Regulatory Impact Analysis for the Final Revisions to the National Ambient Air Quality Standards for Particulate Matter

On February 28, 2013, the PM RIA is being replaced to add Appendix 3.A, which was inadvertently left out and to correct the document number to EPA-452/R-12-005.

EPA-452/R-12-005
December 2012

Regulatory Impact Analysis for the Final Revisions to the National Ambient Air Quality Standards for Particulate Matter

U.S. Environmental Protection Agency
Office of Air Quality Planning and Standards
Health and Environmental Impacts Division
Research Triangle Park, NC 27711

CONTENTS

LIST OF FIGURES

LIST OF TABLES

EXECUTIVE SUMMARY

ES.1 Overview

Based on its review of the air quality criteria and the national ambient air quality standards (NAAQS) for particulate matter (PM), the U.S. Environmental Protection Agency (EPA) is making revisions to the primary standards for PM to provide requisite protection of public health and welfare. The EPA is revising the primary annual (health-based) standard, retaining the primary 24-hour standard, and retaining the secondary (welfare-based) NAAQS for fine particles (generally referring to particles less than or equal to 2.5 micrometers [µm] in diameter—$PM_{2.5}$). The EPA is retaining the current primary and secondary 24-hour PM_{10} standards.

As has traditionally been done in NAAQS rulemakings, the EPA has conducted a Regulatory Impact Analysis (RIA) to provide the public with illustrative estimates of the potential costs and health and welfare benefits of attaining the revised annual standard along with two alternative standards. In NAAQS rulemakings, the RIA is prepared for informational purposes only, and the decisions related to the setting of the PM NAAQS standards are not in any way based on consideration of the information or analyses in the RIA. The RIA fulfills the requirements of Executive Orders 12866 and 13563 and guidelines of the Office of Management and Budget's (OMB) Circular A-4.[1]

The control strategies presented in this RIA are illustrative and represent one set of control strategies States might choose to implement in order to meet the final standards. As a result, benefit and cost estimates provided in this RIA cannot be added to benefits and costs from other regulations because each regulation is based on a different set of analytical assumptions and policy decisions. The costs and benefits identified in this RIA will not be realized until specific controls are mandated by future State Implementation Plans (SIPs) or other Federal regulations.

ES.2 Existing and Revised PM Air Quality Standards

Two primary $PM_{2.5}$ standards provide public health protection from effects associated with fine particle exposures: the annual standard and the 24-hour standard. The current annual standard is set at a level of 15.0 $\mu g/m^3$, based on the 3-year average of annual arithmetic mean $PM_{2.5}$ concentrations. The current 24-hour standard is set at a level of 35 $\mu g/m^3$, based on the

[1] U.S. Office of Management and Budget. Circular A-4, September 17, 2003. Available at http://www.whitehouse.gov/omb/circulars/a004/a-4.pdf.

3-year average of the 98th percentile of 24-hour $PM_{2.5}$ concentration. In the RIA, the current primary $PM_{2.5}$ standards, including both annual and 24-hour standards, are denoted as 15/35 µg/m^3. Attainment of the 24-hour standard is analyzed only in developing the scenario of attainment with the existing standards of 15/35 µg/m^3. All other scenarios evaluate additional emissions reductions needed to attain alternative annual standards only.

In this PM NAAQS review, the EPA has revised and lowered the level of the primary annual $PM_{2.5}$ standard to 12 µg/m^3 in conjunction with retaining the level of the 24-hour standard at 35 µg/m^3 and this standard is denoted as 12 µg/m^3. In addition to the revised annual standard of 12 µg/m^3, the RIA also analyzes the benefits and costs of incremental control strategies for two alternative annual standards of 13 µg/m^3 and 11 µg/m^3.

Currently, the existing secondary (welfare-based) $PM_{2.5}$ standards are identical in all respects to the primary standards. In this PM NAAQS review, the EPA is retaining the current suite of secondary standards for 24-hour and annual $PM_{2.5}$. Thus, while the new primary annual standard will be revised to 12 µg/m^3, the secondary annual standard will remain at 15 µg/m^3. Non-visibility welfare effects are addressed by this suite of secondary standards, and PM-related visibility impairment is addressed by the secondary 24-hour $PM_{2.5}$ standard, which EPA is leaving unchanged at 35 µg/m^3. The secondary standards will thus remain at 15/35 µg/m^3.

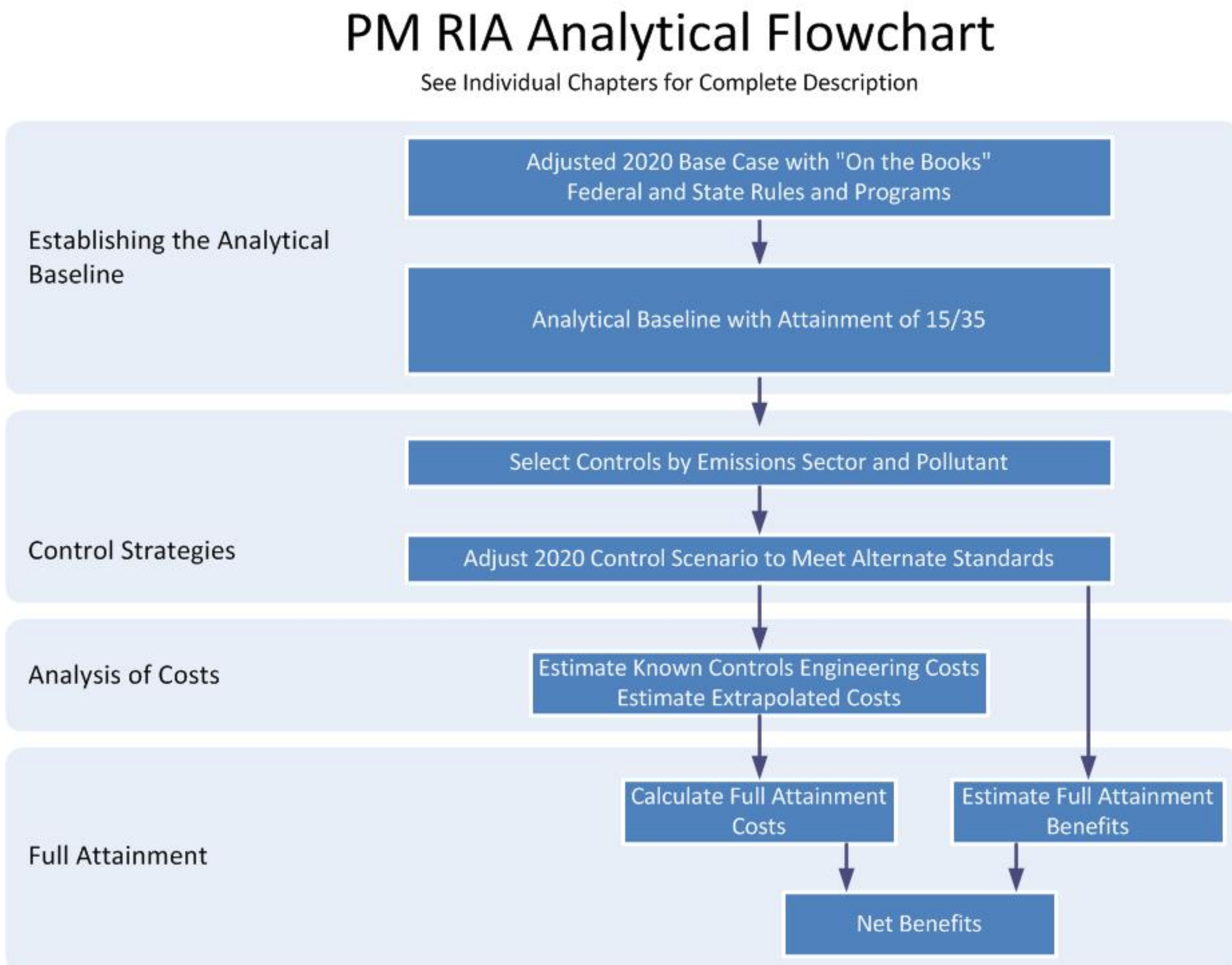
With regard to the primary and secondary standards for particles less than or equal to 10 µm in diameter (PM_{10}), the EPA is retaining the current primary and secondary 24-hour PM_{10} standards, which are both set at a level of 150 µg/m^3, not to be exceeded more than once per year on average over 3 years (U.S. EPA, 1997).[2] Because the benefit-cost analysis of the alternative PM_{10} standards was conducted when the standard was promulgated in 1997, this RIA does not repeat that analysis here.

ES.2.1 Overview of the Analytical Steps in this RIA

The goal of this RIA is to provide the best estimates of the costs and benefits of an illustrative attainment strategy to the meet the revised annual standard. The flowchart below (Figure ES-1) outlines the analytical steps taken to illustrate attainment with the revised annual standard of 12 µg/m^3, and the following discussion, by primary flowchart section, describes each of the steps taken. For important updates and analytical differences between proposal and final, see section ES.5 of this Executive Summary.

[2] U.S. Environmental Protection Agency. 1997. Regulatory Impact Analyses for the Particulate Matter and Ozone National Ambient Air Quality Standards and Proposed Regional Haze Rule. Available at: http://www.epa.gov/ttn/oarpg/naaqsfin/ria.html.

