] The values, e_i, adjust the response rates by the percent of time spent in each microenvironment at the relevant exercise rates (i.e., percent of time at a fast rate is used for Avol et al., 1984, and percent of time at a moderate rate is used for Seal et al., 1993). U.S. EPA (1993) presents information to determine e_i values.

Particulate Matter

Similarly to ozone, particulate matter has been associated with a variety of health effects, ranging from acute respiratory symptoms to mortality. Although detailed discussion of each health endpoint is not provided, this section addresses several specific issues concerning the dose-response functions. These functions pertain to PM-10.

Selection of dose-response functions. The analysis has identified many studies that include dose-response functions for particulate matter. For example, some studies included multiple analyses, yielding both significant and non-significant results, depending on the nature of the input parameters (e.g., for different lag periods or concurrent exposures). In these cases, only significant results were included. Studies were excluded if health endpoints could not be defined in the U.S. population. For example, in Pope and Dockery (1992) the authors developed a unique definition of symptomatic children in Utah which has no correlation in the incidence data bases which were available (e.g., from National Center for Health Statistics, Centers for Disease Control); consequently, the results could not be applied to the general population. Table 71 includes the particulate matter concentration-response functions used in this analysis. In general, the chosen functions come from peer-reviewed literature.

Conversion of coefficients dependent on symptom status during the previous day. Krupnick et al. (1990) employed a Markov process to determine the probability of symptoms that were dependent on symptom status of the previous day. The current analysis adjusts the regression coefficients produced by the model in order to eliminate this dependence on previous day's symptom status.

Within-study metaanalysis. Ostro and Rothschild (1989) report six separate regression coefficients that correspond to regression models run for six separate years. To estimate a single coefficient, this analysis performed a fixed coefficient metaanalysis on the six years of data.

Conversion of exposure measures. Some studies modeled concentration of total suspended particulates (TSP), fine particulates, or a coefficient of haze. This analysis converted these measures to PM-10 based on location-specific information whenever possible.

Issues related to the applied population. As noted earlier, the application of the concentration-response functions was generally carried out for the entire population and the particular age cohort of the study. For particulate matter, the current analysis includes several exceptions to using the concentration-response functions for the entire population. The Schwartz hospital admission studies (1994a, 1994b, 1994c, and 1995) found effects for individuals over 65 years of age; thus the current analysis uses the concentration-response function for this cohort only. Pope et al. (1991) measured respiratory effects in 10-12 year old children. These effects are extrapolated to those under 18 years old, nationwide. Another paper, Ostro et al. (1991), applies only to asthmatics, while Ostro and Rothschild (1989) applies only to working age adults.

Issues related to the overlap of health effects. As with ozone, several of the health endpoints identified in the particulate matter literature may overlap with each other. This occurrence is apparent especially with studies that model various respiratory symptoms. For example, one study models the presence of any of 19 respiratory symptoms (Krupnick et al., 1990), while another models lower or upper respiratory symptom responses (Pope et al., 1991) which may include the same symptoms modeled in Krupnick et al. (1990). Still other studies include dose-response functions for acute respiratory symptoms that result in minor restrictions in daily activity or both minor and more severe restrictions in daily activity (such as days of lost work) (Ostro and Rothschild, 1989; Ostro, 1987). This analysis models incidence of all endpoints.

Table 71. Summary of Dose Response Functions for Particulate Matter

Except where noted otherwise, the functional form is

$$\Delta cases = cases * (e^{\beta * \Delta PM - 10} - 1).$$

where "cases" refers to incidence at the first pollution level.

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas. from Original Study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
mortality	803[b]/yr non-accidental (of U.S. pop.)	1-day lag PM_{10}	all in E. Tenn., nonaccidental	all	$\beta = 0.00160$	s.e. = 0.00149	Dockery et al., 1992 E. Tennessee
mortality	803[b]/yr non-accidental (of U.S. pop.)	2-day lag PM_{10}	all in St. Louis, nonaccidental	all	$\beta = 0.00175$	s.e. = 0.00067	Dockery et al., 1992 St. Louis
mortality	803[b]/yr non-accidental (of U.S. pop.)	same-day COH	all in Santa Clara Co., CA, non-accidental	all	$\beta = 0.000924$	s.e. = 0.00034	Fairley, 1990 Santa Clara
mortality	803[b]/yr non-accidental (of U.S. pop.)	same day PM_{10}	all in Chicago, nonaccidental	all	$\beta = 0.000583$	s.e. = 0.000218	Ito et al., 1995 Cook County
mortality	803[b]/yr non-accidental (of U.S. pop.)	same day	all in Los Angeles, nonaccidental	all	$\beta = 0.000488$	s.e. = 0.000266	Kinney, et al 1995 Los Angeles
mortality	803[b]/yr non-accidental (of U.S. pop.)	previous day average TSP	all in Philadelphia, nonaccidental	all	$\beta = 0.000260$ (The model includes TSP, ozone, and SO_2.)	s.e. = 0.000189	Moolgavkar, et al., 1995 Philadelphia
mortality	803[b]/yr emergency (of U.S. pop.)	average of previous 3 days PM_{10}	all in Santiago, nonaccidental	all	$\beta = 0.000784$	s.e. = 0.000197	Ostro et al, 1995 Santiago, Chile

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas. from Original Study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
mortality	803[b]/yr non-accidental (of U.S. pop.)	5-day moving average PM_{10} levels	all in Utah Co., nonaccidental	all	$\beta = 0.001470$	s.e. = 0.00031	Pope et al., 1992 Utah
mortality	803[b]/yr non-accidental (of U.S. pop.)	previous day TSP	all in Steubenville, nonaccidental	all	$\beta = 0.000556$	s.e. = 0.000237	Schwartz and Dockery, 1992a Steubenville
mortality	803[b]/yr non-accidental (of U.S. pop.)	48-hr avg. TSP	all in Philadelphia, nonaccidental	all	$\beta = 0.00082$	s.e. = 0.000293	Schwartz and Dockery, 1992b Philadelphia
mortality	803[b]/yr non-accidental (of U.S. pop.)	previous day TSP	all in Detroit, non- accidental	all	$\beta = 0.000952$	s.e. = 0.000298	Schwartz, 1991 Detroit
mortality	803[b]/yr non-accidental (of U.S. pop.)	mean of current and previous day TSP	all in Minneapolis /St. Paul, nonaccidental	all	$\beta = 0.000972$	s.e. = 0.000278	Schwartz, 1991/1992 Minneapolis
mortality	803[b]/yr non-accidental (of U.S. pop.)	average of previous 3 days PM_{10}	all in Birmingham, nonaccidental	all	$\beta = 0.001044$	s.e. = 0.000415	Schwartz, 1993a Birmingham
mortality	803[b]/yr non-accidental (of U.S. pop.)	3-day mean PM_{10}	all in Chicago, nonaccidental	all	$\beta = 8.0 \times 10^{-4}$ (no copollutants in model)	s.e. = 4.0×10^{-4}	Styer et al., 1995 Chicago
mortality (long-term exposure)	864[b]/yr total mortality (of U.S. pop.)	annual mean PM_{10}	6 cities, all deaths	all	$\beta = 0.00845$	s.e. = 0.00284	Dockery et al., 1993 6 cities cohort
mortality (long-term exposure)	864[b]/yr total mortality (of U.S. pop.)	annual mean PM_{10}	151 cities, all deaths	all	$\beta = 0.003572$	s.e. = 0.0008326	Pope et al., 1995 American Cancer Society cohort

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas. from Original Study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
hospital admissions--all resp. illnesses (ICD 460-519)	504[c]/year (incidence in pop. > 65 years of total U.S. pop.)	same day PM_{10}	65 and older in New Haven, CT, Tacoma, WA	65 and older	New Haven: 0.00172 Tacoma: 0.00227 average: 0.0020	c.i. = New Haven: 1.00-1.12 s.e. = 0.00093 Tacoma: 0.97-1.29 s.e. = 0.00146	Schwartz, 1995 New Haven and Tacoma
hospital admissions -- all resp. illnesses (ICD 460-519)	n/a	mean monthly PM_{10}	variety of ages in Salt Lake Valley, Utah	all	Δ cases = β * ΔPM_{10} * Pop. where β = 0.8047 monthly admissions / Salt Lake Valley population (780,000). = 3.4×10^{-6} (converted from monthly to daily admissions)	s.e. = 0.28	Pope, 1991 Salt Lake Valley
daily respiratory admissions (total) includes 466, 480, 481, 482, 485, 490, 491, 492, 493	n/a	same-day PM-10	Toronto metro area	all	Δ cases = β * Δ PM-10 * Pop where β = 0.0339 daily admissions / Toronto population (2.4 million) = 1.4×10^{-6} (model also includes O_3)	s.e. = 0.034/2.4 million = 1.4×10^{-6}	Thurston et al. 1994 Toronto
hospital admissions pneumonia (480-487)	229[c]/year (incidence in pop. > 65 years of total U.S. pop.)	same day PM_{10}	over 65, Birmingham AL	over 65	β = 0.00174	c.i. = 1.07 - 1.32 s.e. = 0.000536	Schwartz, 1994a Birmingham
hospital admissions COPD (490-496)	103[c]/year (incidence in pop. > 65 years of total U.S. pop.)	same day PM_{10}	over 65, Birmingham AL	over 65	β = 0.00239	c.i. = 1.08 - 1.50 s.e. = 0.00084	Schwartz, 1994a Birmingham
hospital admissions pneumonia (480-487)	229[c]/year (incidence in pop. > 65 years of total U.S. pop.)	same day PM_{10}	over 65, Detroit	over 65	β = 0.00115	s.e. = 0.00039	Schwartz, 1994b Detroit

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas. from Original Study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
hospital admissions COPD (490-496)	103[e]/year (incidence in pop. > 65 years of total U.S. pop.)	same day PM_{10}	over 65, Detroit	over 65	$\beta = 0.00202$	s.e. = 0.00059	Schwartz, 1994b Detroit
hospital admissions pneumonia (480-487)	229[e]/year (incidence in pop. > 65 years of total U.S. pop.)	same day PM_{10}	65 and over in Mpls	over 65	$\beta = 0.00157$	c.i. = 1.02 - 1.33 s.e. = 0.00068	Schwartz, 1994c Mpls, St. Paul
hospital admissions COPD (490-496)	103[e]/year (incidence in pop. > 65 years of total U.S. pop.)	current and previous day	65 and over in Mpls	over 65	$\beta = 0.00451$	c.i. = 1.20 - 2.06 s.e. = 0.00138	Schwartz, 1994c Mpls, St. Paul
hospital admissions for congestive heart failure (ICD 428)	231[e]/yr (incidence in pop. > 65 years of total U.S. pop.)	avg same and previous day PM_{10}	65 and older in Detroit	65 and older	$\beta = 0.00098$	c.i. = 1.012-1.052 s.e. = 0.00031	Schwartz and Morris, 1995 Detroit
hospital admissions for ischemic heart disease (ICD 410-414)	450[e] (incidence in pop. > 65 years of total U.S. pop.)	24 hr avg PM_{10} same day	65 and older in Detroit	65 and older	$\beta = 0.00056$	c.i. = 1.005-1.032 s.e. = 0.00021	Schwartz and Morris, 1995 Detroit
LRI defined as trouble breathing, dry cough, wheezing	584 (ages 10-12)[d] 2,600 (ages ≤ 12)[d]	same day PM_{10}	10-12 yr old non-symptomatic	12 and under	$\beta = 0.005$	s.e. = 0.002	Pope et al., 1991 Utah
URI defined as runny or stuffy nose, wet cough, burning, aching, or red eyes	1,192 (ages 10-12)[d] 5,307 (ages ≤ 12)[d]	same day PM_{10}	10-12 yr old non-symptomatic	12 and under	$\beta = 0.0036$	s.e. = 0.0015	Pope et al., 1991 Utah

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas. from Original Study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
acute bronchitis (ICD 466)	n/a	PM_{10} annual avg (converted)	10 to 12 year olds	18 and under	$\beta = 0.0330$ $\Delta cases = \frac{P_0(e^{\beta * \Delta PM-10}_1)}{1 - P_0 + P_0(e^{\beta * \Delta PM-10}_1)} * Pop$ P_0 = baseline probability of having bronchitis = 0.065[e]	s.e. = 0.0216	Dockery et al., 1989 6 cities
chronic bronchitis	710/year (of study pop.)	annual mean TSP	Seventh Day Adventists in California	all	$\beta = 0.00512$ convert PM_{10} to TSP: $\Delta\ TSP = \frac{\Delta PM10}{0.56}$ where 0.56 is the specific conversion based on region and initial TSP conc.	not available	Abbey et al., 1993
chronic bronchitis		annual mean PM_{10}	adults	all	Δ cases/day = $(p_1 - p_0)$ * Pop where $p_1 = \frac{1}{1 + e^{-(\ln \frac{p_0}{1 - p_0} + \beta * \Delta O_3)}}$ and p_0 = probability of developing physician-diagnosed chronic bronchitis per individual per day = 0.006 $\beta = 0.012$[f]	s.e. = 0.0043[f]	Schwartz, 1993b

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas. from Original Study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
presence of any of 19 acute respiratory symptoms		24 hour average COH in units/ 100 ft)[g]	adult members of families of elementary school-aged children in Glendora-Covina-Azuza, CA	adults 18-65	Δ $Sympt_{days}$/day = $(p_1 - p_0)$ * Pop where $p_1 = \frac{1}{1 + e^{-(\ln \frac{p_0}{1-p_0} + \beta + \Delta Q_3)}}$ and p_0 = the probability of $Sympt_{days}$ per individual for a 24-hour period in the base case = 0.19 β = 0.00046[h] (Model includes O_3, COH, SO_2)	s.e. = 0.00024[h]	Krupnick et al., 1990
moderate or worse asthma status	n/a	average $PM_{2.5}$ during 9:00 am to 4:00 pm ($\mu g/m^3$)	Denver asthmatics between ages 18 and 70	asthmatic (4%[i] of total pop.)	Δ asthma status = $\beta[\ln(X_1/X_0)]$*Pop where X_0 = PM_{10} concentrations with CAA, X_1 = PM_{10} concentrations without CAA, and β = 0.00038[j] (model includes $PM_{2.5}$ and modeled $PM_{2.5}$ measures for periods where $PM_{2.5}$ measures were missing)	s.e. = 0.00019	Ostro et al., 1991 Denver
Restricted Activity Days (RADs)	400,531 days/year[k](of the total U.S. pop)	2-wk average $PM_{2.5}$ ($\mu g/m^3$)	All adults 18-65 in metropolitan areas in the U.S.	adults aged 18-65	Δ health effects determined over a 2 wk period β = 0.0030[j, l]	s.e. = 0.00018[l]	Ostro, 1987
respiratory and nonrespiratory conditions resulting in a minor restricted activity day (MRAD)	780,000 days/year (cited as 7.68 days per person per year in study)	$PM_{2.5}$ averaged over a 2-wk period	employed adults across the U.S. between the ages of 18-65	adults aged 18-65	number of health effects determined over a 2-week period β = 0.00463[j, l] (Model includes fine particulates and O_3)	s.e. = 0.00044[l]	Ostro and Rothschild, 1989

Health Endpoint (ICD-9 code)	Baseline Incidence (per 100,000)	Expos Meas. from Original Study	Study Pop.	Applied Pop.	Functional form[a]	Uncert & Var.	Sources
respiratory restricted activity days (RRADs)	306,000 days/year (cited as 3.06 days per person per year in study)	$PM_{2.5}$ averaged over a 2 wk period	employed adults across the U.S. between the ages of 18-65	adults aged 18-65	number of health effects determined over a 2-wk period $\beta = 0.00936^{j,l}$ (Model includes fine particulates and O_3)	s.e. = 0.00103[l]	Ostro and Rothschild, 1989
Work Loss Days (WLDs)	150,750[m] (of total U.S. pop)	2-wk average $PM_{2.5}$ ($\mu g/m^3$)	All adults 18-65 in metropolitan areas in the U.S.	adults aged 18-65	Δ health effects determined over a 2 wk period $\beta = 0.0029^{j,l}$	s.e. = 0.00022[l]	Ostro, 1987
Household Soiling Damage (change in dollar valuation)	–	annual mean PM_{10}	households in 20 metropolitan areas and in the Philadelphia area	all house-holds	Soiling Damage = β * Pop/PPH * ΔPM_{10} where β = \$2.52 PPH = people per household (2.68)[a]	Beta distribution with mean = \$2.52 s.e. = \$1.00 interval = [\$1.26 - \$10.08] slope parameters: α = 1.2, β = 7.3	Manuel et al. (1982); McClelland, et al. (1994); Watson and Jaksch (1982); ESEERCO (1994)

NOTES:

[a] Pollutant coefficient is for a change in health effects per change in $\mu g/m^3$ PM-10.

[b] Mortality data for general U.S. population taken from Vital Statistics of the United States, Vol. II - Mortality, Part B, pgs. 198-200 (U.S. Dept. of Health and Human Services). Incidence rates were generated for total mortality and mortality excluding accidental deaths and adverse effects, suicide, homicide, and other external causes (ICD E800-E999). Rates calculated based on 1990 population (City and County Databook, 12th edition, 1994, U.S. Dept. of Commerce, Bureau of the Census, Washington, D.C.).

[c] Centers for Disease Control, 1992. Vital and Health Statistics, Detailed Diagnoses and Procedures, National Hospital Discharge Survey, 1990. Number of 1990 discharges divided by 1990 U.S. population (248,709,873) from City and County Databook, 12th edition, 1994, U.S. Dept. of Commerce, Bureau of the Census, Washington, D.C.

[d] Pope et al., 1991 NOTE: rates were not available from standard incidence sources and so were calculated from incidence in the study of 10-12 year olds. This may not be entirely appropriate for older or younger individuals. Children of this age are less likely to have colds than much younger children and may be more representative of the adult population.

[e] Dockery et al., 1989.

[f] β and s.e. estimated from the reported odds ratio and 95th percentile values for the odds ratios.

[g] Coefficient and standard error are converted from a β and s.e. for COH to a β and s.e. for PM_{10}. This was done by using a ratio of COH to TSP of 0.116 from the study authors (as cited in ESEERCO, 1994) and a ratio of PM_{10} to TSP of 0.55 (U.S. EPA, 1986).

[h] Coefficient and standard error incorporate the stationary probabilities as described in Krupnick et al. (1990). To do this, the calculation used the transitional probabilities supplied by the authors and presented in ESEERCO, 1994.

[i] U.S. EPA, 1994a.

[j] β converted from a change in health effects per change in $\mu g/m^3$ $PM_{2.5}$ to a change per $\mu g/m^3$ PM_{10} using the following relationship: 1 $\mu g/m^3$ PM_{10} = 0.625 $\mu g/m^3$ $PM_{2.5}$ (ESEERCO, 1994)

[k] Number of RADs for all acute conditions from: National Center for Health Statistics. Current Estimates from the National Health Interview Survey: United States, 1990. (Hyattsville, MD). This number is divided by the U.S. population for 50 states for 1990 (248,709,873) and multiplied by 100,000 (to obtain the incidence per 100,000).

[1] Based on fixed-weight meta-analysis of single-year coefficients and standard errors reported in study.

[2] Number of WLDs of 374,933,000 from: National Center for Health Statistics. Current Estimates from the National Health Interview Survey from 1990. (Hyattsville, MD). Series 10; No. 181. This number is divided by the U.S. population for 50 states for 1990 (248,709,873) and multiplied by 100,000 (to obtain the incidence per 100,000).

[3] 1990 Census

[4] Selection of distribution type and parameters is detailed in appendix documenting economic valuation of affects.

Nitrogen Oxides

Nitrogen dioxide (NO_2) is the primary focus of health studies on the nitrogen oxides and serves as the basis for this analysis. The primary pathophysiology of NO_2 in humans involves the respiratory system and the concentration-response function identified for NO_2 describes the relationships between measures of NO_2 and respiratory illness.

A number of epidemiological studies are available; however, most have either confounded exposures (with other pollutants) or insufficient exposure quantification (e.g., exposure assessment indicates only absence or presence of a gas stove). Most studies consider NO_2 generated by gas stoves or other combustion sources in homes. Most studies are not directly usable in concentration-response functions. However, studies by Melia et al, 1980 and Hasselblad et al, 1992 provide a reasonable basis for development of a concentration response function. Table 72 presents the function obtained from their work. The function relates NO_2 to respiratory illness in children.

Applied population. The only concentration-response function readily adaptable to this analysis measured respiratory effects in children aged six and seven. The current analysis applies the concentration-response function to all ages, based on the likelihood that adults are also at risk. Epidemiological studies demonstrate the susceptibility of adults to other pollutants (see PM and ozone analyses) and adult respiratory systems do not differ substantially from those of children. In addition, adults, especially the elderly, have illnesses which may make them more susceptible to effects of respiratory damage. Consequently, if adults were not included in this analysis, serious undercounting would be likely to occur. On the other hand, many of the studies which evaluated very young children in depth have obtained results indicating that the youngest age cohort are not at risk (Melia, et al, 1982 and 1983; Samet et al, 1993); the specific age cut-offs vary from one study to another. However, some studies report positive associations in young children (Ware et al, 1984). Thus, the most inclusive alternative involves evaluating risks to the entire population, applying the notion that all individuals, even very young children and older adults, may be susceptible to nitrogen oxides.

Table 72. Summary of Dose Response Functions for NO_2

Health Endpoint	Exposure Measure from Original Studies	Study Population	Applied Population	Functional Form[a,b]	Uncertainty/ Variab.	Sources
respiratory illness (as indicated by respiratory symptoms)	NO_2 measurements in bedrooms with Palmes tubes (one year time weighted average concentration in $\mu g/m^3$)	children ages 6 to 7	all (combining functions for men and women)	$$logodds\ Resp = -0.536 + 0.0275\ NO_2 - 0.0295\ gender$$ where: gender = 1 for boys and 0 for girls (the term drops out for girls) $$Prob(cases) = \frac{1}{1 + e^{-logodds}}$$ where: Prob(resp) = probability of respiratory illness during a one year period $$\Delta Resp\ cases = \Delta Prob(Resp) * Pop$$	s.e. = 0.0132	Hasselblad, et al., 1992.

NOTES:

[a] This equation was obtained from two sources. The NO_2 coefficient was reported in Hasselblad et al., 1992. The background and gender intercepts were obtained via personal communication with V. Hasselblad 2/28/95 by Abt Associates. The equation was based on an evaluation by Hasselblad et al. of study results obtained by Melia et al. (1980). See text for further discussion.

[b] Pollutant coefficient expressed as a change in health effects per change in ppb NO_2. The coefficient for NO_2 was converted based on the following relationship: 1 $\mu g/m^3$ NO_2 = 0.53 ppb NO_2.

Pollutants Using Dose-response Information from Laboratory Studies

Carbon Monoxide

Three dose-response relationships are available for estimating the health effects of carbon monoxide. The first relates ambient CO levels with hospital admissions for congestive heart failure (Morris et al., 1995). The second equation (Allred et al., 1989a,b, 1991) relates the CO level in the bloodstream to the relative change in time of onset to angina pain upon exertion. The third relates the CO level in the bloodstream to the relative change in time of onset to silent ischemia. Due to the lack of quantitative information relating silent ischemia to a meaningful physical health effect, this analysis uses only the first two dose-response functions.

Table 73 presents these functions. The Morris et al (1995) function represents a straightforward application of a linear function. The relationship presented in Allred et al. (1989a,b,; 1991) requires a relationship between CO exposure (in ppm) to %COHb (the measure of CO in the bloodstream used by Allred et al.). The document *Air Quality Criteria for Carbon Monoxide* presents such a relationship graphically (U.S. EPA, 1991). Relationships for 1-hour and 8-hour exposure times are available for two levels of exertion, at rest (alveolar ventilation rate of 10 L/min) and with light exercise (20 L/min). These functions assume an initial COHb of 0.5 percent and standard physical characteristics for the exposed individual. The current analysis presents the linear functions relating 1-hour CO exposure during light exercise and 8-hour CO exposure at rest to COHb levels.

The analysis requires the following additional data in order to characterize the health effects of CO exposure for angina patients:

- nationwide population of angina patients,
- frequency of angina attacks for individual patients, and
- baseline time to onset of angina from beginning of exertion.

The nationwide population of angina patients was 3,080,000 in 1989 (the American Heart Association, 1991). Data on the frequency of angina attacks or baseline time to onset of angina could not be located; therefore, data in Allred et al. (1991) were used. Allred et al. (1991) give the mean frequency of angina attacks for the study population as 4.6 per week with a range of 0 to 63. The baseline time to onset of angina on beginning of the treadmill test was 515 seconds at a COHb level of 0.63 percent. This number was obtained by taking the mean of the mean times to onset for four treadmill tests: three pre-exposure tests and one post-exposure test in which the exposure was to background laboratory air in a control chamber. The treadmill test used a modified Naughton protocol in which individuals were subjected to an increase of 1 basal metabolic equivalent every two minutes (Naughton and Haider, as cited by Allred et al., 1991).

The study population of Allred et al. (1989a,b, 1991) was 63 males, age 35-75, with stable angina. Only nonsmokers who had quit at least three months before the study were included, but most of the individuals had a history of smoking. Because active smokers have chronically high COHb levels (Allred et al., 1989a), the dose-response relationship used in this analysis can only be applied to nonsmokers. This analysis assumes that all angina patients are nonsmokers, although they may have had a history of smoking. In addition, the analysis assumes that the dose-response relationship is accurate for all male and female angina patients.

Table 73. Summary of Dose Response Functions for Carbon Monoxide

Health Endpoint	Baseline Incidence	Exposure Measure	Study Population	Applied Population	Functional Form	Uncert./ Variability	Sources
Hospital admiss. for congestive heart failure	n/a	average of hourly max CO (ppm)	Medicare population in 7 large U.S. cities (96% of which are ≥65)	65 and over	$\Delta cases = \beta * \Delta CO * Pop$ where $\beta = 1.1 \times 10^{-7}$	s.e. = 1.9×10^{-8}	Morris et al., 1995 7 large U.S. cities
percent change in time to angina	baseline time to onset of angina during treadmill test from Allred et al. studies = 515 seconds at %COHb = 0.63[a]	CO (in ppm) averaged over 1 or 8 hours	men, age 35-75 years, stable angina, nonsmokers (of at least 3 months) at time of study	Angina patients in U.S. = 3,080,000 in 1989[e] Frequency of angina attacks for the study population = 4.6 per week (range = 0 - 63)[d]	percent change in time to angina = $B*\Delta$%COHb where: β = -1.89% and COHb = blood level of carboxyhemoglobin and Δ %COHb = 0.45 * ΔCO^e, where: CO = concentration of CO (ppm), for non-smoking adults undertaking light exercise (alveolar ventilation rate of 20L/min) for one hour at low altitude, with an initial COHb = 0.5%. or Δ %COHb = 0.12 * ΔCO^e, where: conditions are the same as above except that study individuals are at rest (alveolar ventilation rate of 10L/min) for 8 hours.	s.e. = 0.81%	Allred, et al., 1989a,b, 1991

NOTES:
[a] Calculated as the mean of means from 3 pre-exposure treadmill tests and 1 post-exposure test (control exposure to air) (Allred et al., 1991).
[b] American Heart Association (1991)
[c] Allred et al. (1991)
[d] Multiple daily events are not modeled. Although it is possible that angina attacks may occur more than once per day, the average frequency of attacks was 4.6 per week (< 1 per day).
[e] Equation calculated from figure in U.S. EPA (1991), p. 2-7.

Sulfur Dioxide

This analysis estimated one concentration-response function based on an epidemiological study of premature mortality (Moolgavkar et al., 1995). A second function was estimated using the clinical data on the responses of exercising asthmatics to SO_2, as measured by the occurrence of respiratory symptoms. The table that follows summarizes the derivation of these functions. The procedure for estimating a concentration-response function for respiratory symptoms is detailed below.

Symptoms reported for individual subjects are available from two studies (Roger et al., 1985 and Linn et al., 1987). Using the individual subject data, the percentage of subjects experiencing at least moderate symptoms as a result of exposure to SO_2 while exercising can be estimated.

All types of symptoms (chest tightness, shortness of breath, or wheeze) were combined; that is, if a subject had any of these symptoms, he/she was recorded as having a symptom response; if the subject had more than one of these responses, the highest of the symptom scores was used in the analysis.

A multiple regression was performed with the log odds of experiencing at least noticeable symptoms as the dependent variable, and SO_2 exposure level, in ppm and asthma status as the independent variables. A similar regression was performed for log odds of experiencing at least moderate symptoms. To calculate cases from the log odds regression, the log odds are first transformed back to probability:

$$Prob(symp) = \frac{e^{logodds}}{1 + e^{logodds}} \qquad (24)$$

The probability for mild and moderate asthmatics is calculated separately using this equation, and using a value of 0 for mild and 1 for moderate for the status variable. Finally, these probabilities are multiplied by the size of the applicable asthmatic population:

$$Cases = Prob_{mild}(symp) \cdot Pop_{mild} + Prob_{mod}(symp) \cdot Pop_{mod} \qquad (25)$$

where Pop_{mild} = exposed population of exercising mild asthmatics, Pop_{mod} = exposed population of exercising moderate asthmatics and Cases = number of occurrences of at least moderate symptoms.

The size of the relevant exposed population is also needed. The number of asthmatics in the population is estimated to be four percent (US EPA, 1994a). The percent of asthmatics exercising at a given time is uncertain. US EPA (1994a) presents several statistics of studies on both the general population and specific asthmatic populations regarding the percent of time devoted to exercise. The value used in the US EPA (1994a) analysis was 1.7 percent of waking hours, based on a study of the general population. Estimates from studies specifically on asthmatics range from 0.2 to 3.3 percent. We use the point estimate of 1.7 percent and the range of values as the probability of a person in the asthmatic population exercising during a given hour. An additional piece of information is needed regarding the relative proportions of mild and moderate asthmatics in the population. This analysis assumes that two-thirds of the asthmatic population has a mild form of the disease, and one-third are moderate asthmatics.

Table 74. Summary of Dose Response Functions for Sulfur Dioxide

Health Endpoint	Expos Meas. from original study	Study Pop.	Applied Pop.	Functional Form[a]	Uncert & Var.	Sources
mortality	previous day 24-hour SO_2 concentration	Philadelphia	all	$\Delta cases = cases * (e^{\beta * \Delta SO_2} - 1)$ $\beta = 0.001328$	s.e. = 0.000255	Moolgavkar et al., 1995 Philadelphia
Any Symptom (chest tightness, shortness of breath, etc.)	5-minute SO_2 concentration, ppm (using peak to mean ratio from hourly SO_2 concentration of 2:1 to 3:1)	generally young exercising asthmatics (ventilation rate 0.4 m^3/min)	exercising asthmatics - defined as 4% of general population, of whom 1.7% (range 0.2% to 3.3%) are exercising during waking hours	$logodds\ Symp = -5.65 + 0.0059\ SO_2 + 1.10\ status$ where *status* = asthma status (0 for mild, 1 for moderate) $Prob(symp) = \frac{1}{1 + e^{-logodds}}$ $Cases = Prob_{mild}(effects) \cdot Pop_{mild} + Prob_{mod}(effects) \cdot Pop_{mod}$ Cases = number of individuals with occurrences of at least moderate effects for all three measures. where Pop_{mild} = exposed population of exercising mild asthmatics (assumed to be 2/3 of asthmatic population); Pop_{mod} = exposed population of exercising moderate asthmatics (assumed to be 1/3 of asthmatic population)	s.e. for: const. term= 2.60 for SO_2 coeff = 0.0025 for *status* coeff = 1.44	data from Linn *et al.*(1987, 1988, 1990), Roger et al. (1985)

NOTES:
[a]Pollutant coefficients in all functions are expressed as a change in health effect per change in ppb SO_2

Visibility: Selection and Use of Concentration-Response Functions

This analysis estimates benefits due to improved visibility in two ways. First, the change in visibility, measured in deciview, captures the physical change in atmospheric conditions attributable to emissions reductions achieved under the CAA. Second, the direct economic valuation of monetary benefits translates the estimated visibility improvement to dollars based on a willingness to pay estimate for a unit change in visibility.

Deciview represents a visibility measure useful for comparing the effects of air quality on visibility across a range of geographic locations for a range of time periods. It is directly related to two other common visibility measures, visual range (measured in km) and light extinction (measured in km^{-1}); however, it characterizes visibility in terms of perceptible changes independent of baseline conditions. As noted in Table 75, two separate sources provided visibility data, one for the eastern U.S., and one for the southwestern U.S. In the east, the Regional Acid Deposition Model (RADM) generated extinction coefficient estimates for each of 1,330 grid cells in the RADM domain (essentially the eastern half of the country). A conversion was necessary to translate the extinction coefficient into deciview terms. In the western U.S., visibility conditions, measured in deciview terms, were modeled using a linear rollback model and a conventional extinction budget approach (SAI, 1994).

Physical benefits in terms of visibility improvements are measured in terms of a population-weighted deciview change. These benefits are computed separately for the east and west. Monetary benefits are likewise computed for both regions of the country.

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Table 75. Summary of Concentration-Response Functions for Visibility[a]

Endpoint	Expos Meas. From Original Study	Applied Pop.	Functional Form	Uncert & Var.	Sources
Visibility (change in deciview)[b]	Eastern U.S.: Extinction coefficient (Ext) in units of m^{-1} Western U.S.: deciview, dv (unitless)	all	$\Delta\ Visibility = (deciview_{No\text{-}CAA} - deciview_{CAA}) * Pop$ to convert to deciview in East, $deciview = 10 \ln\left(\frac{Ext}{0.01\ km^{-1}}\right)$ and Ext units are converted from m^{-1} to km^{-1}		Pitchford and Malm (1994)
Visibility (direct economic valuation)	Eastern U.S.: Extinction coefficient (Ext) in units of m^{-1} Western U.S.: deciview, dv (unitless)	all	$Benefit = WTP_{HH} * \frac{Pop}{no.\ per\ HH} * \ln\left(\frac{VR_{CAA}}{VR_{No\text{-}CAA}}\right)$ where, VR = visual range (km), which can be translated to deciview: $deciview = 10 \ln\left(\frac{391\ km}{VR}\right)$ and, WTP_{HH} =willingness to pay per household: = \$149/change in VR (East) = \$117/change in VR (West) no. per HH = people per household (2.68 according to '90 Census)	WTP_{HH} triangular distribution: most likely WTP_{HH}=\$149, min WTP_{HH} = \$117, max WTP_{HH} = \$299. Values between the most likely value and each of the extreme values are sampled with decreasing probability the closer to the extremes. Visual Range Constant The visual range constant, used in the deciview conversion, is assumed to vary uniformly between 230 km and 391 km.	Pitchford and Malm (1994)

NOTES:

[a] Visibility is measured in two ways: (1) in terms of extinction coefficient in the eastern U.S. (based on modeling of RADM domain); and (2) as deciview (dv) in the west (modeling of 30 western cities) (SAI, 1994).

[b] Deciview is a haziness index used to characterize visibility through uniform hazes.

Health Effects Model

Section 812 of the 1990 Clean Air Act Amendments charged EPA with assessing the costs and benefits attributable to the original 1970 Clean Air Act (CAA) and its subsequent amendments. In order to prepare such an assessment, EPA identified and estimated the quantifiable health and welfare benefits enjoyed by Americans due to improved air quality resulting from the CAA. Health benefits resulted from an avoidance of air pollution-related health effects, such as mortality, respiratory illness, and heart disease. Welfare benefits accrued where improved air quality averted damage to measurable resources, for example agricultural production (these are described in Appendix F) and visibility. This section describes the analysis of health effects in terms of its three major components:

- Air Quality - estimated changes in exposure concentrations of criteria air pollutants between the factual scenario with the CAA controls and the hypothetical "no-control" scenario without the CAA. Air quality improvements resulting from the Act were evaluated in terms of both their temporal distribution from 1970 to 1990 and as their spatial distribution across the United States.

- Population Distribution - determined the population exposed to the different levels of air quality improvement. This represents a necessary step towards quantifying the change in health effects attributable to CAA control.

- Health and Welfare Effects - quantified the relationship between exposure to a given level of air pollution and the physical health and welfare effects associated with such exposure.

Results of most of the analyses were generated in terms of both point estimates of benefits (avoided incidences and corresponding monetary values) as well as a probability distribution of these estimates based on Monte Carlo uncertainty analysis. The following sections describe the three elements of the benefits analysis in more detail.

Air Quality

The first step towards quantifying benefits attributable to the CAA involved characterizing air quality improvements for the following pollutants:

- Particulate matter, less than 10 microns in diameter (PM_{10})
- Ozone (O_3)
- Nitrogen dioxide (NO_2)
- Sulfur dioxide (SO_2)
- Carbon monoxide (CO)
- Light extinction[113]
- Lead (Pb)

[113] While light extinction is not a criteria air pollutant, it provides an important measure of a significant welfare effect resulting from air pollution, visibility degradation. Light extinction results from light scattered by fine particles in the atmosphere, especially sulfates and ammonium nitrates. As atmospheric concentrations of such particles increases, light is attenuated and visible conditions decrease.

Generally, air pollution monitoring data provided baseline ambient air quality levels for the factual control scenario. Air quality modeling was used to generate estimated ambient concentrations for the hypothetical (no-control) scenario. With the exception of lead data, air quality modeling data were provided by EPA's Office of Air and Radiation. A variety of modeling techniques were applied, depending on the pollutant modeled, requiring different applications of the air quality data to the benefits analysis, as described below.

Data Coverage

PM_{10}

PM_{10} air quality estimates were provided at the county level, synthesizing historical ambient measurement data collected throughout each monitored county. Annual data for each of the two scenarios (control and no-control) were provided from 1970 to 1990. These air quality estimates were comprised of daily averages and annual means for each monitored county (measured in $\mu g/m^3$). In lieu of modeling daily air quality data for each monitor nationwide over two decades, annual concentration profiles were reduced to frequency distributions. That is, the daily averages for a given year were reported as a single distribution, which, for PM_{10}, was summarized using the pollutant levels at every fifth percentile of that distribution. While these descriptive summaries limited analysis of air quality changes to the single day level (i.e., it was not possible to determine pollutant concentrations for sequences of days), they eased the computational burden and data storage requirements tremendously.

Two different measures of ambient concentrations of particulate matter were used in the United States during the period 1970 to 1990. Prior to 1987, the indicator for the National Ambient Air Quality Standard for PM was total suspended particulates (TSP). In 1987, the indicator was changed to PM_{10} (particles less than 10 μM in diameter). Widespread PM_{10} monitoring did not begin until 1985; prior to that only TSP data is available. Because the majority of the PM concentration response functions used in this analysis are in terms of PM_{10}, PM_{10} data is preferred, if available. Where only TSP is available, PM_{10} concentrations were estimated using PM_{10}:TSP ratios that vary by area of the country and for urban/rural adjustments.

In order to compute air quality improvements relative to PM_{10}, differences were taken between the distributions for the control and no-control scenarios at the 20 intervals provided. The result was an approximation of the distribution of PM_{10} concentration changes at the given monitor, reported as 20 concentrations along the probability density function.

In addition to modeling PM_{10} concentrations in counties for which ambient monitoring data was available, EPA also modeled concentrations in unmonitored counties using large-scale regional models. Extending the coverage of PM_{10} monitors to counties beyond the monitor network decreased the confidence associated with the estimates but at the same time extended PM_{10} exposure estimates to a significantly greater proportion of the U.S. population. This issue is discussed further in below.

Ozone, NO_2, SO_2, and CO

Ozone, NO_2, SO_2, and CO concentrations for both the controlled and uncontrolled scenarios were provided for the locations of air quality monitors. Throughout the U.S., measured ambient concentrations and modeled no-control concentrations were reported for a variety of averaging times ranging from one-hour to yearly. With the exception of ozone, estimates of pollutant concentrations for these monitor-level pollutants were provided for 1970, 1975, 1980, 1985, and 1990. Ambient ozone concentrations for both scenarios were provided at each monitor annually from 1970 through 1990.

Similar to PM_{10}, profiles of measured and modeled concentrations at each monitor were reduced to descriptive statistical summaries of the concentration distributions. From the original concentration profile for a given monitor during a given year, distributions were generated for a variety of useful averaging times (e.g., 1-hour, 8-hour, 12-hour, daily). For each averaging time, a frequency distribution of pollutant concentrations was developed. A statistical function was fit to the distribution which was then summarized by two or three fitting parameters (typically the moments of the distribution). For the pollutants described here, the distributional form used was almost always the gamma (one was log normal), which is commonly used to approximate air quality observations. Two parameters are used to describe the gamma distribution, reducing the amount of data required to describe a year's worth of hour concentrations to two values. From these parameters, the approximate distributions can be rebuilt to estimate the changes in health or welfare effects.

For most analyses, translating the reported fitting parameters into air quality changes attributable to the CAA required a transformation of the profile parameters for each of the two scenarios (control and no-control) back into probability density functions. Similar to PM_{10}, the difference between the two distributions was evaluated at five percent intervals, yielding a distribution of concentration changes resulting directly from enactment of the CAA. For the agricultural welfare analysis, an inverse cumulative density function was used to calculate representative ozone concentrations for each hour from the gamma distributions. From these data, an long-term ozone index that can be used in estimating plant response was calculated. Because ozone concentrations during a crop's growing season is used to determine crop yield loss, and the growing seasons vary by crop and region, the agricultural analysis used separate ozone profiles in two month blocks throughout the year.

Light Extinction

Extinction data was provided for 60 kilometer by 60 kilometer gridcells modeled by EPA's Regional Acid Deposition Model (RADM). Extinction coefficients, measured in 1/meters, were reported at five year increments from 1970 to 1990. Only annual mean extinction values for each of the two scenarios are used to estimate the benefits based on changes in visual range.

Lead

Determining the impacts of the CAA on atmospheric lead concentrations was handled differently than the other pollutants. In contrast to using ambient measurements for the control case and modeled concentrations for the no-control case, CAA effects on lead were treated in two ways, depending on the source of lead emissions. Emissions from lead in gasoline (essentially an area source due to its former widespread use in motor fuels) were treated differently than point sources of atmospheric lead. EPA estimated the lead content of gasoline, by far the dominant source of lead emissions, for the period 1974 to 1990. Note that in this case emissions estimates (based on gasoline content) rather than air quality data were used to measure the impact of CAA control. Leaded gasoline was assumed to have remained in use in the no-control scenario. Lead emissions from point sources (combustion sources, electric utilities, and industrial processes and boilers) were obtained from a variety of EPA sources for 1990. Growth and emissions factors were used to estimate emissions back to earlier years. Air dispersion modeling of each point source was used to quantify ambient concentrations in the vicinity of these sources over the 20 year period of the study.

With respect to the distribution of air quality data across the two decades considered, it should be noted that both the number and location of monitors tracking air quality changed over time. Table 76 depicts the number of monitors for each pollutant across the period of this analysis. The number of monitors generally increased throughout the 1970s and leveled off or declined at varying points during the 1980s, depending on the pollutant.

Table 76. Criteria Air Pollutant Monitors in the U.S., 1970 - 1990.

	Pollutant				
Year	PM_{10}	O_3	NO_2	SO_2	CO
1970	245	1	43	86	82
1975	1,120	321	303	827	494
1980	1,131	546	375	1,088	511
1985	970	527	308	916	458
1990	781	627	345	753	493

Population Distribution

Health and some welfare benefits resulting from air quality improvements are distributed to individuals in proportion to the reduction in exposure each enjoys. Predicting individual exposures, then, is a necessary step in estimating health effects. Doing so for the Section 812 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. Described below is the method used to allocate U.S. Census data to a symmetrical grid overlying the country.

Census Data

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 during intermediate years 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.