Figure ES-1. PM RIA Analytical Flowchart



Establishing the Analytical Baseline (Flowchart Section 1)

This section of the flowchart reflects the analytical steps taken to account for Federal and State rules and programs currently underway, as well as to reflect attainment of the current annual and daily standards of 15/35 µg/m^3 for the purpose of estimating the incremental costs and benefits of attaining the revised annual standard. Detailed discussions of the elements of this section of the flowchart are in Chapters 1 and 3 of the final RIA.

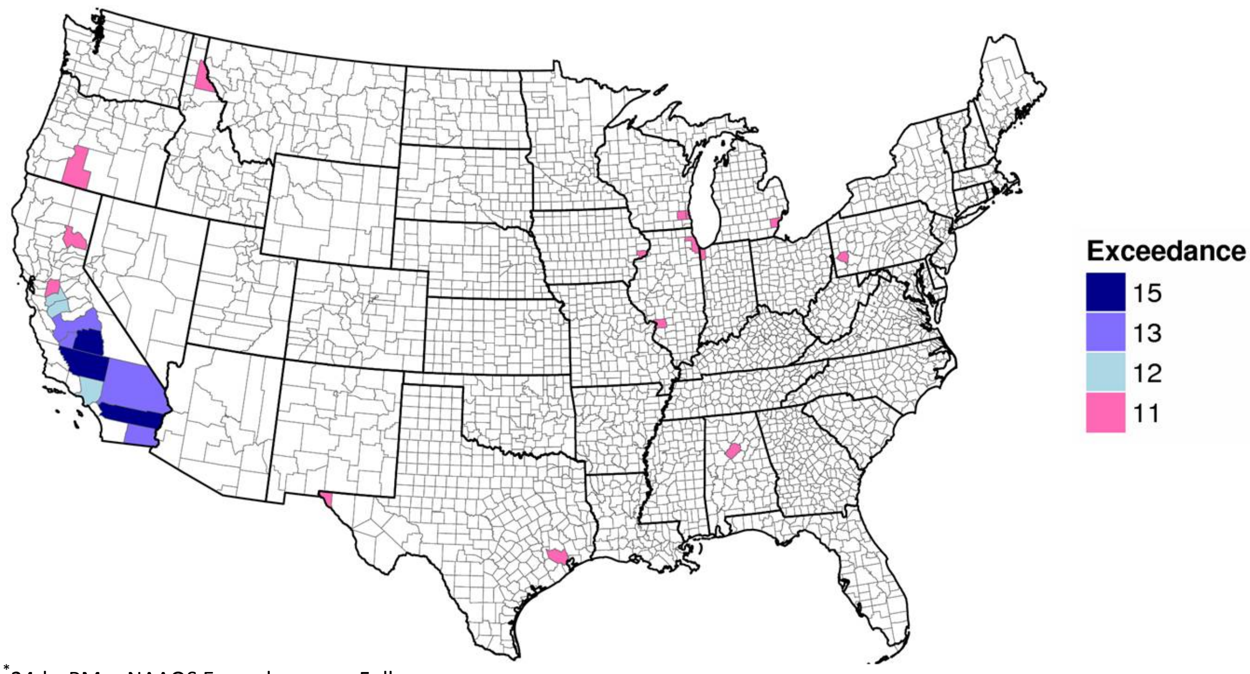
- *Adjusted 2020 Base Case with "On the Books" Federal and State Rules and Programs*—The adjusted 2020 base case includes reductions expected to occur between 2007 and 2020 from existing (i.e., "on-the-books") Federal and State control programs. This projection reflects air quality modeling of 2020 that accounts for major Federal and State programs along with adjustments to these national-level modeling results to account for $PM_{2.5}$ reductions expected, largely in the western

US, from the implementation of episodic burn ban programs in certain counties and to remove the effects of atypical events such as wildfires and fireworks displays. (See Figure ES-2 below for illustration.) Below is a list of some of the major national rules reflected in the base case. Refer to Chapter 3, Section 3.2.1.4 for a more detailed discussion of the rules reflected in the 2020 base case emissions inventory.

- Light-Duty Vehicle Tier 2 Rule (U.S. EPA, 1999)
- Heavy Duty Diesel Rule (U.S. EPA, 2000)
- Clean Air Nonroad Diesel Rule (U.S. EPA, 2004)
- Regional Haze Regulations and Guidelines for Best Available Retrofit Technology Determinations (U.S. EPA, 2005b)
- NO_x Emission Standard for New Commercial Aircraft Engines (U.S. EPA, 2005a)
- Emissions Standards for Locomotives and Marine Compression-Ignition Engines (U.S. EPA, 2008a)
- Control of Emissions for Nonroad Spark Ignition Engines and Equipment (U.S. EPA, 2008b)
- C3 Oceangoing Vessels (U.S. EPA, 2010a)
- Boiler MACT (U.S. EPA, 2011d)
- Hospital/Medical/Infectious Waste Incinerators: New Source Performance Standards and Emission Guidelines: Final Rule Amendments (U.S. EPA, 2009)
- Reciprocating Internal Combustion Engines (RICE) NESHAPs (U.S. EPA, 2010b)
- Mercury and Air Toxics Standards (U.S. EPA, 2011b)
- Cross-State Air Pollution Rule (U.S. EPA, 2011a)[3]

[3] See Chapter 3, Section 3.2.1.4 for a discussion of the role CSAPR plays in the $PM_{2.5}$ RIA and the reasons we believe CSAPR remains an appropriate proxy for this analysis.

Figure ES-2. Annual NAAQS Exceedances* in "Adjusted 2020 Base Case" Scenario



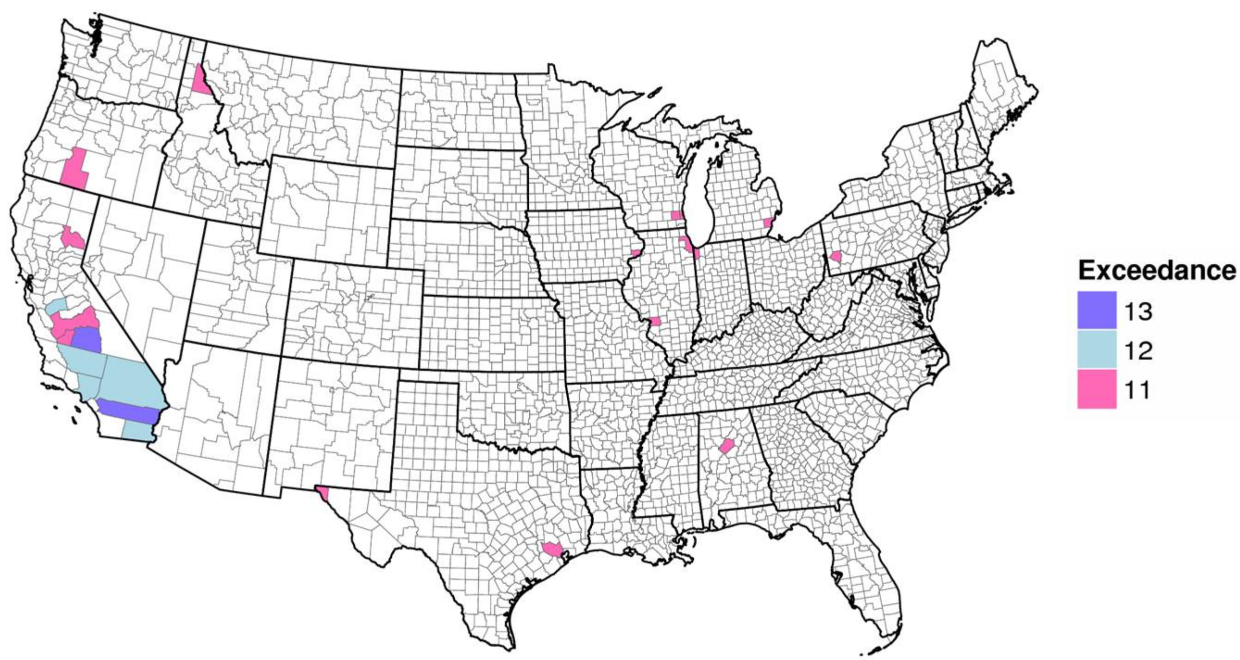
*24-hr $PM_{2.5}$ NAAQS Exceedances as Follows:

- San Joaquin Valley (6 counties); South Coast (2 counties); Imperial, CA; Allegheny, PA; Salt Lake, UT; Lake, OR; and Sacramento, CA

- *Analytical Baseline with Attainment of 15/35*—The analytical baseline includes reductions from additional controls that the EPA estimates are needed to attain the current standards (15/35 μg/m^3) for the purpose of estimating the incremental costs and benefits of attaining the revised annual standard of 12 μg/m^3. Determining the level of emissions reductions needed to meet both the current annual and daily standards is done through adjusting a county's projected design value (DV) concentration using the geographic area and pollutant specific air quality ratios (as described in Chapter 3 of the final RIA). In addition, for each area, it is necessary to determine which of the standards will be "controlling," i.e., which standard will require the most reductions to reach attainment of current standard. This is important for establishing the analytical baseline because when the daily standard is controlling (as is the case in much of California), the emissions reductions required to meet the daily standard will result in reductions in the annual design value to below the annual standard of 15 μg/m^3. As a result, the annual $PM_{2.5}$ increment needed to attain alternative standards for each county will not be relative to an annual design value of 15 μg/m^3, rather, the increment will vary by county based on the annual design value in that county that resulted from applying emissions controls to meet the 35 μg/m^3 daily standard. As shown in the map below, for counties labeled as exceeding the 12 μg/m^3 annual standard, the baseline design values can be any value between 12 and 15 μg/m^3. Similarly, exceedances of the 11 μg/m^3 and 13 μg/m^3 standards can be anywhere between 11 or 13 and 15 μg/m^3.