Gridding U.S. Population

To ease computational burden, block group population estimates were aggregated to a rectangular grid structure. The grid, comprised of ten kilometer by ten kilometer gridcells, spanned the entire area of the contentional United States, plus Alaska and Hawaii. This grid size generated 46,885 populated gridcells throughout the U.S.

The entire population of each block group was assumed to reside at the geographical centroid of the block group area, the coordinates of which were available from the U.S. Bureau of the Census. Block group populations were aggregated to gridcells according to the block group centroids encompassed by each cell. In addition to the population of each gridcell, the state and county names for each gridcell were retained, permitting alternative aggregations of data (i.e., other than nationwide).

Allocating Exposure Estimates to the Population

Two alternative modeling strategies were used to allocate air quality improvements to the U.S. population. They differed both in terms of the certainty of the estimates as well as the geographic coverage:

(1) Air quality improvements (difference between control and no-control scenarios) were applied to individuals living in the vicinity of air quality monitors. For pollutants with monitor-level data, it was assumed that the individuals in a gridcell were exposed to air quality changes estimated at the nearest monitor if it was within 50 kilometers. Likewise, for PM_{10} (for which data was available at the county level) the population of each monitored county was assumed to be exposed to the air quality changes reported for that county.[114] The remainder of the population was excluded from the analysis.

Unfortunately, by limiting the quantitative analysis to populations within 50 km (or within a monitored county, for PM), a significant portion of the U.S. population was left out of the analysis (see Table 77). For most pollutants in most years (excepting lead), less than ¾ of the population lived within 50 km of a monitor (or within a PM-monitored county). Clearly, an analysis that excluded 25 percent of the population from the benefits calculations (thus implicitly assuming that the CAA had no impact on that population) would understate the physical effects of the CAA. Conversely, ascribing air pollution reduction benefits to persons living great distances from air quality monitors is a somewhat speculative exercise, and could overstate benefits.

Table 77. Population Coverage in the "Within 50 km" Model Runs (percent of continental U.S. population).

	1975	1980	1985	1990
CO	67.4%	67.9%	68.4%	70.4%
EXT	73.2%	72.3%	72.3%	72.2%
NO_2	53.3%	58.8%	60.8%	61.5%
O_3	55.5%	70.5%	71.5%	74.4%
PM_{10}	78.5%	79.5%	75.8%	67.8%
SO_2	64.7%	73.3%	73.0%	70.6%
Pb	100%	100%	100%	100%

(2) Air quality improvements were applied to almost all individuals nationwide. Where monitor data was not available within 50 kilometers, data from the closest monitor, regardless of distance, was used. Similarly, PM_{10} concentrations were extrapolated using regional air quality models to all counties (even those for which monitoring data was unavailable) and applied to the populations of those counties.

Although subject to less certain air quality data, the second alternative extrapolates pollutant exposure estimates to almost the entire population using the closest monitoring data available (see Table 78).[115]

Table 78. Population Coverage for "Extrapolated to All U.S." Model Runs (percent of continental U.S. population).

	1975	1980	1985	1990
CO	97.2%	97.2%	98.7%	100.0%
EXT	73.6%	74.8%	74.7%	74.7%
NO_2	97.2%	97.2%	98.7%	100.0%
O_3	96.6%	97.2%	98.7%	100.0%
PM_{10}	95.9%	95.8%	97.2%	98.5%
SO_2	95.4%	95.6%	97.0%	98.4%
Pb	100%	100%	100%	100%

[114] Since the lead (Pb) analysis, which was handled separately from that of the other criteria pollutants, did not require air quality modeling data, the issue of proximity to monitors is irrelevant. The Pb analysis extended to 100 percent of the population.

[115] While this alternative captures the vast majority of the U.S. population, it does not model exposure for everyone. To improve computational efficiency, those gridcells with populations less than 1,000 were not modeled; these cells account for less than three percent of the U.S. population.

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. In order to quantify such benefits, it was necessary to identify concentration-response relationships for each effect being considered. As detailed above, 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 contract a disease per unit change in air quality. By evaluating each concentration response function for every gridcell throughout the country, and aggregating the resulting incidence estimates, it was possible to generate national estimates of avoided incidence.

It is impossible to estimate all of the physical effects that would have occurred without the Clean Air Act. While scientific information is available that makes it possible to estimate certain effects, many other, potentially very important, health and welfare effects cannot be estimated at this time. Other physical effects can be quantified, but it is impossible to assess the economic value of those endpoints. Table 79 shows the health and welfare effects for which quantitative analysis has been prepared, as well as some of the health effects that have not been quantified in the analysis.

Health effects for lead were treated slightly differently. Again, gasoline as a source was addressed separately from conventional point sources. Instead of using ambient concentrations of lead resulting from 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 consumed each year was directly related to mean blood lead levels using a relationship presented in the 1985 Regulatory Impact Analysis (U.S. EPA, 1985). Health effects resulting from exposure to point sources of atmospheric lead were considered using the air concentration distributions modeled around point sources. Concentration response functions were used to relate changes in blood lead levels in nearby populations.

Welfare effects were treated in a similar manner to 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 losses, such as reduction in crop losses due to improved ozone conditions. Welfare effects quantified as part of this analysis included household soiling damage by PM_{10}, visibility degradation measured by light extinction, and agricultural benefits measured in terms of net surplus.

Table 79. 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 Emergency room visits Asthma attacks Decreased income resulting from decreased exercise performance Changes in pulmonary function Chronic Sinusitis & Hay Fever Mortality	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 Lower respiratory illness Upper respiratory illness Chest illness Respiratory symptoms Minor restricted activity days All respirsal activity days Days of work loss Moderate or worse asthma status (asthmatics)	Changes in pulmonary function Chronic bronchitis	Chronic respiratory diseases other than chronic bronchitis Inflammation in the lung
Carbon Monoxide	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: Changes in pulmonary function Respiratory symptoms 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

Health Effects Model Results

Tables 81 and 82 summarize the results of the health (and visibility) effects modeling. For some health endpoints, more than one CR function was used, with each being based on a different health effects study. The alternative CR functions provide differing measures of the health effect. These can be used to derive a range of possible results. In the case of lead (Pb), alternative CR functions were not used; rather, two analytical procedures were used (labeled the "backward-looking" and "forward looking" analyses), giving a range of results for most Pb endpoints.

Table 81 presents the results of the "50 km" model run. Observe that, for those health endpoints where the pollutant of concern is PM_{10} only, incidence declines between 1985 and 1990. Only two factors, besides the CR function itself, should affect the model's output: The difference in ambient air quality between the two scenarios, and the affected population. Between 1985 and 1990, difference in air quality between the scenarios increased, and the U.S. population increased. One should see, then, an unambiguous increase in incidence between 1985 and 1990. The decrease in incidence observed here is due exclusively to the decline in population residing in counties with PM_{10} monitors (i.e., due to the decrease in the number of monitors).

Table 78 presents the results of the "all U.S. population" model run (although, with the exception of Pb, not all of the population is modeled, with up to 5 percent being excluded in a given year), and provides a more accurate depiction of the *pattern* of health effects incidence across years. The accuracy of the *scale* of incidence is less certain. These results are almost certainly more accurate than the "50 km" results, but rely on the assumption that (for a portion of the population) distant air quality monitors provide a reasonable estimate of local air quality conditions. Thus, the results presented here are somewhat speculative. It is likely that the predicted health effects are overstated for that population group (20 to 30 percent of total population in the case of PM_{10}) for which distant monitors are used. Conversely, there is an implied zero health impact for that portion of the population (3 to 4 percent in the case of PM_{10}) excluded from the analysis altogether, an understatement of health impacts for that group.

Table 80 presents estimated incidence reductions for several health effects which could be quantified but not monetized for this analysis.

Table 80. Quantified Benefits Which Could Not Be Monetized -- Extrapolated to the Entire 48 State Population.

Endpoint	Study	Pollutant	1975	1980	1985	1990	Units
Pulmonary Function Decrements							
Decreased FEV by 15 % or more	Avol et al. 1984 & Seal et al. 1993	O3	81,689,960	170,396,436	272,636,667	377,201,857	*person-days with decreased FEV (per year)*
Decreased FEV by 20 % or more	Avol et al. 1984 & Seal et al. 1993	O3	61,953,575	126,046,232	199,217,189	273,313,091	*person-days with decreased FEV (per year)*
Chronic Sinusitis and Hay Fever	Portney and Mullahy, 1990	O3	7,286,310	7,788,364	8,268,812	8,753,723	*cases/year*
Time to Onset of Angina Pain	Allred, et al., 1989a,b, 1991	CO	0.2%	0.4%	0.8%	0.9%	*fractional change in time to onset of angina attack*

Table 81. Criteria Pollutants Health Effects -- Population Within 50 KM of a Monitor Modeled (cases per year).

Endpoint	Study	Pollutant	1975	1980	1985	1990
Mortality	Dockery et al., 1992, St. Louis	PM10 & O3	29,554	72,449	18,704	79,841
	Dockery et al., 1992, E. Tenn.	PM10 & O3	25,674	62,374	70,179	64,993
	Pope et al., 1991	PM10	24,391	59,415	68,086	64,464
	Schwartz, 1994a	PM10	17,228	41,488	47,590	45,308
	Schwartz, 1993/1992	PM10	16,885	38,734	44,197	41,997
	Schwartz, 1991	PM10	15,491	37,816	43,238	41,109
	Pope, 1989	PM10	15,224	36,772	41,945	39,907
	Schwartz and Dockery, 1992b	PM10	13,410	32,336	36,858	35,053
	Ryan et al., 1990	PM10	13,180	31,727	36,161	34,393
	Ozkaynak et al., 1995	PM10	12,894	31,079	35,419	33,689
	Xu et al., 1995	PM10	9,564	22,963	26,159	24,907
	Schwartz and Dockery, 1992a	PM10	9,118	21,902	24,974	23,735
	Moolgavkar et al., 1995	PM10, O3, SO2	7,425	22,307	23,714	23,224
	Kinney et al., 1995	PM10 & O3	7,896	19,188	21,876	20,782
	Backward Looking Analysis	Pb	1,645	10,136	19,825	24,679
	Forward Looking Analysis	Pb	1,220	8,263	17,367	17,118
Coronary Heart Disease & Stroke						
Coronary Heart Disease	Backward Looking Analysis	Pb	1,581	9,169	18,475	24,228
	Forward Looking Analysis	Pb	891	6,415	10,863	12,527
Stroke	Backward Looking Analysis	Pb	775	3,889	10,101	12,715
	Forward Looking Analysis	Pb	393	3,107	3,527	6,604
Hospital Admissions						
All Respiratory Admissions	Schwartz, 1995, Toronto	PM10 & O3	35,563	45,787	78,091	74,400
	Pope, 1991, Salt Lake Valley	PM10	25,481	40,398	48,727	45,819
	Schwartz, 1995, New Haven	PM10 & O3	18,619	43,813	53,349	51,758
	Thurston et al. 1994, Toronto	PM10 & O3	11,487	27,577	33,850	32,575
Pneumonia + COPD Admissions	Schwartz, 1994a	PM10 & O3	17,723	44,946	51,540	49,503
	Schwartz 1994b	PM10 & O3	15,863	34,191	39,908	38,472
	Schwartz 1994b	PM10 & O3	10,251	26,126	31,261	31,009
Ischemic Heart Disease	Schwartz and Morris, 1995	PM10	5,147	12,364	14,970	13,398
Congestive Heart Failure	Schwartz and Morris, 1995 & Morris et al. 19	PM10 & O3	6,573	17,651	26,520	30,642
Respiratory Illness or Symptoms						
Children						
Upper + Lower Resp. Illness	Pope et al., 1991	PM10	487,394	1,203,416	2,064,209	1,859,304
Acute Bronchitis	Dockery et al., 1989	PM10	1,318,106	4,153,260	4,894,129	4,406,121
Adults						
Any of 19 Symptoms	Krupnick et al., 1987	PM10 & O3	34,970,757	89,133,915	96,629,545	92,502,530
Any Symptom	Lipfert et al. (1987, 1988, 1989)	SO2	58,392	144,200	127,454	102,673
Increase in Respiratory Illness	Hasselblad, 1992	NO2	881,581	2,280,701	6,806,320	10,448,806
Asthma Attacks	Ostro et al., 1991	PM10	311,408	408,569	228,599	305,440
	Whittemore and Korn, 1980, US EPA, 1982	O3	126,252	326,243	365,564	761,149
Restricted Activity and Work Loss Days						
RAD	Ostro, 1987	PM10	1,118,203	2,794,748	5,259,914	5,055,589
MRAD	Ostro and Rothschild, 1989	PM10 & O3	3,976,101	9,280,059	11,061,088	10,440,697
RRAD	Ostro and Rothschild, 1989	PM10 & O3	2,987,369	11,074,350	16,888,284	8,442,319
Work Loss Days	Ostro, 1987	PM10	458,397	1,015,261	1,174,943	1,150,550
IQ Changes & Hypertension						
Lost IQ Points	Backward Looking Analysis	Pb	1,174,549	5,754,187	9,788,103	11,363,474
	Forward Looking Analysis	Pb	1,172,694	5,752,419	9,786,134	11,361,070
IQ < 70	Backward Looking Analysis	Pb	3,963	27,461	41,935	49,758
	Forward Looking Analysis	Pb	0,894	23,614	32,912	64,078
Hypertension	Backward Looking Analysis	Pb	950,954	6,252,889	11,240,454	13,561,229
	Forward Looking Analysis	Pb	677,094	4,204,720	7,832,922	9,749,688
Decreased Worker Productivity	Ferm, Crocker & Hunt, 1981 and EPA,1994c	O3	(Direct Economic Valuation)			
Visibility (DeciView change - Res)	Pitchford and Malm, 1994	DeciView	0.4	1.4	1.8	2.0
Visibility (DeciView change - West)	Pitchford and Malm, 1994	DeciView	2.4	4.9	5.9	6.0
Household Soiling	BOD/Ragsdale Reilly, 1994	PM10	(Direct Economic Valuation)			

Table 82. Criteria Pollutants Health Effects -- Extrapolated to Entire U.S. Population (cases per year).

Endpoint	Study	Pollutant	1975	1985	1990	1991
Mortality	Dockery et al., 1992, St. Louis	PM10 & O3	36,103	87,148	104,384	115,443
	Dockery et al., 1992, E. Tenn.	PM10 & O3	30,878	74,956	87,753	98,283
	Pope et al., 1992	PM10	29,674	71,482	89,095	93,394
	Schwartz, 1993a	PM10	20,968	50,195	59,827	65,682
	Schwartz, 1991/1992	PM10	19,505	46,643	55,350	61,030
	Schwartz, 1991	PM10	19,109	45,659	54,217	59,741
	Fairley, 1990	PM10	18,532	44,284	52,577	57,939
	Schwartz and Dockery, 1992a	PM10	16,325	38,949	46,218	50,950
	Styer et al., 1995	PM10	14,021	38,217	45,347	49,992
	Ostro et al., 1995	PM10	13,698	37,437	44,418	48,970
	Ito et al., 1995	PM10	11,646	27,698	32,827	36,215
	Schwartz and Dockery, 1992a	PM10	11,103	26,394	31,281	34,512
	Moolgavkar et al., 1995	PM10,O3,SO2	9,992	27,534	30,167	32,241
	Kinney et al., 1995	PM10 & O3	9,737	23,125	27,399	30,235
	Backward-Looking Analysis	Pb	1,643	10,136	19,625	24,679
	Forward-Looking Analysis	Pb	1,220	6,983	12,367	15,118
Coronary Heart Disease & Stroke						
Coronary Heart Disease	Backward-Looking Analysis	Pb	1,381	9,189	18,473	24,228
	Forward-Looking Analysis	Pb	890	5,418	10,063	12,527
Stroke	Backward-Looking Analysis	Pb	773	5,039	10,101	12,715
	Forward-Looking Analysis	Pb	503	3,052	5,477	6,684
Hospital Admissions						
All Respiratory Admissions	Schwartz, 1995, Tacoma	PM10 & O3	31,706	76,852	94,924	106,591
	Pope, 1991, Salt Lake Valley	PM10	30,582	73,093	86,407	95,486
	Schwartz, 1995, New Haven	PM10 & O3	22,891	55,178	66,958	74,516
	Thurston et al, 1994, Toronto	PM10 & O3	14,293	33,330	41,189	46,839
Pneumonia + COPD Admissions	Schwartz, 1994a	PM10 & O3	21,647	53,829	69,093	71,374
	Schwartz 1994a	PM10 & O3	16,996	41,133	49,897	55,412
	Schwartz 1994b	PM10 & O3	13,177	31,371	39,110	44,230
Ischemic Heart Disease	Schwartz and Morris, 1995	PM10	8,267	14,809	17,658	19,482
Congestive Heart Failure	Schwartz and Morris, 1995 & Morris et al. 95	PM10 & CO	8,809	22,364	35,210	42,757
Respiratory Illness or Symptoms						
Children						
Upper + Lower Resp. Illness	Pope et al., 1991	PM10	834,135	2,114,316	2,544,473	2,759,301
Acute Bronchitis	Dockery et al., 1989	PM10	1,719,361	5,310,174	6,639,827	7,266,270
Adults						
Any of 19 Symptoms	Krupnick et al., 1990	PM10 & O3	42,488,222	100,283,786	120,133,130	133,724,647
Any Symptom	Lipfert et al. (1987, 1988, 1989)	SO2	66,166	187,155	165,439	145,584
Increase in Respiratory Illness	Hausfather, 1992	NO2	1,161,543	4,120,706	9,451,855	14,727,594
Asthma Attacks	Ostro et al., 1991	PM10	264,430	548,906	686,953	841,916
	Whittemore and Korn, 1980, EPA, 1985	O3	201,879	430,813	729,900	969,473
Restricted Activity and Work Loss Days						
RAD	Ostro, 1987	PM10	1,336,294	3,355,260	4,024,442	4,306,546
MRAD	Ostro and Rothschild, 1989	PM10 & O3	4,380,900	11,121,528	13,630,345	14,932,954
RRAD	Ostro and Rothschild, 1989	PM10 & O3	3,307,964	14,236,364	12,173,187	12,151,677
Work Loss Days	Ostro, 1987	PM10	493,585	1,217,051	1,459,105	1,594,835
IQ Changes & Hypertension						
Lost IQ Points	Backward-Looking Analysis	Pb	1,174,549	5,734,187	9,788,035	11,863,674
	Forward-Looking Analysis	Pb	1,172,494	5,732,418	9,786,134	11,861,670
IQ < 70	Backward-Looking Analysis	Pb	9,563	27,461	41,935	49,795
	Forward-Looking Analysis	Pb	6,698	32,616	52,912	64,075
Hypertension	Backward-Looking Analysis	Pb	983,994	6,352,089	12,340,604	15,561,239
	Forward-Looking Analysis	Pb	677,084	4,204,930	7,839,852	9,740,608
Decreased Worker Productivity	From Crocker & Horst, 1981 & EPA,1994c	O3	(Direct Economic Valuation)			
Visibility (DeciView change - East)	Pitchford and Malm, 1994	DeciView	0.4	1.4	1.8	2.0
Visibility (DeciView change - West)	Pitchford and Malm, 1994	DeciView	2.4	4.8	5.8	6.0
Household Soiling	RCG/Hagler Bailly, 1994	PM10	(Direct Economic Valuation)			

Other Results

Additional analyses were conducted in an effort to quantify effects resulting from *long-term* exposure air pollution: premature mortality due to chronic exposure to PM_{10} (where the CR function was inferred from "cohort" mortality studies), chronic bronchitis from long-term PM_{10} exposure, and chronic asthma due to ozone exposure. The results of these analyses are found in Table 83. As with the health effects presented above, results for both the "50 km" model run and the "all U.S." model run are presented.

Table 83. Results from Long-Term Exposure Studies (cases per year).

Endpoint	Study	Pollutant	1975	1980	1985	1990
Applied Only to Populations Living in Counties with PM Monitors or within 50 km of an O_3 Monitor:						
Mortality	Dockery et al. 1993	PM_{10}	148,402	304,782	460,485	428,416
	Pope et al. 1995	PM_{10}	60,189	151,085	171,892	163,189
Development of chr. bronchitis	Schwartz, 1993b	PM_{10}	150,191	416,768	503,390	453,641
	Abbey et al., 1993	PM_{10}	132,948	356,271	420,618	386,883
Development of chr. asthma	Abbey et al., 1995a	O_3	5,248	14,832	24,665	31,845
Applied to Populations in All Counties:						
Mortality	Dockery et al. 1993	PM_{10}	194,014	503,361	622,526	667,293
	Pope et al. 1995	PM_{10}	78,593	192,537	231,216	253,067
Development of chr. bronchitis	Schwartz, 1993b	PM_{10}	196,080	531,559	674,848	708,809
	Abbey et al., 1993	PM_{10}	173,571	454,309	564,753	602,990
Development of chr. asthma	Abbey et al., 1995a	O_3	8,349	18,416	30,884	41,198

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Appendix E: Ecological Effects of Criteria Pollutants

Introduction

Benefits to human welfare from air pollution reductions achieved under the CAA can be expected to arise from likely improvements in the health of aquatic and terrestrial ecosystems and the myriad of ecological services they provide. For example, improvements in water quality stemming from a reduction in acid deposition-related air pollutants (e.g., SO_x and NO_x) could conceivably 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. Increased growth and productivity of U.S. forests could occur from reduced emissions of ozone forming precursors, particularly VOC and NO_x, and thus may result in benefits from increased timber production; greater opportunities for recreational services such as hunting, camping, wildlife observation; and nonuse benefits such as nutrient cycling, CO_2 sequestration, and existence value.

In this Appendix, the potential ecological benefits from CAA pollutant controls are discussed in the context of three types of ecosystems: aquatic and wetland ecosystems and forests. In describing the potential ecological benefits of the CAA, it is clearly recognized that this discussion is far from being comprehensive in terms of the types and magnitude of ecological benefits that may actually occur from the implementation of CAA. Rather, this discussion reflects current limitations in understanding and quantifying the linkages which exist between air quality and ecological services, in addition to limitations in the subsequent valuation of these services in monetary terms. This discussion also does not cover potential benefits from improvements in other ecological services, namely agriculture and visibility, which are discussed and quantified in other sections of this report. This appendix is dedicated to a *qualitative* evaluation of ecological benefits. However, where possible, the existing body of scientific literature is drawn upon in an attempt to provide insights to the possible magnitude of benefits that may have resulted from CAA improvements of selected ecological services.

Benefits From Avoidance of Damages to Aquatic Ecosystems

Aquatic ecosystems (lakes, streams, rivers, estuaries, coastal areas) provide a diverse range of services that benefit the welfare of the human population. Commercially, aquatic ecosystems provide a valuable food source to humans (e.g., commercial fish and shellfish harvesting), are used for the transportation of goods and services, serve as important drinking water sources, and are used extensively for irrigation and industrial process (e.g., cooling water, electrical generation). Recreationally, water bodies provide important services that include recreational fishing, boating, swimming, and wildlife

viewing. They also provide numerous indirect services such as nutrient cycling, and the maintenance of biological diversity.

Clearly, these and other services of aquatic ecosystems would not be expected to be equally responsive to changes in air pollution resulting from the implementation of the CAA. The available scientific information suggests that the CAA-regulated pollutants that can be most clearly linked to effects on aquatic resources include SO_x and NO_x (through acid deposition and increases in trace element bioavailability), NO_x (through eutrophication of nitrogen-limited water bodies), and mercury (through changes in atmospheric deposition). Potential benefits from each of these processes (acid deposition, eutrophication, mercury accumulation in fish) are described separately in the following sections.

Acid Deposition

Background

Acid deposition refers to the depositing of acids (e.g., H_2SO_4, HNO_3) from the atmosphere to the earth's surface. Acid deposition can occur in the wet or dry form and can adversely affect aquatic resources through the acidification of water bodies and watersheds. Acidification of aquatic ecosystems is of primary concern because of the adverse effects of low pH and associated high aluminum levels on fish and other aquatic organisms. Low pH can produce direct effects on organisms, through physiological stress and toxicity processes, and indirect effects, mediated by population and community changes within aquatic ecosystems. Acidification can affect many different aquatic organisms and communities. As pH decreases to 5.5, species richness in the phytoplankton, zooplankton, and benthic invertebrate communities decreases.[116] Additional decreases in pH affect species richness more significantly, and may sometimes affect overall biomass.[117] Table 84 presents descriptions of the biological effects of acidification at different pH levels. In evaluating the severity of biological changes due to acidification, the reversibility of any changes is an important consideration; biological populations and communities may not readily recover from improved water quality under certain circumstances. Researchers have addressed acidification effects through many different experimental protocols, including laboratory bioassays, particularly concerning pH, aluminum, and calcium; manipulative whole-system acidification studies in the field; and comparative, nonmanipulative field studies.

Although acidification affects phytoplankton, zooplankton, benthic invertebrates, fish, amphibians, and waterfowl, most acidification research has concentrated on fish populations.[118] Aluminum, which can be toxic to organisms, is soluble at low pH and is leached from watershed soils by acidic deposition.[119] Acidification may affect fish in several ways. The direct physiological effects of low pH and high aluminum include increased fish mortality, decreased growth, and decreased reproductive potential. The mechanism of toxicity involves impaired ion regulation at the gill.[120]

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[116] J. Baker et al., NAPAP SOS/T 13, 1990; Locke, 1993.

[117] J. Baker et al., NAPAP SOS/T 13, 1990.

[118] NAPAP, 1991.

[119] J. Baker et al., NAPAP SOS/T 13, 1990.

[120] J. Baker et al., NAPAP SOS/T 13, 1990.

Table 84. Summary of Biological Changes with Surface Water Acidification.

pH Decrease	Biological Effects
6.5 to 6.0	Small decrease in species richness of phytoplankton, zooplankton, and benthic invertebrate communities resulting from the loss of some acid-sensitive species, but no measurable change in total community abundance or production. Some adverse effects (decreased reproductive success) may occur for acid-sensitive fish species (e.g., fathead minnow, striped bass).
6.0 to 5.5	Loss of sensitive species of minnows and dace, such as blacknose dace and fathead minnow; in some waters decreased reproductive success of lake trout and walleye. Distinct decrease in the species richness and change in species composition of the phytoplankton, zooplankton, and benthic invertebrate communities. Loss of a number of common invertebrate species from the zooplankton and benthic invertebrate communities, including zooplankton species such as *Diaptomus sicilis*, *Mysis relicta*, *Epischura lacustris*; many species of snails, clams, mayflies, and amphipods, and some crayfish Visual accumulations of filamentous green algae in the littoral zone of many lakes and in some streams.
5.5 to 5.0	Loss of several important sport fish species, including lake trout, walleye, rainbow trout, and smallmouth bass; as well as additional non-game species such as creek chub Continued shift in the species composition and decline in species richness of the phytoplankton, periphyton, zooplankton, and benthic invertebrate communities; decreases in the total abundance and biomass of benthic invertebrates and zooplankton may occur in some waters Loss of several additional invertebrate species common in oligotrophic waters, including *Daphnia galeata mendotae*, *Diaphanosoma leuchtenbergianum*, *Asplanchn priodonta*; all snails, most species of clams, and many species of mayflies, stoneflies, and other benthic invertebrates Inhibition of nitrification Further increase in the extent and abundance of filamentous green algae in lake littoral areas and streams
5.0 to 4.5	Loss of most fish species, including most important sport fish species such as brook trout and Atlantic salmon Measurable decline in the whole-system rates of decomposition of some forms of organic matter, potentially resulting in decreased rates of nutrient cycling Substantial decrease in the number of species of zooplankton and benthic invertebrates, including loss of all clams and many insects and crustaceans; measurable decrease in the total community biomass of zooplankton and benthic invertebrates in most waters Further decline in the species richness of the phytoplankton and periphyton communities Reproductive failure of some acid-sensitive species of amphibians such as spotted salamanders, Jefferson salamanders, and the leopard frog

Source: Baker, J. et al. (NAPAP SOS/T 13, 1990), p. 13-173.

Population losses occur frequently because of recruitment failure,[121] specifically due to increased mortality of early life stages.[122] Changes at other trophic levels may affect fish populations by altering food availability.[123] Fish in poorly buffered, low pH water bodies may accumulate higher levels of mercury, a toxic metal, than in less acidic water bodies, due to increased mercury bioavailability. The primary consequence of mercury accumulation appears to be hazardous levels to humans and wildlife who consume fish, rather than direct harm to aquatic organisms (discussed further below).

The CAA-regulated pollutants that are likely to have the greatest affect on aquatic ecosystems through acid deposition and acidification are SO_2 and NO_x. In the atmosphere, SO_2 and NO_x react to form sulfate and nitrate particulates, which may be dry-deposited; also the pollutants may react with water and be wet-deposited as dilute sulfuric and nitric acids. SO_2 is considered the primary cause of acidic deposition, contributing 75 to 95 percent of the acidity in rainfall in the eastern United States (NAPAP, 1991).

Current Impacts of Acid Deposition

Effects on Water Chemistry

The effects of acid deposition and resulting acidification of water bodies was intensively studied as part of a 10-year, congressionally-mandated study of acid rain problem in the United States.[124] Based on the NAPAP study, it is estimated that 4 percent of the lakes and 8 percent of the streams in acid-sensitive regions of the U.S. are chronically acidic due to natural and anthropogenic causes. NAPAP defines acidic conditions as occurring when the acid neutralizing capacity[125] (ANC) is below 0 μeq/L. Furthermore, approximately 20 percent of the streams and lakes in these regions are considered to be extremely susceptible to acidity (defined as ANC <50 μeq/L) and slightly more than half show some susceptibility to acidification (defined as ANC <200 μeq/L).

In terms of the role of acid deposition as a causal mechanism for the acidification of water bodies, it is estimated that 75 percent of the 1,181 acidic lakes and 47 percent of the 4,668 streams studied under NAPAP receive their dominant source of acid anions from atmospheric deposition (see Table 85). On a regional basis, the importance of acid deposition varies considerably, which is believed to result from regional differences in SO_x and NO_x emissions and differences in the biogeochemistry of individual watersheds. For acidic lakes (ANC <0), the regions that appear most likely to be influenced by acid deposition include the Adirondacks and Mid-Atlantic Highland region, with acid deposition cited as the dominant source of acidity in 100 percent of the acidic lakes studied (Table 85). This is in stark contrast to the West region, where none of the acidic lakes studied were dominated by acid deposition (notably, the sample size of lakes for this region was small to begin with). For acidic streams, the Mid-Atlantic Highland region contains the greatest proportion of streams whose acidic inputs are dominated

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[121] Rosseland, 1986, as cited by J. Baker et al., NAPAP SOS/T 13, 1990.

[122] J. Baker et al., NAPAP SOS/T 13, 1990.

[123] Mills et al., 1987, as cited by J. Baker et al., NAPAP SOS/T 13, 1990.

[124] NAPAP, 1991.

[125] ANC is expressed in units of micro equivalents per liter (μeq/L), where an equivalent is the capacity to neutralize one mole of H^+ ions. Generally, waters with an ANC < 0 have corresponding pH values of less than 5.5 (L. Baker et al., NAPAP SOS/T 9, 1990).

by acid deposition (56 percent). This contrasts with acidic streams of Florida, where the vast majority (79 percent) are acidic primarily due to organic acids, rather than acid deposition.

Table 85. Comparison of Population of Acidic National Surface Water Survey (NSWS) by Chemical Category[1]

Region	Number of Acidic Waters	Deposition Dominated (%)	Organic Dominated (%)	Acid Mine Drainage Dominated (%)	Watershed Sulfate Dominated (%)
		LAKES			
New England	173	79	21	—	—
Adirondacks	181	100	—	—	—
Mid-Atlantic Highlands	88	100	—	—	—
Southeastern Highlands	—	—	—	—	—
Florida	477	59	37	—	4
Upper Midwest	247	73	24	—	3
West	15	—	—	—	100
All Lakes	1,181	75	22	—	3
		STREAMS			
Mid-Atlantic Highlands	2,414	56	—	44	—
Mid-Atlantic Coastal Plain	1,334	44	54	—	2
Southeastern Highlands	243	50	—	50	—
Florida	677	21	79	—	—
All Streams	4,668	47	27	26	<1

[1] Source: NAPAP 1991 (Table 2.2-1, p. 28).

Effects on Fish Habitat Quality

By combining information on relevant water chemistry parameters (pH, aluminum, calcium), fish toxicity models, and historical and current distributions of fish populations in the lakes and streams comprising the National Surface Water Survey (NSWS), NAPAP investigators estimated the proportion of water bodies with water chemistry conditions that are unsuitable for survival of various fish species.[126] In the Adirondack region, where the acidic lakes are dominated by acid deposition, it is estimated that 10 percent of the lakes are unsuitable for the survival of acid-tolerant fish species such as brook trout; 20 percent of the lakes are estimated to be unsuitable for the survival of acid-sensitive species such as

[126] NAPAP, 1991.

minnows. About 2 percent and 6 percent of the lakes in the New England region are estimated to be unsuitable for acid-tolerant and acid-sensitive fish species, respectively. A greater proportion of streams in the Mid-Atlantic Highland region are estimated to be unsuitable for acid-tolerant and acid-resistant fish species (18 percent and 30 percent, respectively); however, about 44 percent of streams surveyed in this region are thought to be heavily influenced by acid mine drainage (Table 85).

Economic Damages to Recreational Fishing

In an effort to assess some of the impacts from *existing* levels of acid deposition to public welfare, NAPAP investigated the current economic damages associated with acid deposition to trout anglers of New York, Maine, Vermont, and New Hampshire. The general approach used consisted of linking the catch per unit effort (CPUE) for four species of trout at individual lakes (estimated using participation survey data) to the relevant water quality conditions at these lakes (namely, the acid stress index or ASI). Using historical water quality data, critical water quality conditions (i.e., the ASI values) were estimated for lakes in the absence of acid deposition and compared current conditions reflecting the presence of acid deposition. Using two types of travel cost models, the Random Utility Model (RUM) and Hedonic travel-cost model (HTCM), estimates of the willingness to pay (WTP) per trip of sampled trout anglers were obtained. Aggregate estimates of the WTP were obtained across the populations of trout anglers using statistical weighting factors. Finally, the difference in total WTP between the current (acid deposition) scenario and the historical (acid deposition-free) scenarios was determined.

The resulting estimates of economic damages to trout anglers in the four state region are relatively small. Specifically, damage estimates range from $0.3 million to $1.8 million (in 1989 dollars) for the hedonic travel-cost and random utility models, respectively. By many accounts, these estimates can be considered to underestimate actual damages to anglers in these states. First, data limitations precluded the development of meaningful WTP estimates for brook trout anglers, which may be a significant component of trout fishing in these areas. Second, resource constraints necessitated exclusion of a large population of trout anglers (i.e., those residing in New York City). Third, the economic damage estimates were limited to trout anglers, thus excluding potentially similar if not greater economic damages to anglers fishing for other coldwater or warmwater fish species. In addition, the NAPAP analysis was performed in the context recreational fishing in lakes, thereby excluding potentially important welfare impacts from recreational fishing in streams. Finally, these estimates do not address non-use values of lakes in this region.

Benefits From Acid Deposition Avoidance Under the CAA

It is currently estimated that in the absence of pollution reductions achieved under the Clean Air Act, total sulfur emissions to the atmosphere would have increased by nearly 16 million tons by 1990, a 40 percent increase above 1990 levels estimated with CAA controls remaining in place.[127] Based on atmospheric transport and deposition modeling, this increase in sulfur emissions corresponds to an approximate 25 to 35 percent increase in total sulfur deposition (wet & dry) in large portions of the northeastern portion of the United States.[128] Given sulfur emission and deposition changes of this

[127] U.S. EPA, 1995; Table B-2.

[128] U.S. EPA 1995, pages 3-10.

magnitude, and the importance of sulfur emissions in contributing to acid deposition, one would expect some benefits to human welfare to be achieved as a result of improved quality of aquatic ecosystems. To date, however, no formal benefits assessment of CAA-avoided acid deposition impacts has been conducted for aquatic ecosystems. Nevertheless, past benefit assessments involving acid deposition impacts on aquatic ecosystems provide some opportunity to gain insights into the relative magnitude of certain aquatic-based benefits that may be achieved through pollution reductions under the CAA.[129]

Recreational Fishing

NAPAP evaluated the impact of changes in acid deposition on use values of aquatic ecosystems (i.e., recreational fishing).[130] In their integrated assessment, NAPAP valued the impacts of three different sulfur-induced acid deposition scenarios to trout anglers from NY, VT, NH and ME.[131] The three scenarios evaluated were:

1. No change in acid deposition,
2. a 50 percent reduction in acid deposition, and
3. a 30 percent increase in acid deposition.

As described above, equations were developed by NAPAP to estimate the catch per hour for species at each lake as a function of the ASI value for each lake and of the technique of the fishers. Baseline and predicted changes in CPUE were evaluated for all lakes modeled in the region. Willingness-to-pay estimates for CPUE per trip were derived for the baseline and sulfur emission scenarios using two travel-cost models, a random utility model and a hedonic travel cost model. These willingness-to-pay estimates were then combined with the results of a participation model that predicted the total number of trips taken by trout anglers. Total welfare changes were determined over a 50 year period (from 1990 to 2040).

At current levels of acid deposition, NAPAP estimates that trout anglers in these four states will experience annual losses by the year 2030 of $5.3 or $27.5 million (in 1989 dollars) for the random utility model and hedonic travel cost model, respectively (see Table 86). If acid deposition *increases* by 30 percent, which roughly corresponds to the 25 to 35 percent increase predicted for the northeast U.S. in the absence of CAA sulfur controls,[132] the resulting economic losses to trout anglers in 2030 would range from $10 million to nearly $100 million annually (in 1989 dollars) for the RUM and HTCM, respectively. If deposition decreases by 50 percent, annual benefits to recreational anglers are estimated to be $14.7 million (RUM) or $4.2 million (HTCM).

[129] See, for example, NAPAP, 1991.

[130] NAPAP, 1991.

[131] NAPAP, 1991; page 383-384.

[132] U.S. EPA 1995.

Table 86. Results from Benefits Assessments of Aquatic Ecosystem Use Values from Acid Deposition Avoidance.

Study	Use Value	Scenario Modeled	Method	Annual Benefits
NAPAP (1991)	Trout Fishing (NY, ME, VT, NH)	No change in acid deposition	RUM	-$3.3 million
			HTCM	-$27.5 million
		50% decrease in acid deposition	RUM	$14.4 million
			HTCM	$4.2 million
		30% decrease in acid deposition	RUM	$10.3 million
			HTCM	-$97.7 million
		No new emission reductions after 1985	RUM	-$3.3 million
			HTCM	-$3.5 million
		10 million ton reduction of SO_2 from 1980 levels by 2000	RUM	$9.7 million
			HTCM	$4.4 million

While an estimation of CAA-related benefits to trout anglers based on the 30 percent increase in acid deposition scenario has some appeal, a strict transfer of these benefits to the 812 retrospective analysis is hindered by several factors. First, the NAPAP benefits estimates are projected for future conditions (the year 2030). Therefore, the extent to which the NAPAP benefits reflect conditions and benefits in 1990 (the focus of the 812 retrospective assessment) is unclear. Second, the NAPAP and CAA 812 analyses operate from different baselines (1990 for the NAPAP study versus 1970-1990 for the 812 study). However, the NAPAP estimates of annual benefits of $10 to $100 million provide a rough benchmark for assessing the likely magnitude of the avoided damages to an important and sensitive recreational fishery in a four state area most impacted by surface water acidification form atmospheric deposition.

Eutrophication

Eutrophication is the process by which aquatic systems respond to nutrient enrichment. The most common nutrients involved in eutrophication are nitrogen and phosphorous (and related chemical species). When water bodies receive excessive amounts of nutrients, adverse impacts on their health can occur from excessive algal growth and the reduction in dissolved oxygen caused by decaying algal biomass. Under highly eutrophic conditions, excessive nutrients can cause depleted oxygen levels that result in subsequent loss of economically important benthic organisms (shellfish), fish kills, and changes in phytoplankton, zooplankton, and fish community composition.[133] Nuisance algal blooms can have numerous economic and biological costs, including water quality deterioration affecting biological resources, toxicity to vertebrates and higher invertebrates, and decreased recreational and aesthetic value of waters.[134] Although severe eutrophication is likely to adversely affect organisms, especially fish, a moderate increase in nutrient levels may also increase fish stocks, by increasing productivity in the food chain.[135]

[133] Paerl, 1993.

[134] Paerl, 1988.

[135] Hansson and Rudstam, 1990; Rosenberg et al., 1990; Paerl, 1993.

Atmospheric Deposition and Eutrophication

The deposition of NO_x in aquatic systems and their watersheds is one source of nitrogen that may contribute to eutrophication. The relative importance of NO_x deposition as a contributor to aquatic eutrophication depends on the extent to which the productivity of an aquatic ecosystem is limited by nitrogen availability and the relative importance of nitrogen deposition compared to other internal and external sources of nitrogen to the aquatic ecosystem. Furthermore, the vulnerability of aquatic ecosystems to eutrophication is known to vary seasonally and spatially, although these systems are affected by nutrient deposition throughout the year. In general, freshwater ecosystems appear to be more often limited by phosphorus, rather than nitrogen, and are not as likely to be heavily impacted by nitrogen deposition compared to some estuarine and coastal ecosystems.[136] In contrast to acidification of streams and lakes, eutrophication from atmospheric deposition of nitrogen is more commonly found in coastal and estuarine ecosystems, which are more frequently nitrogen-limited.[137]

Unfortunately, there is limited information with regard to the relative importance of atmospheric deposition as a nitrogen source in many estuarine and marine ecosystems. Estimates of the importance of atmospheric nitrogen deposition are difficult to make because of uncertainties in estimating deposition, especially dry deposition, as well as watershed nitrogen retention.[138] Paerl (1993) reviews the importance of atmospheric nitrogen deposition as a contributor to eutrophication of coastal ecosystems; he concludes that 10 to 50 percent of the total nitrogen loading to coastal waters is from direct and indirect atmospheric deposition. Estimates of the nitrogen budget for the economically important Chesapeake Bay indicate that about 25 percent of the nitrogen loadings to the bay occur via atmospheric deposition.[139] Hinga et al. (1991) estimate that anthropogenic deposition provides 11 percent of total anthropogenic inputs of nitrogen in Narragansett Bay, 33 percent for the New York Bight, and 10 percent for New York Bay. Fisher and Oppenheimer (1991) estimate that atmospheric nitrogen provides 23 percent of total nitrogen loading to Long Island Sound and 23 percent to the lower Neuse River in North Carolina. Information on the importance of atmospheric nitrogen deposition for most other U.S. coastal ecosystems is not available in the literature. Episodic atmospheric inputs of nitrogen may be an important source of nitrogen to nutrient-poor marine ecosystems, such as the North Atlantic near Bermuda and the North Sea.[140]

Valuing Potential Benefits from Eutrophication Avoidance Under the CAA

It is currently estimated that in the absence of pollution reductions achieved under the Clean Air Act, total nitrogen emissions to the atmosphere would increase by nearly 90 million tons by 1990, a 2-fold increase above 1990 levels estimated with CAA controls remaining in place.[141] However, the ability to determine the potential economic benefit from such a reduction in nitrogen emissions is heavily constrained by gaps in our current biological and economic knowledge base of aquatic ecosystems.

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[136] Hecky and Kilham, 1988; Vitovsek and Howarth, 1991.

[137] U.S. EPA, 1993a; Paerl, 1993.

[138] U.S. EPA, 1993a.

[139] U.S. EPA, 1993a.

[140] Owens et al., 1992.

[141] U.S. EPA, 1995; Table B-3.

One water body that has received much study in the area of nitrogen induced eutrophication is Chesapeake Bay. As previously discussed, it is estimated that atmospheric deposition of nitrogen contributes approximately 25 percent to the total nitrogen loadings to the bay.[142] In deposition terms, an estimated 15 to more than 25 percent increase in total nitrogen deposition has been forecast in the Chesapeake Bay watershed by 1990 in the absence of CAA pollution controls.[143] These results are based on an estimated 40,000 tons of atmospherically deposited nitrogen (as nitrate and ammonia) to Chesapeake Bay in 1985,[144] which means a 20 percent increase in atmospheric deposition would amount to approximately 8,000 additional tons.

One indirect method available to gage the potential economic relevance of avoidance of such atmospheric nitrogen loadings to Chesapeake Bay is through the avoidance cost of nitrogen controls. However, such an assessment is difficult due to the site, facility, and treatment-specific variation in treatment costs. For example, Camacho (1993; as cited by NERA 1994) reviewed nitrogen treatment costs for chemical treatment of water from important point sources (mostly public owned treatment works) and found that costs ranged from $9,600 to $20,600 per ton (annual costs, 1990 dollars), depending on the facility evaluated. Biological treatment of nitrogen from point sources was far more expensive, varying from $4,000 to $36,000 per ton. For control of non-point source loading, values of nitrogen removal practices ranged from $1,000 to $285,000 per ton.[145] Taking chemical addition as one possible example, the avoided costs of treatment of 8,000 tons of nitrogen would range from about $75 million to about $170 million annually (in 1990 dollars).

Mercury

Mercury, in the form of methyl mercury, is a neurotoxin of concern and can accumulate in tissue of fish to levels that are hazardous to humans and aquatic-feeding wildlife in the U.S. In relation to the 812 CAA retrospective analysis, mercury is of interest for two reasons. First, potential benefits to human welfare may have occurred as a result of mercury emission controls implemented under EPA's National Emission Standards for Hazardous Air Pollutants (NESHAP). Second, experimental and observational evidence suggests that acidification of water bodies enhances mercury accumulation in fish tissues.[146] Therefore, CAA-mandated reductions in sulfur and nitrogen oxide emissions and subsequent acid deposition may have resulted in indirect benefits from a reduction in mercury accumulation in fish and subsequent improvements to human health and welfare.

The accumulation of mercury to hazardous levels in fish has become a pervasive problem in the U.S. and Canada. A rapid increase in advisories occurred during the 1980s, including a blanket advisory affecting 11,000 lakes in Michigan.[147] The Ontario Ministries of Environment and Natural Resources (1990, as cited by Spry and Wiener, 1991) recommend fish consumption restrictions for 90 percent of the

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[142] U.S. EPA, 1993a.

[143] U.S. EPA 1995, Figure C-6.

[144] NERA, 1994.

[145] Shyler, 1992; as cited by NERA, 1994.

[146] Bloom et al., 1991; Watras and Bloom, 1992; Miskimmin et al., 1992; Spry and Wiener, 1991; Wiener et al., 1990.

[147] Watras et al., 1994.

walleye populations, 80 percent of smallmouth bass populations, and 60 percent of lake trout populations in 1,218 Ontario lakes because of mercury accumulation. In many instances, mercury has accumulated to hazardous levels in fish in highly remote water bodies that are free from direct water discharges of mercury.[148] Mass balance studies have shown that atmospheric deposition of mercury can account for the accumulation of mercury in fish to high levels in lakes of these remote regions.[149] The potential impacts of mercury on the health of humans and fish-eating (piscivorous) wildlife has lead EPA to recently establish water quality criteria to protect piscivorous species in the Great Lakes.[150]

Although mercury accumulation in fish via atmospheric deposition is now widely recognized as a potential hazard to human health and certain wildlife species, studies establishing quantitative linkages between sources of mercury emissions, atmospheric deposition of mercury, and subsequent accumulation in fish are lacking. Thus at the present time, we are unable to quantify potential benefits from CAA-avoided mercury accumulation in fish of U.S. water bodies. Given the pervasiveness of the mercury problem with CAA-pollution controls, potential benefits to human health and welfare from avoidance of further mercury related damages to aquatic ecosystems could be substantial.

Benefits from Avoided Damages to Wetland Ecosystems

Introduction

This review addresses the effects of air pollutants on wetland ecosystems; the focus is on acidification and nutrient loading. Valuable service flows of wetland ecosystems include flood control, water quality protection, wildlife and fish habitat, and biodiversity. The limited scientific evidence suggests that air pollutants may most affect biodiversity, in particular because of nutrient loading through nitrogen deposition.

Wetlands are broadly characterized as transitional areas between terrestrial and aquatic systems in which the water table is at or near the surface or the land is periodically covered by shallow water.[151] Types of wetlands include swamps (forested wetlands), marshes (herbaceous vegetation), and peatlands, which are wetlands that accumulate partially decayed vegetative matter due to limited decomposition.[152] Peatlands include bogs and fens. Bogs receive water solely from precipitation, are generally dominated by *Sphagnum* moss, and are low in nutrients. Fens receive water from groundwater and precipitation, contain more marsh-like vegetation, and have higher pH and nutrient levels than bogs.[153] Most of the

[148] Glass et al., 1990; Sorenson et al., 1990; Grieb et al. 1990; Schofield et al. 1994.

[149] Fitzgerald et al. 1991.

[150] U.S. EPA, 1995.

[151] Cowardin et al., 1979.

[152] Mitsch and Gosselink, 1986.

[153] Mitsch and Gosselink, 1986.

limited work on the effects of atmospheric deposition on wetlands has been done in peatlands, specifically in Europe, where levels of atmospheric deposition are generally much higher than in the U.S.

The air pollutants of greatest concern with respect to effects on wetland ecosystems are oxides of nitrogen (NO_x) and oxides of sulfur (SO_x), primarily sulfur dioxide (SO_2). Air pollutants may affect wetland ecosystems by acidification of vulnerable wetlands and by increasing nutrient levels. Acidification in vulnerable wetlands may affect vegetation adversely, as appears to have occurred in Europe. In wetlands where nitrogen levels are low, increased nitrogen deposition may alter the dynamics of competition between plant species. Species adapted to low-nitrogen levels, including many endangered species, may decrease in abundance. [154]

Effects of Acidification

Limited evidence suggests that acidic deposition and decreased pH may harm certain wetland plants, alter competitive relations between wetland plants and cause changes in wetland drainage and water retention.

Work concerning the possible acidification of peatlands is inconclusive. Acidic deposition is unlikely to result in displacement of base cations from cation exchange sites in bogs, and therefore it will not cause a drop in pH.[155] Peatland sediments are low in Al^{3+}, so mobilization of toxic aluminum is not a concern as it is in forest soils and aquatic ecosystems.[156] Acidification might affect certain fen ecosystems. Gorham et al. (1984, as cited by Turner et al., NAPAP SOS/T 10, 1990) have hypothesized that acidic deposition could leach base cations from mineral-poor fens and decrease pH levels. This could result in a transition to bog vegetation such as *Sphagnum* and away from sedge meadow vegetation. At this time, this remains a hypothesis; however, pH did not decrease in a mineral-poor Ontario fen during a four-year period in which researchers experimentally increased acidic deposition.[157]

In European wetlands affected by high levels of deposition for many years, acidic deposition has seriously affected wetland vegetation. Roelofs (1986) reports that acidification of heath pools in the Netherlands has caused a change in species composition with *Sphagnum* and rushes replacing the original vegetation. Likewise, significant declines in *Sphagnum* in British bogs have occurred in areas affected by 200 years of atmospheric pollution, including nitrogen deposition.[158] It is unclear how such changes have affected wetland service flows apart from the effects on biodiversity; however, water retention has decreased and significant erosion has occurred in seriously perturbed British bogs near Manchester and Liverpool.[159]

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[154] U.S. EPA, 1993a.

[155] Gorham et al., 1984, as cited by Turner et al., NAPAP SOS/T 10, 1990.

[156] Turner et al., NAPAP SOS/T 10, 1990.

[157] Rochefort et al., 1990.

[158] Lee et al., 1986, as cited by U.S. EPA, 1993a.

[159] Lee et al., 1986, as cited by U.S. EPA, 1993a.

Effects of Nutrient Loading

Atmospheric deposition may affect wetlands by increasing the level of nutrients, particularly nitrogen, in wetlands. Sulfur is not a limiting nutrient in peatlands,[160] but nitrogen commonly limits plant growth.[161] The effects of increased nitrogen levels in wetlands include an increased threat to endangered plant species and possible large-scale changes in plant populations and community structure. Endangered and threatened plant species are common in wetlands, with wetland species representing 17 percent of the endangered plant species in the U.S. (U.S. EPA, 1993a). These plants are often specifically adapted to low nitrogen levels; examples include isoetids[162] and insectivorous plants.[163] In eastern Canadian wetlands, nationally rare species are most common in infertile sites.[164] When nitrogen levels increase, other species adapted to higher levels of nitrogen may competitively displace these species. Thus, NO_x emissions that increase nitrogen levels in nitrogen-poor wetlands may increase the danger to threatened and endangered species.

By changing competitive relations between plant species, increased nitrogen deposition may broadly affect community structure in certain wetlands. Common species that thrive in nitrogen-poor wetlands may become less abundant. Many nitrogen-poor bogs in the northern U.S. are dominated by *Sphagnum* species. These species capture low levels of nitrogen from precipitation. Increased nitrogen levels may directly harm *Sphagnum* and cause increased nitrogen to be available to vascular plants that may out compete *Sphagnum*.[165] Studies in Great Britain have documented large declines in *Sphagnum* moss because of atmospheric pollution;[166] nitrogen loading may play an important role in these declines. However, Rochefort et al. (1990) document limited effects of fertilization from experimentally-increased NO_3^- and SO_4^{2-} deposition on an Ontario mineral-poor fen over a four-year period, apart from initially increased *Sphagnum* growth.

Increases in nitrogen levels due to NO_x emissions will have the greatest effect on wetlands that are extremely nitrogen-limited and that receive small amounts of nitrogen naturally. Since bogs, including *Sphagnum* bogs, receive little surface water runoff, they get most of their nutrient and water loadings through precipitation. These bogs may receive a total of approximately 10 kg nitrogen per hectare per year (N/ha/yr), which is one to two orders of magnitude less nitrogen than other freshwater wetlands and saltmarshes receive.[167] As atmospheric deposition of nitrogen has been estimated to be at least 5.5 to 11.7 kg N/ha/yr,[168] changes in NO_x emissions would most likely affect these bogs. The results of a model by Logofet and Alexandrov (1984) suggest that a treeless, nutrient-poor bog may undergo succession to a forested bog because of the input of greater than 7 kg N/ha/yr.

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[160] Turner et al., NAPAP SOS/T 10, 1990.

[161] U.S. EPA, 1993a.

[162] Boston, 1986, as cited by U.S. EPA, 1993a.

[163] Moore et al., 1989.

[164] Moore et al., 1989; Wisheu and Keddy, 1989, as cited by U.S. EPA, 1993a.

[165] Lee & Woodin 1988, Aerts et al., 1992.

[166] Ferguson et al., 1984; Lee et al., 1986, both as cited by U.S. EPA, 1993a.

[167] U.S. EPA, 1993a.

[168] U.S. EPA, 1993a.

As in freshwater wetlands, significantly increased nitrogen deposition to coastal wetlands will increase productivity and alter the competitive relationships between species (U.S. EPA, 1993a). However, studies showing this increased productivity have used 100 to 3000 kg N/ha/yr.[169] Therefore, limited changes in NO_x emissions may not affect coastal wetland productivity.

Summary of Wetland Ecosystem Effects

The effects of air pollutants on wetlands have received little attention, in contrast to the large body of work on the effects of acid rain on aquatic and forest ecosystems. Little evidence exists that would suggest that acidification due to atmospheric deposition is a major threat to wetlands. In particular, peatlands are naturally acidic, although mineral-poor fens may be at risk from acidification. Nitrogen loading may alter community composition in wetlands naturally low in nutrients, such as bogs. Nitrogen loading may threaten rare species adapted to low nitrogen levels. In Britain and the Netherlands, heavy atmospheric deposition over a long period has caused serious declines in *Sphagnum* in peatlands.

Air pollutants appear to threaten most seriously rare and endangered species, biodiversity, and community composition in wetlands, particularly bogs. These changes are difficult to associate with changes in economic value; even the qualitative nature of the effects is uncertain. Air pollutants may not significantly affect such important wetland service flows as flood control, water quality protection, and wildlife habitat in most wetlands, so the impacts on the economic value of wetlands may be limited.

Benefits from Avoided Damages to Forests

Introduction

Forests occupy 33 percent of the land mass in the U.S. (some 738 million acres) and provide a wealth of services to the U.S. population. Notable services provided by forests include timber production, recreational opportunities such as hunting, camping, hiking, wildlife observation, maintenance of water quality, nutrient removal and cycling, flood control, erosion control, carbon sequestration, preservation of diversity and existence values. In 1991, hunting participation alone accounted for 236 million recreation days that included 214 million person trips with estimated expenditures valued at $12.3 billion.[170]

The Clean Air Act-regulated pollutants of greatest concern with respect to effects on forest ecosystems are oxides of sulfur (SO_x), primarily sulfur dioxide (SO_2), oxides of nitrogen (NO_x), and volatile organic compounds (VOCs). While extremely high ambient concentrations of SO_2 and NO_x may

[169] U.S. EPA, 1993a.

[170] U.S. DOI, 1993.

directly affect vegetation, such effects are uncommon in the U.S.;[171] the indirect effects of these pollutants are of greater concern. Specifically, emissions of SO_2 and NO_x are known to contribute to acid deposition in portions of the United States, with SO_2 contributing 75 to 95 percent of the acidity in rainfall in the eastern U.S.[172] Acid deposition is of concern to forests primarily from the acidification of soils (i.e., by reducing seed germination, altering nutrient availability). Direct foliar damage can occur from precipitation with extremely low pH levels (i.e., 3.0-3.6 and below), although these levels are lower than ambient levels in the U.S.[173] VOCs and NO_x are important precursors to ozone formation, which can affect leaf photosynthesis and senescence and decrease cold hardiness, thereby causing deleterious impacts on tree growth, survival and reproduction. Deposition of NO_x may also alter the nutrient balance of forest soils, which in turn might alter the competitive relationships between tree species and affect species composition and diversity.[174]

Current Air Pollutant Effects on Forests

Acid Deposition Impacts

In 1985, NAPAP organized the Forest Response Program (FRP) in order to evaluate the significance of forest damage caused by acidic deposition, the causal relationships between air pollutants and forest damage, and the dynamics of these relationships regionally. Research was focussed on four forest regions: Eastern Spruce-Fir, Southern Commercial Forests, Eastern Hardwoods, and Western Conifers. With the exception of high-elevation spruce-fir forests, the available evidence suggests that acidic deposition does not currently affect these forests and that observed declines in sugar maple and southern pines are not due to acidic deposition.[175]

Circumstantial evidence suggests that acidic deposition may affect high-elevation spruce-fir forests in the northeastern U.S. These forests have extensive contact with acidic cloud water.[176] Experimental evidence suggests that acidic deposition may affect cold hardiness in red spruce, an important component of the spruce-fir forest. Significant declines in red spruce growth and in its importance in the forest have occurred in New York and northern New England. The proximate cause of death of red spruce in the region is pathogens and insects; acidic deposition may interact with these biological stresses and with weather-induced stress to produce adverse effects in red spruce. Ozone may also play a role in red spruce decline in this region.[177] Available evidence suggests that soil aluminum and soil pH levels have not affected red spruce adversely.[178]

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[171] Shriner et al., NAPAP SOS/T 18, 1990.

[172] NAPAP, 1991.

[173] Shriner et al., NAPAP SOS/T 18, 1990.

[174] U.S. EPA, 1993a.

[175] Barnard et al., NAPAP SOS/T 16, 1990; NAPAP 1991.

[176] Barnard et al., NAPAP SOS/T 16, 1990.

[177] Shriner et al., NAPAP SOS/T 18, 1990.

[178] Barnard et al., NAPAP SOS/T 16, 1990.

Ozone Impacts

Experimental Evidence

For practical reasons, the majority of experimental evidence linking ozone exposure to damage to tree species has been derived from studies of individual plants, especially seedling and branch studies.[179] Results from these studies suggests that ozone exposure can reduce photosynthesis and increase senescence in leaves. Subsequently, such effects from ozone may alter the carbohydrate allocation to plant tissues such as roots, which may affect plant growth and cold hardiness. Decreases in cold tolerance may be particularly important for trees in northern latitudes and high elevations. Recent work on quantifying the relationship between ozone exposure and plant responses suggest that seedlings of aspen, ponderosa pine, black cherry, tulip poplar, sugar maple, and eastern white pine seedlings may experience biomass reductions of approximately 10 percent at or near ambient ozone exposures.[180]

Although indicative of short-term relative response to ozone exposure, results from these experiments are unable to provide reliable information on the long-term effects of ozone on forests. This limitation arises because the effects of ozone on forests will depend on both the response of individual plants to ozone exposure and the response of populations of plants, which interact with their environment. Population response will altered by the varying intraspecific genetic susceptibility to ozone. Individual plant response will also be affected by many environmental factors, including insect pests, pathogens, plant symbionts, competing plants, moisture, temperature, light, and other pollutants. Consistent evidence on the interaction of ozone with other environmental factors is lacking. Furthermore, most experimental studies have only studied exposure for one growing season; effects on forest species may occur over decades.[181] Therefore, considerable uncertainties occur in scaling across individuals of different ages, from individuals to populations and communities, and across time.

Observational Evidence

Studies of the forests of the San Bernardino Mountains provide the strongest case for linking ozone exposure to damages to an entire forest ecosystem. These forests have been exposed to extremely high ambient ozone levels over the past 50 years due to their proximity to the Los Angeles area. The area has been extensively studied regarding the effects of ozone, as described in U.S. EPA (1993b). The ecosystem has been seriously affected by ozone pollution, with the climax-dominant, but ozone-sensitive ponderosa pine and Jeffrey pine declining in abundance at the expense of more ozone-tolerant species. These sensitive species have experienced decreased growth, survival, reproduction, and susceptibility to insects. The effects of ozone on these species have resulted in other ecosystem effects, including the buildup of a large litter layer, due to increased needle senescence. The decline of the fire-tolerant ponderosa and Jeffrey pines may seriously affect the fire ecology of the ecosystem, with fire-sensitive species becoming more common. Ozone concentrations have been declining in recent decades, and crown injury of ponderosa and Jeffrey pine has decreased. However, the two species have continued to

[179] U.S. EPA, 1993b.

[180] Hogsett et al. 1993; as cited by U.S. EPA, 1993b; Table 5-25.

[181] U.S. EPA, 1993b.

decline in abundance, as measured by total basal area, compared with other species over the period 1974 to 1988.[182]

Limited field studies have been completed in other forest ecosystems. Peterson et al. (1991, as cited by U.S. EPA, 1993b) reports foliar damage and decreased radial growth in ponderosa and Jeffrey pine in the Sierra Nevada Mountains of California. Work in the southern Appalachian Mountains shows that foliar injury from ozone is particularly common in white pine and black cherry (and occurs in other species), and that ozone can reduce the growth of tree species and herbaceous vegetation.[183] Evidence of population and community changes caused by ozone has not been found in these areas.

Monitoring by the USDA Forest Service shows that growth rates of yellow pine in the Southeast have been decreasing over the past two decades in natural stands but not in pine plantations.[184] Solid evidence linking this growth reduction to air pollutants is lacking, although ozone, in particular, may be a factor.[185] Ambient ozone levels in the region are high enough to damage sensitive tree species, including pine seedlings during experimental exposure.[186] Due to the commercial importance of yellow pine, the economic impacts of ozone on forest ecosystems in this area could be significant if ozone is affecting growth.

Although the ecosystem effects occurring in the San Bernardino forest ecosystem have occurred at very high ozone exposures, lower ozone exposure elsewhere in the U.S. may still affect forests. Even small changes in the health of ozone-sensitive species may affect competition between sensitive and tolerant species, changing forest stand dynamics.[187] Depending on the sensitivities of individual competing species, this could affect timber production either positively or negatively, and affect community composition and, possibly, ecosystem processes.

Valuation of Benefits From CAA-Avoided Damages to Forests

Background

In order to quantitatively assess the economic benefits of avoided damages of relevant CAA pollutants to forests, it is necessary to link estimated changes in air pollution to measures of forest health and conditions that can be readily quantified in economic terms. For commercial timber production, this would require quantifying the relationship between atmospheric deposition and measures of forest productivity such as timber yield. For assessing recreational benefits, linkages would have to be drawn between air pollution and vulnerable factors that influence forest-based recreation (e.g., site-characteristics such as canopy density, type of tree species, degree of visible tree damage, etc.). While

[182] Miller et al., 1989 and Miller et al., 1991, as cited by U.S. EPA, 1993b.

[183] U.S. EPA, 1993b.

[184] NAPAP, 1991.

[185] NAPAP, 1991.

[186] NAPAP, 1991.

[187] U.S. EPA, 1993b.

important strides have been made in establishing these linkages (e.g., NAPAP modeling of air pollution effects on forest soil chemistry and tree branch physiology), critical gaps in our ability to predict whole tree and forest responses to air pollution changes have precluded the establishment of such quantitative linkages.[188] Critical knowledge gaps exist in our ability to extrapolate experimental results from seedling and branch studies to whole tree and forest responses, to account for key growth processes of mature trees, to integrate various mechanisms by which air pollution can affect trees (e.g., soil acidification, nutrification, and direct foliar damage, winter stress, etc.), and to account for the interaction of other stressors on forest health and dynamics (susceptibility to insect damage, drought, disease, fire, nutrient and light competition, etc.).

Despite these constraints to quantifying economic benefits from air pollution reductions on forest ecosystems, relevant studies that have attempted to value air pollution damages on forests are reviewed and summarized below. In some cases, the relationship between air pollution and forest response is estimated using expert judgement (e.g., for NAPAP assessment from various growth scenarios). In other cases, damage estimates reflect current impacts of air pollution on forests, and the dose-response relationship is absent. In the aggregate, this summary provides some insight into possible CAA-related benefits from avoided damages to a select and narrowly focussed group of forest services, but , because of severe date constraints, is not able to provide even a coarse estimate of the overall range of forest-based benefits possible under the CAA.

Commercial Timber Harvesting

The economic impact of hypothetical growth reductions in northeastern and southeastern trees (both hardwood and softwood species) was intensively studied under NAPAP.[189] Growth reductions ranging from 5 to 10 percent over a 5 to 10 year period, depending on the species and location, were assumed to occur as a result of all forms of air pollution based on expert opinion derived from a survey by deSteigner and Pye (1988, as cited in Haynes and Kaiser, NAPAP SOS/T 27 part B, 1990). Timber market responses to these hypothesized growth declines were modeled until the year 2040 using a revised version of the Timber Assessment Market Model (TAMM90) and the Aggregate Timberland Assessment System (ATLAS), which was used to simulate timber inventories on private timberland in the United States. Economic welfare outputs included changes in consumer and producer surplus and changes in revenue to southeast stumpage owners. Results indicate that annualized reductions in consumer and producer surplus would total $0.5 billion by the year 2000 and $3 billion by the year 2040 (in 1967 dollars). Simulated effects on stumpage owners' revenues were minimal ($10 to $20 million).

In an attempt to estimate the net economic damages from ozone effects on selected U.S. forests, NAPAP studied the effect of various *assumed* reductions in growth rates of commercial southeastern pine forests (both natural and planted).[190] For both planted and natural plus planted pines, the following changes in growth rates were assumed to occur: a 2 percent increase, no change, a 2 percent decrease, a 5 percent decrease, and a 10 percent decrease. The 2 to 5 percent growth reductions were considered as possible outcomes from current ozone induced damage to southeastern forests, although no quantitative

[188] NAPAP, 1991.

[189] Haynes and Kaiser, NAPAP SOS/T 27 part B, 1990.

[190] NAPAP, 1991.

linkage between ozone exposure and damages was established. The 10 percent growth reduction scenario was primarily included for evaluating model sensitivity to severe changes in growth and was considered out of the range of likely ozone damage estimates. The TAMM and ATLAS models were again used to simulate timber market responses under baseline and hypothesized growth change scenarios from 1985 to 2040. Results indicate that *annual* changes in total economic surplus (i.e., the sum of consumer and producer surplus and timber owner revenues in 1989 dollars) would range from an increase of $40 million (for the 2 percent increase in growth scenario) to a decrease of $110 million (for the 10 percent decrease in growth scenario) planted and natural pine model simulations.

In the context of estimated benefits from avoidance of other damages in the absence of the Clean Air Act from 1970 to 1990,[191] the magnitude of economic damages estimated to the commercial timber industry are comparatively small. For example, economic damage estimates range up to $3 billion annually for 5 to 10 percent growth rate reductions in northeast and southeast forests, and just $110 million for southeastern pines. However, in the context of damages to forest-based services as a whole, the NAPAP-derived commercial timber damage estimates should be viewed as representing a lower bound estimate for a variety of reasons. First, these damage estimates exclude other categories of possible forest-based benefits, including recreational and non-use values. Second, even within the context of timber-related damages, the NAPAP forest-damage studies focused on a portion of U.S. forests (northeastern and southeastern U.S.); a much greater geographic range of forests could become susceptible to timber-related damages in the absence of CAA controls. Finally, the NAPAP damage estimates consider only two types of tree species: planted and naturally grown pines, although these species are economically important. Damages to other commercially harvested tree species, such as mixed pine and hardwood forests, are therefore excluded.

Non-marketed Forest Services

In an effort to address the potential benefits resulting from avoidance of acid deposition-induced damages to non-marketed forest-based services (e.g., recreation use, existence value), an extensive review of the economic literature was conducted under the auspices of NAPAP.[192] From their review, NAPAP could not identify any single study or model that could be reliably used to quantify economic benefits from avoided acid deposition-caused damages to non-marketed forest services (such as recreational use) on a regional or national basis. The primary limitation in many of the studies reviewed was the absence of a quantitative linkage between the value of a recreational user day and important site characteristics which could be tied to air pollution effects. In addition, most studies were narrowly focused geographically to specific sites and did not attempt to value system-wide (larger scale) damages that could result from acid deposition over an entire region. Since the availability of nearby substitution sites will affect the recreational value for given site, the benefits from such site-specific studies may not reflect actual economic damages incurred from wide-scale air pollution impacts on forests. The inability of studies to consider additional crowding at unaffected sites in addition to changes in recreational participation rates as a function of air pollution damages was also recognized as an important limitation.

[191] Most notably avoided human health effects, which are estimated on the order of $300 to $800 billion annually.

[192] Rosenthal, NAPAP SOS/T 27 Part B, 1990.

Despite not being able to quantitatively assess the benefits from avoided acid deposition-induced damages to nonmarket forest services, several important concepts emerge from NAPAP's review of recreational benefits, that bear relevance to the 812 retrospective analysis. First, several studies were identified that established a relationship between key forest site characteristics and the value of recreational participation. For example, Brown et al. (1989, as cited in Rosenthal, NAPAP SOS/T 27 Part B, 1990) used contingent valuation to evaluate the relationship between scenic beauty ratings and willingness of recreationalists to pay at pictured sites. Based on their interviews with over 1400 recreationalists at 10 different sites in Arizona, positive correlations were established between scenic beauty rankings determined from one group of recreationalists and willingness to pay to recreate determined by a separate group of recreationalists (r^2 ranged from 0.27 to 0.98 depending on ranking). In another study, Walsh et al. (1989, as cited in Rosenthal, NAPAP SOS/T 27 Part B, 1990) developed a functional relationship between reduction of recreational benefits and tree density changes that reflected varying levels of insect damage at six campgrounds in the Front Range of the Colorado Rockies. By using both contingent valuation and travel cost models, Walsh et al. (1989) were able to show that 10 percent, 20 percent, and 30 percent decreases in tree densities reduces the total recreational benefits at their sites by 7 percent, 15 percent and 24 percent, respectively. Although results from these studies are limited to the sites from which they were derived, they do confirm intuition that the degree of visible damage to forests is to some extent correlated with the magnitude of damages to forest-based recreation expected. This finding supports the notion that the avoidance of damages to forest ecosystems from CAA-induced pollution controls (albeit currently unquantified) have likely benefited forest-based recreation in the U.S.

In addition to establishing relationships between recreational value and visible damage to forest sites, there is evidence linking air pollution (ozone) effects on forests to economic damages to non-use values of forests. For example, Peterson et al. (1987, as cited in Rosenthal, NAPAP SOS/T 27 Part B, 1990) valued ozone-included damages to forests surrounding the Los Angeles area. Using contingent valuation methods, Peterson et al. (1987) surveyed recreationalists (a random survey of households in the San Bernardino, Los Angles and Orange counties) and residence (a sample of property owners within the San Bernardino and Angeles national forests) for their willingness to pay to prevent forest scenes from degrading one step on a "forest quality ladder" depicting various levels of ozone-induced damages. The mean willingness to pay to protect further degradation was $37.61 and $119.48 per household for recreationalists and residents, respectively. Annual damages to Los Angeles area residences from a one-step drop on the forest quality ladder were estimated between $27 million and $147 million.

Although these estimates cannot be directly translated into a rough estimate of the potential non-use values of avoided forest damages, they do provide evidence that the recreational and non-use benefits may substantially exceed the commercial timber values.

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Appendix F: Effects of Criteria Pollutants on Agriculture

Introduction

One potential impact of air pollutants on economic welfare is their effect on agricultural crops, including annual and perennial species. Pollutants may affect processes within individual plants that affect growth and reproduction, thereby affecting yields of agricultural crops. Possible physiological effects of pollutants include the following: decreased photosynthesis; changes in carbohydrate allocation; increased foliar leaching; decreased nutrient uptake; increased sensitivity to climatic stress, pests, and pathogens; decreased competitive ability; and decreased reproductive efficiency. These physiological effects, in conjunction with environmental factors and intraspecies differences in susceptibility, may affect crop yields.

Air pollutants that might damage plants include SO_2, NO_x, peroxyacetyl nitrate (PAN), and volatile organic compounds (VOCs). These pollutants may have direct effects on crops, or they may damage crops indirectly by contributing to tropospheric (ground-level) ozone and/or acid deposition, both of which damage plants. Tropospheric ozone is formed by photochemical reactions involving VOCs and NO_x, while SO_2 and NO_x cause acidic deposition.

While all of these air pollutants may inflict incremental stresses on crop plants, in most cases air pollutants other than ozone are not a significant danger to crops. Based primarily on EPA's National Acid Precipitation Assessment Program (NAPAP) conclusions,[193] this analysis considers ozone to be the primary pollutant affecting agricultural production.

This analysis estimates the economic value of the difference in agricultural production that has resulted due to the existence of the CAA since 1970. The analysis is restricted to a subset of agricultural commodities, and excludes those commodity crops for which ozone response data are not available. Fruits, vegetables, ornamentals, and specialty crops are also excluded from this analysis. To estimate the economic value of ozone reductions under the CAA, agricultural production levels expected from control scenario ozone conditions are first compared with those expected to be associated with ozone levels predicted under the no-control scenario. Estimated changes in economic welfare are then calculated based on a comparison of estimated economic benefits associated with each level of production.

[193] Shriner et al., 1990; NAPAP, 1991.

Ozone Concentration Data

To estimate the nationwide crop damages as a result of ozone exposure, the first step is to estimate the nationwide ozone concentrations under the control and no-control scenarios. This section describes the methodology used to estimate ozone concentrations for each county in each of these two scenarios.

First, historical ozone concentration data at the monitor level were compiled from EPA's AIRS system. Differences between the modeled control and no-control scenario ozone concentrations were then used to scale historical data to derive no-control scenario ozone air quality profiles.[194] Next, the ozone index used in the exposure response evaluation was calculated and applied at the monitor level. For this analysis, the W126 index, a peak-weighted average of cumulative ozone concentrations, was selected to conform with the index currently being used by EPA in ozone NAAQS benefits analysis. The W126 index is one of several cumulative statistics, and may correlate more closely to crop damage than do unweighted indices.[195] EPA has not yet made a final determination of the appropriate index to use in agricultural benefits analysis; thus this analysis should be viewed only as an indicator of the magnitude of potential benefits.

The third step in ozone concentration estimation involved the use of triangulation and planar interpolation to arrive at a W126 statistic at the county, rather than at the monitor, level. For each county centroid, the closest surrounding triangle of monitors is located and the W126 is calculated for that county using a distance-weighted average of the ozone concentration at each of these monitors.

Control and No-control Scenario Ozone Concentration Data

The initial estimation of ozone concentrations in the control and no-control scenarios was performed by Systems Applications International (SAI). To create the control scenario, SAI compiled ozone data from the EPA's Aerometric Information and Retrieval System (AIRS).[196] SAI summarized these data by fitting gamma distributions to them and providing the alpha and the beta parameters to these distributions. Each of these distributions describes a set of ozone concentration levels, and the distributions are categorized by year, season, and averaging time. SAI defines six distinct "seasons," each composed of a two month period in the year. This analysis uses those distributions which describe 1-hour average ozone concentrations taken from 7 AM to 7 PM and separated into seasons. The analysis utilizes only those monitor records that were modeled in both the control and no-control scenarios.

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[194] Derivation of these ozone air quality profiles for the control and no-control scenario is summarized in the following section and described in detail in Appendix C.

[195] Lefohn et al., 1988.

[196] SAI, ICF Kaiser, 1995.

To determine the ozone concentrations for the no-control scenario, SAI utilized the Ozone Isopleth Plotting with Optional Mechanisms-IV (OZIPM4) model. The input data required for OZIPM4 includes air quality data, surface and upper-air meteorological data, and estimates of anthropogenic and biogenic emissions of volatile organic compounds, NO_x and CO.[197] To create these inputs, SAI used (among other sources) outputs from the Regional Acid Deposition Model (RADM) and the SJVAQS/AUSPEX Regional Modeling Adaptation project (SARMAP). Additional detail concerning the development of ozone concentration data is available in Appendix C and in the SAI report to EPA.[198]

Calculation of the W126 Statistic

Using the SAI ozone concentration distributions, we calculated a sigmoidally weighted ozone index for each monitor. The generalized sigmoidal weighting function used in calculating such indices is presented in Lefohn and Runeckles (1987) as:

$$w_i = 1 / [1 + M \cdot \exp(-A \cdot i)] \quad (34)$$

where: w_i = weighting factor for concentration I (unitless)
c_i = concentration I (ppm)
M = an arbitrary constant
A = an arbitrary constant

The constants M and A are chosen to give different weights to higher or lower concentrations. The index used in this analysis is the W126 statistic, which is calculated as follows:[199]

$$w_i = 1 / [1 + 4403 \cdot \exp(-126 \cdot c_i)] \quad (35)$$

and

$$W126 = \sum w_i \quad (36)$$

Missing values are accounted for by multiplying the resulting W126 statistic by the ratio of the number of potential observations to the number of actual observations (i.e., total hours in period/hours of data in period).

[197] SAI, ICF Kaiser, 1995.

[198] SAI, ICF Kaiser, 1995.

[199] Lefohn et al., 1988.

To calculate W126 indices from the monitor level gamma distributions, we used an inverse cumulative density function to calculate a separate representative air concentration for each hour in the two month season. These values are then used in the above equation to obtain a monitor-level W126 statistic.

To ensure that the interpretation of the gamma distributions in this manner does not generate errors, we tested our gamma-derived control-scenario W126s against W126s calculated directly from the AIRS database. We found that insignificant error resulted from the utilization of the gamma distributions to create W126 statistics.

Aggregating Ozone Data to the County Level

Because crop production data are available at the county level, the lowest level of aggregation that could be used for ozone indices is also the county level. Therefore, monitor level data needed to be aggregated to a county level. For each county, we first located the monitors from which we would be interpolating data. To identify these monitors, we searched for the three monitors which formed the closest triangle around the centroid of the county.[200] The closest triangle was defined as that triangle in which the sum of the distances from the three monitors to the county centroid was the least. The algorithm stopped searching for closest triangles of monitors when it had searched all monitors within 500 km of a given county centroid (an arbitrary distance, selected to reduce computational requirements).

For coastal counties and some rural counties in some years, monitor triangles around the county centroid do not exist. We assigned the W126 value from the monitor closest to the centroid to these counties. Approximately 15 percent of all county-years (36,973 of 248,880 records) were assigned W126s in this manner.

For the remaining 85 percent, after the closest triangle of monitors was found, a "planar interpolation" was used to calculate the W126 at that county for that year. One way to picture this process is to plot each of the three monitors as a point in space. For each monitor, the x axis represents longitude, the y axis represents latitude and the z axis represents the W126 statistic. A plane can then be drawn through these three points in space. Furthermore, using the equation for the plane, and given the x and y values (latitude and longitude) for the county centroid, the county centroid's z value (W126 statistic) can be calculated. In essence, this procedure calculates a distance-weighted average of three monitors' W126 index values and assigns this value to the county centroid.

The final result of this data manipulation is a monthly W126 statistic for each county in the continental United States for the years 1971-1990. From these data, yield change estimates were generated, and economic impacts were estimated.

[200] The vast majority of monitors had latitude and longitude data available through AIRS. 1,528 of 1,536 monitors were located in this manner. For the remaining 8 monitors, if in a given year of monitor data another monitor exists in the county of the unfound monitor, we discarded the unlocated monitor's data. Otherwise, we located that monitor at the county's centroid. We located 5 of the remaining 8 monitors in this fashion.

Yield Change Estimates

There are several steps involved in generating yield change estimates. The first is the selection of relevant ozone exposure-response functions (minimum and maximum) for each crop in the analysis. Ozone data, triangulated to the county level, are transformed into an index suitable for use in the selected function(s) to estimate county level predicted yield losses for both the control and no-control scenarios. In the next step, the proportion of each county to the national production of each crop is calculated to permit national aggregation of estimated yield losses. Finally, the control scenario percentage relative yield loss (PRYL) is compared to the minimum and maximum PRYL for the no-control scenario. Each step is discussed in more detail below.

Exposure-Response Functions

To estimate yield impacts from ozone, exposure-response functions are required for each crop to be analyzed. This analysis was restricted to estimating changes in yields for those commodity crops for which consistent exposure-response functions are available and that are included in national agricultural sector models. To maintain consistency with the current ozone NAAQS benefits analysis being conducted by OAQPS, NCLAN-based exposure-response functions using a Weibull functional form and a 12-hour W126 ozone index were used.

Several crops included in the NCLAN research program were not evaluated in this analysis. Non-commodity crops that are not modeled in national agricultural sector models were not included in this analysis: lettuce, tomatoes, potatoes, alfalfa, tobacco, turnips, and kidney beans. In addition, one commodity crop, spring wheat, was excluded because the NCLAN exposure-response function was only developed for winter wheat.

Minimum/Maximum Exposure-Response Functions

Estimated responsiveness of a given crop to ozone varies within the NCLAN data. This range of response is partially explained by the program's evaluation of several cultivars for some crops; ozone sensitivity varies across cultivars. In addition, the conditions for different experiments varied due to variations in location, year, and additional treatments included in some experiments. No one exposure-response function can be assumed to be representative of all cultivars in use, or of all environmental conditions for crop production. To develop a range of benefits estimates that reflects this variation in responsiveness, a minimum responsiveness and a maximum responsiveness function were selected for each crop. In actuality, a number of different cultivars are planted by producers, and so ozone response will be a weighted average of the responsiveness of each cultivar to its ozone condition and its proportion of total acreage. It is important to note that these values do not necessarily bound the analysis, since the number of cultivars evaluated by NCLAN is small relative to the number grown for many crops.

For the crops used in this study, CERL conducted an analysis to identify the ozone concentration required to reduce yields by 10 percent for each crop cultivar using its 12-hour W126 exposure-response

function. For each crop, the function demonstrating the lowest ozone concentration at a 10 percent yield loss represents the maximum response, and the function with the highest concentration at 10 percent yield loss represents the minimum response. Table 87 reports the minimum and maximum exposure-response functions for each crop. Two crops, peanuts and sorghum, did not have multiple NCLAN experiments on which to base a comparison of the responsiveness of different cultivars or the variation in response with different experimental conditions.

Table 87. Agriculture Exposure-Response Functions.

Crop	Cultivar	Equation Type	Yield Function (FRYL, ppm)	Duration (days)
Barley	CM-72	Both	$1-\exp(-(W126/6998.5)^{1.288}$	95
Corn-Field	PAG 397	Min	$1-\exp(-(W126/94.2)^{4.176}$	83
Corn-Field	Pioneer 3780	Max	$1-\exp(-(W126/92.7)^{2.585}$	83
Cotton	McNair 235	Min	$1-\exp(-(W126/113.3)^{1.307}$	125
Cotton	Acala SJ2	Max	$1-\exp(-(W126/74.6)^{1.066}$	98
Grain Sorghum	DeKalb A28 +	Both	$1-\exp(-(W126/285.3)^{1.057}$	89
Peanuts	NC-6	Both	$1-\exp(-(W126/96.8)^{1.000}$	112
Soybeans	Corsoy-79	Min	$1-\exp(-(W126/476.7)^{1.315}$	93
Soybeans	Davis	Max	$1-\exp(-(W126/130.1)^{1.000}$	93
Wheat	AET	Min	$1-\exp(-(W126/76.8)^{2.001}$	54
Wheat	VONA	Max	$1-\exp(-(W126/24.7)^{1.409}$	61

Source: EPA/CERL (unpublished) for all functions.

Calculation of Ozone Indices

Each NCLAN ozone exposure-response experiment exposed each studied crop over a portion of the crop's growing season. The duration of the NCLAN experiments was provided by CERL and was rounded to the nearest month. The W126 index is cumulative, and so is sensitive both to the duration over which it is calculated and to the specific month(s) within a growing season that are included in it. Because cropping seasons vary across the U.S., the ozone index used to calculate county-level changes in yield due to ozone must reflect the local season for each crop. To determine which portion of the growing season a particular exposure period pertains to (in order to calculate an exposure index), we developed state-specific growing seasons based on planting and harvesting data developed by USDA.[201] The W126 index was calculated using the county level ozone data developed in the prior section,

[201] USDA, 1984. Some states did not have explicit growing seasons reported for certain crops due to the low production in these states. In these cases a proxy state growing season was used. In most of these cases the proxy growing season was taken from a state with an adjoining boundary within the same geographic region.

summed for the number of months of NCLAN experimental duration, with the exposure period anchored on the usual harvest month for each crop.[202]

Calculations of County Weights

Because the benefits analysis did not require a regional level of disaggregation and to minimize computational burdens the economic analysis was conducted at a national level. Ozone data and estimated yield responses, however, were developed at a county level. To conduct a national analysis, the county level yield change estimates were weighted to develop a single national percent relative yield loss for each crop relative to the control scenario, for both the minimum and the maximum yield responses. As a part of calculating a percent change in yield at the national level, weights for each county and crop were created for 1975, 1980, 1985, and 1990. The weights for these four years were used to represent the year itself and the four years immediately following it (e.g., 1975 weights were also used for 1976, 1977, 1978, and 1979). Although weather and other conditions may change the proportion of counties' production to the total national production in each year, five year weights should reflect stable periods of agricultural policy between each Farm Bill, and are sufficient for the level of precision needed for this analysis. The weights were calculated by dividing the production level of a crop in a county[203] by the sum of all states' reported production for that crop.[204] These county weights were applied to the percent relative yield loss results for each county, as discussed below.