The analytical baseline reflects attainment of 15/35 µg/m^3 by2020. We also modified the analytical baseline for counties in the South Coast Air Quality Management District and the San Joaquin Valley Air Pollution Control District to reflect reductions in mobile NO_x emissions that these areas are expected to achieve between 2020 and 2025 due to fleet turnover. These reductions in NO_x emissions are not attributable to attainment of the current or revised PM standards, but reflect the impacts of other "on-the-books" mobile programs so that these reductions are not included as either an incremental cost nor benefit to these area's attaining the revised annual standard.

Figure ES-3. Annual NAAQS Exceedances in "Analytical Baseline" Scenario



- 13 Level: 2 Counties (Tulare, CA and Riverside, CA)
- 12 Level: 7 Counties (All in CA)
- 11 Level: 23 Counties

Control Strategies (Flowchart Section 2)

This section of the flowchart reflects analytical steps taken to analyze controls and emissions reductions needed beyond the current standard (15/35 µg/m^3) and other existing major rules to achieve the revised standard (12 µg/m^3). We apply control options that might be available to States for application by 2020. Detailed discussion of the elements of this section of the flowchart is in Chapter 4 of the final RIA.

- *Select Controls by Emissions Sector and Pollutant*—Non-EGU point and nonpoint control measures were applied for the revised and alternative standards' control strategies. These controls were identified using the U.S. EPA's Control Strategy Tool.

Additional control measures were not applied to EGUs because of the extensive nature of controls resulting from the inclusion of MATS.

- *Adjust 2020 Control Scenario to Meet Alternate Standards*—After identifying the known controls in the control scenario that were needed to meet the analytical baseline, additional known controls needed to meet the revised and alternative standards were identified. The EPA used air quality modeling results to determine whether the control scenario was sufficient to meet the revised and alternative standards for each geographic area. Where the control scenario modeling resulted in design value reductions below the level needed for the revised or alternative standards for specific geographic areas, county-specific ratios of air quality response to emission reductions were used to determine the subset of controls that were needed to attain the standard. Where it was determined that the control scenario was not sufficient in attaining the standard, these same response factors were used to calculate the amount of additional emission reductions beyond known controls needed to meet the standard. For the revised and alternative control strategy analysis, known controls for two sectors were used: non-EGU point and area sources. Onroad mobile source controls were not used in the revised and alternative standards analysis because they were applied previously in the analytical baseline analysis where they were deemed to be most cost effective. Emission reductions were calculated for the known control strategy analysis and the cost analysis for emission reductions needed beyond known controls ("extrapolated" costs) for each alternative standard being analyzed. The EPA estimates the national-scale emission reductions for revised annual standard of 12 $\mu g/m^3$ and two alternative annual standards (13 $\mu g/m^3$ and 11 $\mu g/m^3$) as shown in Table ES-1.

  Because the rules listed above and other emissions reductions should have substantially reduced ambient $PM_{2.5}$ concentrations by 2020 in the East, no additional controls are anticipated to be needed outside of California (assuming the absence of new sources). Specifically, our analysis estimates that in 2020 only 7 counties, all in California, will be out of attainment with the revised annual standard of 12ug/m^3. Emissions reductions are needed in more locations for the alternative standard of 11ug/m^3.

Table ES-1. Emission Reductions in Illustrative Emission Reduction Strategies for the Revised and Alternative Annual Primary $PM_{2.5}$ Standards, by Pollutant and Region in 2020 (tons)[a]

	13 µg/m^3	12 µg/m^3	11 µg/m^3
Directly emitted $PM_{2.5}$			
East	0	0	8,200
West	0	0	160
CA	730	4,000	10,600
SO_2			
East	0	0	21,000
West	0	0	43
CA	0	0	0
NOx			
East	0	0	9
West	0	0	0
CA	0	0	0

[a] See Chapter 4 for more information on the illustrative emission reduction strategies. The emissions in this table reflect both known and unknown controls. Estimates are rounded to two significant figure. Estimates are rounded to two significant figures.

Analysis of Costs (Flowchart Section 3)

This section of the flowchart reflects analytical steps taken to estimate the costs of both known and unknown controls. Detailed discussion of the elements of this section of the flowchart is in Chapter 7 of the final RIA.

- *Estimate Known Controls Engineering Costs, Estimate Extrapolated Costs*—We provide engineering cost estimates for the control strategies identified in Chapter 4 that include control technologies on non-EGU point sources and area sources. Engineering costs generally refer to the capital equipment expense, the site preparation costs for the application, and annual operating and maintenance costs. For this analysis, we included known controls for all of the geographic areas likely to exceed the revised and/or alternative standards. We also provide estimates for the engineering costs of the additional emissions reductions that are needed beyond the application of known controls to reach full attainment of the alternative standards analyzed; the cost estimates derived from this approach are referred to as "extrapolated" costs. By definition, no cost data currently exists for the additional emissions reductions needed beyond known controls. We employ two methodologies for estimating the costs of unidentified future controls, and both approaches assume that innovative strategies and new control options make possible the emissions reductions needed for attainment by 2020.

Full Attainment (Flowchart Section 4)

This section of the flowchart reflects analytical steps taken to estimate the costs of attainment, the benefits of attainment and the net benefits for the revised standard of 12 $\mu g/m^3$. Detailed discussions of the elements of this section of the flowchart are in Chapter 5, Chapter 7, and Chapter 8 of the final RIA.

- *Calculate Full Attainment Costs*—In Chapter 7 we present a summary of the total national costs of attaining the revised annual standard of 12 $\mu g/m^3$ and the alternative annual standards of 13 $\mu g/m^3$ and 11 $\mu g/m^3$ in 2020. This summary includes the known and extrapolated costs. The total cost estimates are $53 million (2010$) and $350 million (2010$) for the revised annual standard of 12 $\mu g/m^3$; $11 million and $100 million for the alternative annual standard of 13 $\mu g/m^3$; and $320 million and $1,700 million for the alternative annual standard of 11 $\mu g/m^3$.

- *Calculate Full Attainment Benefits*—Chapter 5 presents the estimated human health benefits for the revised NAAQS. We quantify the health-related benefits of the fine particulate matter ($PM_{2.5}$)-related air quality improvements resulting from the illustrative emissions control scenarios that reduce emissions of directly emitted particles and precursor pollutants including SO_2 and NO_x to reach alternative $PM_{2.5}$ NAAQS levels in 2020. These benefits are relative to an analytical baseline reflecting nationwide attainment of the current primary $PM_{2.5}$ standards (i.e., annual standard of 15 $\mu g/m^3$ and 24-hour standard of 35 $\mu g/m^3$) that includes promulgated national regulations and illustrative emissions controls to simulate attainment with 15/35 as well as a NO_x emission adjustment to reflect expected reductions in mobile NO_x emissions between 2020 and 2025.

  The estimated benefits for the revised and alternative standards are in addition to the substantial benefits estimated for several recent implementation rules. Rules such as the Mercury and Air Toxics Standard (MATS) and other emission reductions will have substantially reduced ambient $PM_{2.5}$ concentrations by 2020 in the East, such that no additional controls would be needed in the East for the revised annual standard of 12 $\mu g/m^3$. Thus, all of the estimated benefits occur in California because this is the only State that needs additional air quality improvement beyond the analytical baseline after accounting for air quality improvements from recent rules.

- *Net Benefits*—Chapter 8 compares estimates of the benefits with costs and summarizes the net benefits of revised annual standard of 12 $\mu g/m^3$ and the alternative annual standards of 13 $\mu g/m^3$ and 11 $\mu g/m^3$ relative to the analytical baseline that includes recently promulgated national regulations and additional emissions reductions needed to attain the existing 15/35 μg/m^3 standards, as well as adjustments to NO_x emissions in the San Joaquin and South Coast areas.

ES.2.2 Health and Welfare Co-Benefits

The EPA estimated impacts on human health (e.g., mortality and morbidity effects) under full attainment of the three alternative annual $PM_{2.5}$ standards. We considered an array of health impacts attributable to changes in $PM_{2.5}$ exposure and estimated these benefits using the BenMAP model (Abt Associates, 2012), which has been used in many recent RIAs (e.g., U.S. EPA, 2006, 2011a, 2011b), and *The Benefits and Costs of the Clean Air Act 1990 to 2020* (U.S. EPA, 2011c). The monetized benefits estimated in the core analysis include avoided premature deaths (derived from effect coefficients in tow cohort studies [Krewski et al. (2009) and Lepeule et al. (2012)] for adults and one for infants [Woodruff et al. (1997)]) as well as avoided morbidity effects for 10 non-fatal endpoints ranging in severity from lower respiratory symptoms to heart attacks. As noted above, because California is the only state that needs additional air quality improvement beyond the analytical baseline after accounting for expected air quality improvements expected from recent rules, all of the benefits associated with the revised standard of 12ug/m3 occur in California.

Since the proposed rule, the EPA has incorporated an array of policy and technical updates to the benefits analysis approach applied in this RIA, including incorporation of the most recent follow-up to the Harvard Six Cities cohort study (Lepeule et al., 2012), more recent demographic data projections, additional hospitalization and emergency department visit studies, inflation adjustment to 2010 dollars, and an expanded uncertainty assessment. Each of these updates is fully described in the health benefits chapter (Chapter 5) and summarized below in section ES.5. Compared with the proposal benefits, the estimated benefits for the revised standard are about double due to a combination of updates to the analytic baseline

Even though the primary standards are designed to protect against adverse effects to human health, the emission reductions will have welfare co-benefits in addition to the direct human health benefits. The term *welfare co-benefits* covers both environmental and societal benefits of reducing pollution, such as reductions in visibility impairment, materials damage, and ecosystem damage. Despite our attempts to quantify and monetize as many of the benefits as possible, the welfare co-benefits associated with meeting the alternative standards are not quantified or monetized in this analysis. Unquantified health benefits are discussed in Chapter 5, and unquantified welfare co-benefits are discussed in Chapter 6.

It is important to note that estimates of the health benefits from reduced $PM_{2.5}$ exposure reported here contain uncertainties, which are described in detail in Chapter 5 and Appendix 5b. Below are two key assumptions in the benefits analysis:

1. We assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This is an important assumption, because $PM_{2.5}$ varies considerably in composition across sources, but the scientific evidence is not yet sufficient to allow differentiation of effect estimates by particle type. The *Integrated Science Assessment for Particulate Matter* (PM ISA), which was twice reviewed by CASAC, concluded that "many constituents of $PM_{2.5}$ can be linked with multiple health effects, and the evidence is not yet sufficient to allow differentiation of those constituents or sources that are more closely related to specific outcomes" (U.S. EPA, 2009). These uncertainties are likely to be magnified in the current analysis to the extent that the emissions controls are less diverse when focusing on one small region of the country rather than a broader geography with more diverse emissions sources and the application of a more diverse set of controls.

2. We assume that health impact functions based on national studies are representative for exposures and populations in California. In addition to the national risk coefficients we use as our core estimates, the EPA considered the cohort studies conducted in California specifically. Although we have not calculated the benefits results using the cohort studies conducted in California, we provided these risk coefficients to show how much the monetized benefits could have changed. Most of the California cohort studies report central effect estimates similar to the (nation-wide) all-cause mortality risk estimate we applied from Krewski et al. (2009) and Lepeule et al. (2012) albeit with wider confidence intervals. Three cohort studies conducted in California indicate statistically significant higher risks than the risk estimates we applied from Lepeule et al. (2012), and four studies showed insignificant results.

3. We assume that the health impact function for fine particles is log-linear without a threshold in this analysis. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5,}$ including both areas that do not meet the fine particle standard and those areas that are in attainment, down to the lowest modeled concentrations.

In general, we are more confident in the magnitude of the risks we estimate from simulated $PM_{2.5}$ concentrations that coincide with the bulk of the observed PM concentrations in the epidemiological studies that are used to estimate the benefits. Likewise, we are less confident in the risk we estimate from simulated $PM_{2.5}$ concentrations that fall below the bulk of the observed data in these studies. As noted in the preamble to the rule, the range from the

25th to 10th percentiles of the air quality data in the epidemiology studies is a reasonable range below which we start to have appreciably less confidence in the magnitude of the associations observed in the epidemiological studies. Concentration benchmark analyses (e.g., 25th percentile, 10th percentile, one standard deviation below the mean,[4] and lowest measured level [LML]) provide some insight into the level of uncertainty in the estimated $PM_{2.5}$ mortality benefits. Most of the estimated avoided premature deaths for this rulemaking occur at or above the lowest measured $PM_{2.5}$ concentration in the two studies that are used to estimate mortality benefits. There are uncertainties inherent in identifying any particular point at which our confidence in reported associations becomes appreciably less, and the scientific evidence provides no clear dividing line. However, the EPA does not view these concentration benchmarks as a concentration threshold below which we would not quantify health benefits of air quality improvements. Rather, the core benefits estimates reported in this RIA (i.e., those based on Krewski et al. [2009] and Lepeule et al. [2012]) are the best measures because they reflect the full range of modeled air quality concentrations associated with the emission reduction strategies and because the current body of scientific literature indicates that a no-threshold model provides the best estimate of PM-related long-term mortality. It is important to emphasize that “less confidence” does not mean “no confidence.”

The estimated benefits reflect illustrative control measures and emission reductions to lower $PM_{2.5}$ concentrations at monitors projected to exceed the revised and alternative annual standards. The result is that air quality is expected to improve in counties that exceed these standards as well as surrounding areas that do not exceed the alternative standards. In order to make a direct comparison between the benefits and costs of the emission reduction strategies, it is appropriate to include all the benefits occurring as a result of the emission reduction strategies applied regardless of where they occur. Therefore, it is not appropriate to estimate the fraction of benefits that occur only in the counties that exceed the standards because it would omit benefits attributable to emission reduction in exceeding counties. In addition, we estimate benefits using modeled air quality data with 12 km grid cells, which is important because the grid cells are often substantially smaller than counties, and $PM_{2.5}$ concentrations can vary spatially within a county. Some grid cells in a county can be below the level of the alternative standard even though the highest monitor exceeds the alternative standard. Thus, emission reductions can lead to benefits in grid cells that are below the alternative standards within an exceeding county.

[4] A range of one standard deviation around the mean represents approximately 68% of normally distributed data and below the mean falls between the 25th and 10th percentiles.

ES.2.3 Cost Analysis Approach

The EPA estimated total costs under partial and full attainment of the alternative $PM_{2.5}$ standards. The engineering costs generally include the costs of purchasing, installing, and operating the referenced control technologies. The technologies and control strategies selected for analysis are illustrative of one way in which nonattainment areas could meet a revised standard. There are numerous ways to construct and evaluate potential control programs that would bring areas into attainment with alternative standards, and the EPA anticipates that State and local governments will consider programs that are best suited for local conditions.

The partial attainment cost analysis reflects the costs associated with applying known controls. Costs for full attainment include estimates for the engineering costs of the additional tons of emissions reductions that are needed beyond identified controls, referred to as extrapolated costs. By definition, no cost data currently exist for the additional emissions reductions needed beyond known controls. We employ two methodologies for estimating the costs of unidentified future controls: a fixed-cost methodology and a hybrid methodology; both approaches assume either that existing technologies can be applied in particular combinations or to specific sources that we currently can't predict or that innovative strategies and new control options make possible the emissions reductions needed for attainment by 2020. The two approaches, however, implicitly reflect different assumptions about technological progress and innovation in emissions reductions strategies. The fixed-cost methodology uses a \$15,000/ton estimate for each ton of $PM_{2.5}$ reduced, and the hybrid methodology generates a total annual cost curve for $PM_{2.5}$ for unknown future controls that might be applied in order to move toward 2020 attainment. The hybrid methodology has the advantage of incorporating information about how significant the needed reductions from unspecified control technologies are relative to the known control measures and matching that information with expected increasing per-ton cost for applying unknown controls. Employing the fixed-cost methodology, approximately 90% of total costs for attaining the revised annual standard of 12 $\mu g/m^3$ are from unspecified control technologies. Employing the hybrid methodology, approximately 97% of total costs for attaining the revised annual standard of 12 $\mu g/m^3$ are from unspecified control technologies. The EPA recognizes that the extrapolated portion of the engineering cost estimates reflects substantial uncertainty about which sectors and which technologies might become available for cost-effective application in the future.

The engineering cost estimates are limited in their scope. Our analysis focuses on the emissions reductions needed for attainment of the revised and alternative standards. Also, the amendments to the ambient air monitoring regulations will revise the network design requirements for $PM_{2.5}$ monitoring sites, resulting in moving 21 monitors to established near-road monitoring stations by January 1, 2015. The incremental cost associated with moving these 21 monitors is a one-time cost of $28,570. Lastly, the EPA understands that some States will incur costs designing SIPs and implementing new control strategies to meet the revised standard. However, the EPA does not know what specific actions States will take to design their SIPs to meet the revised standards; therefore, we do not include estimated costs that government agencies may incur for managing the requirement, implementing these (or other) control strategies, or for offering incentives that may be necessary to encourage the implementation of specific technologies, especially for technologies that are not necessarily market driven.

ES.2.4 Comparison of Benefits and Costs

The EPA's illustrative analysis has estimated the health and welfare benefits and costs associated with the revised annual PM NAAQS. The results for 2020 suggest there will be significant health and welfare benefits and these benefits will outweigh the costs associated with the illustrative control strategies in 2020. In the analysis, we estimate the net benefits of the revised annual $PM_{2.5}$ standard of 12 µg/m^3 and alternative annual standards of 13 µg/m^3 and 11 µg/m^3, incremental to the 2020 analytical baseline. For the revised annual standard of 12 µg/m^3, net benefits are estimated to be $3.7 billion to $9 billion at a 3% discount rate and $3.3 billion to $8.1 billion at a 7% discount rate in 2020 (2010 dollars). For an alternative annual standard of 13 µg/m^3, net benefits are estimated to be $1.2 billion to $2.9 billion at the 3% discount rate and $1.1 billion to $2.6 billion at the 7% discount rate. Net benefits of an alternative annual $PM_{2.5}$ standard of 11 µg/m^3 are estimated to be $11 billion to $29 billion at a 3% discount rate and $10 billion to $26 billion at a 7% discount rate in 2020. See Table ES-2.

For the revised annual standard of 12 µg/m^3, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 12 to 171 times at a 3% discount rate. For the alternative annual standard of 13 µg/m^3, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 13 to 272 times at a 3% discount rate. For the alternative annual standards of 11 µg/m^3, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 8 to 90 times at a 3% discount rate.