Calculation of Percent Change in Yield

Ozone exposure-response functions are expressed in terms of percent relative yield loss (PRYL); the ozone level being analyzed is compared with "clean" (charcoal filtered/zero ozone) air. To calculate a percent change in yield between the control and no-control scenarios, we first calculate a PRYL based on the county-level control scenario W126 ozone index, and a PRYL based on the no-control scenario W126 index. Next, the county weights are applied to the PRYLs. The change in yield, measured relative to the hypothetical zero-ozone crop production, is then:

$$(PRYL_{CAA} - PRYL_{NCAA}) \qquad (37)$$

To obtain the change in terms of our (non-zero) baseline yield, we divide by that yield, and get:

[202] This analysis required "rounding" some months: if a harvest date was specified to be from the 15th to the end of a month, the W126 index was calculated using that month's data; if the harvest date was specified to be from the first to the 14th of a month, the W126 index was calculated using the prior month's data as the final month in the exposure period.

[203] USDA, 1995.

[204] The national total does not include USDA areas designated "other counties". These areas are groups of counties that for one reason or another (disclosure rules, low amount of production, etc.) are not individually listed. Because we did not have ozone values for these groups, we did not use their production levels in the calculation of the total national production.

$$(PRYL_{CAA} - PRYL_{NCAA}) / (100 - PRYL_{CAA}) \quad (38)$$

To create the national percent change in yield for each crop, the results of this equation are summed for each scenario (maximum and minimum) and for each year. Tables 88 and 89 present the resulting percent yield changes that were used as inputs to the economic model.

Table 88. Relative No-control to Control Percent Yield Change (harvested acres) for the Minimum Scenario.

	Crop						
Year	Barley	Corn	Cotton	Peanuts	Soybeans	Sorghum	Winter Wheat
1975	-0.000020	-0.000171	-0.011936	-0.006635	-0.001166	-0.000717	-0.005631
1976	-0.000013	-0.000329	-0.017505	-0.024048	-0.002171	-0.001841	-0.004841
1977	-0.000013	-0.000169	-0.013114	-0.015150	-0.001562	-0.001118	-0.005464
1978	-0.000019	-0.000291	-0.018692	-0.017606	-0.002480	-0.001844	-0.005894
1979	-0.000027	-0.000100	-0.017217	-0.013067	-0.001898	-0.001389	-0.004998
1980	-0.000019	-0.000200	-0.021315	-0.022761	-0.002397	-0.002222	-0.005385
1981	-0.000016	-0.000071	-0.018552	-0.014269	-0.001951	-0.000802	-0.003964
1982	-0.000020	-0.000070	-0.017295	-0.014200	-0.002107	-0.001050	-0.004773
1983	-0.000023	-0.000617	-0.020842	-0.028601	-0.003901	-0.002368	-0.005904
1984	-0.000027	-0.000111	-0.023552	-0.019225	-0.002919	-0.002881	-0.006121
1985	-0.000025	-0.000132	-0.020947	-0.017965	-0.002643	-0.001726	-0.007316
1986	-0.000029	-0.000158	-0.027968	-0.031605	-0.002899	-0.001564	-0.007597
1987	-0.000033	-0.000358	-0.034584	-0.043854	-0.003776	-0.001812	-0.009669
1988	-0.000027	-0.000662	-0.035069	-0.038085	-0.004563	-0.002922	-0.019873
1989	-0.000024	-0.000150	-0.031245	-0.022094	-0.003769	-0.001359	-0.007605
1990	-0.000024	-0.000210	-0.037988	-0.047395	-0.003819	-0.001567	-0.006449

Note: There is only one scenario for barley, peanuts, and sorghum, because there was only one exposure-response function.

Table 89. Relative No-control to Control Percent Yield Change (harvested acres) for the Maximum Scenario.

Year	Crop						
	Barley	Corn	Cotton	Peanuts	Soybeans	Sorghum	Winter Wheat
1975	-0.000020	-0.001139	-0.021059	-0.006635	-0.005808	-0.000717	-0.034803
1976	-0.000013	-0.002281	-0.032063	-0.024048	-0.010298	-0.001841	-0.040303
1977	-0.000013	-0.001232	-0.025773	-0.015130	-0.007764	-0.001118	-0.049636
1978	-0.000019	-0.002015	-0.033075	-0.017606	-0.011803	-0.001844	-0.050308
1979	-0.000027	-0.001092	-0.031433	-0.013067	-0.009592	-0.001389	-0.052211
1980	-0.000019	-0.001537	-0.037278	-0.022761	-0.011845	-0.002222	-0.054128
1981	-0.000016	-0.000923	-0.035058	-0.014269	-0.009902	-0.000802	-0.053470
1982	-0.000020	-0.000974	-0.034101	-0.014200	-0.010815	-0.001050	-0.058409
1983	-0.000023	-0.003888	-0.040405	-0.028601	-0.018597	-0.002366	-0.063556
1984	-0.000027	-0.001443	-0.043890	-0.019225	-0.014502	-0.002881	-0.067612
1985	-0.000025	-0.001377	-0.040845	-0.017965	-0.013384	-0.001726	-0.072177
1986	-0.000029	-0.001451	-0.052426	-0.031605	-0.014754	-0.001564	-0.081225
1987	-0.000033	-0.002565	-0.061295	-0.043854	-0.018578	-0.001812	-0.089042
1988	-0.000027	-0.004318	-0.061660	-0.038085	-0.021767	-0.002922	-0.120703
1989	-0.000024	-0.001987	-0.059573	-0.022094	-0.018739	-0.001359	-0.086958
1990	-0.000024	-0.002056	-0.071659	-0.047395	-0.018670	-0.001567	-0.082309

Note: There is only one scenario for barley, peanuts, and sorghum, because there was only one exposure-response function.

Economic Impact Estimates

To estimate the economic benefits of controls on ozone precursor pollutants under the Clean Air Act, changes in yields due to those controls need to be evaluated in terms of their effect on agricultural markets. To do this, yield changes can be incorporated into an economic model capable of estimating the associated changes in economic surpluses within the agricultural economy, preferably one that reflects changes in producers' production decisions and demand substitution between crops. This type of dynamic analysis is needed because even small changes in yield or price expectations can cause large shifts in the acreage allocated to specific crops, and the degree to which alternative crops will be substituted (particularly for feed uses).

The modeling approach used in this analysis is to use an econometric model of the agricultural sector, which estimates demand and supply under different production technologies and policy conditions. The AGricultural SImulation Model (AGSIM) has been used extensively to evaluate air pollution impacts, as well as a number of other environmental policy analyses. Its author, Dr. C. Robert

Taylor, modified AGSIM for this analysis to reflect production conditions and policies as they changed through the 20-year span of the Clean Air Act, from 1970 to 1990. During this period, U.S. farm policy parameters changed every five years with the enactment of each Farm Bill, and producer participation varied significantly over the period. Over this time, due to policy, weather, technological development, and other variations, production levels and prices have varied, as have production technologies, costs of production, and relevant cultivars. To reflect these changes, Dr. Taylor re-estimated demand relationships for three periods (1975-79; 1980-84; and 1985-89) based on the farm policies in effect in each period, and structured the model to run on a national level rather than a regional level. The period from 1970-1975 was not modeled because of data limitations and because there was limited impact from the CAA on ozone levels during that period.

The AGSIM baseline production and price data serve as the control scenario baseline. Percent relative yield losses (PRYLs) between the control and no-control scenarios are the relevant input parameter for this analysis, from which AGSIM calculates new yield per planted acre values. Based on these values (as well as on lagged price data, ending stocks from the previous year, and other variables), AGSIM predicts acreage, production, supply, and price parameters for each crop for each year, as well as calculating yield per harvested acre. From these results and the demand relationships embedded in the model, AGSIM calculates the utilization of each crop (i.e., exports, feed use, other domestic use, etc.), as well as the change in consumer surplus, net crop income, deficiency payments and other government support payments. Net surplus is calculated as net crop income plus consumer surplus, less government payments. The first year of results is 1976 because AGSIM must have one year (1975) of lagged data.

Table 90 presents the net *changes* in economic surpluses in both nominal terms and as a cumulative present value (discounted at 3 percent) over the period 1976-1990 due to the Clean Air Act. The positive surpluses exhibited in almost all years are a result of the increase in yields associated with lower ozone levels than those predicted to occur under the no-control scenario. The present value of the estimated agricultural benefits of the CAA ranges between $4.6 billion in the minimum response case to approximately $22 billion in the maximum response case. It should be reiterated that this range represents the impacts that would occur if all of the acreage planted to a given crop had an ozone response function similar to either the minimum *available* response function or the maximum *available* response function. The available response functions do not necessarily bracket the true range of potential crop responses, and it is unrealistic to anticipate that all acreage will be planted in cultivars with a uniform response to ozone exposure. These considerations notwithstanding, these values do indicate the likely magnitude of agricultural benefits associated with control of ozone precursors under the CAA, but not the precise value of those benefits.

Table 90. Change in Farm Program Payments, Net Crop Income, Consumer Surplus, and Net Surplus Due to the CAA (millions nominal $).

	Change in Farm Program Payments		Change in Net Crop Income		Change in Consumer Surplus		Change in Net Surplus	
Year	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum
1976/77	0	0	112	224	109	458	220	682
1977/78	0	0	-48	-128	172	768	125	640
1978/79	23	184	16	159	209	878	202	852
1979/80	0	0	-81	-235	260	1158	179	923
1980/81	0	0	5	-113	249	1302	254	1188
1981/82	78	361	-69	-283	307	1808	161	1164
1982/83	124	726	47	79	279	2020	202	1373
1983/84	118	776	178	737	243	1515	304	1475
1984/85	-146	650	66	450	-224	1356	-11	1156
1985/86	241	1076	151	733	514	1713	424	1371
1986/87	232	1163	197	829	396	1943	362	1609
1987/88	415	1799	283	1243	626	2645	494	2089
1988/89	511	1855	290	1383	730	2698	510	2226
1989/90	316	1345	155	589	506	2469	344	1713
1990/91	414	1927	180	473	618	3047	384	1593
Cumulative Present Value of Net Surplus at 3 percent ($ 1990)							4554	21892

Conclusions

Agricultural benefits associated with control of ozone precursors under the Clean Air Act are likely to be fairly large. Because it is possible that over time producers have adopted more ozone-resistant cultivars, it may be appropriate to consider the lower end of the range of predicted benefits to be more indicative of the likely total benefits. The estimates developed in this analysis, however, do not represent all of the likely benefits accruing to agriculture, in that many high-value and/or ozone sensitive crops could not be included in the analysis due to either exposure-response data limitations or agricultural sector modeling limitations. The second consideration implies that benefits will likely be larger than estimated. The minimum case may be the most appropriate starting point, however, due to the first consideration: the current crop mix is probably biased toward lower ozone responsiveness. Therefore, we anticipate that cumulative total agricultural benefits from the Clean Air Act are on the order of under ten billion dollars (real terms).

Table 91. AGSIM Output: Barley Acreage.

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
1976/77	9.3	0.00%	NA	8.4	0.00%	NA	0.00	NA	NA	0.0	NA	NA
1977/78	10.8	0.00%	16.13%	9.7	0.00%	15.48%	0.00	NA	NA	0.0	NA	NA
1978/79	10.0	0.00%	-7.41%	9.3	0.00%	-4.12%	0.10	0.0%	NA	1.0	0.0%	NA
1979/80	8.1	0.00%	-19.00%	7.5	0.00%	-19.35%	0.20	0.0%	100.0%	1.2	0.0%	20.0%
1980/81	8.3	0.00%	2.47%	7.3	0.00%	-2.67%	0.00	NA	-100.0%	0.0	NA	-100.0%
1981/82	9.6	0.00%	15.66%	9.0	0.00%	23.29%	0.00	NA	NA	0.0	NA	NA
1982/83	9.5	-1.04%	-1.04%	9.0	0.00%	0.00%	0.10	0.0%	NA	0.4	0.0%	NA
1983/84	10.4	0.00%	9.11%	9.7	0.00%	6.67%	0.10	0.0%	-100.0%	1.1	0.0%	-100.0%
1984/85	12.0	0.84%	29.03%	11.3	0.00%	34.52%	0.05	-50.0%	NA	0.4	-33.3%	NA
1985/86	13.1	0.00%	9.17%	11.6	0.00%	2.65%	0.13	30.0%	160.0%	1.0	11.1%	150.0%
1986/87	13.0	0.00%	-0.76%	12.0	0.00%	3.45%	0.17	0.0%	30.8%	2.4	0.0%	140.0%
1987/88	10.9	0.00%	-16.15%	10.0	0.00%	-16.67%	0.20	0.0%	17.6%	3.6	0.0%	50.0%
1988/89	9.8	0.00%	-10.09%	7.6	0.00%	-24.00%	0.20	0.0%	0.0%	3.4	0.0%	-5.6%
1989/90	9.1	0.00%	-7.14%	8.3	0.00%	9.21%	0.10	0.0%	-50.0%	2.4	0.0%	-29.4%
1990/91	8.2	0.00%	-9.89%	7.5	0.00%	-9.64%	0.10	0.0%	0.0%	3.1	0.0%	29.2%

Table 92. AGSIM Output: Barley Production, Supply, and Stocks.

Year	Production			Total Supply			Ending Stocks		
	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year
976/77	392	0.00%	NA	520	0.00%	NA	126	0.00%	NA
977/78	435	0.00%	10.97%	561	0.00%	7.88%	175	0.00%	37.30%
978/79	462	0.00%	6.21%	635	0.00%	13.19%	228	0.00%	31.79%
979/80	390	-0.26%	-15.58%	619	0.00%	-2.52%	192	0.00%	-15.79%
980/81	364	0.00%	-6.67%	556	0.00%	-10.18%	137	0.00%	-28.65%
981/82	481	0.00%	32.14%	618	0.00%	11.15%	148	0.00%	8.03%
982/83	520	0.00%	8.11%	668	0.00%	8.09%	217	0.00%	46.62%
983/84	508	0.00%	-2.31%	724	-0.14%	8.38%	189	0.00%	-12.90%
984/85	608	1.00%	19.69%	796	0.63%	9.94%	230	1.21%	22.28%
985/86	586	-0.34%	-3.62%	837	0.24%	5.15%	328	0.31%	31.20%
986/87	615	-0.16%	4.95%	944	0.11%	12.78%	337	0.30%	2.74%
987/88	533	0.00%	-13.33%	869	0.00%	-7.94%	321	0.00%	-4.75%
988/89	300	0.00%	-43.71%	621	0.00%	-28.54%	196	0.00%	-38.94%
989/90	416	-0.24%	38.67%	612	-0.16%	-1.45%	160	-0.62%	-18.37%
990/91	435	-0.23%	4.57%	595	-0.17%	-2.61%	134	-0.74%	-16.25%

Table 93. AGSIM Output: Barley Prices.

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/bu)	Difference From Baseline	Change From Previous Year	Price ($/bu)	Difference From Baseline	Change From Previous Year	Price ($/bu)	Change From Previous Year	Price ($/bu)	Change From Previous Year
976/77	$2.25	0.00%	NA	$1.31	0.00%	NA	$1.28	NA	$1.22	NA
977/78	$1.78	0.00%	-20.89%	$0.97	0.00%	-25.95%	$2.15	68.0%	$1.63	33.6%
978/79	$1.92	0.00%	7.87%	$0.96	0.00%	-1.03%	$2.25	4.7%	$1.63	0.0%
979/80	$2.27	0.00%	18.23%	$1.05	0.00%	9.38%	$2.40	6.7%	$1.71	4.9%
980/81	$2.79	0.00%	22.91%	$1.18	0.00%	12.36%	$2.55	6.2%	$1.83	7.0%
981/82	$2.48	0.00%	-11.11%	$0.95	0.00%	-19.49%	$2.60	2.0%	$1.95	6.6%
982/83	$2.18	0.00%	-12.10%	$0.79	0.00%	-16.84%	$2.60	0.0%	$2.08	6.7%
983/84	$2.47	0.00%	13.30%	$0.86	0.00%	8.86%	$2.60	0.0%	$2.16	3.8%
984/85	$2.26	-1.31%	-8.50%	$0.75	-1.32%	-13.79%	$2.60	0.0%	$2.08	-3.7%
985/86	$1.98	0.00%	-12.39%	$0.64	0.00%	-14.67%	$2.60	0.0%	$2.08	0.0%
986/87	$1.61	0.00%	-18.69%	$0.50	0.00%	-21.88%	$2.60	0.0%	$1.56	-25.0%
987/88	$1.81	0.00%	12.42%	$0.55	0.00%	10.00%	$2.60	0.0%	$1.49	-4.5%
988/89	$2.80	0.00%	54.70%	$0.82	0.00%	49.09%	$2.51	-3.5%	$1.44	-3.4%
989/90	$2.43	0.41%	-13.21%	$0.69	0.00%	-17.07%	$2.44	-2.8%	$1.34	-6.9%
990/91	$2.15	0.47%	-11.52%	$0.59	1.75%	-14.71%	$2.36	-3.3%	$1.28	-4.5%

Table 94. AGSIM Output: Corn Acreage (Minimum Scenario).

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
1976/77	84.7	0.12%	NA	71.6	0.14%	NA	0.00	NA	NA	0.0	NA	NA
1977/78	84.3	0.00%	-0.47%	71.6	0.00%	0.00%	0.00	NA	NA	0.0	NA	NA
1978/79	81.7	0.00%	-3.08%	71.9	0.00%	0.42%	0.10	0.0%	NA	8.3	0.0%	NA
1979/80	81.4	0.00%	-0.37%	72.4	0.00%	0.70%	0.10	0.0%	0.0%	4.1	0.0%	-50.6%
1980/81	84.0	0.00%	3.19%	73.0	0.00%	0.83%	0.00	NA	-100.0%	0.0	NA	-100.0%
1981/82	84.1	0.00%	0.12%	74.5	0.00%	2.05%	0.00	NA	NA	0.0	NA	NA
1982/83	81.8	-0.12%	-2.73%	72.7	0.00%	-2.42%	0.10	0.0%	NA	2.1	0.0%	NA
1983/84	60.2	0.00%	-26.41%	51.5	0.00%	-29.16%	0.10	0.0%	0.0%	32.2	0.0%	1433.3%
1984/85	82.2	2.11%	36.54%	73.6	2.36%	42.91%	0.05	-50.0%	-50.0%	3.9	-43.5%	-87.9%
1985/86	82.1	-1.56%	-0.12%	73.9	-1.73%	0.41%	0.13	30.0%	160.0%	11.4	23.9%	192.3%
1986/87	76.5	-0.13%	-6.82%	68.8	-0.15%	-6.90%	0.17	0.0%	30.8%	20.9	0.0%	83.3%
1987/88	66.2	0.00%	-13.46%	59.5	0.00%	-13.52%	0.20	0.0%	17.6%	33.1	0.0%	58.4%
1988/89	67.2	0.00%	1.51%	57.8	0.00%	-2.86%	0.20	0.0%	0.0%	30.9	0.0%	-6.6%
1989/90	72.1	0.00%	7.29%	64.5	-0.15%	11.59%	0.10	0.0%	-50.0%	13.1	0.0%	-57.6%
1990/91	74.3	0.00%	3.05%	66.9	0.00%	3.72%	0.10	0.0%	0.0%	15.0	-0.7%	14.5%

Table 95. AGSIM Output: Corn Acreage (Maximum Scenario).

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
976/77	84.8	0.24%	NA	71.8	0.42%	NA	0.00	NA	NA	0.0	NA	NA
977/78	84.4	0.12%	-0.47%	71.7	0.14%	-0.14%	0.00	NA	NA	0.0	NA	NA
978/79	81.6	-0.12%	-3.32%	71.9	0.00%	0.28%	0.10	0.0%	NA	8.3	0.00%	NA
979/80	81.4	0.00%	-0.25%	72.4	0.00%	0.70%	0.10	0.0%	0.0%	4.1	0.00%	-50.60%
980/81	84.0	0.00%	3.19%	72.9	-0.14%	0.69%	0.00	NA	-100.0%	0.0	NA	-100.00%
981/82	84.1	0.00%	0.12%	74.5	0.00%	2.19%	0.00	NA	NA	0.0	NA	NA
982/83	81.7	-0.24%	-2.85%	72.6	-0.14%	-2.55%	0.10	0.0%	NA	2.1	0.00%	NA
983/84	60.1	-0.17%	-26.44%	51.5	-0.39%	-29.34%	0.10	0.0%	0.0%	32.2	0.00%	1433.33%
984/85	82.3	2.24%	36.94%	73.6	2.36%	43.47%	0.05	-50.0%	-50.0%	3.9	-43.48%	-87.89%
985/86	83.2	-0.24%	1.09%	75.0	-0.27%	1.90%	0.10	0.0%	100.0%	9.4	2.17%	141.03%
986/87	76.3	-0.59%	-8.29%	68.7	-0.39%	-8.40%	0.17	0.0%	70.0%	21.1	0.96%	124.47%
987/88	66.0	-0.30%	-13.50%	59.3	-0.34%	-13.68%	0.20	0.0%	17.6%	29.2	0.30%	37.35%
988/89	67.1	-0.15%	1.67%	57.7	-0.17%	-2.70%	0.20	0.0%	0.0%	31.0	0.32%	6.61%
989/90	72.0	-0.14%	7.30%	64.5	-0.15%	11.79%	0.10	0.0%	-50.0%	13.1	0.00%	-57.74%
990/91	74.1	-0.13%	2.92%	66.2	-0.15%	2.72%	0.10	0.0%	0.0%	15.0	-0.06%	14.50%

Table 96. AGSIM Output: Corn Production, Supply, and Stocks (Minimum Scenario).

Year	Production			Total Supply			Ending Stocks		
	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year
976/77	6252	0.05%	NA	6928	0.06%	NA	1137	0.18%	NA
977/78	6506	0.00%	3.40%	7645	0.03%	10.35%	1438	0.14%	26.47%
978/79	7265	-0.04%	11.67%	8704	-0.01%	13.85%	1709	-0.06%	18.85%
979/80	7925	-0.04%	9.08%	9635	-0.04%	10.70%	2031	-0.15%	18.84%
980/81	6638	-0.02%	-16.24%	8669	-0.06%	-10.03%	1368	-0.29%	-31.66%
981/82	8118	-0.02%	22.30%	9507	-0.05%	9.67%	2533	-0.16%	82.49%
982/83	8231	-0.04%	1.39%	10769	-0.07%	13.22%	3517	-0.17%	38.85%
983/84	4172	-0.07%	-49.31%	7691	-0.12%	-28.55%	1001	-0.69%	-71.54%
984/85	7851	2.52%	88.18%	8823	1.97%	15.11%	1776	7.64%	77.42%
985/86	8731	-1.72%	11.08%	10506	-0.27%	19.07%	4026	-0.35%	126.69%
986/87	8216	-0.11%	-5.79%	12244	-0.19%	16.54%	4864	-0.37%	20.81%
987/88	7132	0.00%	-13.19%	11999	-0.15%	-2.00%	4246	-0.33%	-12.71%
988/89	4885	-0.06%	-31.51%	9134	-0.19%	-23.88%	1887	-0.68%	-55.56%
989/90	7508	-0.09%	53.68%	9394	-0.21%	2.85%	1294	-1.22%	-31.43%
990/91	7952	-0.01%	5.96%	9250	-0.18%	-1.53%	1480	-0.94%	14.37%

Table 97. AGSIM Output: Corn Production, Supply, and Stocks (Maximum Scenario).

Year	Production			Total Supply			Ending Stocks		
	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year
976/77	6296	0.11%	NA	6931	0.10%	NA	1140	0.44%	NA
977/78	6505	-0.03%	3.32%	7647	0.05%	10.33%	1439	0.21%	26.23%
978/79	7246	-0.30%	11.39%	8687	-0.21%	13.60%	1696	-0.82%	17.86%
979/80	7916	-0.15%	9.25%	9613	-0.27%	10.66%	2014	-0.98%	18.75%
980/81	6628	-0.17%	-16.27%	8642	-0.37%	-10.10%	1367	-1.80%	-32.13%
981/82	8108	-0.15%	22.33%	9476	-0.38%	9.65%	2509	-1.10%	83.54%
982/83	8212	-0.27%	1.28%	10722	-0.46%	13.15%	3485	-1.08%	38.90%
983/84	4148	-0.65%	-49.49%	7635	-0.84%	-28.79%	958	-4.86%	-72.51%
984/85	7547	2.27%	89.18%	8858	1.45%	15.36%	1741	5.32%	81.73%
985/86	8539	-0.39%	12.64%	10869	0.32%	20.22%	4085	1.11%	134.64%
986/87	8184	-0.30%	-7.41%	12270	0.02%	12.87%	4896	0.08%	19.61%
987/88	7092	-0.36%	-13.34%	11982	-0.29%	-2.35%	4233	-0.63%	-13.56%
988/89	4857	-0.63%	-31.51%	9094	-0.62%	-24.10%	1856	-2.32%	-56.15%
989/90	7483	-0.39%	54.07%	9342	-0.76%	2.73%	1254	-4.27%	-32.44%
990/91	7915	-0.35%	5.91%	9187	-0.92%	-1.71%	1428	-4.43%	13.88%

Table 98. AGSIM Output: Corn Prices (Minimum Scenario).

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/bu)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/bu)	Change From Previous Year	Price ($/bu)	Change From Previous Year
976/77	$2.15	0.00%	NA	$1.25	0.00%	NA	$1.57	NA	$1.50	NA
977/78	$2.02	0.00%	-6.05%	$1.10	0.00%	-12.00%	$2.00	27.4%	$2.00	33.3%
978/79	$2.25	0.00%	11.39%	$1.13	0.00%	2.73%	$2.10	5.0%	$2.00	0.0%
979/80	$2.48	0.00%	10.22%	$1.15	0.00%	1.77%	$2.20	4.8%	$2.10	5.0%
980/81	$3.12	0.00%	25.81%	$1.32	0.00%	14.78%	$2.35	6.8%	$2.25	7.1%
981/82	$2.47	0.00%	-20.83%	$0.95	0.00%	-28.03%	$2.40	2.1%	$2.40	6.7%
982/83	$2.33	0.00%	-5.24%	$0.92	0.00%	-3.16%	$2.70	12.5%	$2.53	5.2%
983/84	$3.21	0.00%	25.88%	$1.12	0.00%	21.74%	$2.86	5.9%	$2.65	3.9%
984/85	$2.37	-2.28%	-19.94%	$0.86	-2.27%	-23.21%	$3.03	5.9%	$2.55	-3.8%
985/86	$2.26	1.35%	-12.06%	$0.73	1.39%	-15.12%	$3.03	0.0%	$2.55	0.0%
986/87	$1.51	0.67%	-33.19%	$0.47	0.00%	-35.62%	$3.03	0.0%	$1.92	-24.7%
987/88	$1.94	0.00%	28.48%	$0.59	0.00%	25.53%	$3.03	0.0%	$1.82	-5.2%
988/89	$2.36	0.00%	21.96%	$0.75	0.00%	27.12%	$2.93	-3.3%	$1.77	-2.7%
989/90	$2.38	0.42%	7.03%	$0.66	0.00%	-12.00%	$2.84	-3.1%	$1.65	-6.8%
990/91	$2.29	0.00%	-3.78%	$0.68	1.64%	3.06%	$2.75	-3.2%	$1.57	-4.8%

Table 99. AGSIM Output: Corn Prices (Maximum Scenario).

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/bu)	Difference From Baseline	Change From Previous Year	Price ($/Bu)	Difference From Baseline	Change From Previous Year	Price ($/bu)	Change From Previous Year	Price ($/bu)	Change From Previous Year
976/77	$2.15	0.00%	NA	$1.25	0.00%	NA	$1.57	NA	$1.50	NA
977/78	$2.02	0.00%	-6.05%	$1.10	0.00%	-12.00%	$2.00	27.4%	$2.00	33.3%
978/79	$2.25	0.00%	11.39%	$1.13	0.00%	2.73%	$2.10	5.0%	$2.00	0.0%
979/80	$2.49	0.40%	10.67%	$1.15	0.00%	1.77%	$2.20	4.8%	$2.10	5.0%
980/81	$3.13	0.32%	25.70%	$1.32	0.00%	14.78%	$2.35	6.8%	$2.25	7.1%
981/82	$2.48	0.40%	-20.77%	$0.95	0.00%	-28.03%	$2.40	2.1%	$2.40	6.7%
982/83	$2.56	0.39%	3.23%	$0.93	1.09%	-2.11%	$2.70	12.5%	$2.55	6.2%
983/84	$3.23	0.62%	26.17%	$1.12	0.00%	20.43%	$2.86	5.9%	$2.65	3.9%
984/85	$2.58	-1.90%	-20.12%	$0.86	-2.27%	-23.21%	$3.00	5.0%	$2.55	-3.8%
985/86	$2.22	-0.45%	-13.95%	$0.71	-1.39%	-17.44%	$3.00	0.0%	$2.55	0.0%
986/87	$1.50	0.00%	-32.43%	$0.47	0.00%	-33.80%	$3.00	0.0%	$1.92	-24.7%
987/88	$1.95	0.32%	30.00%	$0.59	0.00%	25.53%	$3.00	0.0%	$1.82	-5.2%
988/89	$2.54	0.78%	32.31%	$0.75	0.00%	27.12%	$2.93	-2.3%	$1.77	-2.7%
989/90	$2.40	1.27%	-6.98%	$0.67	1.52%	-10.67%	$2.84	-3.1%	$1.65	-6.8%
990/91	$2.32	1.31%	-3.33%	$0.62	1.64%	-7.46%	$2.75	-3.2%	$1.57	-4.8%

Table 100. AGSIM Output: Cotton Acreage (Minimum Scenario).

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
976/77	11.5	-0.86%	NA	10.8	-0.92%	NA	0.00	NA	NA	0.0	NA	NA
977/78	13.6	0.00%	18.26%	13.2	0.00%	22.22%	0.00	NA	NA	0.0	NA	NA
978/79	13.3	0.00%	-2.21%	12.3	0.00%	-6.82%	0.00	NA	NA	0.3	0.0%	NA
979/80	13.9	0.00%	4.51%	12.8	0.79%	4.07%	-0.05	NA	NA	0.0	NA	-100.0%
980/81	14.5	0.00%	4.32%	13.2	0.00%	3.12%	0.00	NA	NA	0.0	NA	NA
981/82	14.3	0.00%	-1.38%	13.8	0.00%	4.55%	0.00	NA	NA	0.0	NA	NA
982/83	11.4	0.00%	-20.28%	9.8	-1.03%	-28.99%	0.15	0.0%	NA	1.6	0.0%	NA
983/84	7.9	0.00%	-30.70%	7.3	0.00%	10.20%	0.20	0.0%	-100.0%	7.3	0.0%	-100.0%
984/85	11.0	-0.90%	39.24%	10.3	-0.96%	4.43%	0.25	0.0%	NA	4.1	0.0%	NA
985/86	10.7	0.00%	-2.73%	10.2	0.00%	-0.97%	0.20	0.0%	-20.0%	4.2	0.0%	2.4%
986/87	10.0	0.00%	-6.54%	8.4	-1.18%	-17.68%	0.25	0.0%	25.0%	5.4	0.0%	28.6%
987/88	10.3	-0.96%	3.00%	10.0	0.00%	19.05%	0.25	0.0%	0.0%	5.2	0.0%	-3.7%
988/89	12.5	0.00%	21.36%	12.0	0.00%	20.00%	0.13	0.0%	-48.0%	1.9	0.0%	-63.5%
989/90	10.7	0.94%	-14.40%	9.6	-1.05%	-20.00%	0.25	0.0%	92.3%	4.8	0.0%	152.6%
990/91	12.3	0.00%	14.95%	11.7	0.00%	21.87%	0.13	0.0%	-48.0%	3.0	0.0%	-37.5%

Table 101. AGSIM Output: Cotton Acreage (Maximum Scenario).

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
976/77	11.5	-0.86%	NA	10.7	-1.83%	NA	0.00	NA	NA	0.0	NA	NA
977/78	13.6	0.00%	18.26%	13.2	0.00%	23.36%	0.00	NA	NA	0.0	NA	NA
978/79	13.3	0.00%	-2.21%	12.9	0.00%	-6.82%	0.00	NA	NA	0.3	0.00%	NA
979/80	13.9	0.00%	4.51%	12.8	0.79%	4.07%	-0.05	NA	NA	0.0	NA	-100.00%
980/81	14.6	0.69%	5.04%	13.2	0.00%	3.12%	0.00	NA	NA	0.0	NA	NA
981/82	14.3	0.00%	-2.05%	13.8	0.00%	4.35%	-0.05	NA	NA	0.0	NA	NA
982/83	11.4	0.00%	-20.28%	9.8	1.03%	-28.99%	0.15	0.0%	NA	1.6	0.00%	NA
983/84	7.9	0.00%	-30.70%	7.3	0.00%	9.18%	0.20	0.0%	-100.0%	7.3	0.00%	-100.00%
984/85	11.0	-0.90%	39.24%	10.2	-1.92%	4.57%	0.25	0.0%	NA	4.1	0.00%	NA
985/86	10.7	0.00%	-2.73%	10.2	0.00%	0.00%	0.20	0.0%	-20.0%	4.2	0.00%	2.44%
986/87	10.0	0.00%	-6.54%	8.4	-1.18%	-17.65%	0.25	0.0%	25.0%	5.4	0.00%	28.57%
987/88	10.3	-0.96%	3.00%	9.9	-1.00%	17.86%	0.25	0.0%	0.0%	5.3	-1.92%	-1.85%
988/89	12.5	0.00%	21.36%	12.0	0.00%	21.21%	0.13	0.0%	-48.0%	1.9	0.00%	-64.15%
989/90	10.7	0.94%	-14.40%	9.7	2.11%	-19.17%	0.25	0.0%	93.3%	4.8	0.00%	152.63%
990/91	12.3	0.00%	14.95%	11.7	0.00%	20.62%	0.15	0.0%	-40.0%	3.0	0.00%	-37.50%

Table 102. AGSIM Output: Cotton Lint Production, Supply, and Stocks (Minimum Scenario).

Year	Production			Total Supply			Ending Stocks		
	Amount (Thousands of bales)	Difference From Baseline	Change From Previous Year	Amount (Thousands of bales)	Difference From Baseline	Change From Previous Year	Amount (Thousands of bales)	Difference From Baseline	Change From Previous Year
976/77	10659	-2.47%	NA	14072	-1.88%	NA	2705	-6.04%	NA
977/78	14158	-1.41%	32.77%	16864	-2.19%	19.87%	5048	-4.36%	86.62%
978/79	10923	-1.79%	-22.82%	16007	-2.62%	-5.10%	3661	-6.25%	-27.48%
979/80	14561	-1.53%	32.92%	18227	-2.51%	13.87%	2711	-8.47%	-25.95%
980/81	11203	-2.08%	-23.06%	13943	-4.38%	-23.50%	2378	-9.03%	-12.28%
981/82	13470	-1.83%	20.19%	17874	-2.85%	28.19%	6329	-3.62%	166.15%
982/83	11940	-1.14%	-23.46%	18190	-2.01%	1.77%	7727	-1.49%	22.09%
983/84	7346	-2.19%	-57.62%	15125	-1.83%	-16.85%	2643	-1.86%	-65.80%
984/85	12639	-3.24%	71.12%	15306	-3.00%	1.20%	3832	-4.27%	45.74%
985/86	13315	-2.03%	5.35%	17201	-2.53%	12.38%	9146	-1.54%	137.44%
986/87	9482	-3.26%	-28.79%	18832	-2.42%	9.52%	4799	-2.89%	-47.53%
987/88	14380	-4.08%	51.66%	19182	-3.78%	2.95%	5397	-5.61%	12.46%
988/89	14726	-3.53%	2.41%	20129	-3.98%	4.94%	6678	-4.95%	23.74%
989/90	11970	-4.09%	-18.72%	18651	-3.13%	-7.34%	2612	-6.58%	-60.89%
990/91	15271	-4.05%	27.58%	17899	-4.41%	-4.09%	1939	-14.17%	-25.77%

Table 103. AGSIM Output: Cotton Lint Production, Supply, and Stocks (Maximum Scenario).

Year	Production			Total Supply			Ending Stocks		
	Amount (Thousands of bales)	Difference From Baseline	Change From Previous Year	Amount (Thousands of bales)	Difference From Baseline	Change From Previous Year	Amount (Thousands of bales)	Difference From Baseline	Change From Previous Year
976/77	10263	-4.30%	NA	13876	-3.22%	NA	2576	-16.52%	NA
977/78	13957	-2.81%	35.99%	16540	-4.09%	19.20%	4844	-8.22%	88.04%
978/79	10811	-3.08%	-22.54%	15680	-4.73%	-5.32%	3464	-11.29%	-28.49%
979/80	14377	-2.77%	32.98%	17847	-4.33%	13.97%	2959	-15.29%	-27.57%
980/81	11031	-3.58%	-23.27%	13570	-4.97%	-23.96%	2202	-15.76%	-12.24%
981/82	15197	-3.54%	37.77%	17425	-4.29%	28.41%	6119	-6.82%	177.88%
982/83	11677	-3.50%	-23.16%	17817	-4.02%	2.25%	7537	-3.28%	23.59%
983/84	7236	-4.18%	-38.04%	14834	-3.72%	-16.74%	2567	-4.68%	-66.17%
984/85	12317	-5.70%	70.24%	14909	-5.51%	0.51%	3712	-7.75%	44.60%
985/86	13027	-4.15%	5.76%	16773	-4.96%	12.50%	8998	-3.13%	142.40%
986/87	9232	-5.82%	-29.13%	18233	-4.51%	8.70%	4680	-5.30%	-47.99%
987/88	13933	-7.06%	50.92%	18618	-6.40%	2.10%	5175	-9.50%	10.58%
988/89	14325	-5.96%	2.81%	19906	-6.92%	6.78%	6433	-8.44%	24.31%
989/90	11664	-4.40%	-18.58%	18100	-4.99%	-7.21%	2436	-13.23%	-62.29%
990/91	14680	-7.70%	25.84%	17178	-8.53%	-5.43%	1693	-28.15%	-33.10%

Table 104. AGSIM Output: Cotton Lint Prices (Minimum Scenario).

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Change From Previous Year	Price ($/lb)	Change From Previous Year
976/77	$0.65	1.56%	NA	$0.38	2.70%	NA	$0.42	NA	$0.37	NA
977/78	$0.53	1.92%	-18.46%	$0.29	3.57%	-23.68%	$0.48	14.3%	$0.43	16.2%
978/79	$0.59	1.72%	11.32%	$0.30	3.45%	3.45%	$0.52	8.3%	$0.48	11.6%
979/80	$0.64	1.59%	8.47%	$0.30	3.45%	0.00%	$0.58	11.5%	$0.50	4.2%
980/81	$0.76	1.33%	18.75%	$0.32	0.00%	6.67%	$0.58	0.0%	$0.48	-4.0%
981/82	$0.56	3.70%	-26.32%	$0.21	0.00%	-34.38%	$0.71	22.4%	$0.52	8.3%
982/83	$0.60	1.69%	7.14%	$0.22	4.76%	4.76%	$0.71	0.0%	$0.57	9.6%
983/84	$0.66	0.00%	10.00%	$0.23	0.00%	4.55%	$0.76	7.0%	$0.55	-3.5%
984/85	$0.59	1.72%	-10.61%	$0.20	9.26%	-13.04%	$0.81	6.6%	$0.55	0.0%
985/86	$0.57	1.79%	-3.39%	$0.18	0.00%	-10.00%	$0.81	0.0%	$0.57	3.6%
986/87	$0.53	1.92%	-7.02%	$0.17	6.25%	-5.56%	$0.81	0.0%	$0.55	-3.5%
987/88	$0.67	4.49%	26.42%	$0.20	5.26%	17.65%	$0.79	-2.5%	$0.52	-5.5%
988/89	$0.60	3.26%	-10.45%	$0.18	5.88%	-10.00%	$0.76	-3.8%	$0.52	0.0%
989/90	$0.68	3.03%	13.33%	$0.19	5.56%	5.56%	$0.73	-3.9%	$0.50	-3.8%
990/91	$0.70	4.48%	2.94%	$0.19	5.56%	0.00%	$0.73	0.0%	$0.50	0.0%

Table 105. AGSIM Output: Cotton Lint Prices (Maximum Scenario).

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Change From Previous Year	Price ($/lb)	Change From Previous Year
976/77	$0.65	1.56%	NA	$0.38	2.70%	NA	$0.42	NA	$0.37	NA
977/78	$0.54	3.85%	-16.92%	$0.29	3.57%	-23.68%	$0.48	14.3%	$0.43	16.2%
978/79	$0.60	3.45%	11.11%	$0.30	3.45%	3.45%	$0.52	8.3%	$0.48	11.6%
979/80	$0.66	4.76%	10.00%	$0.30	3.45%	0.00%	$0.58	11.5%	$0.50	4.2%
980/81	$0.76	4.00%	15.15%	$0.33	3.13%	10.00%	$0.58	0.0%	$0.48	-4.0%
981/82	$0.57	5.56%	-26.92%	$0.22	4.76%	-33.33%	$0.71	22.4%	$0.52	8.3%
982/83	$0.61	3.39%	7.02%	$0.22	4.76%	0.00%	$0.71	0.0%	$0.57	9.6%
983/84	$0.67	1.52%	9.84%	$0.23	0.00%	4.55%	$0.76	7.0%	$0.55	-3.5%
984/85	$0.60	3.45%	-10.45%	$0.20	5.26%	-13.04%	$0.81	6.6%	$0.55	0.0%
985/86	$0.58	3.57%	-3.33%	$0.19	5.56%	-5.00%	$0.81	0.0%	$0.57	3.6%
986/87	$0.54	3.85%	-6.90%	$0.17	6.25%	-10.53%	$0.81	0.0%	$0.55	-3.5%
987/88	$0.69	7.81%	27.78%	$0.21	10.53%	23.53%	$0.79	-2.5%	$0.52	-5.5%
988/89	$0.62	8.77%	-10.14%	$0.18	5.88%	-14.29%	$0.76	-3.8%	$0.52	0.0%
989/90	$0.69	4.55%	11.29%	$0.19	5.56%	5.56%	$0.73	-3.9%	$0.50	-3.8%
990/91	$0.73	8.96%	5.80%	$0.20	11.11%	5.26%	$0.73	0.0%	$0.50	0.0%

Table 106. AGSIM Output: Cottonseed Production, Supply, and Stocks (Minimum Scenario).

Year	Production			Total Supply			Ending Stocks		
	Amount (millions of tons)	Difference From Baseline	Change From Previous Year	Amount (Millions of tons)	Difference From Baseline	Change From Previous Year	Amount (Millions of tons)	Difference From Baseline	Change From Previous Year
976/77	4092	-0.73%	NA	4295	-0.69%	NA	279	-1.41%	NA
977/78	5515	-0.11%	34.78%	5794	-0.17%	34.90%	815	-0.24%	192.11%
978/79	4272	0.07%	-22.54%	5087	0.02%	-12.20%	518	-0.38%	-36.44%
979/80	5789	0.19%	35.51%	6308	0.16%	24.00%	1057	-0.09%	104.05%
980/81	4473	0.04%	-22.73%	5531	0.04%	-12.32%	397	-0.25%	-62.44%
981/82	6599	-0.03%	43.04%	6794	0.00%	22.85%	780	-0.13%	96.47%
982/83	4773	0.61%	-27.41%	5552	0.49%	-18.29%	373	0.54%	-52.18%
983/84	3073	-0.10%	-35.62%	3446	-0.03%	-37.93%	114	-1.72%	-69.44%
984/85	5102	-0.51%	66.03%	5216	-0.51%	51.36%	399	-1.72%	250.00%
985/86	5283	0.08%	3.55%	5682	-0.05%	8.93%	347	0.00%	-13.03%
986/87	3788	-0.47%	-28.30%	4129	-0.46%	-27.33%	186	-1.59%	-46.40%
987/88	5732	-0.64%	51.32%	5918	-0.67%	43.33%	353	-1.67%	89.78%
988/89	6073	0.18%	5.95%	6426	0.08%	8.58%	663	-0.30%	87.82%
989/90	4727	1.07%	-22.16%	5390	0.90%	-16.12%	368	0.55%	-44.49%
990/91	5903	-0.25%	24.88%	6323	-0.21%	17.31%	647	-0.46%	75.82%

Table 107. AGSIM Output: Cottonseed Production, Supply, and Stocks (Maximum Scenario).

Year	Production			Total Supply			Ending Stocks		
	Amount (Millions of tons)	Difference From Baseline	Change From Previous Year	Amount (Millions of tons)	Difference From Baseline	Change From Previous Year	Amount (Millions of tons)	Difference From Baseline	Change From Previous Year
976/77	4075	-1.14%	NA	4278	-1.09%	NA	275	-2.83%	NA
977/78	5508	-0.24%	35.17%	5782	-0.38%	35.16%	809	-0.98%	194.18%
978/79	4279	0.23%	-22.31%	5086	0.04%	-12.00%	512	-1.54%	-36.71%
979/80	5800	0.38%	35.55%	6312	0.22%	24.06%	1051	-0.66%	105.27%
980/81	4478	0.16%	-22.79%	5529	0.01%	-12.40%	390	-2.01%	-62.89%
981/82	6393	-0.06%	42.76%	6784	-0.16%	22.70%	774	-0.90%	98.46%
982/83	4789	0.95%	-25.09%	5562	0.67%	-18.01%	368	-0.81%	-52.45%
983/84	3071	-0.16%	-35.87%	3439	-0.23%	-38.17%	106	-8.02%	-71.20%
984/85	5078	-1.36%	65.35%	5184	-1.84%	50.74%	389	-4.19%	266.98%
985/86	5275	-0.06%	3.88%	5664	-0.37%	9.26%	339	-2.31%	-12.85%
986/87	3778	-0.61%	-28.38%	4117	-0.75%	-27.31%	181	-4.23%	-46.61%
987/88	5712	-0.99%	51.19%	5893	-1.09%	43.14%	344	-4.18%	90.06%
988/89	6075	0.21%	6.36%	6419	-0.03%	8.93%	655	-1.50%	90.41%
989/90	4745	1.45%	-21.89%	5399	1.07%	-15.89%	360	-1.64%	-45.04%
990/91	5926	-0.87%	24.90%	6296	-0.63%	16.61%	636	-2.15%	76.67%

Table 108. AGSIM Output: Cottonseed Prices (Minimum Scenario).

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/ton)	Difference From Baseline	Change From Previous Year	Price ($/ton)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Change From Previous Year	Price ($/lb)	Change From Previous Year
976/77	$103.68	0.66%	NA	$60.28	0.67%	NA	$0.42	NA	$0.37	NA
977/78	$70.68	0.54%	-31.83%	$38.41	0.52%	-36.28%	$0.48	14.3%	$0.43	16.2%
978/79	$114.32	0.28%	61.74%	$57.45	0.28%	49.57%	$0.52	8.3%	$0.48	11.6%
979/80	$121.16	0.13%	5.98%	$56.09	0.12%	-2.37%	$0.58	11.5%	$0.50	4.2%
980/81	$129.32	0.25%	6.73%	$54.57	0.26%	-2.71%	$0.58	0.0%	$0.48	-4.0%
981/82	$86.31	0.36%	-33.26%	$33.20	0.36%	-39.16%	$0.71	22.4%	$0.52	8.3%
982/83	$76.54	-0.60%	-11.32%	$27.73	-0.61%	-16.48%	$0.71	0.0%	$0.57	9.6%
983/84	$166.35	0.33%	117.40%	$58.05	0.33%	109.27%	$0.76	7.0%	$0.55	-3.5%
984/85	$102.05	2.02%	-38.73%	$34.02	2.07%	-41.38%	$0.81	6.6%	$0.55	0.0%
985/86	$66.11	0.17%	-35.22%	$21.26	0.19%	-37.51%	$0.81	0.0%	$0.57	3.6%
986/87	$80.83	1.04%	22.27%	$25.26	1.04%	18.81%	$0.81	0.0%	$0.55	-3.5%
987/88	$85.00	2.41%	5.16%	$25.76	2.43%	1.98%	$0.79	-2.5%	$0.52	-5.5%
988/89	$118.59	0.50%	39.52%	$34.68	0.52%	34.68%	$0.76	-3.8%	$0.52	0.0%
989/90	$104.28	-0.82%	-12.07%	$29.13	-0.82%	-16.00%	$0.73	-3.9%	$0.50	-3.8%
990/91	$122.27	1.06%	17.25%	$32.85	1.08%	12.77%	$0.73	0.0%	$0.50	0.0%

Table 109. AGSIM Output: Cottonseed Prices (Maximum Scenario).

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/ton)	Difference From Baseline	Change From Previous Year	Price ($/ton)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Change From Previous Year	Price ($/lb)	Change From Previous Year
976/77	$104.43	1.39%	NA	$60.71	1.39%	NA	$0.42	NA	$0.37	NA
977/78	$71.68	1.96%	-31.36%	$38.96	1.96%	-35.83%	$0.48	14.3%	$0.43	16.2%
978/79	$115.66	1.46%	61.36%	$58.12	1.45%	49.18%	$0.52	8.3%	$0.48	11.6%
979/80	$122.52	1.26%	5.93%	$56.72	1.25%	-2.41%	$0.58	11.5%	$0.50	4.2%
980/81	$130.77	1.37%	6.73%	$55.18	1.36%	-2.72%	$0.58	0.0%	$0.48	-4.0%
981/82	$87.88	2.19%	-32.80%	$33.80	2.18%	-38.75%	$0.71	22.4%	$0.52	8.3%
982/83	$77.75	0.97%	-11.53%	$28.17	0.97%	-16.66%	$0.71	0.0%	$0.57	9.6%
983/84	$168.92	1.76%	117.26%	$58.86	1.76%	108.95%	$0.76	7.0%	$0.55	-3.5%
984/85	$105.13	3.13%	-37.76%	$35.04	3.13%	-40.47%	$0.81	6.6%	$0.55	0.0%
985/86	$68.31	3.50%	-35.02%	$21.96	3.49%	-37.33%	$0.81	0.0%	$0.57	3.6%
986/87	$82.41	3.01%	20.64%	$25.75	3.00%	17.26%	$0.81	0.0%	$0.55	-3.5%
987/88	$87.79	5.77%	6.53%	$26.60	5.77%	3.30%	$0.79	-2.5%	$0.52	-5.5%
988/89	$121.40	2.88%	38.28%	$35.50	2.90%	33.46%	$0.76	-3.8%	$0.52	0.0%
989/90	$107.06	1.83%	-11.81%	$29.90	1.80%	-15.77%	$0.73	-3.9%	$0.50	-3.8%
990/91	$126.54	4.38%	18.20%	$33.98	4.40%	13.65%	$0.73	0.0%	$0.50	0.0%

Table 110. AGSIM Output: Peanut Acreage.

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
976/77	1.5	-6.25%	NA	1.5	0.00%	NA	0.00	NA	NA	0.0	NA	NA
977/78	1.5	0.00%	0.00%	1.5	0.00%	0.00%	0.00	NA	NA	0.0	NA	NA
978/79	1.5	0.00%	0.00%	1.5	0.00%	0.00%	0.00	NA	NA	0.0	NA	NA
979/80	1.6	0.00%	6.67%	1.5	0.00%	0.00%	0.00	NA	NA	0.0	NA	NA
980/81	1.5	0.00%	-6.25%	1.4	0.00%	-6.67%	0.00	NA	NA	0.0	NA	NA
981/82	1.5	0.00%	0.00%	1.5	0.00%	7.14%	0.00	NA	NA	0.0	NA	NA
982/83	1.3	0.00%	-13.33%	1.3	0.00%	-13.33%	0.00	NA	NA	0.0	NA	NA
983/84	1.4	0.00%	7.69%	1.4	0.00%	15.38%	0.00	NA	NA	0.0	NA	NA
984/85	1.6	0.00%	14.29%	1.5	0.00%	0.00%	0.00	NA	NA	0.0	NA	NA
985/86	1.5	0.00%	-6.25%	1.5	0.00%	0.00%	0.00	NA	NA	0.0	NA	NA
986/87	1.6	0.00%	6.67%	1.5	0.00%	0.00%	0.00	NA	NA	0.0	NA	NA
987/88	1.6	0.00%	0.00%	1.6	6.67%	6.67%	0.00	NA	NA	0.0	NA	NA
988/89	1.6	0.00%	0.00%	1.6	0.00%	0.00%	0.00	NA	NA	0.0	NA	NA
989/90	1.7	0.00%	6.25%	1.7	6.25%	6.25%	0.00	NA	NA	0.0	NA	NA
990/91	1.8	0.00%	5.88%	1.8	0.00%	5.88%	0.00	NA	NA	0.0	NA	NA

Table 111. AGSIM Output: Peanuts Production, Supply, and Stocks.

Year	Production			Total Supply			Ending Stocks		
	Amount (Millions of lbs)	Difference From Baseline	Change From Previous Year	Amount (Millions of lbs)	Difference From Baseline	Change From Previous Year	Amount (Millions of lbs)	Difference From Baseline	Change From Previous Year
976/77	3627	-3.02%	NA	4687	-2.35%	NA	598	-5.29%	NA
977/78	3666	-1.32%	1.08%	4245	-1.60%	-9.22%	571	-1.72%	-2.89%
978/79	3877	-1.90%	5.76%	4449	-1.67%	4.56%	573	-2.22%	0.35%
979/80	3921	-1.18%	1.13%	4404	-1.34%	1.01%	619	-1.43%	8.03%
980/81	2248	-2.39%	-42.67%	3268	-1.52%	-27.28%	408	-2.42%	-34.89%
981/82	3927	-1.38%	74.69%	4332	-1.48%	32.56%	747	-1.32%	83.36%
982/83	3393	-1.37%	-13.60%	4142	-1.36%	-4.39%	855	-1.04%	14.46%
983/84	3197	-3.00%	-5.78%	4035	-2.57%	-2.10%	594	-2.78%	-30.53%
984/85	4333	-1.66%	35.53%	4929	-1.79%	21.85%	1411	-0.91%	137.54%
985/86	4035	-1.65%	-6.92%	5467	-1.48%	10.91%	833	-1.42%	-40.96%
986/87	3582	-3.11%	-11.68%	4418	-2.77%	-19.19%	984	-1.89%	18.13%
987/88	3458	-4.37%	-3.46%	4444	-3.83%	0.59%	835	-5.36%	-18.19%
988/89	3849	-3.32%	11.31%	4656	-4.32%	4.77%	819	-2.85%	1.74%
989/90	3919	-1.78%	1.82%	4740	-1.96%	1.80%	687	-2.00%	-16.12%
990/91	3451	-4.77%	-11.95%	4145	-4.29%	-12.55%	652	-4.54%	-5.09%

Table 112. AGSIM Output: Peanut Prices.

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Change From Previous Year	Price ($/lb)	Change From Previous Year
1976/77	$0.20	0.00%	NA	$0.12	0.00%	NA	$0.21	NA	$0.00	NA
1977/78	$0.21	0.00%	5.00%	$0.12	9.09%	0.00%	$0.22	4.8%	$0.00	NA
1978/79	$0.21	5.00%	0.00%	$0.10	0.00%	-16.67%	$0.21	-4.5%	$0.00	NA
1979/80	$0.20	5.26%	-4.76%	$0.09	0.00%	-10.00%	$0.21	0.0%	$0.00	NA
1980/81	$0.21	5.00%	5.00%	$0.09	0.00%	0.00%	$0.23	9.5%	$0.00	NA
1981/82	$0.23	4.55%	9.52%	$0.09	0.00%	0.00%	$0.23	0.0%	$0.00	NA
1982/83	$0.17	0.00%	-26.09%	$0.06	0.00%	-33.33%	$0.28	21.7%	$0.00	NA
1983/84	$0.18	5.88%	5.88%	$0.06	0.00%	0.00%	$0.28	0.0%	$0.00	NA
1984/85	$0.21	0.00%	16.67%	$0.07	0.00%	16.67%	$0.28	0.0%	$0.00	NA
1985/86	$0.13	0.00%	-38.10%	$0.04	0.00%	-42.86%	$0.28	0.0%	$0.00	NA
1986/87	$0.19	5.56%	46.15%	$0.06	0.00%	50.00%	$0.30	7.1%	$0.00	NA
1987/88	$0.11	10.00%	-42.11%	$0.03	0.00%	-50.00%	$0.30	0.0%	$0.00	NA
1988/89	$0.16	6.67%	45.45%	$0.05	25.00%	66.67%	$0.31	3.1%	$0.00	NA
1989/90	$0.17	6.25%	6.25%	$0.05	25.00%	0.00%	$0.31	0.0%	$0.00	NA
1990/91	$0.18	5.88%	5.88%	$0.05	25.00%	0.00%	$0.32	3.2%	$0.00	NA

Table 113. AGSIM Output: Soybean Acreage (Minimum Scenario).

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
976/77	50.3	0.00%	NA	49.4	0.00%	NA	0.00	NA	NA	0.0	NA	NA
977/78	59.0	0.00%	17.30%	57.8	0.00%	17.00%	0.00	NA	NA	0.0	NA	NA
978/79	64.7	0.00%	9.66%	63.6	-0.16%	10.08%	0.00	NA	NA	0.0	NA	NA
979/80	71.4	0.00%	10.36%	70.3	0.00%	10.53%	0.00	NA	NA	0.0	NA	NA
980/81	70.0	0.14%	-1.96%	67.8	0.00%	-3.56%	0.00	NA	NA	0.0	NA	NA
981/82	67.6	0.15%	-3.43%	66.2	0.00%	-2.34%	0.00	NA	NA	0.0	NA	NA
982/83	70.9	0.00%	4.88%	69.5	0.14%	4.98%	0.00	NA	NA	0.0	NA	NA
983/84	63.8	0.00%	-10.01%	62.5	0.00%	-10.07%	0.00	NA	NA	0.0	NA	NA
984/85	67.8	0.00%	6.27%	66.1	0.00%	5.76%	0.00	NA	NA	0.0	NA	NA
985/86	63.4	0.32%	-6.49%	61.8	0.32%	-6.51%	0.00	NA	NA	0.0	NA	NA
986/87	60.5	0.17%	-4.57%	58.4	0.17%	-5.50%	0.00	NA	NA	0.0	NA	NA
987/88	58.2	0.00%	-3.80%	57.2	0.00%	-2.05%	0.00	NA	NA	0.0	NA	NA
988/89	58.8	0.00%	1.03%	57.4	0.00%	0.35%	0.00	NA	NA	0.0	NA	NA
989/90	60.7	-0.16%	3.23%	59.4	-0.17%	3.48%	0.00	NA	NA	0.0	NA	NA
990/91	57.9	0.17%	-4.61%	56.6	0.18%	-4.71%	0.00	NA	NA	0.0	NA	NA

Table 114. AGSIM Output: Soybean Acreage (Maximum Scenario).

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
976/77	50.2	-0.20%	NA	49.4	0.00%	NA	0.00	NA	NA	0.0	NA	NA
977/78	58.9	-0.17%	17.33%	57.7	-0.17%	16.80%	0.00	NA	NA	0.0	NA	NA
978/79	64.5	-0.11%	9.51%	63.4	-0.07%	9.88%	0.00	NA	NA	0.0	NA	NA
979/80	71.4	0.00%	10.70%	70.4	0.14%	11.04%	0.00	NA	NA	0.0	NA	NA
980/81	70.1	0.29%	-1.82%	68.0	0.29%	-3.41%	0.00	NA	NA	0.0	NA	NA
981/82	67.8	0.44%	-3.28%	66.4	0.30%	-2.35%	0.00	NA	NA	0.0	NA	NA
982/83	71.0	0.14%	4.72%	69.5	0.14%	4.67%	0.00	NA	NA	0.0	NA	NA
983/84	63.7	-0.16%	-10.28%	62.4	-0.16%	-10.22%	0.00	NA	NA	0.0	NA	NA
984/85	67.9	0.15%	6.59%	66.2	0.13%	6.09%	0.00	NA	NA	0.0	NA	NA
985/86	63.7	0.79%	-6.19%	62.1	0.81%	-6.19%	0.00	NA	NA	0.0	NA	NA
986/87	60.9	0.63%	-4.40%	59.8	0.66%	-5.31%	0.00	NA	NA	0.0	NA	NA
987/88	58.4	0.34%	-4.11%	57.4	0.38%	-4.36%	0.00	NA	NA	0.0	NA	NA
988/89	59.0	0.34%	1.03%	57.5	0.17%	0.17%	0.00	NA	NA	0.0	NA	NA
989/90	60.9	0.16%	3.22%	59.6	0.17%	3.65%	0.00	NA	NA	0.0	NA	NA
990/91	58.2	0.05%	-4.43%	56.8	0.53%	-4.70%	0.00	NA	NA	0.0	NA	NA

Table 115. AGSIM Output: Soybeans Production, Supply, and Stocks (Minimum Scenario).

Year	Production			Total Supply			Ending Stocks		
	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year
976/77	1286	-0.23%	NA	1591	-0.20%	NA	182	-0.97%	NA
977/78	1764	-0.17%	37.17%	1886	-0.21%	21.88%	159	-1.24%	-55.88%
978/79	1863	-0.32%	5.61%	2022	-0.39%	8.36%	172	-2.27%	8.18%
979/80	2257	-0.18%	21.15%	2429	-0.33%	20.15%	354	-1.12%	105.81%
980/81	1794	-0.17%	-20.51%	2148	-0.32%	-11.57%	308	-1.28%	-12.99%
981/82	1947	-0.15%	10.76%	2294	-0.26%	6.80%	251	-1.18%	-18.51%
982/83	2186	-0.18%	10.02%	2438	-0.25%	6.18%	342	-0.87%	36.25%
983/84	1630	-0.37%	-25.43%	1972	-0.45%	-19.11%	172	-2.27%	-49.71%
984/85	1857	-0.27%	13.93%	3029	-0.44%	2.69%	312	-1.27%	81.40%
985/86	2103	0.14%	13.25%	2415	-0.04%	19.02%	337	0.00%	72.12%
986/87	1938	-0.15%	-7.85%	2475	-0.12%	2.48%	435	-0.23%	-18.99%
987/88	1931	-0.36%	-0.36%	2266	-0.38%	-4.40%	298	-1.32%	-31.49%
988/89	1542	-0.45%	-20.15%	1844	-0.59%	-22.06%	177	-2.75%	-40.60%
989/90	1913	-0.42%	24.19%	2095	-0.68%	13.61%	232	-3.52%	31.07%
990/91	1908	-0.76%	-0.26%	2155	-0.51%	2.86%	323	-1.52%	39.22%

Table 116. AGSIM Output: Soybeans Production, Supply, and Stocks (Maximum Scenario).

Year	Production			Total Supply			Ending Stocks		
	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year
976/77	1275	-1.09%	NA	1520	-0.91%	NA	96	-6.80%	NA
977/78	1751	-0.91%	37.33%	1846	-1.28%	21.45%	149	-7.45%	55.21%
978/79	1841	-1.50%	5.14%	1990	-1.97%	7.80%	137	-10.80%	-8.37%
979/80	2241	-0.88%	21.73%	2397	-1.64%	20.45%	339	-5.31%	115.92%
980/81	1781	-0.89%	-20.53%	2119	-1.67%	-11.60%	295	-5.45%	-12.98%
981/82	1977	-0.65%	11.01%	2272	-1.30%	7.22%	240	-5.51%	-18.64%
982/83	2170	-0.91%	9.76%	2410	-1.39%	6.07%	330	-4.35%	37.50%
983/84	1604	-1.56%	-26.08%	1933	-2.42%	-19.79%	153	-11.95%	-53.03%
984/85	1838	-1.29%	14.59%	1993	-2.21%	3.10%	297	-6.01%	91.61%
985/86	2089	-0.52%	13.66%	2386	-1.24%	19.72%	525	-2.23%	76.77%
986/87	1928	-0.67%	-7.71%	2453	-1.01%	2.81%	426	-2.29%	-18.86%
987/88	1910	-1.44%	-0.93%	2337	-1.60%	-4.73%	285	-5.63%	-33.10%
988/89	1519	-1.94%	-20.47%	1808	-2.53%	-22.64%	160	-12.09%	-43.86%
989/90	1890	-1.72%	24.42%	2053	-2.61%	13.55%	212	-10.92%	32.50%
990/91	1902	-1.19%	0.63%	2117	-2.26%	3.12%	305	-7.01%	43.87%

Table 117. AGSIM Output: Soybean Prices (Minimum Scenario).

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/bu)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/bu)	Change From Previous Year	Price ($/bu)	Change From Previous Year
976/77	$6.02	0.15%	NA	$3.96	0.00%	NA	$0.00	NA	$2.50	NA
977/78	$5.89	0.17%	-13.64%	$3.20	0.00%	-19.19%	$0.00	NA	$3.50	40.0%
978/79	$6.68	0.30%	13.41%	$3.36	0.30%	5.00%	$0.00	NA	$4.50	28.6%
979/80	$6.30	0.32%	-5.69%	$2.92	0.34%	-13.10%	$0.00	NA	$4.50	0.0%
980/81	$7.59	0.26%	20.48%	$3.20	0.31%	9.59%	$0.00	NA	$5.02	11.6%
981/82	$6.09	0.33%	-19.76%	$2.34	0.43%	-26.88%	$0.00	NA	$5.02	0.0%
982/83	$5.73	0.35%	-5.91%	$2.08	0.48%	-11.11%	$0.00	NA	$5.02	0.0%
983/84	$7.86	0.38%	37.17%	$2.74	0.37%	31.73%	$0.00	NA	$5.02	0.0%
984/85	$5.67	0.51%	-25.32%	$1.96	0.51%	-28.47%	$0.00	NA	$5.02	0.0%
985/86	$5.05	0.00%	-13.97%	$1.62	0.00%	-17.35%	$0.00	NA	$5.02	0.0%
986/87	$4.79	0.21%	-5.15%	$1.50	0.67%	-7.41%	$0.00	NA	$4.77	-5.0%
987/88	$5.91	0.51%	23.38%	$1.79	0.56%	19.33%	$0.00	NA	$4.77	0.0%
988/89	$7.47	0.67%	26.40%	$2.18	0.46%	21.79%	$0.00	NA	$4.77	0.0%
989/90	$5.75	0.88%	-23.03%	$1.61	1.26%	-26.13%	$0.00	NA	$4.53	-5.0%
990/91	$5.80	0.57%	0.87%	$1.55	0.65%	-3.73%	$0.00	NA	$4.50	-0.7%

Table 118. AGSIM Output: Soybean Prices (Maximum Scenario).

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/bu)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/bu)	Change From Previous Year	Price ($/bu)	Change From Previous Year
976/77	$6.84	0.44%	NA	$3.96	0.51%	NA	$0.00	NA	$2.50	NA
977/78	$5.93	0.85%	-13.30%	$3.23	0.94%	-18.84%	$0.00	NA	$3.50	40.0%
978/79	$6.76	1.50%	14.00%	$3.40	1.49%	5.26%	$0.00	NA	$4.50	28.6%
979/80	$6.39	1.75%	-5.47%	$2.96	1.72%	-12.94%	$0.00	NA	$4.50	0.0%
980/81	$7.07	1.52%	20.03%	$3.24	1.57%	9.46%	$0.00	NA	$5.02	11.6%
981/82	$6.16	1.48%	-19.69%	$2.37	1.72%	-26.85%	$0.00	NA	$5.02	0.0%
982/83	$5.82	1.93%	-5.52%	$2.11	1.93%	-10.97%	$0.00	NA	$5.02	0.0%
983/84	$7.99	2.04%	37.29%	$2.78	1.83%	31.75%	$0.00	NA	$5.02	0.0%
984/85	$5.99	2.57%	-25.03%	$2.00	2.56%	-28.06%	$0.00	NA	$5.02	0.0%
985/86	$5.14	1.78%	-14.19%	$1.68	1.85%	-17.50%	$0.00	NA	$5.02	0.0%
986/87	$4.86	1.67%	-5.45%	$1.52	2.01%	-7.88%	$0.00	NA	$4.77	-5.0%
987/88	$6.03	2.55%	24.07%	$1.83	2.81%	20.39%	$0.00	NA	$4.77	0.0%
988/89	$7.02	2.70%	26.37%	$2.23	2.76%	21.86%	$0.00	NA	$4.77	0.0%
989/90	$5.94	4.21%	-22.05%	$1.46	4.40%	-25.56%	$0.00	NA	$4.53	-5.0%
990/91	$5.97	3.83%	0.51%	$1.40	3.90%	-3.61%	$0.00	NA	$4.50	-0.7%

Table 119. AGSIM Output: Sorghum Acreage.

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
1976/77	18.2	0.55%	NA	14.5	0.00%	NA	0.00	NA	NA	0.0	NA	NA
1977/78	16.6	0.00%	-8.79%	13.8	0.00%	-4.83%	0.00	NA	NA	0.0	NA	NA
1978/79	16.2	0.00%	-2.41%	13.4	0.00%	-2.90%	0.10	0.0%	NA	2.0	0.0%	NA
1979/80	15.3	0.00%	-5.56%	12.9	0.00%	-3.73%	0.10	0.0%	0.0%	1.7	0.0%	-15.0%
1980/81	19.6	0.00%	1.96%	12.5	0.00%	-3.10%	0.00	NA	-100.0%	0.0	NA	-100.0%
1981/82	15.9	0.00%	1.92%	13.7	0.00%	9.40%	0.00	NA	NA	0.0	NA	NA
1982/83	16.0	0.00%	0.63%	14.1	0.00%	2.92%	0.10	0.0%	NA	0.9	0.0%	NA
1983/84	11.9	0.00%	-25.63%	10.0	0.00%	-29.08%	0.10	0.0%	0.0%	4.7	0.0%	422.2%
1984/85	17.4	0.58%	46.22%	15.6	1.30%	56.00%	0.05	-50.0%	-50.0%	0.7	-30.0%	-85.1%
1985/86	18.1	-1.09%	4.02%	16.6	-1.19%	6.41%	0.13	30.0%	160.0%	1.7	21.4%	142.9%
1986/87	15.5	1.31%	-14.36%	14.0	0.72%	-15.66%	0.17	0.0%	30.8%	3.8	0.0%	123.5%
1987/88	11.4	-1.69%	-25.16%	10.4	-0.95%	-25.71%	0.20	0.0%	17.6%	5.7	1.8%	50.0%
1988/89	10.5	-1.94%	-9.48%	9.2	2.22%	-11.54%	0.20	0.0%	0.0%	5.4	-1.8%	-5.3%
1989/90	12.5	-0.79%	19.05%	10.9	-1.80%	18.48%	0.10	0.0%	-50.0%	3.6	0.0%	-33.3%
1990/91	10.6	0.95%	-15.20%	9.2	1.10%	-15.60%	0.10	0.0%	0.0%	3.9	0.0%	8.3%

Table 120. AGSIM Output: Sorghum Production, Supply, and Stocks.

Year	Production			Total Supply			Ending Stocks		
	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year
976/77	710	0.00%	NA	793	0.00%	NA	117	0.00%	NA
977/78	780	-0.26%	9.86%	897	-0.22%	13.11%	216	-0.46%	84.62%
978/79	729	-0.27%	-6.54%	943	-0.32%	5.35%	206	-0.96%	-4.63%
979/80	806	-0.25%	10.56%	1012	-0.39%	7.09%	175	-1.69%	-15.05%
980/81	578	-0.34%	-28.29%	753	-0.66%	-25.59%	127	-3.05%	-27.43%
981/82	875	0.00%	51.38%	1002	-0.40%	33.07%	316	-0.94%	148.82%
982/83	833	-0.36%	-4.80%	1149	-0.52%	14.67%	436	-0.91%	37.97%
983/84	486	0.00%	-41.66%	921	-0.54%	-19.84%	283	-1.39%	-35.09%
984/85	875	0.92%	80.04%	1158	0.35%	25.73%	302	0.67%	6.71%
985/86	1108	-1.16%	26.63%	1410	-0.77%	21.76%	544	-1.27%	80.13%
986/87	947	0.96%	-14.53%	1491	0.13%	5.74%	743	0.00%	36.58%
987/88	722	-1.30%	-23.76%	1465	-0.76%	-1.74%	656	-1.06%	-11.71%
988/89	586	1.21%	-18.84%	1242	0.00%	-15.22%	439	-0.23%	-33.08%
989/90	625	-1.63%	5.24%	1044	-1.04%	-15.94%	211	-3.65%	-51.94%
990/91	579	1.03%	-6.30%	790	-0.25%	-24.33%	139	-1.42%	-34.12%

Table 121. AGSIM Output: Sorghum Prices.

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/bu)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/bu)	Change From Previous Year	Price ($/bu)	Change From Previous Year
976/77	$2.03	0.00%	NA	$1.18	0.00%	NA	$1.49	NA	$1.43	NA
977/78	$1.82	0.00%	-10.34%	$0.99	0.00%	-16.10%	$2.28	53.0%	$1.90	32.9%
978/79	$2.01	0.00%	10.44%	$1.01	0.00%	2.02%	$2.28	0.0%	$1.90	0.0%
979/80	$2.35	0.00%	16.92%	$1.09	0.00%	7.92%	$2.34	2.6%	$2.00	5.3%
980/81	$2.92	0.34%	24.26%	$1.23	0.00%	12.84%	$2.50	6.8%	$2.14	7.0%
981/82	$2.25	0.00%	-22.96%	$0.87	0.00%	-29.27%	$2.55	2.0%	$2.28	6.5%
982/83	$2.48	0.40%	10.22%	$0.90	1.12%	3.45%	$2.60	2.0%	$2.42	6.1%
983/84	$2.75	0.36%	10.89%	$0.86	1.05%	-4.67%	$2.72	4.6%	$2.52	4.1%
984/85	$2.30	-0.86%	-16.36%	$0.77	0.00%	-19.79%	$2.88	5.9%	$2.42	-4.0%
985/86	$1.96	1.55%	-14.78%	$0.63	1.61%	-18.18%	$2.88	0.0%	$2.42	0.0%
986/87	$1.36	-0.73%	-30.61%	$0.42	-2.33%	-33.33%	$2.88	0.0%	$1.74	-28.1%
987/88	$1.75	1.76%	27.21%	$0.52	0.00%	23.81%	$2.88	0.0%	$1.74	0.0%
988/89	$2.26	-0.44%	30.64%	$0.66	0.00%	26.92%	$2.78	-3.5%	$1.68	-3.4%
989/90	$2.13	1.43%	-5.75%	$0.60	1.69%	-9.09%	$2.70	-2.9%	$1.57	-6.5%
990/91	$2.12	0.00%	-0.47%	$0.57	0.00%	-5.00%	$2.61	-3.3%	$1.49	-5.1%

Table 122. AGSIM Output: Wheat Acreage (Minimum Scenario).

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
976/77	80.4	0.00%	NA	70.9	0.00%	NA	0.00	NA	NA	0.0	NA	NA
977/78	75.4	0.00%	-6.22%	66.7	0.00%	-5.92%	0.00	NA	NA	0.0	NA	NA
978/79	66.0	0.00%	-12.47%	56.5	0.00%	-15.29%	0.20	0.0%	NA	12.2	0.0%	NA
979/80	71.5	0.14%	8.33%	62.5	0.00%	10.62%	0.20	0.0%	0.0%	10.2	-1.0%	-16.4%
980/81	80.8	0.00%	13.01%	71.1	0.00%	13.76%	0.00	NA	-100.0%	0.0	NA	-100.0%
981/82	88.3	0.00%	9.28%	80.7	0.12%	13.50%	0.00	NA	NA	0.0	NA	NA
982/83	86.3	0.12%	-2.27%	78.0	0.13%	-3.35%	0.15	0.0%	NA	5.7	-1.7%	NA
983/84	76.5	0.13%	-6.84%	61.5	0.16%	-9.10%	0.15	0.0%	-100.0%	26.8	0.0%	-100.0%
984/85	79.3	0.13%	-1.57%	67.0	0.13%	-3.50%	0.20	0.0%	NA	21.0	-0.5%	NA
985/86	75.6	0.13%	-4.67%	64.8	0.13%	-3.28%	0.20	0.0%	0.0%	21.8	-0.5%	3.8%
986/87	72.1	0.14%	-4.63%	60.8	0.16%	-6.17%	0.23	0.0%	15.0%	23.1	0.0%	6.0%
987/88	65.9	0.15%	-8.60%	56.1	0.18%	-7.73%	0.28	0.0%	21.7%	29.6	-0.3%	28.1%
988/89	65.6	0.15%	-0.46%	53.2	0.00%	-5.17%	0.28	0.0%	0.0%	31.6	0.0%	6.8%
989/90	76.7	0.13%	16.92%	62.3	0.16%	17.11%	0.10	0.0%	-64.3%	12.7	-0.8%	-59.8%
990/91	77.3	0.13%	0.78%	69.4	0.14%	11.40%	0.05	0.0%	-50.0%	7.5	0.0%	-40.9%

Table 123. AGSIM Output: Wheat Acreage (Maximum Scenario).

Year	Acreage Planted			Acreage Harvested			Set Aside Rate			Acreage Set Aside		
	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year	Amount	Difference From Baseline	Change From Previous Year
1976/77	80.1	-0.37%	NA	70.7	-0.28%	NA	0.00	NA	NA	0.0	NA	NA
1977/78	75.2	-0.27%	-6.12%	66.5	-0.30%	-5.94%	0.00	NA	NA	0.0	NA	NA
1978/79	66.0	0.00%	-12.23%	56.5	0.00%	-15.04%	0.20	0.0%	NA	12.1	-0.82%	NA
1979/80	71.8	0.56%	8.79%	62.7	0.32%	10.97%	0.20	0.0%	0.0%	9.5	-7.77%	-21.49%
1980/81	80.9	0.12%	12.67%	71.3	0.28%	13.72%	-0.05	NA	-125.0%	0.0	NA	-100.00%
1981/82	88.4	0.11%	9.27%	80.8	0.25%	13.32%	-0.05	NA	NA	0.0	NA	NA
1982/83	86.5	0.35%	-2.15%	78.2	0.39%	-3.22%	0.15	0.0%	NA	5.6	-3.45%	NA
1983/84	78.3	2.75%	-7.40%	69.2	2.93%	-9.59%	0.10	-33.3%	-100.0%	23.1	-13.81%	-100.00%
1984/85	80.6	1.77%	0.62%	68.2	1.94%	-3.54%	0.20	0.0%	NA	20.2	-4.27%	NA
1985/86	76.8	-1.72%	-4.71%	64.8	-1.70%	-5.52%	0.20	0.0%	0.0%	20.9	-4.57%	3.47%
1986/87	72.8	1.11%	-5.21%	61.4	1.15%	-6.69%	0.23	0.0%	15.0%	22.9	-0.87%	9.57%
1987/88	67.0	1.82%	-7.97%	56.9	1.61%	-7.33%	0.28	0.0%	21.7%	28.5	-4.04%	24.43%
1988/89	66.2	1.07%	-1.19%	53.6	1.13%	-5.45%	0.28	0.0%	0.0%	31.5	-0.32%	10.53%
1989/90	77.0	0.52%	16.31%	62.5	0.48%	16.17%	0.10	0.0%	-64.3%	13.2	3.12%	-58.10%
1990/91	72.2	-2.52%	-6.26%	71.0	-2.55%	13.60%	0.00	-100.0%	-100.0%	5.1	-45.55%	-61.54%

Table 124. AGSIM Output: Wheat Production, Supply, and Stocks (Minimum Scenario).

Year	Production			Total Supply			Ending Stocks		
	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year
976/77	2137	-0.51%	NA	2806	-0.39%	NA	1108	-0.45%	NA
977/78	2025	-0.54%	-4.77%	3144	-0.54%	12.05%	1169	-0.85%	5.51%
978/79	1754	-0.62%	-13.81%	2926	-0.65%	-6.93%	903	-1.20%	-22.76%
979/80	2135	-0.47%	21.72%	3040	-0.69%	3.90%	890	-1.33%	-1.44%
980/81	2368	-0.30%	10.91%	3261	-0.73%	7.27%	975	-1.22%	9.55%
981/82	2778	-0.36%	17.31%	3756	-0.61%	15.18%	1148	-1.12%	17.74%
982/83	2752	-0.40%	-0.94%	3907	-0.64%	4.02%	1501	-0.92%	30.75%
983/84	2408	-0.50%	-12.50%	3914	-0.63%	0.18%	1384	-1.07%	-7.79%
984/85	2581	-0.50%	7.18%	3975	-0.67%	1.56%	1409	-1.12%	1.81%
985/86	3410	-0.50%	32.11%	4836	-0.75%	21.66%	1888	-0.84%	34.00%
986/87	2077	-0.67%	-39.09%	3988	-0.72%	-17.54%	1805	-0.88%	-4.40%
987/88	2091	-0.81%	0.67%	3912	-0.88%	-1.91%	1243	-1.43%	-31.14%
988/89	1778	-1.53%	-14.97%	3045	-1.68%	-22.16%	675	-3.85%	-45.70%
989/90	2024	-0.59%	13.84%	2723	-1.38%	-10.57%	513	-4.11%	-24.00%
990/91	2772	-0.55%	36.96%	3272	-1.12%	20.16%	844	-2.43%	64.52%

Table 125. AGSIM Output: Wheat Production, Supply, and Stocks (Maximum Scenario).

Year	Production			Total Supply			Ending Stocks		
	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year	Amount (Millions of bu)	Difference From Baseline	Change From Previous Year
976/77	2053	-4.62%	NA	2723	-3.84%	NA	1068	-4.04%	NA
977/78	1940	-5.18%	-5.50%	3011	-4.75%	10.58%	1098	-6.87%	2.81%
978/79	1676	-5.04%	-13.61%	2778	-5.67%	-7.74%	819	-10.39%	-25.41%
979/80	2042	-4.80%	21.84%	2866	-6.57%	3.17%	790	-12.42%	-3.54%
980/81	2255	-5.25%	10.43%	3051	-7.12%	6.45%	855	-13.46%	8.23%
981/82	2643	-5.20%	17.21%	3504	-7.28%	14.85%	1005	-13.44%	17.54%
982/83	2611	-5.50%	-1.21%	3628	-7.75%	3.54%	1342	-11.42%	33.53%
983/84	2334	-5.55%	-10.61%	3684	-6.47%	1.54%	1249	-10.72%	-6.93%
984/85	2463	-5.05%	5.53%	3726	-6.90%	1.14%	1270	-10.88%	1.68%
985/86	2287	-5.45%	-7.15%	3578	-7.43%	-3.97%	1745	-8.35%	37.40%
986/87	1943	-7.08%	-15.04%	3712	-7.62%	3.88%	1656	-9.06%	-5.10%
987/88	1953	-7.55%	0.51%	3631	-7.94%	-2.26%	1088	-13.72%	-34.30%
988/89	1612	-11.09%	-17.46%	2729	-11.85%	-24.84%	502	-28.49%	-53.86%
989/90	1908	-4.25%	15.88%	2400	-13.07%	-12.04%	332	-37.94%	-33.86%
990/91	2574	-5.96%	37.79%	2949	-10.85%	22.88%	601	-33.70%	80.80%

Table 126. AGSIM Output: Wheat Prices (Minimum Scenario).

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/bu)	Difference From Baseline	Change From Previous Year	Price ($/lb)	Difference From Baseline	Change From Previous Year	Price ($/bu)	Change From Previous Year	Price ($/bu)	Change From Previous Year
976/77	$2.74	0.57%	NA	$1.60	0.63%	NA	$2.29	NA	$2.25	NA
977/78	$2.35	0.86%	-14.23%	$1.28	0.79%	-20.00%	$2.90	26.6%	$2.25	0.0%
978/79	$3.00	0.67%	27.66%	$1.51	0.67%	17.97%	$3.40	17.2%	$2.35	4.4%
979/80	$3.82	0.53%	27.33%	$1.77	0.57%	17.22%	$3.40	0.0%	$2.50	6.4%
980/81	$4.02	0.75%	5.24%	$1.69	0.60%	-4.52%	$3.63	6.8%	$3.00	20.0%
981/82	$3.72	0.81%	-7.46%	$1.43	0.70%	-15.38%	$3.81	5.0%	$3.20	6.7%
982/83	$3.48	0.87%	-6.45%	$1.26	0.80%	-11.89%	$4.05	6.3%	$3.55	10.9%
983/84	$3.54	0.85%	1.72%	$1.23	0.82%	-2.38%	$4.30	6.2%	$3.65	2.8%
984/85	$3.42	0.88%	-3.39%	$1.14	0.88%	-7.32%	$4.38	1.9%	$3.30	-9.6%
985/86	$3.12	1.30%	-8.77%	$1.00	1.01%	-12.28%	$4.38	0.0%	$3.30	0.0%
986/87	$2.46	1.65%	-21.15%	$0.77	1.32%	-23.00%	$4.38	0.0%	$2.40	-27.3%
987/88	$2.61	1.56%	6.10%	$0.79	1.28%	2.60%	$4.38	0.0%	$2.28	-5.0%
988/89	$3.81	2.42%	45.98%	$1.11	1.83%	40.51%	$4.23	-3.4%	$2.21	-3.1%
989/90	$3.76	1.08%	-1.31%	$1.05	0.96%	-5.41%	$4.10	-3.1%	$2.06	-6.8%
990/91	$2.66	1.22%	-29.26%	$0.71	1.43%	-32.38%	$4.00	-2.4%	$1.95	-5.3%

Table 127. AGSIM Output: Wheat Prices (Maximum Scenario).

Year	Market Price			Real Price (in 1967 dollars)			Target Price		Loan Rate	
	Price ($/bu)	Difference From Baseline	Change From Previous Year	Price ($/Bu)	Difference From Baseline	Change From Previous Year	Price ($/bu)	Change From Previous Year	Price ($/bu)	Change From Previous Year
976/77	$2.65	4.40%	NA	$1.66	4.40%	NA	$2.29	NA	$2.25	NA
977/78	$2.49	6.87%	-12.63%	$1.35	6.30%	-18.67%	$2.90	26.6%	$2.25	0.0%
978/79	$3.14	5.37%	26.10%	$1.58	5.33%	17.04%	$3.40	17.2%	$2.35	4.4%
979/80	$4.00	5.26%	27.39%	$1.85	5.11%	17.09%	$3.40	0.0%	$2.50	6.4%
980/81	$4.25	6.52%	6.25%	$1.80	7.14%	-2.70%	$3.63	6.8%	$3.00	20.0%
981/82	$4.03	9.21%	-5.18%	$1.65	9.15%	-13.89%	$3.81	5.0%	$3.20	6.7%
982/83	$3.82	10.72%	-5.21%	$1.59	11.20%	-10.32%	$4.05	6.3%	$3.55	10.9%
983/84	$3.76	7.12%	-1.57%	$1.51	7.38%	-5.76%	$4.30	6.2%	$3.65	2.8%
984/85	$3.73	10.03%	-0.80%	$1.24	9.73%	-5.34%	$4.38	1.9%	$3.30	-9.6%
985/86	$3.44	11.69%	-7.77%	$1.11	12.12%	-10.48%	$4.38	0.0%	$3.30	0.0%
986/87	$2.80	15.70%	-18.60%	$0.88	15.79%	-20.72%	$4.38	0.0%	$2.40	-27.3%
987/88	$2.99	14.34%	6.79%	$0.91	16.07%	3.41%	$4.38	0.0%	$2.28	-5.0%
988/89	$4.28	13.08%	43.14%	$1.25	14.68%	37.36%	$4.23	-3.4%	$2.21	-3.1%
989/90	$4.24	13.98%	-0.93%	$1.19	14.42%	-4.80%	$4.10	-3.1%	$2.06	-6.8%
990/91	$3.15	20.69%	-25.71%	$0.84	20.00%	-29.41%	$4.00	-2.4%	$1.95	-5.3%

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Appendix G: Lead Benefits Analysis

Methods Used to Measure and Value Health Effects

The scientific understanding of the relationship between lead and human health is rapidly expanding. This expansion is documented in numerous EPA studies on the health effects associated with lead exposure. In a pioneering study, Schwartz et al. (U.S. EPA, 1985) quantified a number of health benefits that would result from reductions in the lead content of gasoline. The work was extended by EPA's analysis of lead in drinking water (U.S. EPA, 1986a) and by an EPA-funded study of alternative lead National Ambient Air Quality Standards (U.S. EPA, 1987).