Table ES-2. Total Monetized Benefits, Total Costs, and Net Benefits in 2020 (millions of 2010$)—Full Attainment[a]

Alternative Annual Standard ($\mu g/m^3$)	Total Costs[b]		Monetized Benefits[d]		Net Benefits	
	3% Discount Rate[c]	7% Discount Rate	3% Discount Rate	7% Discount Rate	3% Discount Rate[b]	7% Discount Rate
13	$11 to $100	$11 to $100	$1,300 to $2,900	$1,200 to $2,600	$1,200 to $2,900	$1,100 to $2,600
12	$53 to $350	$53 to 350	$4,000 to $9,100	$3,600 to $8,200	$3,700 to $9,000	$3,300 to $8,100
11	$320 to $1,700	$320 to $1,700	$13,000 to $29,000	$12,000 to $26,000	$11,000 to $29,000	$10,000 to $26,000

[a] These estimates reflect incremental emissions reductions from an analytical baseline that gives an "adjustment" to the San Joaquin and South Coast areas in California for NO_x emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

[b] The two cost estimates do not represent lower- and upper-bound estimates but represent estimates generated by two different methodologies. The lower estimate is generated using the fixed-cost methodology, which assumes that technological change and innovation will result in the availability of additional controls by 2020 that are similar in cost to the higher end of the cost range for current, known controls. The higher estimate is generated using the hybrid methodology, which assumes that while additional controls may become available by 2020, they become available at an increasing cost and the increasing cost varies by geographic area and by degree of difficulty associated with obtaining the needed emissions reductions.

[c] Due to data limitations, we were unable to discount compliance costs for all sectors at 3%. See Chapter 7, Section 7.2.2 for additional details on the data limitations. As a result, the net benefit calculations at 3% were computed by subtracting the costs at 7% from the monetized benefits at 3%.

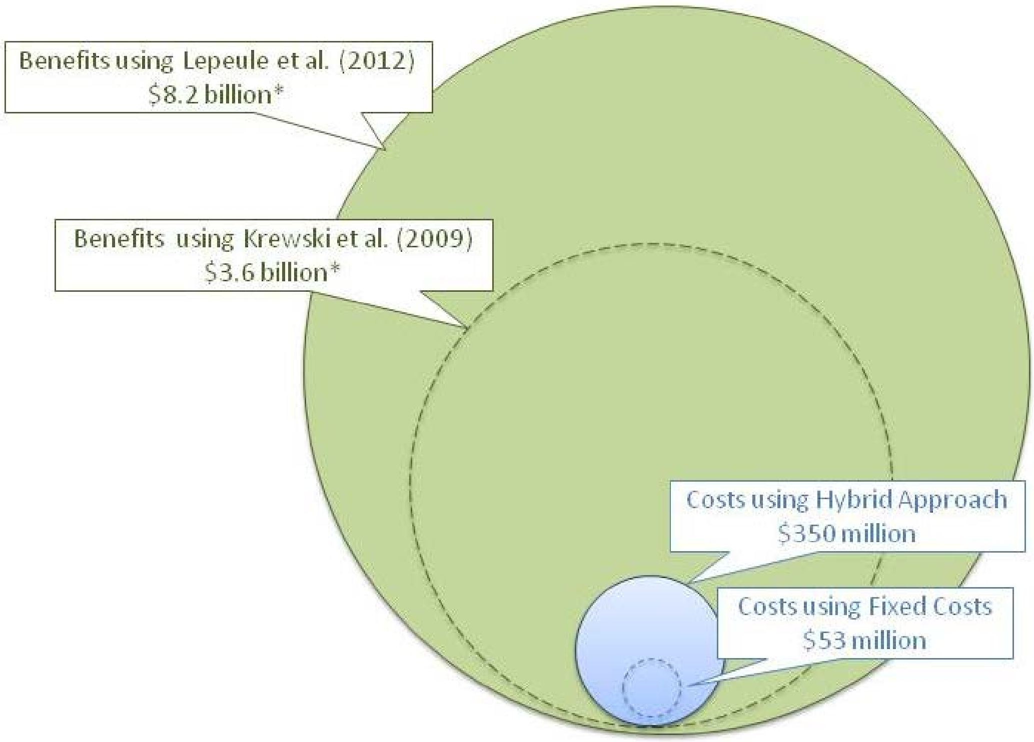
[d] The reduction in premature deaths each year accounts for over 90% of total monetized benefits. Mortality risk valuation assumes discounting over the SAB-recommended 20-year segmented lag structure. Not all possible benefits or disbenefits are quantified and monetized in this analysis. B is the sum of all unquantified benefits. Data limitations prevented us from quantifying these endpoints, and as such, these benefits are inherently more uncertain than those benefits that we were able to quantify. The range of benefits reflects the range of the central estimates from two mortality cohort studies (i.e., Krewski et al., 2009 and Lepeule et al., 2012).

Table ES-3. Benefit-to-Cost Ratios for Alternative Standards at 3% and 7% Based on Projected Benefits and Costs in 2020

	13 $\mu g/m^3$	12 $\mu g/m^3$	11 $\mu g/m^3$
Benefit-Cost Ratio 3%[a]	13 to 272	12 to 171	8 to 90
Benefit-Cost Ratio 7%	11 to 246	11 to 154	7 to 81

[a] Due to data limitations, we were unable to discount compliance costs for all sectors at 3%. See Chapter 7, Section 7.2.2 for additional details on the data limitations. As a result, the net benefit calculations at 3% were computed by subtracting the costs at 7% from the monetized benefits at 3%.

Figure ES-4 reflects the range of costs based on the calculation of costs using the fixed-cost approach and the hybrid approach. Additionally, we see the difference in the calculation of benefits based on using various studies.



Note: Relative size of benefits and costs are to scale.

Figure ES-4. Monetized Benefit to Cost Comparison for 12 μg/m^3 in 2020 (7% Discount Rate)

Table ES-4. Estimated Number of Avoided $PM_{2.5}$ Health Impacts for Standard Alternatives—Full Attainment in 2020[a]

Health Effect	Alternative Annual Standards		
	13 µg/m^3	12 µg/m^3	11 µg/m^3
Adult Mortality			
Krewski et al. (2009) (adult)	140	460	1,500
Lepeule et al. (2012) (adult)	330	1,000	3,300
Woodruff et al. (1997) (infant)	0	1	4
Non-Fatal Heart Attacks (age >18)			
Peters et al. (2001)	160	480	1,600
Pooled estimate of 4 studies	17	52	170
Hospital admissions—respiratory (all ages)	31	110	380
Hospital admissions—cardiovascular (age > 18)	43	140	480
Emergency department visits for asthma (all ages)	67	230	810
Acute bronchitis (age 8–12)	280	870	2,700
Lower respiratory symptoms (age 7–14)	3,500	11,000	34,000
Upper respiratory symptoms (asthmatics age 9–11)	5,100	16,000	49,000
Asthma exacerbation (age 6–18)	13,000	40,000	120,000
Lost work days (age 18–65)	22,000	71,000	230,000
Minor restricted-activity days (age 18–65)	130,000	420,000	1,300,000

[a] Incidence estimates are rounded to whole numbers with no more than two significant figures.

ES.3 Discussion and Conclusions

An extensive body of scientific evidence documented in PM ISA indicates that $PM_{2.5}$ can penetrate deep into the lungs and cause serious health effects, including premature death and other non-fatal illnesses (U.S. EPA, 2009). As described in the preamble to the rule, the revisions to the standards are based on an integrative assessment of an extensive body of new scientific evidence (U.S. EPA, 2009). Health studies published since the PM ISA (e.g., Pope et al. [2009]) confirm that recent levels of $PM_{2.5}$ have had a significant impact on public health. Based on the air quality analysis in this RIA, the EPA projects that nearly all counties with $PM_{2.5}$ monitors in the United States would meet an annual standard of 12 µg/m^3 without additional Federal, State, or local PM control programs. This demonstrates the substantial progress that the United States has made in reducing air pollution emissions over the last several decades.

Regulations such as the EPA's recent Mercury and Air Toxics Standards (MATS) and other Federal programs such as diesel standards will provide substantial improvements in regional concentrations of $PM_{2.5}$. Our analysis shows a few areas would still need additional emissions reductions to address local sources of air pollution, including ports and uncontrolled industrial emissions. For this reason, we have designed the RIA analysis to focus on local controls in these few areas. We estimate that these additional local controls would yield benefits well in excess of costs.

The setting of a NAAQS does not compel specific pollution reductions and as such does not directly result in costs or benefits. For this reason, NAAQS RIAs are merely illustrative. The NAAQS RIAs illustrate the potential costs and benefits of additional steps States could take to attain a revised air quality standard nationwide beyond rules already on the books. We base our illustrative estimates on an array of emission control strategies for different sources. The costs and benefits identified in this RIA will not be realized until specific controls are mandated by SIPs or other Federal regulations. In short, NAAQS RIAs hypothesize, but do not prescribe, the control strategies that States may choose to enact when implementing a revised NAAQS.

It is important to emphasize that the EPA does not "double count" the costs or the benefits of our rules. Emission reductions achieved under rules that require specific actions from sources—such as MATS—are in the baseline of this NAAQS analysis, as are emission reductions needed to meet the current NAAQS. For this reason, the cost and benefits estimates provided in this RIA and all other NAAQS RIAs should not be added to the estimates for implementation rules.