Despite this substantial research, much uncertainty remains. While the health effects of very high levels of Blood Lead (PbB) are quite severe (including convulsions, coma and death from lead toxicity) and have been known for many years, the effects of lower lead doses continue to be the subject of intensive scientific investigation. Dose-response functions are available for only a handful of health endpoints associated with elevated blood lead levels. Other known or strongly suspected health endpoints cannot be quantified due to a lack of information on the relationship between dose and effect. Table 128 presents the health effects that are quantified in this analysis, as well as important known health effects that are not quantified.

Table 128. Quantified and Unquantified Health Effects of Lead.

Population Group	Quantified Health Effect	Unquantified Health Effect
Adult Male	*For men in specified age ranges:* Hypertension Non-fatal coronary heart disease Non-fatal Strokes Mortality	Quantified health effects for men in other age ranges Possible cancer Other cardiovascular diseases
Adult Female	*For women in specified age ranges:* Non-fatal coronary heart disease Non-fatal stroke Mortality	Quantified health effects for women in other age ranges Possible cancer Other cardiovascular diseases Reproductive effects
Children	IQ loss effect on lifetime earnings IQ loss effects on special educational needs Neonatal mortality due to low birth weight caused by maternal exposure to lead	Fetal effects from maternal exposure (including diminished IQ) Other neurological & metabolic effects

Some of the health effects that are quantified in this analysis have not been estimated in previous EPA analyses. This is largely due to more recent information about the dose-response functions that

makes it possible to expand the health effect coverage beyond what was done previously. Recent information is available for previously unquantified health effects, and new information on previously estimated dose-response functions is also available.

The following sections present relevant dose-response relationships for three population groups: children, men, and women. These sections also discuss data sources used for the dose-response relationships, although an extensive review of the literature is not given.[205] In addition, each section includes the methods used to value the changes in health effects determined using these dose-response relationships.

Health Benefits to Children

Changes in IQ

Elevated PbB levels may induce a number of effects on the human nervous system. Generally, these neurobehavioral effects are more serious for children than for adults because of children's rapid rate of development. It is believed that neurobehavioral deficits in children may result from both pre-natal and early post-natal exposure. These nervous system effects may include hyperactivity, behavioral and attentional difficulties, delayed mental development, and motor and perceptual skill deficits. Quantification of certain manifestations of these effects is possible because sufficient data exist to estimate a dose-response relationship and IQ loss. The relationship used in the analysis is discussed below.

Quantifying the Relationship Between Blood Lead Levels and IQ

A dose-response relationship for IQ decrements has been estimated by a meta-analysis of seven research studies.[206] Regression coefficients for each study were used to determine a weighted average linear regression coefficient for the relationship between lead and IQ. Each regression coefficient was weighted by the inverse of the variance of each estimate. To determine an overall coefficient, the regression coefficients for studies that used natural logarithms of lead as the exposure index were linearized. In general, the coefficient was linearized in the blood lead range of 10 to 20 µg/dL. However, in one study (Bellinger et al., 1991), 70 percent of the data was below 10 µg/dL; thus, the Bellinger data was linearized in the 5 to 15 µg/dL range. For the studies that did not transform lead concentrations, the regression coefficients were used directly. Given the typical uncertainty within individual studies, the variation in the regression coefficients among studies was not more than would be expected. The relationship determined by Schwartz (1993) suggests that for a 1 µg/dL increase in lead, a decrease of 0.25 IQ points can be expected. The p-value (< 0.0001) indicates that this relationship is highly significant.

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[205] For a detailed review of this literature see U.S. Environmental Protection Agency, (1986) *Air Quality Criteria Document for Lead*, and 1989 Addendum. Environmental Criteria and Assessment Office, Office of Research and Development, March.

[206] Schwartz, 1993.

To obtain the total change in number of IQ points for a population of children, the 0.25 points lost per μg/dL change in blood lead is multiplied by the average blood lead level for that population. The average blood lead level modeled in this analysis is a geometric mean, not an arithmetic mean. To adjust for this, we use a relationship between the expected value and the geometric mean of a lognormally distributed random variable:

$$E[x] = \exp\left[\ln(GM) + \frac{(\ln(GSD))^2}{2}\right] \quad (39)$$

where E(X) is the expected value (mean) of the distribution, GM is the geometric mean, and GSD is the geometric standard deviation. Taking the natural logarithm of Equation 39 and rearranging gives the ratio between the expected value and the GM:

$$\ln(E(X)) - \ln(GM) = \frac{(\ln(GSD))^2}{2} \quad (40)$$

$$\ln\left[\frac{E(X)}{GM}\right] = \frac{(\ln(GSD))^2}{2} \quad (41)$$

$$\frac{E(X)}{GM} = \exp\left[\frac{(\ln(GSD))^2}{2}\right] \quad (42)$$

For a GSD of 1.6 (the assumed GSD of children's blood lead levels[207]), the resulting ratio between E(X) and GM is 1.117. This ratio is used in equation 43.

The total lost IQ points for each group was estimated as:

$$(TOTAL\ LOST\ IQ)_k = \Delta GM_k \times 1.117 \times 0.25 \times (Pop)_k / 7 \quad (43)$$

where $(Pop)_k$ represents the number of children (up to age six) around a given industrial source (in the case of estimating benefits from reduced industrial emissions) or the total U.S. population of children (in the case of estimating benefits from reductions in gasoline lead emissions).

[207] Suggested value for sub-populations provided by IEUBK guidance manual. (EPA, 1994)

As shown in equation 43, the population of children up to age six is divided by seven to avoid double counting. If we assume that children are evenly distributed by age, this division applies this equation to only children age 0-1. If we did not divide, this equation would count a child who is age zero in the first year of the analysis and count that same child 6 more times in successive years. Dividing by seven does creates some undercounting because in the first year of the analysis children from age 1 to 6 are not accounted for, while presumably they are affected by the lead exposure.

The current model assumes a permanent loss of IQ based on blood lead levels estimated for children six years and younger. EPA, 1986b (Vol IV) states that in a series of studies on a group of lead-exposed children, "IQ deficits . . . were no longer evident at the five year follow up." More recent studies,[208] though, provide concrete evidence of long-term effects from childhood lead exposure. Future exploration of this assumption is probably warranted.

Valuing Changes in Children's Intelligence

Available economic research provides little empirical data for society's willingness to pay (WTP) to avoid a decrease in an infant's IQ. As an alternative measure, it was assumed that IQ deficits incurred through lead exposure will persist throughout the exposed infant's lifetime. Two consequences of this IQ decrement, representing a portion of society's full willingness to pay, are then considered: the decreased present value of expected lifetime earnings for the infant, and the increased educational resources expended for a infant who becomes mentally handicapped or is in need of compensatory education as a consequence of lead exposure. The value of foregone earnings is addressed in this section.

The reduction in IQ has a direct and indirect effect on earnings. The direct effect is straightforward -- lower IQS decrease job attainment and performance. Reduced IQ also results in reduced educational attainment, which, in turn, affects earnings and labor force participation. Note that these effects on earnings are additive since the studies used for this analysis have controlled for these effects separately.

Direct Effect of IQ on Wage Rate

Henry Aaron, Zvi Griliches, and Paul Taubman have reviewed the literature examining the relationship between IQ and lifetime earnings.[209] They find that the direct effect, (schooling held constant) of IQ on wage rates ranged from 0.2 percent to 0.75 percent. Perhaps the best of these studies is Griliches (1977).[210] He finds that the direct effect of IQ on wage rates to be slightly more than 0.5 percent per IQ point. Because this is roughly the median estimate of the USEPA review of the literature, this estimate is used.

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[208] For example, Bellinger 1992.

[209] USEPA 1984.

[210] Griliches used a structural equations model to estimate the impact of multiple variables on an outcome of interest. This method has conceptual advantages over other empirical estimates used in the literature because it successfully controls for the many confounding variables that can affect earnings.

Indirect Effect of IQ on Earnings

From Needleman et al. (1990) it is possible to estimate the change in years of schooling attained per one IQ point change. Their regression coefficients for the effect of tooth lead on achieved grade provide an estimate of current grade achieved. However, many of these children were in college at the time and are expected to achieve a higher grade level. Following Schwartz (1990), after adjusting the published results for the fact that a higher percentage of children with low tooth lead were attending college, a 0.59 year difference in expected maximum grade achieved between the high and low exposure groups was estimated. It is assumed that educational attainment relates with blood lead levels in proportion to IQ. The difference in IQ score between the high and low exposure group was 4.5 points. Dividing 0.59/4.5 = 0.131 suggests that the increase in lead exposure which reduces IQ by one point may also reduce years of schooling by 0.131 years.[211]

Studies that estimate the relationship between educational attainment and wage rates (while controlling for IQ and other factors) are less common. Chamberlain and Griliches (1977) estimate that a one year increase in schooling would increase wages by 6.4 percent. In a longitudinal study of 799 subjects over 8 years, Ashenfelter and Ham (1979) reported that an extra year of education increased the average wage rate over the period by 8.8 percent. Conservatively, we use a lower bound by assuming one year of additional schooling increases the wage rate by 6 percent.[212] To arrive at the indirect effect of increased schooling, increased wages per IQ point is calculated using: (6 percent wage increase/school year) x (0.131 school years/IQ) = 0.786 percent increase in wages per IQ point.

There is one final indirect effect on earnings. Changes in IQ affect labor force participation. Failure to graduate high school, for example, correlates with participation in the labor force, principally through higher unemployment rates and earlier retirement ages. Lead is also a strong correlate with attention span deficits, which likely reduce labor force participation. The results of Needleman et al. (1990) relating lead to failure to graduate high school can be used to estimate changes in earnings due to labor force participation. Using the odds ratio from Needleman et al., it was estimated that a one IQ point deficit would also result in a 4.5 percent increase in the risk of failing to graduate. Krupnick and Cropper (1989) provide estimates of labor force participation between high school graduates and non-graduates, controlling for age, marital status, children, race, region, and other socioeconomic status factors. Based on their data, average participation in the labor force is reduced by 10.6 percent for persons failing to graduate from high school. Because labor force participation is only one component of lifetime earnings (i.e., earnings = wage rate X years of work), this indirect effect of schooling is additive to the effect on wage rates. Combining this estimate with the Needleman result of 4.5 percent increase in the risk of failing to graduate high school per IQ point, indicates that the mean impact of one IQ point loss is a (10.6 percent x 4.5 percent) = 0.477 percent decrease in expected earnings from reduced labor force participation.

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[211] Following Schwartz (1990a), this analysis uses the Needleman (1990) to quantify the change in grade achievement from lead exposure.

[212] A conservative rather than a best estimate (e.g., such as 7.6 percent, which is the average of the two cited studies) is used herein to attempt to account for mitigating factors that we may have missed. For example, some have suggested that the cost of additional schooling should be netted out of the benefits calculation. The average cost of one year of schooling in the U.S. in 1990 ($5,000 per year to $6,000 per year) was approximately ½ the difference in changed lifetime earnings implied by using 6 percent rather than 7.6 percent.

Combining the direct effect of 0.5 percent with the two indirect effects (0.786 percent for less schooling and 0.477 percent for reduced labor force participation) yields a total of 1.76 percent decrease in earnings for every loss of one IQ point.

Value of Foregone Earnings

In the next step to monetize intelligence effects, the percent earnings loss estimate must be combined with an estimate of the present value of expected lifetime earnings. Data on expected lifetime earnings as a function of educational attainment and sex was reported for the U.S. population in 1979 by the Bureau of the Census.[213] Given the distribution of the 1979 population with respect to age, educational attainment, and sex, the Census used age specific employment rates and average wage rates to estimate annual earnings as a function of age, sex and education. Assuming various rates of real wage growth (productivity effect) and discount factors, the annual earnings stream from age 18 to age 64 was collapsed to a series of estimates of the present value of lifetime earnings using an assumption of one percent real wage growth and a three percent discount rate. Men tend to earn more than women because of higher wage rates and higher labor force participation. However, for both men and women, expected lifetime earnings increase greatly with education.

The Census estimates were expressed in 1981 dollars and assumed that the age/education specific employment and average wage rates would remain constant over time. A number of issues must therefore be addressed in updating the Census estimates to 1990 dollars. First, educational attainment has changed since 1979, with a greater proportion of the population attending college (especially a greater proportion of women). Second, wage rates have increased both due to productivity effects (real wage growth) and inflation. Third, age-specific employment rates may have changed. Women, in particular, are likely to have higher rates of labor force participation than in 1979.

In revising the Census estimates, the first issue was addressed by using more recent data on education. USDOC (1992) provides data on educational attainment for the 1991 population. For this analysis, data on the population over age 25 were used in order to remove the influence of those individuals too young to have completed schooling. The population data were used as weighting factors to derive a sex and education weighted average of expected lifetime earnings. So constructed, the weighted average adjusts the estimate based on 1979 data to current levels of educational attainment. The present value of lifetime earnings were thus calculated to be $349,000 (discounted at three percent) for the average work force participant. The next step in adjusting the earnings estimate is to apply an adjustment for wage growth to update from 1981 to 1990 dollars. The Bureau of Labor Statistic's Employment Cost Index rose from a level of 67.2 in 1981 to 105.4 in 1990, an increase of about 57 percent. Updating the average lifetime earnings to 1990 dollars yields a revised estimate of $547,000.

While more recent age, sex, and education-specific employment rates could be used to re-estimate labor force participation, a complete analysis requires steps to dampen the effects of cyclical unemployment. Such an exercise would require considerable effort and is beyond the scope of this analysis. To the extent that labor force participation has increased for specific groups since 1979, the adjusted value presented here underestimates the true expected lifetime earnings. For example, if the

[213] USDOC 1983.

percentage of female children eventually joining the permanent workforce is greater than the percentage of women over age 25 that worked in 1979, the expected lifetime earnings of female children would be greater than estimated in this analysis.

Note that use of earnings is an incomplete measure of an individual's value to society. Those individuals who choose not to participate in the labor force for all of their working years must be accounted for, since the lost value of their productive services may not be accurately measured by wage rates. The largest group are those who remain at home doing housework and child rearing. Also, volunteer work contributes significantly to social welfare and rates of volunteerism tend to increase with educational attainment and income.[214] If the opportunity cost of non-wage compensated work is assumed to be the average wage earned by persons of the same sex, age, and education, the average lifetime earnings estimates would be significantly higher and could be approximated by recalculating the tables using full employment rates for all age/sex/education groups. To be conservative, only the value of lost wages is considered in this analysis.

The adjusted value of expected lifetime earnings obtained above is a present value for an individual entering the labor force at age 18 and working until age 64. Because a lead-induced IQ decrement occurs in childhood, the $547,000 figure must be further discounted to the specific age at which the health effect is measured and adjusted for the probability that the infant would survive to age 18.[215] For an infant less than one year old, the present value of lifetime earnings discounted at three percent from age 18 and adjusted for survival would be $315,370. Combining this value with the estimate of percent wage loss per IQ point yields: $315,370 x 1.76 percent = $5,551 per lost IQ point.

Children with IQs Less Than 70

Quantifying the Number of Children with IQs Less than 70

Unlike the dose-response function used for IQ point loss, the dose-response function used to estimate the lead-induced incidence of IQ values below 70 is not a constant function across all blood lead levels. The dose-response function used in this analysis is derived from Wallsten and Whitfield (1986). Based on encoded expert opinion, Wallsten and Whitfield provide estimates of expected percentages of children with IQs below 70 for specified levels of blood lead. The opinions of the experts were averaged to develop a dose-response relationship for the risk of IQ falling below a threshold of 70 points. The expert judgements relating blood lead levels to mean IQ decrements were averaged to come up with a mean (over all experts) response rate of children having IQ below 70 points for discrete PbB levels ranging from 5 to 50 μg/dL (the specified levels varied by expert). Next, the PbB distribution was divided into 200 intervals from 0 to 50 μg/dL (i.e., 0.25, 0.5, 0.75, etc.). By interpolation of the mean values obtained from Wallsten and Whitfield, a mean response rate was associated with each of the 200 PbB intervals. Then, within specific ranges of the PbB distribution, all the data points were used to estimate a linear regression equation relating PbB value to mean response rate. The result is a piecewise-linear function of mean response rate for IQ below 70 points (see Table 129).

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[214] Statistical Abstract of the United States, 1986. Table No. 651, p. 383.

[215] Special education costs for children who do not survive to age 18 are not counted, which results in some underestimation of benefits. However, most child mortality occurs before the age of 7, when the special education begins, so this under-counting is not substantive.

This function quantifies only the change in the number of children who pass below the IQ=70 threshold. Any other changes in children's IQ are quantified using the IQ point loss function. Using these two endpoints additively does not result in double counting, because the value associated with the IQ point loss function is the change in worker productivity while the value associated with IQs less than 70 is the increased educational costs for the individual.

Table 129. Elements of Piecewise Linear Function for Estimating Probability of IQ < 70 as a Function of Blood Lead (PbB) Range.

PbB Range (µg/dl)	Slope	Intercept
0 - 5.0	2.04×10^{-4}	3.60×10^{-3}
5.1 - 7.5	4.88×10^{-4}	2.18×10^{-3}
7.6 - 10.0	1.068×10^{-3}	-2.17×10^{-3}
10.1 - 12.5	1.044×10^{-3}	-1.93×10^{-3}
12.6 - 15.0	9.76×10^{-4}	-1.08×10^{-3}
15.1 - 17.5	1.26×10^{-3}	-5.43×10^{-3}
17.6 - 22.5	1.328×10^{-3}	-6.53×10^{-3}
22.6 - 25.0	1.532×10^{-3}	-1.112×10^{-2}
> 25.0	1.464×10^{-3}	-9.42×10^{-3}

The piecewise linear function is used in conjunction with the PbB distribution of a population to estimate the response rate associated with each percentile group. The PbB distribution is developed for the population given geometric mean PbB, the assumed GSD. Once the PbB value is calculated, the slope and intercept coefficients for the appropriate linear segment in Exhibit 2-2 are used to calculate the mean IQ response rate for that PbB percentile. This response rate is multiplied by the exposed population in each PbB percentile grouping to estimate the proportion of infants in that group having IQ less than 70 points. Summing the number of infants below 70 points for each PbB percentile group yields the total number of cases for the exposed population.

As in the IQ point loss equation, the results of this function are applied to children age 0-6 and divided by seven to avoid double counting. (See discussion under equation 43).

Valuing the Reduction in Number of Children with IQs less than 70

To value the reduction in the number of children with IQs less than 70, the reduction in education costs were measured - a clear underestimate of the total benefits.[216] Kakalik et al. (1981), using data from a study prepared for the Department of Education's Office of Special Education Programs, estimated that part-time special education costs for children who remained in regular classrooms cost $3,064 extra per child per year in 1978. Adjusting for inflation and real income growth using the GNP price deflator yields an estimate of $6,318 per child in 1990 dollars. For the calculations, this incremental estimate of the cost of part-time special education was used to estimate the cost per year per child needing special education as a result of impacts of lead on mental development. Costs would be incurred from grades one through twelve. Discounting future expenses at a rate of three percent yields an expected present value cost of approximately $52,700 per infant (assuming compensatory education begins at age 7 and continues through age 18). Note that this underestimates the cost, since

[216] The largest part of this benefit is the parents' willingness to pay to avoid having their child become mentally handicapped, above and beyond the increased educational costs.

Kakalik et al. measured the increased cost to educate children attending regular school -- not a special education program.

Changes in Neonatal Mortality

Quantifying the relationship between PbB levels and neonatal mortality

U.S. EPA (1990) cites a number of studies linking fetal exposure to lead (via *in utero* exposure from maternal intake of lead) to several adverse health effects. These effects include decreased gestational age, reduced birth weight, late fetal death, and increases in infant mortality. The Centers for Disease Control (CDC, 1991a) presents a method to estimate changes in infant mortality due to changes in maternal blood lead levels during pregnancy.[217] The analysis links two relationships. The first relationship, between maternal blood lead level and gestational age of the newborn, was estimated by Dietrich et al. (1987). CDC then estimated infant mortality as a function of gestational age, using data from the Linked Birth and Infant Death Record Project from the National Center for Health Statistics. The resulting association is a decreased risk of infant mortality of 10^{-4} (or 0.0001) for each 1 μg/dL decrease in maternal blood lead level during pregnancy. This is the relationship used in the current analysis.

Valuing changes in neonatal mortality

Fisher et al. (1989) conducted a review of studies quantifying individuals' willingness to pay to avoid risks to life. In the reviewed studies, value estimates were obtained from hedonic wage studies, consumer market studies or contingent valuation studies. Based on that review, a range of $1.6 million to $8.5 million (1986 dollars) was recommended when valuing statistical deaths for policy evaluation. The Fisher et al. range has frequently been used by the U.S. EPA for a wide variety of policy analyses.

The above values are expressed in 1986 dollars and must be adjusted to be consistent with other benefits estimates, which are expressed in 1990 dollars. One potential concern in simply adjusting the above estimates for inflation is that society's willingness to pay is likely to increase as national income increases. Since risk reductions are a normal good, willingness to pay is likely to increase with national income. This is clearly an empirical issue but casual examination of the literature suggests that the larger willingness to pay estimates come from studies in later years when incomes were higher. Since the regulatory costs were expressed in 1990 dollars, the statistical life values above are adjusted using the relative change in nominal, per capita GNP from 1986 to 1990 (approximately a 25 percent increase). Using the change in nominal GNP accounts for changes in income as well as inflation. The use of per capita GNP eliminates the influence of population growth on national income. The adjustment results in a range of $2 million to $11 million as the value of a statistical life.

The Fisher et al. range, adjusted to 1990 dollars, is consistent with other recent reviews of the range of value of a statistical life. Viscusi (1992) reviews the literature on value of life estimates,

[217] The estimated change in infant mortality due to change in birthweight was not modeled because the data relating prenatal lead exposure to birth weight is not as strong as data relating lead exposure and gestational age.

including all of the research included in Fisher et al. as well as additional studies published through 1991. He concludes that–

> *"most of the reasonable estimates of the value of life are clustered in the $3 to $7 million range (in 1990 dollars). Moreover, these estimates are for the population of exposed workers, who generally have lower incomes than the individuals being protected by broadly based risk regulations. Recognition of the income elasticity of the value of life will lead to the use of different value of life depending on the population being protected." (p.74)*

Viscusi thus recommends a narrower range that is wholly contained within the adjusted Fisher et al. range, and supports the nature of the adjustment to 1990 dollars described for Fisher et al.

The value of avoiding a statistical death used in this analysis is $4.8 million. This is the proposed value by Industrial Economics Incorporated in their Section 812 valuation work.[218] There is some EPA precedent for using this figure as well. The U.S. EPA Office of Indoor Air (1994) identified 26 acceptable "value of life" studies and calculated the geometric mean estimated willingness to pay to avoid a "statistical death" as $4.8 million (1990 dollars). This same estimated value is used for all premature mortality calculations in this report regardless of age, gender, racial or other demographic characteristics of the population at risk.

Health Benefits to Men

The health effects in adults that are quantified and included in the benefits analysis are all related to the effects of lead on blood pressure.[219] These quantified health effects include increased incidence of hypertension, heart attack, stroke and mortality. Both men and women suffer from these lead-induced health effects, but the estimated relationships differ for men and women. This section describes the quantified health effects for men, and Section 2.4 describes the health effects for women.

Hypertension

Quantifying the relationship between PbB levels and hypertension

Elevated blood lead has been linked to elevated blood pressure (BP) in adult males, especially men aged 40-59 years [220] Further studies have demonstrated a dose-response relationship for hypertension (defined as diastolic blood pressure above 90 mm Hg for this model) in males aged 20-74 years.[221] This relationship is:

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[218] IEc 1992, IEc 1993.

[219] Citing laboratory studies with rodents, U.S. EPA (1990) also presents evidence of the genotoxicity and/or carcinogenicity of lead compounds. While such animal toxicological evidence suggests that human cancer effects are possible, dose-response relationships are not currently available.

[220] Pirkle et al., 1985.

[221] Schwartz, 1988.

$$\Delta Pr(HYP) = \frac{1}{1 + e^{2.744 - .793*(\ln PbB_1)}} - \frac{1}{1 + e^{2.744 - 0.793*(\ln PbB_2)}} \quad (44)$$

where:

ΔPr(HYP)	=	the change in the probability of hypertension;
PbB_1	=	blood lead level in the control scenario; and
PbB_2	=	blood lead level in the no-control scenario.

Valuing reductions in hypertension

The best measure of the social costs of hypertension, society's willingness to pay to avoid the condition, cannot be quantified without basic research well beyond the scope of this project. Ideally, the measure would include all the medical costs associated with treating hypertension, the individual's willingness to pay to avoid the worry that hypertension could lead to a stroke or heart attack, and the individual's willingness to pay to avoid changes in behavior that may be required to reduce that probability that hypertension leads to a stroke or heart attack. Medical costs of hypertension can be divided into four categories: physician charges, medication costs, hospitalization costs and lost work time.

This analysis uses recent research results to quantify two components of this benefit category. Cropper and Krupnick (1989), using data from the National Medical Care Expenditure Survey, have estimated the medical costs of hypertension. These costs include physician care, drugs and hospitalization costs. In addition, hypertensives have more bed disability days and work loss days than others of their age and sex. Krupnick and Cropper estimated the increase in work loss days at 0.8 per year, and these were valued at the mean daily wage rate. Adjusting the above costs to 1990 dollars gives an estimate of the annual cost of each case of hypertension of $681. The estimate is likely to be an underestimate of the true social benefit of avoiding a case of hypertension for several reasons. First, a measure of the value of pain, suffering and stress associated with hypertension is not included. Second, the direct costs (out-of-pocket expenses) of diet and behavior modification (e.g., salt-free diets, etc.) are not valued. These costs are likely to be significant, since modifications are typically severe. Third, the loss of satisfaction associated with the diet and behavior modifications are ignored. Finally, the medication for hypertension produces side effects including drowsiness, nausea, vomiting, anemia, impotence, cancer, and depression. The benefits of avoiding these side effects are not included in this estimate.

Changes In Coronary Heart Disease

Quantifying the relationship between blood lead and blood pressure

Because blood pressure has been identified as a risk factor in a number of cardiovascular illnesses,[222] it is useful to quantify the effect of changes in blood lead levels on changes in blood pressure

[222] Shurtleff, 1974; McGee and Gordon, 1976; Pooling Project, 1978.

for reasons other than predicting the probability of hypertension. A meta-analysis of several studies estimated that a 1.4 mm Hg change in blood pressure is expected for a one μg/dL change in PbB.[223] The benefits analysis uses the following equation based on the Schwartz relationship:

$$\Delta DBP_{men} = 1.4 \times (PbB_1 - PbB_2) \quad (45)$$

where:

ΔDBP_{men}	=	the change in men's diastolic blood pressure expected from a change in PbB;
PbB_1	=	blood lead level in the control scenario; and
PbB_2	=	blood lead level in the no-control scenario.

This blood lead to blood pressure relationship is used to estimate the incidence of heart attacks (initial coronary heart disease), strokes (atherothrombotic brain infarctions and initial cerebrovascular accidents) and mortality in men.

Quantifying the relationship between blood pressure and coronary heart disease

The relationship between blood pressure and other health effects can be used to predict increased probabilities of the initial occurrence of heart attack and stroke.[224] Increased blood pressure would also increase the probability of reoccurrences of heart attacks and strokes, but these quantified relationships are not available. First-time heart attacks (coronary heart disease events) in men can be predicted using an equation with different coefficients for each of three age groups. For men between 40 and 59 years old, information from a 1978 study by the Pooling Project Research Group (PPRG) is used. PPRG (1978) presents a multivariate model (controlling for smoking and serum cholesterol) that relates the probability of coronary heart disease (CHD) to blood pressure. The model used data from five different epidemiological studies. From this study, the equation for the change in 10-year probability of occurrence of CHD is:

$$\Delta Pr(CHD_{40-59}) = \frac{1}{1 + e^{4.996 - 0.030365 \cdot DBP_1}} - \frac{1}{1 + e^{4.996 - 0.030365 \cdot DBP_2}} \quad (46)$$

where:

$\Delta Pr(CHD_{40-59})$	=	change in 10-year probability of occurrence of CHD event for men between 40-59 years old,
DBP_1	=	mean diastolic blood pressure in the control scenario; and
DBP_2	=	mean diastolic blood pressure in the no-control scenario.

The relationship between BP and first time heart attacks in older men was determined from information presented in Shurtleff (1974). This study also uses data from the Framingham Study to estimate univariate relationships between BP and a variety of health effects by sex and for each of the

[223] Schwartz, 1992a.

[224] U.S. EPA, 1987.

following age ranges: 45-54, 55-64, and 65-74 years. Single composite analyses for ages 45-74 were also performed for each sex. For every equation, t-statistics on the variable blood pressure are significant at the 99th percent confidence interval. For men aged 60 to 64 years old, first-time heart attacks can be predicted from the following equation:

$$\Delta Pr(CHD_{60-64}) = \frac{1}{1 + e^{5.19676 - 0.02351 * DBP_1}} - \frac{1}{1 + e^{5.19676 - 0.02351 * DBP_2}} \quad (47)$$

where:

$\Delta Pr(CHD_{60-64})$ = change in 2 year probability of occurrence of CHD event for men from 60 to 64 years old;
DBP_1 = mean diastolic blood pressure in the control scenario; and
DBP_2 = mean diastolic blood pressure in the no-control scenario.

For men aged 65 to 74 years old, the following equation uses data from Shurtleff (1974) to predict the probability of first-time heart attacks:

$$\Delta Pr(CHD_{65-74}) = \frac{1}{1 + e^{4.90723 - 0.02031 * DBP_1}} - \frac{1}{1 + e^{4.90723 - 0.02031 * DBP_2}} \quad (48)$$

where:

$\Delta Pr(CHD_{65-74})$ = change in 2 year probability of occurrence of CHD event for men from 65 to 74 years old;
DBP_1 = mean diastolic blood pressure in the control scenario; and
DBP_2 = mean diastolic blood pressure in the no-control scenario.

Valuing reductions in CHD events

There are two categories of benefits associated with reductions in CHD events: the medical costs and foregone earnings associated with the treatment of and recovery from a heart attack, and the additional willingness to pay to avoid the pain, discomfort, and potentially reduced quality of life after suffering a CHD event. While estimates of the medical costs and foregone earnings associated with CHD events are given in USEPA (1987), these values are likely a small component of the total value of avoiding a statistical case of CHD. There are no studies that explicitly measure people's willingness to pay to avoid a CHD event. However, the EPA economic research program funded a series of studies to estimate the willingness to pay to avoid health risks.

While a WtP value for avoiding a CHD event is not available, several studies estimate the WtP to avoid a statistical case of chronic bronchitis. The present study uses the valuation for chronic bronchitis as a surrogate for CHD based on the assumption that people believe that the pain, suffering and activity restrictions associated with a non-fatal CHD event are at least as severe as those associated with chronic bronchitis. The results of the chronic bronchitis study valuation studies are summarized on a consistent basis, and the estimated values are reported in 1990 dollars in Viscusi (1992). Viscusi et al. (1991) and Krupnick and Cropper (1992) estimated the WtP for chronic bronchitis avoidance in terms of willingness to trade the risk of chronic bronchitis for the risk of a fatal auto accident (a risk-risk

tradeoff). Four central estimates result from the studies: (1) a mean value of $883,000 (Viscusi et al., 1991); (2) a median value of $457,000 which captures the skewness of the response distribution (Viscusi et al., 1991; (3) $210,000 recommended by Rowe et al based on Viscusi et al. (1991) with an adjustment for the severity of the chronic bronchitis case; and (4) a value of $800,000 derived from the two-step approach of Viscusi et al. (1991) but adjusting the skewness of auto death using the median value of auto death. To derive a single point estimate of the value of an avoided case of lead exposure-induced CHD, the present study uses the mean of the four estimates outlined above, which equals $587,500, as the WtP for avoidance of CHD.

The uncertainty surrounding the point estimate is also derived from the four valuations given above. It is assumed that individuals are WtP one of the four valuation estimates to avoid CHD (equated to 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 this endpoint.

The valuation for hypertension is additive with the valuation for CHD despite the fact that the conditions often occur together because the two values represent different costs associated with the conditions. The valuation for hypertension is based on loss of work days as a result of hypertension and some of the medical costs associated with treating hypertension. Alternatively, the valuation for a heart attack is based on the willingness to pay to avoid the pain and suffering of the heart attack itself. Therefore, these two valuations can be separated and added together.

Changes in Initial Cerebrovascular Accidents and Initial Atherothrombotic Brain Infarctions

Quantifying the relationship between blood pressure and first-time stroke

Two types of health events are categorized as strokes: initial cerebrovascular accidents (CA) and initial atherothrombotic brain infarctions (BI). The risk has been quantified for the male population between 45 and 74 years old.[225] For initial cerebrovascular accidents, the logistic equation is:

$$\Delta Pr(CA) = \frac{1}{1 + e^{8.58889 - 0.04066 * DBP_1}} - \frac{1}{1 + e^{8.58889 - 0.04066 * DBP_2}} \qquad (49)$$

where:

$\Delta Pr(CA_{men})$	=	change in 2 year probability of cerebrovascular accident in men;
DBP_1	=	mean diastolic blood pressure in the control scenario; and
DBP_2	=	mean diastolic blood pressure in the no-control scenario.

[225] Shurtleff, 1974.

For initial atherothrombotic brain infarctions, the logistic equation is:

$$\Delta Pr(BI) = \frac{1}{1 + e^{9.9516 - 0.04840 * DBP_1}} - \frac{1}{1 + e^{9.9516 - 0.04840 * DBP_2}} \quad (50)$$

where:

$\Delta Pr(BI_{men})$	=	change in 2 year probability of brain infarction in men;
DBP_1	=	mean diastolic blood pressure in the control scenario; and
DBP_2	=	mean diastolic blood pressure in the no-control scenario.

Valuing reductions in strokes

As with hypertension, very few data exist for estimating society's WtP to avoid a lead exposure-induced stroke. Data on medical costs are available, but a value for pain, suffering, and reduced quality of life has not been rigorously quantified. Thus, as in the case of CHD, the best option is to use the chronic bronchitis value ($587,500 per case avoided) as a surrogate. Again, the activity restrictions associated with strokes are assumed to be at least as severe as those for chronic bronchitis. Since strokes often include lingering paralysis, actual WtP is likely to be quite substantial.

Although the foregone medical costs of averted strokes are not explicitly included in the measure of benefits, it is worth discussing the available data. Hartunian et al. (1981) estimated the medical expenses and foregone earnings associated with three types of strokes: hemorrhagic, infarctive, and transient ischemic attacks. Weighing these cost estimates by the distribution of these types of strokes by age yields an average estimate of $48,000. Given that these treatment costs are based on old data, and represent only a fraction of the WtP to avoid pain, suffering and reduced quality of life, they are not included in the estimate of benefits per case avoided used in this analysis. Given that the chronic bronchitis valuation is used as a surrogate, the uncertainty surrounding the estimate is the same as that presented for CHD events above.

Changes in Premature Mortality

Quantifying the relationship between blood pressure and premature mortality

Information also exists to predict the increased probability of premature death from all causes as a function of elevated blood pressure. U.S. EPA (1987) used population mean values for serum cholesterol and smoking to reduce results from a 12 year follow-up of men aged 40-54 in the Framingham Study (McGee and Gordon, 1976) to an equation in one explanatory variable:

$$\Delta Pr(MORT_{40-54}) = \frac{1}{1 + e^{5.3158 - 0.03516 * DBP_1}} - \frac{1}{1 + e^{5.3158 - 0.03516 * DBP_2}} \quad (51)$$

where:

$\Delta Pr(MORT_{40\text{-}54})$ = the change in 12 year probability of death for men aged 40-54;
DBP_1 = mean diastolic blood pressure in the control scenario; and
DBP_2 = mean diastolic blood pressure in the no-control scenario.

Information from Shurtleff (1974) can be used to estimate the probability of premature death in men older than 54 years old. This study has a 2 year follow up period, so a 2 year probability is estimated. For men aged 55 to 64 years old, mortality can be predicted by the following equation:

$$\Delta Pr(MORT_{55-64}) = \frac{1}{1 + e^{4.89528 - 0.01866 * DBP_1}} - \frac{1}{1 + e^{4.89528 - 0.01866 * DBP_2}} \quad (52)$$

where:

$\Delta Pr(MORT_{55\text{-}64})$= the change in 2 year probability of death in men aged 55-64;
DBP_1 = mean diastolic blood pressure in the control scenario; and
DBP_2 = mean diastolic blood pressure in the no-control scenario.

For men aged 65 to 74 years old, premature mortality can be predicted by the following equation:

$$\Delta Pr(MORT_{65-74}) = \frac{1}{1 + e^{3.05723 - 0.00547 * DBP_1}} - \frac{1}{1 + e^{3.05723 - 0.00547 * DBP_2}} \quad (53)$$

where:

$\Delta Pr(MORT_{65\text{-}74})$ = the change in 2 year probability of death in men aged 55-64;
DBP_1 = mean diastolic blood pressure in the control scenario; and
DBP_2 = mean diastolic blood pressure in the no-control scenario.

Valuing reductions in premature mortality

As discussed above, all premature mortality is valued at $4.8 million in this report. Because this valuation is based on the willingness to pay to avoid death itself, and the CHD valuation is based on the willingness to pay to avoid the pain and suffering of a heart attack, these two endpoints are additive as well.

Health Benefits to Women

Available evidence suggests the possibility of health benefits from reducing women's exposure to lead. Recent expanded analysis of data from the second National Health and Nutrition Examination

Survey[226] (NHANES II) by Schwartz (1990) indicates a significant association between blood pressure and blood lead in women. Another study, by Rabinowitz et al. (1987), found a small but demonstrable association between maternal blood lead and pregnancy hypertension and blood pressure at time of delivery.

A quantitative estimate of female blood pressure related to PbB can be estimated from a recent review of ten published studies.[227] All of the reviewed studies included data for men, and some included data for women. A concordance procedure was used to combine data from each study to predict the decrease in diastolic BP associated with a decrease from 10 μg/dL to 5 μg/dL PbB. The results suggest that this decrease in PbB would decrease diastolic BP by 1 mm Hg in adult males, and about 0.6 mm Hg in adult females. Thus, lead's effect on BP in women is estimated to be 60 percent of the effect seen in men. Applying this value to Equation 2-2 for men, the resulting equation is:

$$\Delta DBP_{women} = (0.6 \times 1.4) \times (PbB_1 - PbB_2) \quad (54)$$

where:

ΔDBP_{women}	=	the change in women's diastolic blood pressure expected from a change in PbB;
PbB_1	=	blood lead level in the control scenario; and
PbB_2	=	blood lead level in the no-control scenario.

Although women are at risk of having lead-induced hypertension, there is not a dose-response function for hypertension in women available at this time. Omitting the hypertension benefits for women creates an underestimate of the total benefits, but the impact on the total benefits estimation will likely be small. Lead raises blood pressure in women less than in men, so the probability of causing hypertension is likely to be less than in men, and the total value of hypertension in men is a small portion of the overall estimated benefits.

Changes in Coronary Heart Disease

Quantifying the relationship between blood pressure and coronary heart disease

Elevated blood pressure in women results in the same effects as for men (the occurrence of heart attack, two types of stroke, and premature death). However, the general relationships between BP and these health effects are not identical to the dose-response functions estimated for men. All relationships presented here have been estimated for women aged 45 to 74 years old using information from Shurtleff (1974). First-time heart attacks in women can be estimated from the following equation:

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[226] The Second National Health and Nutrition Examination Survey (NHANES II) was conducted by the U.S. Department of Health and Human Services from 1976 to 1980 and provides researchers with a comprehensive set of nutritional, demographic and health data for the U.S. population.

[227] Schwartz, 1992b.

$$\Delta Pr(CHD_{women}) = \frac{1}{1 + e^{6.9401 - 0.03072 * DBP_1}} - \frac{1}{1 + e^{6.9401 - 0.03072 * DBP_2}} \quad (55)$$

where:

$\Delta Pr(CHD_{women})$ = change in 2 year probability of occurrence of CHD event for women aged 45-74;
DBP_1 = mean diastolic blood pressure in the control scenario; and
DBP_2 = mean diastolic blood pressure in the no-control scenario.

Valuing reductions in CHD events

Values of reducing CHD events for women are assumed to be equal those calculated for men (above): $587,500 per CHD event.

Changes in Atherothrombotic Brain Infarctions and Initial Cerebrovascular Accidents

Quantifying the relationship between blood pressure and first-time stroke

For initial atherothrombotic brain infarctions in women, the logistic equation is:

$$\Delta Pr(BI_{women}) = \frac{1}{1 + e^{10.6716 - 0.0544 * DBP_1}} - \frac{1}{1 + e^{10.6716 - 0.0544 * DBP_2}} \quad (56)$$

where:

$\Delta Pr(BI_{women})$ = change in 2 year probability of brain infarction in women aged 45-74;
DBP_1 = mean diastolic blood pressure in the control scenario; and
DBP_2 = mean diastolic blood pressure in the no-control scenario.

The relationship between BP and initial cerebrovascular accidents can be predicted by the following logistic equation:

$$\Delta Pr(CA_{women}) = \frac{1}{1 + e^{9.07737 - 0.04287 * DBP_1}} - \frac{1}{1 + e^{9.07737 - 0.04287 * DBP_2}} \quad (57)$$

where:

$\Delta Pr(CA_{women})$ = change in 2 year probability of cerebrovascular accident in women aged 45-74;
DBP_1 = mean diastolic blood pressure in the control scenario; and
DBP_2 = mean diastolic blood pressure in the no-control scenario.

Valuing reductions in strokes

Values of reducing stroke events for women are assumed to be equal those calculated for men (above): $587,500 per BI and CA event.

Changes in Premature Mortality

Quantifying the relationship between blood pressure and premature mortality

The risk of premature mortality in women can be estimated by the following equation:

$$\Delta Pr(MORT_{women}) = \frac{1}{1 + e^{5.40374 - 0.01511 * DBP_1}} - \frac{1}{1 + e^{5.40374 - 0.01511 * DBP_2}} \quad (58)$$

where:

$\Delta Pr(MORT_{women})$ = the change in 2 year probability of death for women aged 45-74;
DBP_1 = mean diastolic blood pressure in the control scenario; and
DBP_2 = mean diastolic blood pressure in the no-control scenario.

Valuing reductions in premature mortality

Values of reducing premature mortality for women are assumed to be equal those calculated for all premature mortality (above): $4.8 million per premature mortality.[228]

Industrial Processes and Boilers and Electric Utilities

This section describes the methods and data sources used to estimate changes in blood lead levels due to changes in lead emissions from industrial processes and boilers between 1970 and 1990 and from electric utilities between 1975 and 1990. It also presents estimates of the resulting changes in health effects and the monetary value of these effects.