In calculating the costs, the EPA assumed the application of a significant number of unidentified future controls that would make possible the additional emissions reductions needed for attainment in 2020. EPA used two methodologies—the fixed-cost and hybrid methodologies—for estimating the costs of unidentified future controls, and both approaches assume either that existing technologies can be applied in particular combinations or to specific sources that we currently can't predict or that innovative strategies and new control options make possible the emissions reductions needed for attainment by 2020. Estimates generated by the two approaches do not represent lower- and upper-bound estimates but simply represent estimates generated by two different methodologies. The fixed-cost methodology implicitly assumes that technological change and innovation will result in the availability of additional controls by 2020 that are similar in cost to the higher end of the cost range for current controls. The hybrid methodology implicitly assumes that while additional controls become available by 2020, they become available at an increasing cost and the increasing cost varies by geographic area and by degree of difficulty associated with obtaining the needed emissions reductions.

For the revised annual standard of 12 μg/m^3, the total cost estimates comprise between 90 and 97% extrapolated cost estimates, and the estimated total cost using the hybrid methodology is roughly 6.5 times more than the estimated total cost using the fixed-cost methodology. Because the hybrid methodology reflects increasing marginal costs in areas needing a higher ratio of emissions reductions from unknown to known controls, it could be more representative of total costs. In an effort to consider the potential fitness of the extrapolated cost estimates, we reviewed the South Coast Air Quality Management District's (SCAQMD) 2012 Air Quality Management Plan (AQMP), and we located data on recent emission reduction credit (ERC) transactions in both the SCAQMD and San Joaquin Valley Air Pollution Control District (SJV APCD). While this information provides context for the extrapolated cost estimates, the current relationship between available controls and costs to reduce emissions may or may not be applicable in 2020 because of changes in innovation and advances in technology.

The SCAQMD's 2012 AQMP includes information on control measures to meet the current 24-hour standard of 35 μg/m^3, including further $PM_{2.5}$ controls for under-fired charbroilers at a cost per ton reduced of $15,000. This control cost matches the parameter used in the fixed-cost methodology, as well as the initial value used for the hybrid methodology and is supportive of our selection of that value. In addition, the California Air Resources Board's 2009 and 2010 *Emission Reduction Offset Transaction Costs, Summary Report* included PM_{10} ERC prices in both the SCAQMD and the SJV APCD. To some degree, ERC transaction prices reflect a choice between installing a more stringent control and purchasing ERCs. Between 2009 and 2010 PM_{10} ERC prices in SJV APCD ranged from $40,000 per ton per year (tpy) to $70,000/tpy, and PM_{10} ERC prices in the SCAQMD ranged from $575,000/tpy to more than $1.9 million/tpy. These prices reflect both marginal costs that are higher than the fixed-cost estimates and marginal costs that are not inconsistent with the higher cost estimates generated using the hybrid methodology. For further discussion of the total cost estimates, refer to Section 7.2.4 in Chapter 7 of this RIA.

Furthermore, the monetized benefits estimates presented in this RIA are not intended to capture the full burden of PM to public health but rather represent the incremental benefits expected upon attaining the revised annual primary standard of 12 μg/m^3. In comparison, modeling by Fann et al. (2012) estimated that 2005 levels of air pollution were responsible for between 130,000 and 320,000 $PM_{2.5}$-related deaths, or between 6.1% and 15% of total deaths from all causes in the continental United States. The monetized benefits associated with attaining the proposed range of standards appear modest when viewed within the context of the potential overall public health burden of $PM_{2.5}$ and ozone air pollution estimated by Fann et al. (2012), but this is primarily because regulations already on the books will make great strides toward reducing future levels of PM. One important distinction between the total public health burden estimated for 2005 air pollution levels and the estimated benefits in this RIA is that ambient levels of $PM_{2.5}$ will have improved substantially by 2020, due to major emissions reductions resulting from implementation of Federal regulations. For example, we estimate that SO_2 emissions (an important $PM_{2.5}$ precursor) in the United States would fall from 14 million tons in 2005 to less than 5 million tons by 2020 (a reduction of 66%). For this reason, States will only need to achieve small air quality improvements to reach the proposed PM standards. As shown in the recent RIA for MATS (U.S. EPA, 2011b), implementing other Federal and State air quality actions will address a substantial fraction of the total public health burden of $PM_{2.5}$ and ozone air pollution.

The NAAQS are not set at levels that eliminate the risk of air pollution. Instead, the Administrator sets the NAAQS at a level requisite to protect public health with an adequate margin of safety, taking into consideration effects on susceptible populations based on the scientific literature. The risk analysis prepared in support of this PM NAAQS reported risks below these levels, while acknowledging that the confidence in those effect estimates is higher at levels closer to the standard (U.S. EPA, 2010c). While benefits occurring below the standard are assumed to be more uncertain than those occurring above the standard, the EPA considers these to be legitimate components of the total benefits estimate. Although there are greater uncertainties at lower $PM_{2.5}$ concentrations, there is no evidence of a population-level threshold in $PM_{2.5}$-related health effects in the epidemiology literature.

Lastly, the EPA recognizes that there are uncertainties in both the cost and benefit estimates provided in this RIA. The EPA was unable to monetize fully all of the benefits associated with reaching these standards in this RIA, including other health effects of PM, visibility effects, ecosystem effects, and climate effects. If the EPA were able to monetize all of the benefits, the benefits would exceed the estimated costs by an even greater margin.

ES.4 Caveats and Limitations

EPA acknowledges several important limitations of this analysis. These include the following:

ES.4.1 Benefits Caveats

- $PM_{2.5}$ mortality benefits represent a substantial proportion of total monetized benefits (over 98%). To characterize the uncertainty in the relationship between $PM_{2.5}$ and premature mortality, we include a set of 12 estimates of the concentration-response function based on results of the $PM_{2.5}$ mortality expert elicitation study in addition to our core estimates. Even these multiple characterizations omit the uncertainty in air quality estimates, baseline incidence rates, populations exposed, chemical composition, transferability of the effect estimate to diverse locations, and additional uncertainty around the mean estimates expressed by the experts. As a result, the reported confidence intervals and range of estimates give an incomplete picture about the overall uncertainty in the $PM_{2.5}$ estimates. This information should be interpreted within the context of the larger uncertainty surrounding the entire analysis.

- Most of the estimated avoided premature deaths occur at or above the lowest measured $PM_{2.5}$ concentration in the two studies used to estimate mortality benefits. In general, we have greater confidence in risk estimates based on $PM_{2.5}$ concentrations where the bulk of the health and air quality data reside and somewhat less confidence where data density is lower.

- We analyzed full attainment in 2020, and projecting key variables introduces uncertainty. Inherent in any analysis of future regulatory programs are uncertainties in projecting atmospheric conditions and source-level emissions, as well as population, health baselines, incomes, technology, and other factors.

- There are uncertainties related to the health impact functions used in the analysis. These include within-study variability; pooling across studies; the application of C-R functions nationwide and for all particle species; extrapolation of impact functions across populations; and various uncertainties in the C-R function, including causality and shape of the function at low concentrations. Therefore, benefits may be under- or over-estimates.

- This analysis omits certain unquantified effects due to lack of data, time, and resources. These unquantified endpoints include other health and ecosystem effects. The EPA will continue to evaluate new methods and models and select those most appropriate for estimating the benefits of reductions in air pollution.

- Full benefits of the revised standards in San Joaquin and South Coast will not be realized until 2025 when those areas are expected to demonstrate attainment with the revised standards. If we were to estimate the monetized benefits for 2025, those benefits would be higher due to population growth, aging of the population, and income growth over time.

ES.4.2 Control Strategy and Cost Analysis Caveats and Limitations

- The control technologies applied as part of the illustrative control strategies represent technologies that are available currently and may not reflect emerging devices that may be available in future years to aid in attainment of the revised standard. In addition, the emission reductions calculated from the known controls (control efficiencies) assume that the control devices are properly installed and maintained. There is also variability in scale of application that is difficult to reflect for small area sources of emissions.

- The illustrative control strategy analysis estimates only one potential pathway to attainment. The control strategies are not recommendations for how the revised $PM_{2.5}$ standard should be implemented, and States will make all final decisions regarding implementation strategies for the revised NAAQS.

- The application of known controls is based on source information obtained from the emissions inventory. To the extent the inventory is lacking data on baseline controls from SIPs, we may analyze control options that are currently in place.

- The future-year emissions used as a basis for developing the control strategies in this RIA have implicit assumptions regarding emissions growth and control, which differ by sector. For some emission sectors, these future-year emissions may not reflect new sources locating in these areas.

- For two areas in California (South Coast Air Quality Management District and San Joaquin Valley Air Pollution Control District) the degree of projected nonattainment with the revised annual standard of 12 $\mu g/m^3$ is such that these areas are not expected to be able to demonstrate attainment with the new standard by 2020. These areas may qualify for up to a 5-year extension of their attainment date and are likely to have until 2025 to demonstrate attainment with the revised annual standard.

- The control technologies applied do not reflect potential effects of technological change that may be available in future years and the effects of "learning by doing" are not accounted for in the emissions reduction estimates. In our analysis, we do not have the necessary data for cumulative output, fuel sales, or emission reductions for all sectors included in order to properly generate control costs that reflect learning-curve impacts. We believe the effect of including these impacts would be to lower our estimates of costs for our control strategies in 2020.

- In addition to the application of known controls, the EPA assumes the application of unidentified future controls that make possible the additional emission reductions needed for attainment in 2020. By definition, no cost data currently exist for unidentified future technologies or innovative strategies and the cost estimates for unidentified future controls reflect some uncertainty.