[228] IEc 1992, IEc 1993.

Methods Used to Determine Changes in Lead Emissions from Industrial Processes from 1970 to 1990

This analysis used several sources to determine the changes in facility-specific emissions of lead from industrial processes. To summarize, the analysis extracted 1990 facility-specific lead emissions data from the Toxics Release Inventory (TRI), which provides recent emissions data for over 20,000 U.S. manufacturing facilities. This study then adjusted these data by the relative changes in lead emissions between 1970 and 1990; these relative changes were derived from several data sources described below. This method yielded facility-specific emissions for five year intervals between 1970 and 1990 for both the controlled and uncontrolled scenarios. The five-year values were interpolated to derive annual changes for each year between 1970 and 1990. Specific details on this approach are given below.

TRI Data

The Toxics Release Inventory (TRI) is mandated by the Superfund Amendment Reauthorization Act (SARA) Title III Section 313 and requires that U.S. manufacturing facilities with more than 10 employees file annual reports documenting multimedia environmental releases and off-site transfers for over 300 chemicals. Facilities report both stack and fugitive releases to air. Reported releases are generally estimates rather than precise quantifications. Emissions data can be presented as numerical point estimates, or, if releases are below 1,000 pounds, as an estimated range of emissions.

TRI data are available for the year 1988-1990. From the TRI data base, this analysis extracted data from the reporting year 1990 for all facilities reporting emissions of lead to air, as either stack or fugitive emissions. Data were reported as annual emissions (in pounds per year). Where emissions are reported as a range, this analysis used the upper bound of the range to represent the emissions.[229] TRI facilities also report their location by latitude and longitude. In order to later match facilities emitting lead with Census data on surrounding exposed populations, this analysis uses the latitudes and longitudes of lead-emitting facilities.

Derivation of Industrial Process Emissions Differentials 1970-1990

The TRI database is the Agency's single best source of consistently reported release data; however, the database does not include information for most of the years modeled in this analysis. Furthermore, this analysis required estimates of hypothetical emissions in the absence of the CAA. Therefore, estimates were created for the emissions of lead from industrial sources under the CAA, and, in the absence of the CAA, for the years 1970, 1975, 1980, 1985, and 1990. The percent changes, or differentials, reflected by these estimates were then applied to the 1990 TRI data to obtain facility-level release estimates for the years of interest for the controlled and uncontrolled scenarios.

[229] Ranges are infrequently reported and are either reported as 0-500 lbs. or 500-1000 lbs. The infrequency of the incidence of a facility reporting a range and the relatively small quantities of lead released by those facilities means any overestimation of benefits that results from using the upper limit of the range is extremely minor.

The method for creating these differentials captured the two potential causes of the differences between emissions from industrial sources regulated by the CAA and those emissions in the absence of the CAA. The first cause of the difference in emissions is a change in overall industrial output, resulting from the macroeconomic impact of the CAA. The second element is a change in emissions per unit of output, which results from the adoption of cleaner processes and the application of emissions control technology mandated by the CAA. The methods used to project the effects of these two causes, described below, were designed to be as consistent as possible with other emissions projection methods for other segments of the CAA retrospective analysis.

Data sources

Data for the differentials estimates were taken from the following sources:

- the Jorgenson/Wilcoxen (J/W) model projections, conducted as part of the Section 812 analysis. This data source addresses the first cause of changes in emissions: the macroeconomic changes that resulted from the implementation of the 1970 CAA. The J/W model calculated the change in economic output for each of thirty-five industrial sectors, roughly analogous to 2-digit SIC codes, that resulted from the CAA's implementation. The specific output used from the J/W model in this analysis was the percentage change in economic output for the various industrial sectors, rather than any absolute measure of economic activity.

- the 1991 OAQPS Trends database. This data base is an emissions projection system that was used to produce the report, "The National Air Pollutant Emission Estimates, 1940-1990." It contains information on economic activity, national level emissions and emission controls, by industrial process, from 1970 through 1990. Three different elements were extracted from the Trends database: the emissions of lead per unit economic output for various industrial processes for the years 1970-1990; annual economic output data for these industrial processes; and the emission calculation formula.

- the National Energy Accounts (NEA), compiled by the Bureau of Economic Analysis. This database records the historical levels of industrial energy consumption, disaggregated by fuel type at the approximately three-digit SIC code level.

The manner in which these data were combined to derive lead emissions estimates is described below.

Estimates of industrial process emissions in the controlled scenario

Emissions data for industrial processes were estimated for the years 1970, 1975, 1980, 1985, and 1990. For each of these years, this analysis extracted an emission factor and a control efficiency for each lead-emitting industrial process in the Trends database. Emissions factors are expressed as amount of lead emitted per unit of economic activity, and control efficiencies are reported as the percent that emissions are reduced through the application of pollution control technology to the process. The year-specific emission factors and control efficiencies were multiplied by the economic activity data for that

year, for that process, as reported in the Trends database, using the following equation found in the Trends report:

$$Emissions = (Economic\ Activity) \times (Emission\ Factor) \times (1 - Control\ Efficiency) \quad (59)$$

This calculation yielded the estimated controlled scenario emissions, by industrial process. Industrial processes were then assigned to an NEA code. Finally, all processes assigned to a given NEA code were summed to give a total emissions estimate for that NEA code.

Estimates of industrial process emissions in the uncontrolled scenario

The results from the J/W model were used to estimate process emissions in the uncontrolled scenario. As stated above, the J/W model provides percent changes in economic outputs by industrial sector. To use these values, lead-emitting industrial processes (in the Trends database) were assigned to a J/W sector. The percent change for that sector from the J/W model was then used to adjust the economic activity data for that process from the Trends database. These adjusted economic output figures were used together with 1970 emission factors and control efficiencies to derive the estimated lead emissions for each industrial process in the uncontrolled scenario. The 1970 emission factors and control efficiencies were used for all years in the analysis (1970, 1975, 1980, 1985 and 1990) in the uncontrolled scenario; this assumes that emissions per unit economic output and control efficiencies would have been constant over time in the absence of the CAA. This is the same approach that was used to project the changes in emissions from industrial processes for other criteria pollutants in other portions of the CAA retrospective analysis. The process-level emissions were then aggregated to the NEA-code level, as in the controlled scenario.

Matching TRI Data to Industrial Process Emissions Differentials

The methods described in the preceding section yielded emissions estimates from industrial processes in the controlled and uncontrolled scenarios, by NEA code. We used these estimates to derive percent changes in emissions between controlled and uncontrolled scenarios, by NEA code, for application to the TRI emissions data. However, since TRI data are reported by SIC code, we first "mapped" NEA codes to the appropriate SIC codes, and used the percent change for each NEA code to represent the percent change for all SIC codes covered by that NEA code.

It should be noted that the Trends data base covers only the most important sources of lead in air, not all sources; as a result, not all SIC codes reporting lead emissions in TRI correspond to an NEA code for which emission differentials have been estimated. However, we assume that the TRI emissions sources that have a match are the most important sources of lead air emissions. In fact, although only 48 out of 519 legitimate SIC codes reporting lead emissions in TRI have matching differentials, these SIC codes account for over 69 percent of the lead emissions reported in TRI. The remaining 31 percent of the emissions are distributed relatively evenly among the remaining 471 SIC codes, each of which contributes a small amount to total emissions.

For the 31 percent of the emissions without differentials, this analysis has no information regarding the change in the lead emissions over time or between the controlled and uncontrolled

scenarios; therefore, we are unable to predict benefits attributable to the CAA for these emission sources. Although excluding these sources may lead us to underestimate total benefits, we believe these sources are unlikely to contribute significantly to the difference between controlled and uncontrolled scenarios. The Trends data focuses on the point sources of lead emissions of greatest concern to the Project Team and of greatest regulatory activity. If a process within an SIC code does not appear in the Trends, it is unlikely to have had specific CAA controls instituted over the past 20 years. A lack of control efficiencies for smaller sources prevents them from being included.

It should also be noted that the total industrial process emissions of lead estimated in the 1990 Trends report actually exceeds the reported lead emissions in TRI, despite the fact that TRI covers more SIC codes. This is probably attributable in part to the fact that TRI covers only a subset of the facilities contributing to economic output in an SIC code. TRI reporting rules only require facilities with greater than 10 employees and who use certain amounts of lead in their processes to submit information to TRI, while the Trends report attempted to estimate emissions from all sources contributing to the economic output for the industrial sector, regardless of size. However, the components of the Trends data base used in this analysis (i.e., emissions factors, economic output data) represent typical conditions at average facilities; they do not allow for the representation of the distribution of emissions across particular facilities. In contrast, a major strength of the TRI is its match of emissions data with geographical information. Because the distribution of emissions geographically determines the size of exposed populations, this analysis used the TRI data, rather than Trends data, to characterize lead release quantities, and used the Trends figures only to characterize relative emissions and changes over time, rather than to estimate total quantities.

Because the Trends data are intended only as an estimate of emissions using typical conditions at average facilities, and do not capture the differences in facility-level emissions, the data do not provide sufficient information to make specific quantitative adjustments to the TRI-based benefits estimates to account for the overall higher emissions estimates in Trends. However, since Trends does generally suggest that there are many more sources than are accounted for by TRI, it is possible that our benefits calculations may be underestimated.

Some additional assumptions were necessary when matching the TRI lead release data and the differentials from the Trends data. Ideally, we would know whether the facilities present at a given location, as reported in the 1990 TRI, were present and operating in earlier years; whether facilities operating in 1970 have ceased to operate; and whether new facilities would have been constructed in the uncontrolled situation. Unfortunately, data do not exist in an accessible form at this level of detail for the years 1970 through 1990. Therefore, for the purposes of this exercise, we have assumed that the locations and numbers of the 1990 sources are the same as they were in 1970.

Methods Used to Determine Changes in Lead Emissions from Industrial Boilers from 1970 to 1990

Several sources were used to determine the change in lead emissions from industrial boilers. TRI locational data, Trends database national fuel consumption levels and emissions factors, and NEA and SIC codes were used to derive the emissions for the controlled and uncontrolled scenarios.

TRI Data

The TRI does not appear generally to contain combustion emissions data. In general, the emissions data is from process sources. We reached this conclusion based on two pieces of information:

(1) TRI reporting requirements: TRI has three reporting requirements: (a) the facility must fall in SIC codes 20-39; (b) the facility must employ more than 10 persons; and (c) the facility must manufacture or process >25,000 lbs. of a TRI chemical, or otherwise use >10,000 lbs. Firms must submit reports only for the chemical that exceeds the thresholds given in item (c), but they must report all releases of that chemical, including releases from uses that would not qualify alone. If the TRI chemical is part of a blended substance and the quantity of the TRI chemical in the blend exceeds the threshold, it must be reported. In our case, if the amount of lead in fuel were to exceed the 10,000 lbs. threshold, then the firm would be required to report all emissions of lead from combustion of fuel. There is an exemption, however, for ingredients present in small proportions. If the amount of lead in the oil were less than 0.1 percent (1000 ppm), then the firm is not required to report the emissions.

The conclusion from the above information is that most firms burning used oil are probably not reporting lead combustion emissions to TRI because these releases fall outside the TRI reporting requirements. The concentration at which lead is typically found is used oil (100 ppm) (NRDC, 1991) is much less than the minimum concentration required for reporting (1000 ppm).

(2) Use data from the TRI data base: The hypothesis that firms do not report lead combustion was confirmed by an analysis of the data submitted by the firms reporting lead use to TRI. On the TRI submission forms, firms must indicate how the chemical is used. Our analysis of category codes submitted by firms reporting lead emissions showed the following four use category reports: as a formulation component; as a reactant; as an article component; and repackaging only. None of these category codes suggest that the source of the reported lead release is combustion. Therefore, we may conclude that all of the lead emissions reported in TRI are process emissions.

Based on these analyses, this analysis could not use the TRI release data to evaluate releases of lead from industrial combustion. However, this study still used the TRI geographical information to locate industrial facilities by longitude and latitude in order to combine combustion data with population information. For combustion emissions, the calculations included all TRI reporting facilities, not just those who report lead emissions. The assignment of combustion emissions to these facilities is described below.

Derivation of Industrial Combustion Emissions 1970-1990

As with industrial process emissions, estimates were created for the emissions of lead from industrial combustion under the CAA, and in the absence of the CAA, for the years 1970, 1975, 1980, 1985, and 1990. These emissions estimates were used, in combination with the TRI data base geographic information, to obtain facility-level release estimates for the years of interest for the controlled and uncontrolled scenarios. The method for deriving these emissions estimates included both the macroeconomic impact of the CAA and the change in emissions per unit of output that resulted from specific pollution control mandates of the CAA. The same data sources were used to derive combustion differentials as were used to derive process differentials. The particular data elements and the methods

by which these data were combined to derive lead emissions estimates from industrial combustion are described below.

Estimates of combustion emissions in the controlled scenario

The Trends database contains a national aggregate industrial fuel consumption estimate, by fuel type (coal, natural gas, oil). For each fuel type, the fuel consumption estimate was disaggregated by the share of that fuel used by each NEA industrial category, using the NEA data base. It should be noted that the NEA includes data only for the years 1970 through 1985. For 1990, the 1985 figures were used to disaggregate the national-level consumption figure into NEA industrial categories.

The Trends database also contains emissions factors for industrial fuel use, by fuel type, as well as control efficiencies. The lead emissions from industrial combustion for each NEA category was 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. By using the NEA data to disaggregate the industrial fuel consumption figures, the analysis assumes that the industrial combustion emissions are the same among all industries covered by a given NEA code, an assumption which may bias the analysis.

Estimates of combustion emissions in the uncontrolled scenario

As in the controlled scenario, the national aggregate industrial fuel consumption estimate, by fuel type (coal, natural gas, oil), was disaggregated by the share of that fuel used by each NEA industrial category. The fuel use was then adjusted by one of two factors: (1) seven of the NEA codes were specifically modeled by the Industrial Combustion Emissions (ICE) model -- for these sectors, the ICE modeled percent changes were used instead of J/W percent changes; or (2) the remaining NEA codes were matched to J/W sectors -- the J/W percent changes were then applied to those matched NEA codes. These fuel use estimates were then combined with the 1970 emission factors and control efficiencies for industrial combustion by fuel type from the Trends database to obtain combustion-related lead emissions from industrial boilers in the uncontrolled scenario, by NEA code.

The process-specific data in the Trends database, and the energy use data in the NEA, are much more disaggregated than the J/W sectoral projections. For the purpose of the analysis, it was assumed that all of the specific industrial processes in the Trends database and industrial categories in the NEA dataset assigned to a given J/W sector changed at the same rate as the entire J/W sector. For example, if the economic activity in the J/W Sector 20, "Primary Metals," changed by one percent between the controlled and uncontrolled scenarios, then the analysis assumed that economic activity in each industrial process assigned to the Primary Metals sector also increased by one percent. This approach assumes that the economic activity of specific industries within a sector are equally affected by the imposition of the CAA. This assumption is consistent with the projection of the change in emissions from industrial processes for the other criteria air pollutants, which were calculated using a similar process.

Matching TRI Data to Industrial Combustion Emissions Data

Because of the structure of the TRI reporting requirements, it does not appear that TRI generally contains releases from combustion sources. Although TRI may incidentally contain lead combustion emissions, TRI would contain data on such releases only if the reporting facility also used more than 10,000 pounds of lead per year for manufacturing or processing. As a result, the combustion releases, estimated using the methods described above, do not have corresponding data in the TRI data base. Therefore, we devised a different method for estimating benefits from changes in combustion releases.

The first step in the method was to divide the estimates of total releases of lead from industrial combustion, by NEA code, by an estimate of the number of facilities in each NEA code. The number of facilities in each NEA category was estimated using the 1987 Census of Manufactures. This Census, conducted by the U.S. Department of Commerce, tallies the number of facilities by 4 digit SIC code; these SIC codes were matched to the NEA codes.

Dividing total lead emissions emitted by number of facilities yielded the average yearly lead emissions from industrial combustion for each SIC code. We then assigned this average value to *all reporting TRI facilities* in the SIC code. The consequence of this approach is that the modeling of combustion from industrial facilities includes substantially more sources than the modeling of industrial process emissions; combustion emissions are assigned to essentially all facilities reporting to TRI, while the process emissions are only evaluated for facilities actually reporting lead air emissions from processes.

One unavoidable drawback to this approach is that it cannot capture differences in release quantities among facilities within an SIC code. Furthermore, this approach does not capture all combustion emissions because we assign average emissions only to facilities that report to TRI. TRI facilities account for between two percent and 50 percent of all facilities listed in the Census of Manufacturers, depending on the SIC code. Because of the inability to place the remaining facilities geographically, this analysis excludes the consideration of emissions from non-TRI facilities.

Methods Used to Determine Changes in Lead Emissions from Electric Utilities from 1975 to 1990

The estimation of lead emissions from electric utilities required data from three different sources. Energy use data for the control and no-control cases was obtained from the national coal use estimates prepared for the Section 812 analysis by ICF Incorporated. The OAQPS Trends Database provided emissions factors and control efficiencies. Individual plant latitudes, longitudes, and stack information were collected from the EPA Interim Emissions Inventory. This analysis combines these three pieces of data and creates yearly lead emissions at the plant level for coal burning electric utilities in the control and no-control scenarios. This section describes the sources and the methods used to create the final dataset.

Coal-Use Data

The energy use data obtained provides plant level energy consumption data for 822 electric utilities. The dataset was broken into four distinct sets for each of the years 1975, 1980, 1985, and 1990. Each set of data provided the state where the plants are located, the plant names, and the amount of coal consumed, for both the control and no-control cases. The four data sets were combined into one comprehensive set by matching the plants' names and states.

The EPA Interim Emissions Inventory

The EPA Office of Air Quality Planning and Standards Technical Support Division provided the 1991 EPA Interim Emissions Inventory. The Interim Inventory contains data for all electric utility and industrial plants in the United States including latitude, longitude, stack height, stack diameter, stack velocity, and stack temperature. The additional stack parameter data allowed the use of plant-specific parameters in the air modeling for electric utilities rather than average parameters for all facilities as was done for industrial emissions.

Matching the Coal-Use Data to the Interim Emissions Inventory

The combination of the Interim Emissions Inventory and the coal-use data required two steps. First, the Interim Emissions Inventory needed to be pared down to include only electric utility data and to narrow down the information provided for each utility. Secondly, the two datasets needed to be combined. One difficulty in the combination of the two sets was the lack of a common data field that would allow a quick and complete matching process between the two.

Electric utility plants were identified in the Interim Emissions Inventory by SIC code (code 4911). The associated stack information file, which lists every stack on every plant, was reduced to include only the largest stack for each plant. This provides a reasonable estimate of the stack height at which most emissions occur. The air modeling assumes that each electric utility releases its emissions from the largest stack that exists at that plant.

Next, the procedure matched the abridged Interim Emissions Inventory file with the coal use data. Due to the lack of a common data field between the two sets, this process required several phases. Both data sets had name fields, but these fields utilized different naming conventions for the plants. Therefore the name fields were matched directly, with individual words in the names, and then with abridged words from the names. Abridged word matches were double checked by ensuring that the names were indeed similar and by verifying that the state fields matched. Finally some matches were made by hand.

Only 27 unmatched plants with positive coal use remained. There were 493 matched plants with positive coal usage and these were included in the final dataset.[230] To eliminate under-counting of emissions, the emissions from the 27 unmatched plants were allocated to matched plants within the states

[230] Plants with zero coal usage were not immediately excluded from the analysis due to the possibility of analyzing lead emissions from oil combustion at these plants. However, OAQPS has suggested that oil combustion comprises under two percent of the total lead emitted from electric utilities. For this reason, the electric utility analysis focused entirely on coal.

where the unmatched plants were located. Allocations were weighted according to the emission level for each matched plant within that state in the year in which the allocation was being made.

Emissions Factors and Control Efficiencies

At this stage, the electric utilities data set contained coal consumption by plant by year in the control and no-control cases as well as air modeling parameters. Using emissions factors for lead and control efficiencies for electric utilities, estimates of lead emissions per plant per year could now be calculated. As in the industrial source analysis, the emissions factors and control efficiencies come from the 1991 OAQPS Trends database.

Control efficiencies are available for coal-fired electric utilities in each year between 1975 and 1990. As in the industrial source analysis, it is assumed that pollution control on coal-burning power plants without the CAA would be the same as the pollution control level in 1970. Therefore, the control efficiency from 1970 is used in the no-control analysis.

The emissions factor obtained from the Trends database is expressed in terms of lead emitted per ton of coal burned (6050 grams per 1,000 tons).[231] The combined dataset, though, contains quantity of coal burned per plant per year in energy units (trillions of Btus). To reconcile this difference, a conversion factor was obtained from a 1992 DOE report titled *Cost and Quality of Fuels for Electric Utility Plants 1991*. The conversion factor used (20.93 million Btus per ton of coal) is the average Btu per pound of coal burned for all domestic electric utility plants in 1990. Data for a small subset of other years were also provided in the DOE report, but they did not differ significantly from the 1990 number. Therefore, the 1990 conversion factor (637.3 pounds of lead per trillion BTU) is assumed valid over the entire study period. The final equation for lead emissions looks quite similar to the equation used in the industrial source analysis.[232] The only change is that "Economic Activity" has been replaced by "Coal Consumed" for this particular analysis:

$$Emissions = (CoalConsumed) \times (EmissionFactor) \times (1 - ControlEfficiency) \quad (60)$$

This equation produces estimates of the emissions per plant per year in both the control and the no-control scenarios.

Use of Air Dispersion Modeling to Estimate Ambient Air Lead Levels

To link estimates of lead emissions to blood lead levels of populations living in the vicinity of facility, the lead benefits model first uses air dispersion modeling to estimate air lead concentrations surrounding facilities that emit lead into the air. The air concentrations are then linked to blood lead levels.

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[231] The actual figure cited is 12.1 metric pounds per 1000 tons. A metric pound is one two-thousanth of a metric ton.

[232] EPA 1991.

This analysis uses the Industrial Source Complex Long Term (ISCLT) air dispersion model, a steady-state Gaussian plume model, to estimate long-term lead concentrations downwind of a source. The concentration is modeled as a function of site parameters (stack height, stack velocity).[233] The general form of the concentration equation from a point source at a distance r greater than one meter away is shown in Equation 61:[234]

$$C_{air,r_{\theta}} = \frac{2K}{\sqrt{2\pi}\, r\theta} \times \frac{Q\, f\, S\, V\, D}{u\sigma_z} \qquad (61)$$

where,

C_{air}	=	concentration at distance r ($\mu g/m^3$),
Q	=	pollutant emission rate (g/sec),
f	=	frequency of occurrence of wind speed and direction,
θ	=	sector width (radians),
S	=	smoothing function used to smooth discontinuities at sector boundaries,
u	=	mean wind speed (m/sec),
σ_z	=	standard deviation of vertical concentration distribution (m),
V	=	vertical term (m),
K	=	scaling coefficient for unit agreement.

For each facility modeled in the lead benefits model, a 21 by 21 kilometer grid around the source is specified. The model stores data in 1 km by 1 km cells and calculates the air lead concentrations for each of the 441 cells surrounding a given facility. Fugitive sources are modeled similarly, the only difference being a modified form of Equation 333.

For facility-specific weather data, the model used Stability Array (STAR) data. The STAR data contains information on typical wind speed and direction for thousands of weather stations in the U.S. For each facility, the model accesses the STAR data for the weather station nearest the source. Standard default assumptions are used for the other parameters because facility-specific data for these parameters are not available (except for utilities). For these parameters, we instead use standard default parameters. Table 130 lists default parameters for the ISCLT, and summarizes sources for other parameters.

Industrial process emissions were modeled as either point or fugitive sources, depending on how they were reported in TRI. All industrial combustion emissions were modeled as "fugitive" emissions. This is a more appropriate model scenario for boiler emissions than a 10 meter stack scenario. All electric utility sources were modeled as point sources.

The model tracks all lead emissions to a given cell. That is, if the plumes of two or more sources overlap in a given cell, the air concentration in the given cell is determined from the sum of all of the contributing sources.

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[233] Ideally, reported stack and fugitive air releases would be modeled using site-specific data (such as source area or stack height). However, since TRI does not contain such facility-specific information, default values are used to model TRI facilities.

[234] This equation is from EPA (1992). The equation is for a specific wind speed, direction, and category (*ijk*). Each facility has several combinations of these that must be added to arrive at a total concentration at that point. The equation for area sources is similar.

Table 130. Air Modeling Parameters.

Parameter	Industrial Source Value	Electric Utility Value	Source/ Comment
Stack height	10 m	site-specific or 115.0 m*	Industrial – U.S. EPA (1992) Utilities – U.S. EPA (1991)
Exit velocity	0.01 m/s	site-specific or 22.5 m/s*	Industrial – U.S. EPA (1992) Utilities – U.S. EPA (1991)
Stack diameter	1 m	site-specific or 5.15 m*	Industrial – U.S. EPA (1992) Utilities – U.S. EPA (1991)
Exit gas temperature	293° K	site-specific or 427.5*	Industrial – U.S. EPA (1992) Utilities – U.S. EPA (1991)
Area source size	10 m^2	10 m^2	U.S. EPA (1992)
Area source height	3 m	3 m	U.S. EPA (1992)
Lead emission rate	site-specific	site-specific	Industrial – TRIS (lbs/yr) Utilities – SAI & OAQPS (lbs/yr)
Frequency of wind speed and direction	site-specific	site-specific	STAR data
Sector width	22.5°	22.5°	360° divided by 16 wind directions
Wind speed	site-specific	site-specific	STAR data (m/sec)
Smoothing function	calculated	calculated	
Vertical term	calculated	calculated	

* average value for electric utilities, utilized for utilities without this information

Determination of Blood Lead Levels from Air Lead Concentrations

Once the air lead concentrations surrounding a given plant are estimated, the model estimates blood lead levels for children and adults living in those areas. This section describes the methods and data sources used to derive blood lead levels from estimated air lead concentrations.

Relationship Between Air Lead Concentrations and Blood Lead Levels

The rates at which lead is absorbed from air depend on the age of the exposed individual, distance from the facility, the initial concentration of blood lead, and other factors. In addition, rates determined from empirical data may differ depending on whether or not the analyses from which rates are derived have controlled for factors such as lead exposure through deposition on dust and soil (i.e., "indirect exposure"). Especially when children constitute the exposed group, the inclusion of indirect

exposure results in higher air lead to blood lead slopes. In both cases, the slope relationship is expressed as the change in blood lead (μg/dL) per change in air concentration (μg/m^3).

In performing this analysis, the choice was made between the use of air lead:blood lead relationships that account for inhalation exposure ("direct" slopes) and those that account for exposure through inhalation and for exposure to lead deposited from air onto soil and dust ("indirect" slopes). The choice of which slopes to use considered both the effects on the estimate of benefits over time (from 1970 to 1990) as well as the estimate of the difference in benefits between the controlled and uncontrolled scenarios. The indirect slope is more comprehensive in its coverage of the types of exposures that will result from air releases, and thus captures more of the health effects predicted to occur from lead exposures, especially to children. For this reason, indirect slopes are preferred to direct slopes, especially when comparing the controlled to uncontrolled scenarios: using only the direct slope would underestimate the benefits of avoiding deposition that controls confer. However, indirect slopes may capture effects from exposure to soil and dust lead deposited from both current air releases and historic air releases. Since lead's dissipation from soil is slow relative to its removal from air, the reservoir of lead in soil and dust is unlikely to change at the same rate as the reductions in air lead concentrations. Therefore, using indirect slopes to represent a change in blood lead over time due to reduced air emissions may overestimate the change in blood lead, and thus overestimate the benefits of reductions over time, to the extent that the indirect slope captures exposure to the total reservoir of soil and dust lead, rather than only recently deposited lead.

Given that the focus of this analysis is the *difference* between the controlled and uncontrolled scenarios, it is important to capture both the benefits from reduced lead deposition that result from the CAA, and the direct benefits from reduced air concentrations. Therefore, this analysis modeled changes in blood lead levels using indirect slopes. It should be kept in mind that this choice may overestimate blood lead changes over time, for both the controlled and uncontrolled scenarios.

The relationship between concentrations of lead in ambient air and blood lead concentrations has been evaluated by a variety of methods. These include experimental studies of adult volunteers, as well as epidemiological studies of different populations of children and adults. The discussion below describes the slopes used in this analysis for children and adults, and for individuals with blood lead values greater than 30 ug/dl.

Children

U.S. EPA (1986) reports that slopes which include both direct (inhalation) and indirect (via soil, dust, etc.) air lead contributions vary widely, but typically range from three to five μg/dl increment in children's blood lead per μg/m^3 increment in air lead concentration (roughly double the slope due to inhaled air lead alone). Since hand dust levels can play a significant role in blood lead levels (U.S. EPA, 1986), this higher slope may be due to mouthing behavior of children that brings them into contact with dust and soil.

Specific values for estimating contribution of air lead to blood lead, including indirect pathways, are cited in U.S. EPA (1986); slope values (ranging from -2.63 to 31.2) and data sources for these values are presented in Table 11-36 of U.S. EPA (1986). The median of these values is 4.0 μg/dl per μg/m^3, which matches the midpoint of the range of typical slope values. This analysis used this value to

represent the relationship between air lead concentrations and blood lead concentrations for children living in the vicinity of point sources of lead emissions.

The use of this slope assumes that "indirect exposure" principally measures indirect effects of lead emissions to air (through deposition to dust and soil). However, it is possible that these slopes include other exposures not related to air lead. In many cases authors have measured other possible exposures, such as water and food, and have confirmed that the most significant contribution comes from soil and dust lead, which is assumed to result from air deposition of lead. Those studies that measured lead in tapwater showed that mean levels were generally low or not significantly related to blood lead. Landrigan et al. (1975) measured lead in pottery and food; lead in pottery was found in only 2.8 percent of homes, and food and water made no more than a negligible contribution to lead uptake. Lead in paint was measured in some studies.[235] Landrigan and Baker (1981) measured lead in paint at levels greater than one percent in about one fourth to one third of the houses in each area studied. Brunekreef et al. (1981) measured high levels of paint in some houses, but excluded these data points from the analysis.

Despite the possibility of confounding factors, this analysis uses the median value determined above (4.0 μg/dl per μg/m^3) as the appropriate slope for children living within five kilometers of the point source. Five kilometers is chosen as the cut off point because the data from most of the studies cited collected the majority of their data points near lead smelters.[236] Furthermore, these slopes, although measured primarily in the vicinity of smelters, are assumed applicable to all point sources that emit lead into the air.

Adults

For adult males and females, the air lead/ blood lead slopes that include indirect effects due to soil and dust differ very little from slopes that include only direct effects. This result is expected since the higher indirect slope values estimated for children are assumed to be as a result of mouthing behavior typical of young children.

U.S. EPA (1986) describes several population studies that estimate indirect slopes for men; these slopes range from -0.1 to 3.1 μg/dl per μg/m^3.[237] Snee (1981) determined a weighted average of these studies and one other study.[238] The average slope, weighted by the inverse of each study's variance, is 1.0 μg/dl per μg/m^3. However, the Azar study measured the direct relationship between air lead and blood lead. Excluding the Azar study form the weighted average, the average slope is 1.1 μg/m^3. Excluding the highest and lowest slopes from this group (from Goldsmith, 1974 and Tsuchiya et al., 1975), both of which had difficulties,[239] the resulting slope is 1.4 μg/dl per μg/m^3.

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[235] Landrigan and Baker, 1981; Brunekreef et al., 1981.

[236] EPA 1986, Table 11-36.

[237] Johnson et al., 1976; Nordman, 1975; Goldsmith, 1974; Tsuchiya et al., 1975; Fugas et al., 1973.

[238] Azar et al., 1975.

[239] Goldsmith (1974) refrigerated (rather than froze) the blood samples, and did not analyze the samples until 8 or 9 months after they were taken, and restricted the analysis to one determination for each blood sample. Tsuchiya et al. (1975) measured air lead concentrations after blood samples were taken; blood was drawn in August and September of 1971, whereas air samples were taken during the 13 month period from September 1971 to September 1972.

Slopes for females range from 0.6 to 2.4 for general atmospheric conditions.[240] Snee determined an average slope for women of 0.9 μg/dl per μg/m^3, weighted by the inverse of the variances of the studies. Excluding the slope for women from Goldsmith (1974), the resulting slope for women is 1.0 μg/dl per μg/m^3.

These values are adjusted by a factor of 1.3 to account for the resorption of lead from bone tissue (according to Chamberlain, 1983), thus deriving an adjusted slope estimate of 1.8 μg/dl blood lead per μg/m^3 increment in air lead concentration for men and 1.3 for women. These are the slope estimates used in this report.

Individuals with initial blood lead levels of 30 μg/dl and greater

For individuals with high blood lead levels, the air lead to blood lead uptake slopes have been shown to be much shallower, as described by U.S. EPA (1986). An appropriate change in blood lead per change in air lead is 0.5 μg/dl per μg/m^3 for individuals that have initial blood lead levels in the range of 30 to 40 μg/dl. This value is based on cross-sectional and experimental studies.[241] For individuals with initial blood lead levels greater than 40 μg/dl, an appropriate range of slopes is 0.03 to 0.2, as determined by occupational studies listed in Table 11-37 of U.S. EPA (1986). The median value of these studies is 0.07. These two slopes (0.5 for the population with blood lead levels between 30 and 40 μg/dl and 0.07 for blood lead levels greater than 40 μg/dl) are used for both children and adults in this analysis. These relationships are summarized in Table 131.

Table 131. Estimated Indirect Intake Slopes: Increment of Blood Lead Concentration (in μg/dL) per Unit of Air Lead Concentration (μg/m^3).

	Individuals with blood lead levels < 30 μg/dl	Individuals with blood lead levels 30-40 μg/dl	Individuals with blood lead levels > 40 μg/dl
Adult Males	1.8	0.5	0.07
Adult Females	1.3	0.5	0.07
Children	4.0	0.5	0.07

Estimates of Initial Blood Lead Concentrations

The benefits model requires an initial distribution of blood lead levels in the exposed populations to model health benefits of reducing lead air emissions. The model estimates the new distribution of blood lead levels that would exist after a given change in air concentrations using the slopes described above. Finally, the model estimates the difference between the two distributions. This analysis begins

[240] Tepper and Levin, 1975; Johnson et al., 1976; Nordman, 1975; Goldsmith, 1974; Daines et al., 1972.

[241] U.S. EPA, 1986.

with an initial 1970, uncontrolled blood lead distribution from which all subsequent changes are modeled. This approach requires an estimate of the blood lead distributions in the U.S. population in 1970. Unfortunately, there are no actual national blood lead distribution estimates for 1970. Although the first NHANES study covered 1970, blood lead data were not collected in this study.[242] Nonetheless, a 1970 distribution of blood lead was estimated using NHANES II data (from 1976-1980), combined with estimates of typical changes in blood lead levels from 1970-1976 observed in localized screening studies.

A major drawback to this approach is the uncertainty in deriving the 1970 estimates. Another drawback to beginning with the 1970 level and modeling changes from that point is the analysis only represents changes in lead exposure from air; reductions from other sources of lead exposure are not accounted for. The purpose of this analysis is to identify changes attributable to the CAA mandates; changes from other sources of lead exposure should not be considered. However, due to nonlinear nature of the lead concentration-response functions (see above), the overall exposure context in which the air lead exposure reductions take place will influence the estimate of benefits from those reductions. Specifically, at higher blood lead levels, the slope of the concentration-response curve is shallower than at lower levels. As a result, a given change in the mean blood lead level may result in a smaller change in the health effect if the change occurs from a relatively high starting level. On the contrary, if one accounts for the fact that other sources of lead exposure are reduced at the same time that the given air reductions occur, then those air emissions reductions may result in greater changes in health risk.

This issue is of concern even though the analysis focuses on the difference between the controlled and uncontrolled states of the world, since the health benefit implications of the emissions differentials between the controlled and uncontrolled scenarios will depend on the point on the blood lead distribution curve at which the differences are considered. That is, a difference between a mean blood lead of 25 μg/dl and one of 20 μg/dl may have different health implications than a difference between 15 μg/dl and 10 μg/dl, even though the absolute value of the difference is the same (5 μg/dl).

An alternative method is to "start" with a 1990 blood lead level and to "backcalculate" benefits by representing the differentials as increases over the 1990 levels, rather than decreases from 1970 levels. The advantage of this approach is that it accounts for reductions in lead exposure from other sources, as represented by current blood lead levels. Its disadvantage is that it holds other sources constant to (lower) 1990 levels, and thus the modeling may underestimate actual blood lead distributions in earlier years, and thereby overestimate benefits from controls during those years. This analysis presents the results of both approaches.

Combination of Air Concentration Estimates with Population Data

The modeled air lead concentrations at various distances from the source were combined with population data from the Census Bureau to arrive at an estimate of the number of cases of health effects for each of the years from 1970 to 1990 in both the controlled and uncontrolled scenarios. The primary census information was accessed from the Graphical Exposure Modeling System Database (GEMS), an

[242] NCHS, 1993.

EPA mainframe database system. The following data were obtained from GEMS for the years 1970, 1980, and 1990: total population for each Block Group/Enumeration District (BG/ED); state and county FIPS codes associated with each BG/ED; latitude and longitude of each BG/ED; and population of males under 5, and females under 5 for each BG/ED. The intervening five year intervals (1975 and 1985) were estimated using the Intercensal County Estimates from the Census, which estimate annual populations on a county by county basis. The decennial Census data and the Intercensal County Estimates data sets were related by county FIPS codes; the population in each BG/ED was assumed to grow or shrink at the same rate as the county population as a whole.

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.[243] 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,[244] birth rates for 1970, 1980 and 1990 were obtained from the Census Bureau.[245] 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 132 shows estimated lead emissions from industrial processes, industrial combustion and electric utilities. Tables 133 and 134 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.

[243] Karl Kuellmer, Abt Associates, and the Bureau of Census, Population, Age and Sex telephone staff, March, 1994.

[244] 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.

[245] Personal communication, Karl Kuellmer, Abt Associates and the Bureau of Census, Population, Fertility/Births telephone staff.

Table 132. 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 133. 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 134. 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:

$$LEAD = \left(\frac{SOLD}{365\,days} \right) \times [FRAC_{Pb} \times PB_{leaded} + (1 - FRAC_{Pb}) \times PB_{unleaded}] \qquad (62)$$

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.[246] 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.[247]

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).[248] 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 ($p < 0.0001$).[249]

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.[250] 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

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[246] U.S. EPA, 1985; Billick et al., 1979; Billick et al., 1982.

[247] Janney, 1982; Annest et al., 1983; Center for Disease Control, 1993; National Center for Health Statistics, 1993.

[248] U.S. EPA, 1985.

[249] U.S. EPA, 1985.

[250] 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 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.

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.[251] 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.

[251] Dept. of Commerce, 1976.

Changes in Leaded Gasoline Emissions and Resulting Decreased Blood Lead Levels and Health Effects

Table 135 shows the estimated quantity of lead burned in gasoline in the five year intervals from 1970 to 1990. Tables 136 and 137 show the difference in health impacts between the two scenarios (for lead in gasoline only) for the "forward-looking" and "backward-looking" analyses.

Table 135. 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 136. 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 137. 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*[252] 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.

[252] U.S. EPA. Office of Policy Planning and Evaluation. *Unfinished Business: A Comparative Assessment of Environmental Problems.* February 1987.

Table 138. 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 138 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 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."*[253]

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."*[254]

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.

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[253] 42 U.S.C. §1857(a)(1).

[254] 42 U.S.C. §1857(b).

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 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"[255] 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."[256]

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.[257]

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)

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[255] 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.

[256] 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.

[257] 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.

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 for 1980.[258] 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.

[258] 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.

With the above caveats in mind, Table 139 provides information about maximum individual risk taken from the Federal Register notices for the NESHAPs promulgated before the 1990 amendments to

Table 139. 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

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 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.

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.[259] 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.[260] All baseline incidence estimates were ultimately expressed relative to a 1985 base year.[261] The assumption was then made that source-category incidence rates were proportional to the level of emissions from that source category.

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[259] 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.).

[260] 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.)

[261] See PES, March 22, 1993 memorandum, p. 3.

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).[262] 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 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.[263] The formula used for these calculations was as follows:[264]

$$I_{ty} = I_{by} \times \left[\frac{A_{ty}}{A_{by}}\right] \times \left[\frac{P_{ty}}{P_{by}}\right] \times \left[\frac{(1-C_{ty})}{(1-C_{by})}\right] \qquad (63)$$

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.[265] In fact, roughly 75 percent of the total reduction in cancer cases averaged over the

[262] 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.)

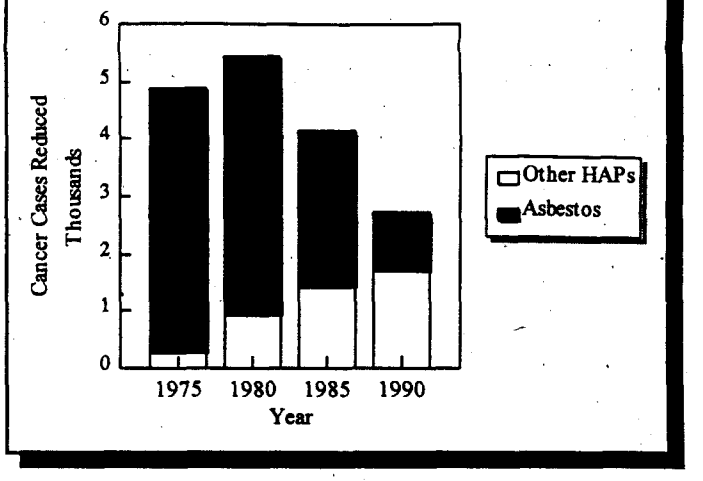
[263] More detailed descriptions of the methodology and associated uncertainties are provided in "Retrospective Analysis for Section 812(a) Benefits Study," a September 30, 1992 memorandum from Ken Meardon, PES to Vasu Kilaru, US EPA.

[264] See PES, March 22, 1993 memorandum, p. 4.

[265] PES, "Cancer Risk Estimates from Stationary Sources," memorandum from Ken Meardon, PES to Vasu Kilaru, US EPA, March 5, 1993.

1970 to 1990 period were attributed to asbestos control.[266] Figure 52 summarizes the PES study overall cancer incidence reductions and the relative contribution of asbestos-related reductions over the study period.

Figure 52. 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 with actual historical activity patterns.[267] 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.[268] 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.

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[266] ICF, "Direct Inhalation Incidence Benefits," Draft Report, November 11, 1994, p. 10.

[267] 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.

[268] 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.

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.[269] 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.[270] 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.[271] ICF was unable, however, to perform an extensive evaluation of the activity level indicators used in the PES study.[272]

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, 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.[273]

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.

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[269] 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.

[270] See ICF Draft Report, p. 12.

[271] See ICF Draft Report, p. 9.

[272] See ICF Draft Report, p. 13.

[273] Additional details of the ICF Re-analysis methodology can be found in ICF, "Direct Inhalation Incidence Benefits," Draft Report, November 11, 1994.

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 53 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 54 presents a comparable compilation reflecting the upper bound estimates for all HAPs.

Figure 53. ICF Estimated Reductions in Total HAP-Related Cancer Cases Using Upper Bound Asbestos Incidence and Lower Bound Non-Asbestos HAP Incidence.

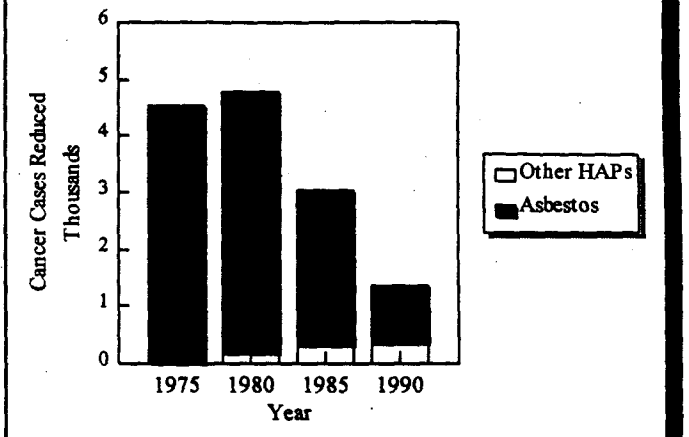
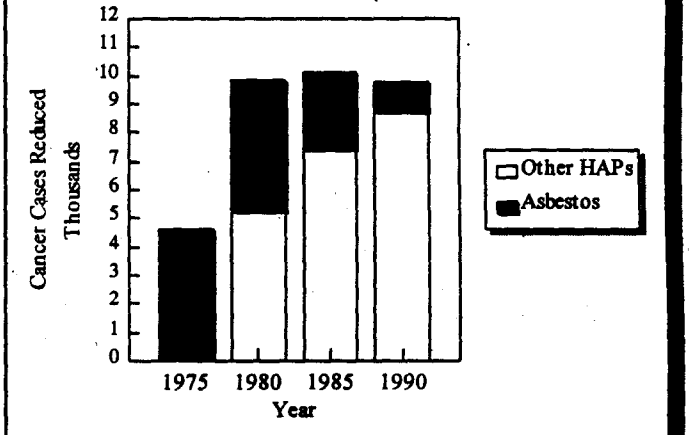


Figure 54. ICF Estimated Reduction in Total HAP-Related Cancer Cases Using Upper Bound Incidence for All HAPs.



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.

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.[274] 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.[275] 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.

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).[276] 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[277] and non-mobile source[278] CO emissions. The HAP exposure function used in the ICF/SAI analysis is summarized by the following equation:

$$E = ((C \times A) - B) \times S \times M \times \frac{(VOC \times HAP)}{CO} \qquad (64)$$

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

[274] Cancer Risk report, Page ES-12.

[275] See US EPA/OAR/OMS, "Motor Vehicle-Related Air Toxics Study," EPA 420-R-93-005. April 1993.

[276] ICF/SAI, "Retrospective Analysis of Inhalation Exposure to Hazardous Air Pollutants from Motor Vehicles," October 1995, p. 4.

[277] Background CO is produced by the oxidation of biogenic hydrocarbons. See ICF/SAI, p. 7.

[278] 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.

VOC	=	VOC emissions by year, county, and scenario
HAP	=	VOC speciation factor by mobile source HAP
CO	=	CO emissions by year, county, and scenario

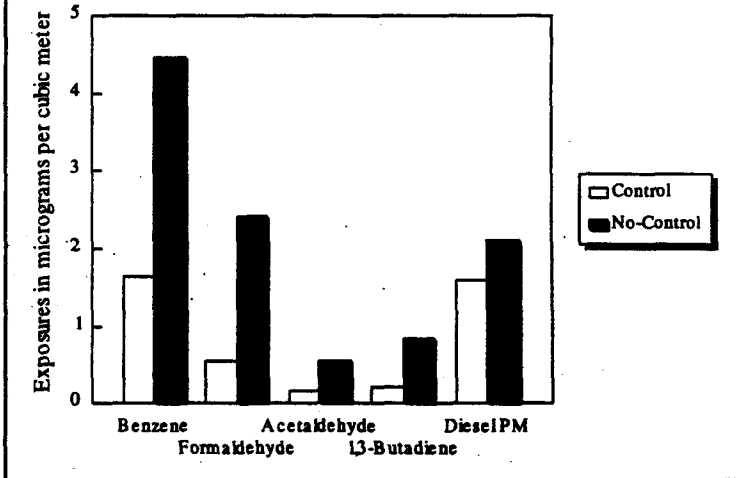
Details of the derivation of each of the variables applied in the above equation are provided in the ICF/SAI report. However, in essence, the calculation involves the following basic steps.

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.[279] 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 emissions.[280] 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 55 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 55. 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.

[279] See ICF/SAI, p. 3.

[280] 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.

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[281] found that ambient concentrations for a substantial number of monitored and modeled HAPs exceeded one or more health benchmarks.[282] 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.

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

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[281] U.S. Environmental Protection Agency, "Toxic Air Pollutants and Noncancer Risks: Screening Studies," External Review Draft, September, 1990.

[282] 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.

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."[283] 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 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."*[284]

Because of the risk of significant exposure to infants and other high-risk groups, such as "sport anglers, Native Americans, and the urban poor,"[285] 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.

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[283] USEPA/OAR/OAQPS, "Deposition of Air Pollutants to the Great Waters, First Report to Congress," EPA-453/R-93-055, May 1994.

[284] EPA-453/R-93-055, May 1994, p. ix.

[285] EPA-453/R-93-055, May 1994, p. x.

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.

- 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 analysis used several sources to obtain dollar values (indicated in 1990$) for the morbidity and mortality effects modeled in the 812 analysis. Memos from Industrial Economics (IEc) to EPA's Office of Air and Radiation provide most of the values (Unsworth et al., 1992; Neumann and Unsworth, 1993; Unsworth and Neumann, 1993; Neumann et al., 1994). A few values, however, were obtained from other sources of information. For example, work by Abt Associates provides cost of illness estimates for several types of hospital admissions (Currier, 1995). In addition, a recent EPA draft report provides unit values for several of the lead-induced health effects (USEPA, 1995).

When using information from the IEc memos, some estimates have been used directly, whereas others have been combined in a variety of ways to relate to the health effects modeled by the dose-response functions. For example, IEc has developed estimates for shortness of breath and mortality, which directly relate to health effects modeled by dose-response functions. However, in several cases, dose-response functions have been developed for a variety of groups of respiratory symptoms. In these instances, the analysis makes judgements about how to combine individual economic estimates recommended by IEc. In other cases, the dose-response study does not clearly define the modeled health effect, and thus, the analysis makes a judgement about the valuations to apply to the study. In cases

where the current analysis combined economic estimates from IEc or applied the estimates to health effects other than those listed specifically in the IEc memos, the analysis generally parallels the methods used by IEc. For example, the current analysis estimates midpoint estimates in a manner according to IEc recommendations when central values are not given by IEc (i.e., the analysis uses the midpoint of the high and low valuations to represent central estimates).

Several points should be made about the choice of economic valuations for a few of the individual endpoints. First, although economic values for hospital admissions were closely matched to the type of hospital admissions included in the concentration-response studies (e.g., COPD, pneumonia), some studies included age groups for which economic values were not available in Currier (1995). These cases are noted in the table below. Second, the low and high estimates for both upper and lower respiratory illness are based on the assumption that at the low end, only one symptom would occur, and at the high end, a group of relevant symptoms would occur. Values for each individual symptom are midpoint estimates recommended by IEc.

Third, Restricted Activity Days (RADs) and Respiratory Restricted Activity Days (RRADs) include days of restricted activity that are more severe than those for Minor Restricted Activity Days (MRADs). Further, RADs include Work Loss Days, creating the potential for double counting. MRADs, by the definition in the study providing the dose response function, do not include WLDs. Therefore, the current analysis develops two alternative composite measures of loss-of days:

$$WorkLossDays + MRADs \tag{65}$$

$$WorkLossDays + (RAD_{incidence} - WLD_{incidence}) \times MRAD\ unitvalue \tag{66}$$

The valuations for WLD and MRADs are from Unsworth and Neumann, 1993.

Hasselblad et al. (1992), which provides the dose-response function for nitrous oxides, models occurrences of "respiratory illness," which the authors do not further define. The current analysis uses an economic value for cough and upper and lower respiratory illness developed by IEc to represent an economic value for respiratory illness modeled by Hasselblad et al. (1992). Table 140 includes additional notes specific to each of the valuations.

The 812 analysis uses the midpoint estimates listed in Table 140 for the point estimate of benefits. For the sensitivity analysis, two types of sampling have been predominantly used when endpoints have more than one valuation estimate. For example, in cases where IEc memoranda recommended low, midpoint, and high values, the 812 analysis gave each of the recommended values equal probabilities of occurring. When IEc memos recommended only high and low estimates, the 812 analysis assumed that all values between the high and low estimates would be equally likely. A few exceptions to the use of these distributions include hospital admissions for pneumonia and mortality (as well as health endpoints that have values based on the mortality valuation). In the case of hospital admissions for pneumonia, the analysis assumes that costs vary according to the normal "bell-shaped" curve.

Values for mortality vary according to a lognormal distribution. However, it was necessary to truncate the upper end value so that estimates would not be extremely high. Thus, the highest possible estimate of mortality to be used in the sensitivity analysis ($13.5 million) is the highest value from all mortality-valuation studies evaluated by IEc. Several endpoints (e.g., hospital admissions for several health effects) have only midpoint estimates of values, and thus, the analysis does not model uncertainty and variability for these effects.

Table 140. Unit Values Used for Economically Valuing Endpoints

Health or Welfare Endpoint	Unit Valuation (1990$)				Comments on Unit Valuation
	Midpoint Estimate	Uncertainty and Variability			
		Low Estimate	High Estimate	Distribution Type[a]	
Mortality	4.8 million IEc-recommended expected value for mortality (Unsworth et al., 1992)	--	--	Lognormal distribution (geometric mean = 3.6 million and geometric standard deviation = 2.3 million from Unsworth et al., 1992); distribution is truncated at the upper end value of 13.5 million	Use for ozone, PM-10, lead
Hospital and emergency admissions for all respiratory illnesses	5,986 Cost of Illness value for hospital admissions for all respiratory illnesses for all ages (Currier, 1995)	5,986	5,986	Midpoint estimate only	Use for ozone, PM-10 for elderly patients and all individuals
Hospital admissions for pneumonia	7,555 Cost of Illness value for pneumonia hospital admissions for all ages (Currier, 1995)	7,555	7,555	Normal distribution (mean = 7,555 and standard deviation = 1888.75)	Use for ozone, PM-10 for elderly patients

Health or Welfare Endpoint	Unit Valuation (1990$)				Comments on Unit Valuation
	Midpoint Estimate	Uncertainty and Variability			
		Low Estimate	High Estimate	Distribution Type[a]	
Hospital admissions for COPD	7,751 for > 65 year olds Weighted average Cost of Illness value for hospital charges for COPD and asthma (Currier, 1995)	7,751	7,751	Midpoint estimate only	Use for PM-10, ozone studies measuring hospital admissions for health effects in ICD codes 490-496 (includes COPD and asthma) for elderly patients
Emergency room admissions for COPD	331 Weighted average Cost of Illness value for emergency room visits for asthma and COPD (Currier, 1995)	331	331	331	Use for PM-10 for ICD codes 490-496 (Sunyer et al., 1993)
Emergency admissions for asthma	7,016 recommended Cost of Illness value for asthma admissions for elderly patients (Currier, 1995)	7,016	7,016	Midpoint estimate only	Use for ozone, PM-10 for studies of admissions of all individuals
Emergency room visits for asthma	292 Cost of Illness value for emergency room visits for asthma from Currier (1995)	292	292	Midpoint estimate only	Use for ozone, PM-10 for studies of admissions of (1) all individuals and (2) individuals less than 65 years old

Health or Welfare Endpoint	Unit Valuation (1990$)				Comments on Unit Valuation
	Midpoint Estimate	Uncertainty and Variability			
		Low Estimate	High Estimate	Distribution Type[a]	
Hospital admissions for congestive heart failure	7,906 Cost of Illness value for hospital admissions for congestive heart failure from Currier (1995)	7,906	7,906	Midpoint estimate only	Use for PM-10 for admissions of elderly individuals
Hospital admissions for ischemic heart disease	10,017 Cost of Illness value for hospital admissions for ischemic heart disease from Currier (1995)	10,017	10,017	Midpoint estimate only	Use for PM-10 for admissions of elderly individuals
Upper Respiratory Illness defined as either (1) one or more of following: runny or stuffy nose, wet cough, burning, aching, or red eyes (Pope et al., 1991) or (2) fever combined with earache and sore throat (Braun-Fahrlander, 1992)	18.22 Midpoint of low and high values	3.72 IEc-recommended low value for cough, and upper and lower respiratory illness. Value represents a midpoint estimate of having one symptom (Neumann et al., 1994)	32.72 sum of IEc-recommended midpoint estimates of the following symptoms: cough, head congestion/ sinus, eye irritation (Unsworth and Neumann, 1993)	Low, midpoint, and high estimates are equally likely[b]	Use for PM-10
Lower Respiratory Illness -- defined as one or more of following: trouble breathing, dry cough, wheezing	13.85 Midpoint of low and high values	3.72 IEc-recommended low value for cough, and upper and lower respiratory illness. Value represents a midpoint estimate of having one symptom (Neumann et al., 1994)	23.97 sum of IEc-recommended midpoint estimates of the following symptoms: PDI, wheeze (multiplied by three according to IEc recommendations), cough, and shortness of breath (Neumann et al., 1994)	Low, midpoint, and high estimates are equally likely	Use for PM-10

Health or Welfare Endpoint	Unit Valuation (1990$)				Comments on Unit Valuation
	Midpoint Estimate	Uncertainty and Variability			
		Low Estimate	High Estimate	Distribution Type[a]	
Acute bronchitis in children	45.02 Midpoint of low and high values	13.29 IEc-recommended low value for acute bronchitis (Neumann et al., 1994)	76.74 IEc-recommended high value for acute bronchitis (Neumann et al., 1994)	Uniform distribution between low and high values[c]	Use for PM-10
Presence of any of 19 acute respiratory symptoms	29.33 Midpoint of low and high values	3.72 IEc-recommended low value for cough, and upper and lower respiratory illness (Neumann et al., 1994)	54.94 IEc-recommended high value for cough, and upper and lower respiratory illness (Neumann et al., 1994)	Uniform distribution between low and high values	Use for ozone, PM-10 (Krupnick et al., 1990)
Any symptom in exercising asthmatics (chest tightness, shortness of breath, etc.)	29.33 Midpoint of low and high values	3.72 IEc-recommended low value for cough, and upper and lower respiratory illness (Neumann et al., 1994)	54.94 IEc-recommended high value for cough, and upper and lower respiratory illness (Neumann et al., 1994)	Uniform distribution between low and high values	Use for SO_2
Combined responses of FEV_1, symptoms, measured as scored combination of responses equaling 4 or higher in exercising asthmatics (score of 4 when both categories are moderate). Symptoms: chest tightness, shortness of breath, wheeze	32.48 IEc-recommended midpoint value for "avoiding a bad asthma day" (Unsworth and Neumann, 1993)	11.81 IEc-recommended low value for "avoiding a bad asthma day" (Neumann et al., 1994)	53.80 IEc-recommended high value for avoiding a "bad asthma day" (Neumann et al., 1994)	Low, midpoint, and high estimates are equally likely	Use for SO_2

Health or Welfare Endpoint	Unit Valuation (1990$)				Comments on Unit Valuation
	Midpoint Estimate	Uncertainty and Variability			
		Low Estimate	High Estimate	Distribution Type[a]	
Combined responses of FEV_1, symptoms, and SR_{aw} in exercising asthmatics measured as scored combination of responses equaling 6 or higher (score of 6 when all categories are moderate). Symptoms = chest tightness, shortness of breath, wheeze	32.48 IEc-recommended midpoint value for "avoiding a bad asthma day" (Unsworth and Neumann, 1993)	11.81 IEc-recommended low value for "avoiding a bad asthma day" (Neumann et al., 1994)	53.80 IEc-recommended high value for avoiding a "bad asthma day" (Neumann et al., 1994)	Low, midpoint, and high estimates are equally likely	Use for SO_2
Respiratory illness in children (one or more respiratory symptoms)	29.33 Midpoint of low and high values	3.72 IEc-recommended low value for cough, and upper and lower respiratory illness from Neumann et al. (1994)	54.94 IEc-recommended high value for cough, and upper and lower respiratory illness from Neumann et al. (1994)	Uniform distribution between low and high values	Use for NO_2
Moderate or worse asthma status (wheezing, shortness of breath, chest tightness, sputum). Not sure whether all symptoms must be moderate or worse before this definition is applied	32.48 IEc-recommended midpoint value for "avoiding a bad asthma day" (Unsworth and Neumann, 1993)	11.81 IEc-recommended low value for "avoiding a bad asthma day" (Neumann et al., 1994)	53.80 IEc-recommended high value for avoiding a "bad asthma day" (Neumann et al., 1994)	Uniform distribution between low and high values	Use for PM-10 (Ostro et al., 1991)
Self-reported asthma attacks	32.48 IEc-recommended midpoint value for "avoiding a bad asthma day" (Unsworth and Neumann, 1993)	11.81 IEc-recommended low value for "avoiding a bad asthma day" (Neumann et al., 1994)	53.80 IEc-recommended high value for avoiding a "bad asthma day" (Neumann et al., 1994)	Uniform distribution between low and high values	Use for ozone (Whittemore and Korn, 1980)

Health or Welfare Endpoint	Unit Valuation (1990$)				Comments on Unit Valuation
	Midpoint Estimate	Uncertainty and Variability			
		Low Estimate	High Estimate	Distribution Type[a]	
Restricted activity days (RADs)	38.37 IEc-recommended midpoint value for MRRADs (Unsworth and Neumann, 1993)	38.37 IEc-recommended low value for MRRADs (Unsworth and Neumann, 1993)	82.52 IEc-recommended high value for MRRADs (Unsworth and Neumann, 1993)	Low, midpoint, and high estimates are equally likely	Use for PM-10 (Ostro, 1987)
Minor restricted activity days (MRADs)	38.37 IEc-recommended midpoint value for MRRADs (Unsworth and Neumann, 1993)	38.37 IEc-recommended low value for MRRADs (Unsworth and Neumann, 1993)	82.52 IEc-recommended high value for MRRADs (Unsworth and Neumann, 1993)	Low, midpoint, and high estimates are equally likely	Use for ozone, PM-10
Respiratory restricted activity days (RRADs)	38.37 IEc-recommended midpoint value for MRRADs (Unsworth and Neumann, 1993)	38.37 IEc-recommended low value for MRRADs (Unsworth and Neumann, 1993)	82.52 IEc-recommended high value for MRRADs (Unsworth and Neumann, 1993)	Low, midpoint, and high estimates are equally likely	Use for ozone, PM-10
Work Loss Days (WLDs)	83 Median weekly wage for 1990 divided by 5 (USDC, 1992)	--	--	Midpoint estimate only	Use for PM-10
Worker productivity	--	--	--	--	Use for ozone functions; Valuation already included in dose-response function
Household soiling damage	--	--	--	--	Use for PM_{10} functions; Valuation already included in dose-response function

Health or Welfare Endpoint	Unit Valuation (1990$)				Comments on Unit Valuation
	Midpoint Estimate	Uncertainty and Variability			
		Low Estimate	High Estimate	Distribution Type[a]	
Visibility	--	--	--	--	Valuation already included in dose-response function
IQ points	Discounted at 3%: 6,442 Discounted at 7%: 1,480 (USEPA, 1995)			Midpoint estimate only	Use for lead
IQ < 70	Discounted at 3%: 52,700 Discounted at 7%: 33,438 (USEPA, 1995)			Midpoint estimate only	Use for lead
Infant mortality	4.8 million IEc-recommended expected value for mortality (Unsworth et al., 1992)	--	--	Lognormal distribution (geometric mean = 3.6 million and geometric standard deviation = 2.3 from Unsworth et al., 1992); distribution truncated at the upper end value of 13.5 million	Use for lead
Hypertension in males	681/year/case (USEPA, 1995)	--	--	Midpoint estimate only	Use for lead
Heart disease	1.54 million (0.32*midpoint mortality value)	--	--	Lognormal distribution (geometric mean = 1.15 million (0.32*mortality value and geometric standard deviation = 2.3 from Unsworth et al., 1992); distribution truncated at the upper end value of 4.32 million (0.32*mortality value)	Use for lead

Health or Welfare Endpoint	Unit Valuation (1990$)				Comments on Unit Valuation
	Midpoint Estimate	Uncertainty and Variability			
		Low Estimate	High Estimate	Distribution Type[a]	
Strokes (Cerebrovascular accidents)	1.54 million (0.32*midpoint mortality value)	--	--	Lognormal distribution (geometric mean = 1.15 million and geometric standard deviation = 2.3 from Unsworth et al., 1992); distribution truncated at the upper end value of 4.32 million	Use for lead
Strokes (Brain infarctions)	1.54 million (0.32*midpoint mortality value)	--	--	Lognormal distribution (geometric mean = 1.15 million and geometric standard deviation = 2.3 from Unsworth et al., 1992); distribution truncated at the upper end value of 4.32 million	Use for lead

[a] Mean and standard deviations used in the sensitivity analysis are reported here if applicable.
[b] The analysis samples only the point estimate, low, and high values in the sensitivity analysis. Each value is equally likely to occur.
[c] All values between the low and high estimates have equal chances of occurring.

Results of Valuation of Health and Welfare Effects

Tables 142 and 143 apply the unit valuations to the health and welfare effects presented in the "Health Effects Appendix." As noted in the Health Effects Appendix, there are several estimates for some health and welfare impacts, and thus there are several benefit estimates. These can be used to derive a range of possible results.

Table 142 presents the results of the "50 km" model run. Observe that, for those health endpoints where the pollutant of concern is PM_{10} only, annual benefits decline between 1985 and 1990. Only two factors, besides the CR function itself, should affect the model's output: The difference in ambient air quality between the two scenarios, and the affected population. Between 1985 and 1990, difference in air quality between the scenarios increased, and the U.S. population increased. One should see, then, an unambiguous increase in economic benefits between 1985 and 1990. The decrease in benefits observed here is due exclusively to the decline in population residing in counties with PM_{10} monitors (i.e., due to the decrease in the number of monitors).

Table 143 presents the results of the "all U.S. population" model run (although, with the exception of Pb, not all of the population is modeled, with 2 to 5 percent being excluded), and 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 are almost certainly more accurate than the "50 km" results, but rely on the assumption that (for a portion of the population) distant air quality monitors provide a reasonable estimate of local air quality conditions. Thus, the results presented here are somewhat speculative. It is likely that the predicted economic benefits are overstated for that population group (20 to 30 percent of total population in the case of PM_{10}) for which distant monitors are used. Conversely, there is an implied zero economic benefit for that portion of the population (3 to 4 percent in the case of PM_{10}) excluded from the analysis altogether, an understatement of economic impacts for that group.

The results for total monetized benefits were also calculated using a Monte Carlo simulation model to capture the combined uncertainties in the selection of physical effects function and unit values for effects reductions. The Monte Carlo methodology used in this analysis and the summary results are presented in Chapter 7. The results of the Monte Carlo simulations for each of the target years, plus the total 1990 value of monetized benefits and costs derived from the Monte Carlo analysis, are presented in Table 141.

Table 141. 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 ($Billions)	1975	1980	1985	1990	Present Value ($B)
5th percentile	$35	$125	$186	$216	$2,744
50th percentile	$108	$317	$435	$489	$6,786
95th percentile	$249	$687	$921	$1,020	$14,629

Notes:

"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. Present value reflects compounding of benefits from 1971 to 1990.

Uncertainty estimates for intermediate years computed based on ratios of 5th to 50th percentile and 95th to 50th percentile for given years. Ratios interpolated between given years and applied to point estimates for intermediate years.

Table 142. Criteria Pollutants Health and Welfare Benefits -- Population Within 50KM of a Monitor Modeled (in millions of 1990 dollars per year).

Endpoint	Study	Pollutant	1975	1980	1985	1990
Mortality	Dockery et al., 1992, St. Louis	PM10&O3	$141,860	$347,754	$401,780	$383,234
	Dockery et al., 1992, E. Tenn.	PM10&O3	$123,234	$299,393	$336,858	$311,774
	Pope et al., 1992	PM10	$117,078	$285,191	$326,526	$309,378
	Schwartz, 1993a	PM10	$82,694	$200,102	$228,433	$216,976
	Schwartz, 1991/1992	PM10	$76,920	$185,925	$212,147	$201,587
	Schwartz, 1991	PM10	$75,318	$181,998	$207,638	$197,324
	Fairley, 1990	PM10	$73,076	$176,506	$201,336	$191,364
	Schwartz and Dockery, 1992b	PM10	$64,366	$155,212	$176,920	$168,255
	Styer et al., 1995	PM10	$63,169	$152,292	$173,574	$165,086
	Ostro et al., 1995	PM10	$61,893	$149,180	$170,009	$161,709
	Ito et al., 1995	PM10	$45,907	$110,319	$125,561	$119,554
	Schwartz and Dockery, 1992a	PM10	$43,766	$105,131	$119,637	$113,929
	Moolgavkar et al., 1995	PM10&O3&SO2	$35,640	$107,507	$113,829	$111,475
	Kinney et al., 1995	PM10 & O3	$38,380	$92,102	$104,765	$99,800
	Backward-Looking Analysis	Pb	$7,895	$48,652	$94,201	$118,460
	Forward-Looking Analysis	Pb	$5,856	$33,425	$59,362	$72,568
Coronary Heart Disease & Strokes						
Coronary Heart Disease	Backward-Looking Analysis	Pb	$811	$5,351	$10,970	$14,234
	Forward-Looking Analysis	Pb	$523	$3,180	$5,912	$7,360
Strokes	Backward-Looking Analysis	Pb	$454	$2,960	$5,934	$7,470
	Forward-Looking Analysis	Pb	$295	$1,781	$3,247	$3,927
Hospital Admissions						
All Respiratory Admissions	Schwartz, 1995, Tacoma	PM10 & O3	$157	$392	$469	$460
	Pope, 1991, Salt Lake Valley	PM10	$156	$372	$421	$402
	Schwartz, 1995, New Haven	PM10 & O3	$114	$281	$329	$319
	Thurston et al, 1994, Toronto	PM10 & O3	$70	$170	$202	$204
Pneumonia + COPD Admissions	Schwartz, 1994c	PM10 & O3	$142	$359	$421	$397
	Schwartz 1994a	PM10 & O3	$111	$273	$319	$307
	Schwartz 1994b	PM10 & O3	$85	$209	$250	$248
Ischemic Heart Disease	Schwartz and Morris, 1995	PM10	$53	$127	$145	$138
Congestive Heart Failure	Schwartz and Morris, 1995 & Morris et al. 95	PM10 & CO	$57	$146	$220	$254
Respiratory Illness or Symptoms						
Children						
Upper + Lower Resp. Illness	Pope et al., 1991	PM10	$10	$27	$31	$29
Acute Bronchitis	Dockery et al., 1989	PM10	$59	$187	$220	$207
Adults						
Any of 19 Symptoms	Krupnick et al., 1990	PM10 & O3	$598	$1,437	$1,653	$1,599
Any Symptom	Linn et al. (1987, 1988, 1990)	SO2	$1	$3	$2	$2
Increase in Respiratory Illness	Hasselblad, 1992	NO2	$12	$52	$118	$181
Asthma Attacks	Ostro et al., 1991	PM10	$7	$14	$17	$18
	Whittemore and Korn, 1980 and US EPA ,19	O3	$4	$11	$19	$25
Restricted Activity and Work Loss Days						
WLDs+(RAD-WLD)*$MRAD	Ostro, 1987	PM10	$61	$153	$177	$166
Work Loss Days + MRAD	Ostro, 1987 & Ostro and Rothschild, 1989	PM10 & O3	$171	$440	$523	$492
IQ Changes & Hypertension						
Lost IQ Points	Backward-Looking Analysis	Pb	$6,520	$31,830	$54,333	$65,855
	Forward-Looking Analysis	Pb	$6,510	$31,821	$54,323	$65,844
IQ < 70	Backward-Looking Analysis	Pb	$314	$1,447	$2,210	$2,624
	Forward-Looking Analysis	Pb	$353	$1,719	$2,788	$3,377
Hypertension	Backward-Looking Analysis	Pb	$670	$4,326	$8,404	$10,597
	Forward-Looking Analysis	Pb	$461	$2,864	$5,339	$6,633
Decreased Worker Productivity	Based on Crocker & Horst, 1981 and EPA,	O3	$40	$146	$294	$346
Visibility - East	Pitchford and Malm, 1994	deciview	$373	$1,321	$1,750	$2,027
Visibility - West	Pitchford and Malm, 1994	deciview	$276	$625	$716	$960
Household Soiling	RCG/Hagler Bailly, 1994	PM10	$1,779	$4,303	$4,815	$4,631
Agriculture (Net Surplus)	Minimum estimate	O3	$220	$254	$424	$384
	Maximum estimate	O3	$682	$1,188	$1,371	$1,593

Table 143. Criteria Pollutants Health and Welfare Benefits -- Extrapolated to Entire 48 State Population (in millions of 1990 dollars per year).

Endpoint	Study	Pollutant	1975	1980	1985	1990
Mortality	Dockery et al., 1992, St. Louis	PM10 & O3	$173,297	$418,313	$502,002	$554,126
	Dockery et al., 1992, E. Tenn.	PM10 & O3	$148,216	$359,790	$421,215	$457,360
	Pope et al., 1992	PM10	$142,437	$343,113	$408,457	$449,251
	Schwartz, 1993a	PM10	$100,648	$240,928	$286,210	$315,274
	Schwartz, 1991/1992	PM10	$93,626	$223,887	$265,874	$292,944
	Schwartz, 1991	PM10	$91,678	$219,165	$260,241	$286,757
	Fairley, 1990	PM10	$88,952	$212,563	$252,368	$278,107
	Schwartz and Dockery, 1992b	PM10	$78,358	$186,955	$221,847	$244,562
	Styer et al., 1995	PM10	$76,902	$183,442	$217,664	$239,961
	Ostro et al., 1995	PM10	$75,350	$179,698	$213,205	$235,058
	Ito et al., 1995	PM10	$55,899	$132,934	$157,570	$173,831
	Schwartz and Dockery, 1992a	PM10	$53,293	$126,689	$150,149	$165,658
	Moolgavkar et al., 1995	PM10&O3&SO2	$47,962	$132,162	$144,803	$154,756
	Kinney et al., 1995	PM10 & O3	$46,738	$111,000	$131,514	$145,128
	Backward-Looking Analysis	Pb	$7,895	$48,652	$94,201	$118,460
	Forward-Looking Analysis	Pb	$5,856	$33,425	$59,362	$72,568
Coronary Heart Disease & Strokes						
Coronary Heart Disease	Backward-Looking Analysis	Pb	$811	$5,351	$10,970	$14,234
	Forward-Looking Analysis	Pb	$523	$3,180	$5,912	$7,360
Strokes	Backward-Looking Analysis	Pb	$454	$2,960	$5,934	$7,470
	Forward-Looking Analysis	Pb	$295	$1,781	$3,247	$3,927
Hospital Admissions						
All Respiratory Admissions	Schwartz, 1995, Tacoma	PM10 & O3	$195	$473	$585	$658
	Pope, 1991, Salt Lake Valley	PM10	$190	$448	$530	$585
	Schwartz, 1995, New Haven	PM10 & O3	$141	$339	$412	$458
	Thurston et al, 1994, Toronto	PM10 & O3	$88	$205	$254	$289
Pneumonia + COPD Admissions	Schwartz, 1994c	PM10 & O3	$173	$431	$522	$572
	Schwartz 1994a	PM10 & O3	$136	$329	$398	$442
	Schwartz 1994b	PM10 & O3	$105	$252	$313	$353
Ischemic Heart Disease	Schwartz and Morris, 1995	PM10	$65	$154	$182	$201
Congestive Heart Failure	Schwartz and Morris, 1995 & Morris et al. 95	PM10 & CO	$73	$185	$292	$354
Respiratory Illness or Symptoms						
Children						
Upper + Lower Resp. Illness	Pope et al., 1991	PM10	$13	$32	$38	$42
Acute Bronchitis	Dockery et al., 1989	PM10	$77	$239	$299	$327
Adults						
Any of 19 Symptoms	Krupnick et al., 1990	PM10 & O3	$734	$1,733	$2,076	$2,311
Any Symptom	Linn et al. (1987, 1988, 1990)	SO2	$1	$3	$3	$3
Increase in Respiratory Illness	Hasselblad, 1992	NO2	$20	$72	$163	$254
Asthma Attacks	Ostro et al., 1991	PM10	$9	$18	$22	$27
	Whittemore and Korn, 1980 and US EPA ,19	O3	$7	$14	$24	$31
Restricted Activity and Work Loss Days						
WLDs+(RAD-WLD)*$MRAD	Ostro, 1987	PM10	$74	$183	$220	$340
Work Loss Days + MRAD	Ostro, 1987 & Ostro and Rothschild, 1989	PM10 & O3	$209	$528	$644	$705
IQ Changes & Hypertension						
Lost IQ Points	Backward-Looking Analysis	Pb	$6,520	$31,830	$54,333	$65,855
	Forward-Looking Analysis	Pb	$6,510	$31,821	$54,323	$65,844
IQ < 70	Backward-Looking Analysis	Pb	$314	$1,447	$2,210	$2,624
	Forward-Looking Analysis	Pb	$353	$1,719	$2,788	$3,377
Hypertension	Backward-Looking Analysis	Pb	$670	$4,326	$8,484	$10,597
	Forward-Looking Analysis	Pb	$461	$2,964	$5,339	$6,633
Decreased Worker Productivity	Based on Crocker & Horst, 1981 and EPA,	O3	$70	$184	$364	$454
Visibility - East	Pitchford and Malm, 1994	deciview	$373	$1,321	$1,750	$2,027
Visibility - West	Pitchford and Malm, 1994	deciview	$276	$625	$716	$960
Household Soiling	RCG/Hagler Bailly, 1994	PM10	$2,323	$5,480	$6,478	$7,159
Agriculture (Net Surplus)	Minimum estimate	O3	$220	$254	$424	$384
	Maximum estimate	O3	$687	$1,188	$1,371	$1,593

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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.

In recent years, the EPA's extramural grants program has grown considerably. Several grants that were awarded in fiscal year 1995 are directly relevant to future Section 812 analyses. Examples of topic areas include:

- Health effects of particulate matter
- Biologically-based risk assessment models for cancer
- Non-cancer risk assessment statistical methods
- Valuation of damages to forest ecosystems
- Valuation of mortality risk

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.[286] The analytical issues list and strawman design was presented to the Science Advisory Board Clean Air Act Compliance Analysis Council (CAACAC), 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.

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 CAACAC, 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.

[286] Copies of the Prospective Study planning briefing materials are available from EPA.

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.

NEIL GOLDSCHMIDT
GOVERNOR

Department of Environmental Quality

811 SW SIXTH AVENUE, PORTLAND, OREGON 97204-1390 PHONE (503) 229-5696

September 29, 1989

CERTIFIED MAIL

Oddvar K. Aurdal, Chief
Grants Administration Section
U.S. Environmental Protection Agency
Region 10
1200 Sixth Avenue, MD-100
Seattle, WA 98101

Re: V-000332-01-C
Extension of
Project/Budget Periods

Dear Mr. Aurdal:

We are pleased to accept your approval extending the Project/Budget Period to September 30, 1990.

As requested, we have reviewed the amendment and find it to be satisfactory.

Enclosed are the signed original and two copies.

Sincerely,

Fred Hansen
Director

FH:c
C1057
Enclosures
cc: Kenneth Brooks, EPA
Michael Downs, ECD Administrator, DEQ
Judy Hatton, Business Office, DEQ
John Loewy, Acting MSD Administrator, DEQ
John Rist, Budget Office, DEQ

RECEIVED
OCT 4 1989
GRANTS ADMINISTRATION

NEIL GOLDSCHMIDT
GOVERNOR

Department of Environmental Quality

811 SW SIXTH AVENUE, PORTLAND, OREGON 97204-1390 PHONE (503) 229-5696

CERTIFIED MAIL NO. P 915 446 265
RETURN RECEIPT REQUESTED

Oddvar K. Aurdal, Chief
Grants Administration Section
U.S. Environmental Protection Agency
Region 10
1200 Sixth Avenue
Seattle, WA 98101

RE: PA/SI V-000332-01-B
SUPERFUND MULTISITE

Dear Mr. Aurdal:

Enclosed are a signed original and two copies of the amendment for the PA/SI portion of the above referenced agreement.

Our records confirm the decrease of $11,651 is consistent with the Final Financial Status Report (FSR) for the PA/SI portion of the award completed 03-31-89. It is understood the amendment applies only to the portion of the award completed, and does not affect the remaining $360,596 in the Superfund Multisite Agreement.

Sincerely,

Fred Hansen
Director

RN:m
PPD\SM2501
Enclosures
cc: John Rist: MSD, DEQ
Rich Nourse: ECD, DEQ

RECEIVED
SEP 2 1999
GRANTS ADMINISTRATION

UNITE ATES ENVIRONMENTAL PROTECTIOI ENCY

FEB 26 1990

Reply To
Attn Of: MD-100

P 539 661 780

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NO INSURANCE COVERAGE PROVIDED
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Street and No.	
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Postage	$
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Return Receipt showing to whom and Date Delivered	
Return Receipt showing to whom, Date, and Address of Delivery	
TOTAL Postage and Fees	$
Postmark or Date	

PS Form 3800, June 1985 U.S.G.P.O. 1989-234-555

Fred Hansen, Director
Oregon Department Environmental Quality
811 Southwest Sixth Avenue
Portland, Oregon 97204

Re: V-000332-01-E
Superfund Multi-Site Agreement

Dear Mr. Hansen:

I am pleased to approve an increase of management assistance activities at the Jos Previously we awarded $360,596; the revisec $390,596.

Please review the enclosed amendment. please sign, date, and return an original mail to Grants Administration Section, MD- receipt.

If you wish to discuss this amendment Robinson, EPA Project Officer at (206) 442-2104.

Sincerely,

Charles E. Findley, Director
Hazardous Waste Division

Enclosures

7118/mr/
bcc: Robinson, HW-113 Nourse, ODEQ Hatton, ODEQ

CONCURRENCES								
SYMBOL								
SURNAME			Robinson					
DATE		2/23/90	2-23-90	2/23/90				

EPA Form 1320-1 (12-70) Rudial Larsen Robinson Millam OFFICIAL FILE COPY
02-23-90

★U.S.GPO:1988-0-206-471

United States
Environmental Protection
Agency

Region 10
1200 Sixth Avenue
Seattle WA 98101

Alaska
Idaho
Oregon
Washington

EPA

July 31, 1990

MEMORANDUM

SUBJECT: Cooperative Agreement V-000332-01

FROM: Kirk Robinson

TO: Mel Rozier

The Oregon State Department of Environmental Quality has requested that $8,820 be transferred from the Martin Marietta component of the above referenced agreement to the Teledyne component. A copy of their letter of request is attached. The letter contains the proposed distribution of funds within the Teledyne component of the agreement. The State has also requested that the Martin Marietta component of the agreement be administratively closed once the funds transfer is completed.

I have reviewed and approve the request.

Thank you for your attention to this matter.

NEIL GOLDSCHMIDT
GOVERNOR

RECEIVED
JUN 25 1990
SUPERFUND BRANCH

Department of Environmental Quality

811 SW SIXTH AVENUE, PORTLAND, OREGON 97204-1390 PHONE (503) 229-5696

June 22, 1990

Mr. Kirk Robinson, Program Officer
U. S. Environmental Protection Agency
Region X
1200 Sixth Avenue
Seattle, WA 98101

Re: Federal Cooperative Agreement
V-000332-01 - Martin Marietta

Dear Mr. Robinson:

The Oregon Department of Environmental Quality (DEQ) is currently a signature party to the Consent Decree for the Martin Marietta Superfund Project. The state is receiving direct reimbursement from the company for its participation in project oversight, and is no longer charging expenditures to the portion of the above referenced cooperative agreement approved for Martin Marietta.

According to the most recent fiscal reports, there is an unexpended balance of $8,820 remaining in the Martin Marietta Management Assistance project fund, and no outstanding obligations. The Oregon DEQ requests the existing balance of $8,820 currently on deposit with Martin Marietta be transferred to the Teledyne Wah Chang project which has the same activity under the cooperative agreement between EPA and the State of Oregon. Please transfer the Martin Marietta funds to the Teledyne project funds in accordance with the following distribution.

PERSONNEL:.....	$ 4,700.00
BENEFITS:.....	1,645.00
INDIRECT COSTS:.....	2,475.00
	$ 8,820.00

Once the transfer of funds has been completed, please initiate the administrative completion and closeout of the Martin Marietta project.

Thank you for your assistance and cooperation. If you have questions, or require additional information, please contact Rich Nourse at 503-229-6801.

Sincerely,

Jeff Christensen
Policy & Program Development
Environmental Cleanup Division

RN:m
PPD\SM3105
cc: Chip Humphrey, Oregon Operations, EPA
Judy Hatton, DEQ Business Office

NEIL GOLDSCHMIDT
GOVERNOR

Environmental Quality Commission

811 SW SIXTH AVENUE, PORTLAND, OR 97204 PHONE (503) 229-5696

March 21, 1990

CERTIFIED MAIL

Oddvar K. Aurdal, Chief
Grants Administration Section
U.S. Environmental Protection Agency
Region 10
1200 Sixth Avenue, MD-100
Seattle, WA 98101

Re: V-000332-01-E
Superfund Multi-Site
Agreement

Dear Mr. Aurdal:

We are pleased to accept your increase for the above referenced cooperative agreement, dated February 26, 1990, of $30,000 for participation in the management assistance activities at the Joseph Forest Products site.

As requested, we have reviewed the amendment and find it to be satisfactory.

Enclosed are the signed original and two copies.

Sincerely,

Fred Hansen
Director

RECEIVED
MAR 2 3 1990
GRANTS ADMINIST

FH:c
C1119
Enclosures
cc: Kenneth Brooks, EPA
Kirk Robinson, Project Officer, EPA
Michael Downs, Environmental Cleanup Division, DEQ
Richard Nourse, Environmental Cleanup Division, DEQ
Judy Hatton, Business Office, DEQ
Peter Dalke, MSD Administrator, DEQ
John Rist, Budget Office, DEQ
Peggy Halferty, Budget Analyst, DEQ

United States
Environmental Protection
Agency

Region 10
1200 Sixth Avenue
Seattle WA 98101

Alaska
Idaho
Oregon
Washington

EPA

Reply To
Attn. Of: MD-100

April 20, 1990

Judy Hatton, Accounting Manager
Oregon Department of Environmental Quality
811 SW Sixth Avenue
Portland, Oregon 97204

Dear Ms. Hatton:

According to our records, the unpaid balance for Superfund agreements are as follows:

	Unpaid Balances thru No. 239	Agreement Expires
V-000332-01		
Teledyne	7,350	9/30/90
Gould	27,050	9/30/90
United Chrome	12,100	9/30/90
Martin Marietta	8,820	9/30/90
Umatilla	7,833	9/30/90
Joseph Forest Products	30,000	2/28/91
Allied Plating	22,150	9/30/90
V-000350-02 CORE	0	2/28/90
V-000350-03 CORE	290,000	2/28/91
V-000363-01 SARA	50,583	6/30/90
V-000399-01 PA/SI	35,580	5/31/91

A final Financial Status Report FSR) on the 2nd segment of the Core program is due by May 31, 1990.

An annual FSR is due by June 30, 1990 for the budget period ending March 31, 1990 for all sites except Joseph Forest Products covered in the multi-site agreement, the 3rd segment of the Core program, and the SARA and PA/SI agreements.

Please contact Mel Rozier, of my staff, on (206) 442-2919 if our records are not in agreement.

Sincerely,

Oddvar K. Aurdal

Oddvar K. Aurdal, Chief
Grants Administration Section

cc: Rich Nourse, ODEQ
Kirk Robinson, HW-113
Debbie Flood, HW-093