- Because we obtain control cost data from many sources, we are not always able to obtain consistent data across original data sources. If disaggregated control cost data are unavailable (i.e., where capital, equipment life value, and operation and maintenance [O&M] costs are not separated out), the EPA typically assumes that the estimated control costs are annualized using a 7% discount rate. When disaggregated control cost data are available (i.e., where capital, equipment life value, and O&M costs are explicit), we can recalculate costs using a 3% discount rate. In general, we have some disaggregated data available for non-EGU point source controls, and we do not have any disaggregated control cost data for area source controls. In this analysis, for the revised annual standard of 12 µg/m^3 and the alternative standard of 13 µg/m^3 we did not have any disaggregated known control cost data; therefore, we were not able to recalculate known control costs using a 3% discount rate.

- The EPA understands that some States will incur costs designing SIPs and implementing new control strategies to meet the revised annual standard. However, the EPA does not know what specific actions States will take to design their SIPs to meet the revised standards; therefore we do not present estimated costs that government agencies may incur for managing the requirement, implementing these (or other) control strategies, or for offering incentives that may be necessary to encourage the implementation of specific technologies, especially for technologies that are not necessarily market driven.

ES.5 Important Updates and Analytical Differences Between the PM NAAQS Proposal RIA and the Final RIA

There have been several major improvements in the analytical components the EPA used to estimate benefits and costs between the proposal RIA (June 2012) and this RIA accompanying the final PM NAAQS. Important updates to our emissions, air quality modeling and ambient data, air quality ratios, population projections, as well as currency year valuation resulted in an improved analytical base for our analysis for the final rule. Based on the complexity and magnitude of the updates and improvements made between the PM RIA proposal and final RIA, it would not be appropriate to perform a simple direct comparison of results. Therefore, each analysis stands alone and must be evaluated independently as such.

Below is a list summarizing some of the analytical changes between the proposal RIA and the final RIA. Between the proposal RIA and the final RIA, we developed the control strategies based on an improved modeling platform and updated the approach in designing the control strategies. The improved modeling platform updated the current and projected air quality levels for each area across the nation and the updated approach allowed for more effective emissions reductions for each area in both attaining the current annual and daily standards for our analytical baseline and in attaining the revised annual standard. For example, in the proposal RIA we controlled precursor emissions of NO_x and SO_2 over a broader region and these emissions reductions were not as effective in reducing design value for each area as the direct $PM_{2.5}$ emissions reductions targeted in the final RIA. These analytical improvements resulted in different estimates of costs and benefits between the proposal RIA and the final RIA in which we have more confidence in reflecting the approaches that States will pursue to attain the current and revised standards.

- ***New modeling platform***—In the modeling platform for the final rule we included key updates to the current ambient data that generally show improved air quality when compared to modeling for the proposed rule, although daily design values (DVs) for some areas in California increased due largely to wildfires in 2008. To address the increases in these areas we adjusted the ambient data for these atypical events.

- ***Future air quality with "on-the-books" controls***—We also project $PM_{2.5}$ air quality levels between 2007 and the 2020 base case which were generally lower in the final RIA compared to the proposal RIA largely because the starting values for the ambient data were lower (i.e., final RIA air quality projections showed more improvements than proposal RIA).

- ***Analytical baseline with attainment of 15/35***—For the final RIA, EPA's approach to attaining the existing standards of 15/35 $\mu g/m^3$ was improved with "air quality adjustment ratios" that were based on more focused sensitivity model runs for (i) specific areas like California counties; (ii) influential sources like residential wood combustion; and (iii) specific PM emission species like directly emitted $PM_{2.5}$. In the proposal RIA, we conducted the analysis using more general air quality ratios that reflected multiple sources such as point and area and precursor PM emissions like NO_x and SO_2. As a result, in the final RIA, the daily standard of 35 $\mu g/m^3$ was attained more effectively and had less impact on annual DVs because of episodic, direct $PM_{2.5}$ reductions, while in the proposal RIA, the daily standard was attained less effectively because we pursued year-around NO_x and SO_2 reductions that necessitated more emissions reductions and had more impact on the annual DVs.

- ***Incremental air quality changes***—In the final RIA, the use of improved air quality adjustment ratios resulted in more incremental air quality improvement needed to attain the annual standard of 12 $\mu g/m^3$ in California. Thus, these larger incremental air quality improvements needed to attain the revised standard resulted in higher estimated health benefits in the final rule compared to the proposal RIA. As described above, this is because the daily standard of 35 $\mu g/m^3$ was attained more effectively and had less impact on annual DVs across the counties in California. In addition, the focus on direct $PM_{2.5}$ emissions reductions allowed for more effective controls to attain the revised standard with fewer $PM_{2.5}$ emissions reductions thereby resulting in similar fixed costs estimates.[5]

- ***Benefits analysis***—EPA incorporated several updates that affected the core benefits estimates in this RIA. Specifically, the EPA incorporated the most recent follow-up to the Harvard Six Cities cohort study (Lepeule et al., 2012), which decreases the high end of the monetized benefits range by 13%. The EPA also updated the demographic data projections to reflect the 2010 Census, which increased the monetized benefits by 4% percent for the revised standard. Additional epidemiology studies for hospitalizations and emergency department visits and updated survival rates for non-fatal heart attacks did not affect the rounded benefits estimates.

- ***Cost estimates***—In the final RIA, the EPA presents a range of costs using both the fixed and hybrid methodologies to estimate the costs associated with unknown controls.

- ***Inflation***—The EPA updated the currency year in this RIA to use 2010 dollars, which increased both the costs and the monetized benefits by approximately 8% since the proposal.

ES.6 References

Abt Associates, Inc. 2012. *BenMAP User's Manual Appendices*. Prepared for U.S. Environmental Protection Agency Office of Air Quality Planning and Standards. Research Triangle Park, NC. September.

Fann, N., A. Lamson, K. Wesson, D. Risley, S.C. Anenberg, and B.J. Hubbell. 2012. "Estimating the National Public Health Burden Associated with Exposure to Ambient $PM_{2.5}$ and Ozone. Risk Analysis." *Risk Analysis* 32(1): 81-95.

Krewski, D., R.T. Burnett, M.S. Goldbert, K. Hoover, J. Siemiatycki, M. Jerrett, M. Abrahamowicz, and W.H. White. 2009. "Reanalysis of the Harvard Six Cities Study and the American Cancer Society Study of Particulate Air Pollution and Mortality." Special Report to the Health Effects Institute. Cambridge, MA. July.

Lepeule, J., F. Laden, D. Dockery, and J. Schwartz. 2012. "Chronic Exposure to Fine Particles and Mortality: An Extended Follow-Up of the Harvard Six Cities Study from 1974 to 2009." *Environ Health Perspect.* In press. Available at: http://dx.doi.org/10.1289/ehp.1104660.

Peters, A., D. W. Dockery, J. E. Muller and M. A. Mittleman. 2001. "Increased particulate air pollution and the triggering of myocardial infarction." *Circulation* 103(23): 2810-5.

[5] The hybrid methodology cost estimates increased between the proposal RIA and the final RIA largely because a large amount of emissions reductions were needed from one county with a low amount of known controls.

Pope, C.A. III, E. Majid, and D. Dockery. 2009. “Fine Particle Air Pollution and Life Expectancy in the United States.” *New England Journal of Medicine* 360: 376-386.

U.S. Environmental Protection Agency (U.S. EPA). 1997. *Regulatory Impact Analyses for the Particulate Matter and Ozone National Ambient Air Quality Standards and Proposed Regional Haze Rule*. Available at: http://www.epa.gov/ttn/oarpg/naaqsfin/ria.html.

U.S. Environmental Protection Agency (U.S. EPA). 1999. *Regulatory Impact Analysis—Control of Air Pollution from New Motor Vehicles: Tier 2 Motor Vehicle Emissions Standards and Gasoline Sulfur Control Requirements*. EPA 420-R-99-023. Available at: http://www.epa.gov/otaq/regs/ld-hwy/tier-2/documents/r99023.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2000. *Regulatory Impact Analysis: Control of Emissions of Air Pollution from Highway Heavy-Duty Engines*. EPA 420-R-00-010. Available at: http://www.epa.gov/otaq/regs/hd-hwy/2000frm/r00010.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2004. *Final Regulatory Analysis: Control of Emissions from Nonroad Diesel Engines*. EPA420-R-04-007. Available at: http://www.epa.gov/otaq/documents/nonroad-diesel/420r04007.pdf.

U.S. Environmental Protection Agency (U.S. EPA). November 17, 2005a. *Federal Register*. Vol. 70, No. 221. “Control of Air Pollution From Aircraft and Aircraft Engines; Emission Standards and Test Procedures.” 40 CFR Part 87. Available at: http://www.epa.gov/fedrgstr/EPA-AIR/2005/November/Day-17/a22704.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2005b. *Regulatory Impact Analysis for the Final Clean Air Visibility Rule or the Guidelines for Best Available Retrofit Technology (BART) Determinations Under the Regional Haze Regulations*. EPA-452/R-05-004. Available at: http://www.epa.gov/oar/visibility/pdfs/bart_ria_2005_6_15.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2006. *Regulatory Impact Analysis, 2006 National Ambient Air Quality Standards for Particulate Matter, Chapter 5*. Office of Air Quality Planning and Standards, Research Triangle Park, NC. October. Available at: http://www.epa.gov/ttn/ecas/regdata/RIAs/Chapter%205—Benefits.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2008a. *Regulatory Impact Analysis: Control of Emissions of Air Pollution from Locomotive Engines and Marine Compression Ignition Engines Less than 30 Liters Per Cylinder*. Available at: http://www.regulations.gov/#!documentDetail;D=EPA-HQ-OAR-2003-0190-0938.

U.S. Environmental Protection Agency (U.S. EPA). 2008b. *Control of Emissions from Marine SI and Small SI Engines, Vessels, and Equipment. Final Regulatory Impact Analysis*. Available at: http://www.regulations.gov/#!documentDetail;D=EPA-HQ-OAR-2004-0008-0929.

U.S. Environmental Protection Agency (U.S. EPA). 2009a. *Economic Impacts of Revised MACT Standards for Hospital/Medical/Infectious Waste Incinerators.* Available at: http://www.epa.gov/ttn/atw/129/hmiwi/hmiwi_eia_090803.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2009b. *Integrated Science Assessment for Particulate Matter (Final Report)*. EPA-600-R-08-139F. National Center for Environmental Assessment—RTP Division. December. Available at: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=216546.

U.S. Environmental Protection Agency (U.S. EPA). 2010a. *Regulatory Impact Analysis: Control of Emissions of Air Pollution from Category 3 Marine Diesel Engines*. Available at: http://www.epa.gov/oms/regs/nonroad/marine/ci/420r09019.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2010b. *Regulatory Impact Analysis: Regulatory Impact Analysis (RIA) for Existing Stationary Spark Ignition (SI) RICE NESHAP.* Available at: http://www.epa.gov/ttn/atw/rice/fnl_si_rice_ria.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2010c*. Quantitative Health Risk Assessment for Particulate Matter—Final Repor*t. EPA-452/R-10-005. Office of Air Quality Planning and Standards, Research Triangle Park, NC. September. Available on the Internet at <http://www.epa.gov/ttnnaaqs/standards/pm/data/PM_RA_FINAL_June_2010.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2011a. *Regulatory Impact Analysis for the Federal Implementation Plans to Reduce Interstate Transport of Fine Particulate Matter and Ozone in 27 States; Correction of SIP Approvals for 22 States*. June. Available at: http://www.epa.gov/airtransport/pdfs/FinalRIA.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011b. *Regulatory Impact Analysis for the Final Mercury and Air Toxics Standards.* EPA-452/R-11-011. December. Available at: http://www.epa.gov/ttn/ecas/regdata/RIAs/matsriafinal.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011c. *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress*. Office of Air and Radiation, Office of Policy, Washington, DC. March. Available at: http://www.epa.gov/oar/sect812/feb11/fullreport.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011d. *Regulatory Impact Analysis: National Emission Standards for Hazardous Air Pollutants for Industrial, Commercial, and Institutional Boilers and Process Heaters.* February. Available at: http://www.epa.gov/ttnecas1/regdata/RIAs/boilersriafinal110221_psg.pdf.

U.S. Office of Management and Budget. Circular A-4, September 17, 2003. Available at: http://www.whitehouse.gov/omb/circulars/a004/a-4.pdf.

Woodruff, T.J., J. Grillo, and K.C. Schoendorf. 1997. "The Relationship Between Selected of postneonatal infant mortality and particulate air pollution in the United States." *Environmental Health Perspectives* 105(6): 608-612.

CHAPTER 1
INTRODUCTION AND BACKGROUND

1.1 Synopsis

This chapter summarizes the purpose and results of this Regulatory Impact Analysis (RIA). This RIA estimates the human health and welfare benefits and costs of attaining the revised and two alternative annual particulate matter (PM) National Ambient Air Quality Standards (NAAQS) nationwide. According to the Clean Air Act ("the Act"), the Environmental Protection Agency (EPA) must use health-based criteria in setting the NAAQS and cannot consider estimates of compliance cost. The EPA is producing this RIA both to provide the public a sense of the benefits and costs of meeting a revised annual PM NAAQS and to meet the requirements of Executive Orders 12866 and 13563.

1.2 Background

1.2.1 NAAQS

Two sections of the Act govern the establishment and revision of NAAQS. Section 108 (42 U.S.C. 7408) directs the Administrator to identify pollutants that "may reasonably be anticipated to endanger public health or welfare" and to issue air quality criteria for them. These air quality criteria are intended to "accurately reflect the latest scientific knowledge useful in indicating the kind and extent of all identifiable effects on public health or welfare which may be expected from the presence of [a] pollutant in the ambient air." PM is one of six pollutants for which the EPA has developed air quality criteria.

Section 109 (42 U.S.C. 7409) directs the Administrator to propose and promulgate "primary" and "secondary" NAAQS for pollutants identified under section 108. Section 109(b)(1) defines a primary standard as "the attainment and maintenance of which in the judgment of the Administrator, based on [the] criteria and allowing an adequate margin of safety, [are] requisite to protect the public health." A secondary standard, as defined in section 109(b)(2), must "specify a level of air quality the attainment and maintenance of which in the judgment of the Administrator, based on [the] criteria, [are] requisite to protect the public welfare from any known or anticipated adverse effects associated with the presence of [the] pollutant in the ambient air." Welfare effects as defined in section 302(h) [42 U.S.C. 7602(h)] include but are not limited to "effects on soils, water, crops, vegetation, manmade materials, animals, wildlife, weather, visibility and climate, damage to and deterioration of property, and hazards to transportation, as well as effects on economic values and on personal comfort and well-being."

Section 109(d) of the Act directs the Administrator to review existing criteria and standards at 5-year intervals. When warranted by such review, the Administrator is to retain or revise the NAAQS. After promulgation or revision of the NAAQS, the standards are implemented by the States.

1.2.2 2006 PM NAAQS

In 2006, the EPA's final PM rule established a 24-hour standard of 35 μg/m^3 and retained the annual standard of 15 μg/m^3. The EPA revised the secondary standards for fine particles by making them identical in all respects to the primary standards. Following promulgation of the final rule in 2006, several parties filed petitions for its review. On February 24, 2009, the U.S. Court of Appeals for the District of Columbia Circuit remanded the primary annual $PM_{2.5}$ NAAQS to the EPA citing that the EPA failed to adequately explain why the standards provided the requisite protection from both short- and long-term exposures to fine particles, including protection for at-risk populations. The court remanded the secondary standards to the EPA citing that the Agency failed to adequately explain why setting the secondary PM standards identical to the primary standards provided the required protection for public welfare, including protection from visibility impairment. In 2006, the EPA also retained the primary and secondary 24-hour PM_{10} standards, both set at a level of 150 μg/m^3, not to be exceeded more than once per year on average over 3 years (U.S. EPA, 1997).

1.3 Role of this RIA in the Process of Setting the NAAQS

1.3.1 Legal Requirement

In setting primary ambient air quality standards, the EPA's responsibility under the law is to establish standards that protect public health, regardless of the costs of implementing those new standards. The Act requires the EPA, for each criteria pollutant, to set standards that protect public health with "an adequate margin of safety." As interpreted by the Agency and the courts, the Act requires the EPA to create standards based on health considerations only.

The prohibition against the consideration of cost in the setting of the primary air quality standards, however, does not mean that costs or other economic considerations are unimportant or should be ignored. The Agency believes that consideration of costs and benefits is essential to making efficient, cost-effective decisions for implementing these standards. The impact of cost and efficiency is considered by States during this process, as they decide what timelines, strategies, and policies make the most sense. This RIA is intended to inform the public about the potential costs and benefits that may result when new standards are implemented, but it is not relevant to establishing the standards themselves.

1.3.2 Role of Statutory and Executive Orders

This RIA is separate from the NAAQS decision-making process, but several statutes and executive orders still apply to any public documentation. The analysis required by these statutes and executive orders is presented in Chapter 9.

The EPA presents this RIA pursuant to Executive Orders 12866 and 13563 and the guidelines of Office of Management and Budget (OMB) Circular A-4.[1] In accordance with these guidelines, the RIA analyzes the benefits and costs associated with emissions controls to attain the revised annual PM standard, incremental to attainment of the existing standards. In addition, this RIA analyses two alternative primary annual $PM_{2.5}$ standards: one that is more stringent than the existing standards but less stringent than the revised annual standard and another that is more stringent than the revised annual standard.

In the current PM NAAQS review, the EPA is revising and lowering the level of the primary annual $PM_{2.5}$ standard from 15 $\mu g/m^3$ to 12 $\mu g/m^3$ in conjunction with retaining the level of the 24-hour standard at 35 $\mu g/m^3$. Thus, the incremental benefits and costs analyzed in this RIA result from emissions controls needed to attain a more protective annual standard, rather than the 24-hour standard of 35 $\mu g/m^3$. In addition to the revised annual standard of 12 $\mu g/m^3$, the RIA also analyzes the benefits and costs of incremental control strategies for two alternative annual standards (13 $\mu g/m^3$ and 11 $\mu g/m^3$).

The control strategies presented in this RIA are illustrative and represent one set of control strategies States might choose to implement in order to meet the final standards. As a result, benefit and cost estimates provided in the RIA are cannot be added to benefits and costs from other regulations, and, further, the costs and benefits identified in this RIA will not be realized until specific controls are mandated by State Implementation Plans (SIPs) or other Federal regulations.

The analytical baseline for this analysis does not assume emissions controls that might be implemented to meet the other NAAQS for O_3, NO_x, or SO_2. To the extent that some of the estimated emissions reductions needed to meet the revised annual PM standard would be needed to meet the current standards for O_3, NO_x, or SO_2, the costs and benefits of meeting the revised PM annual PM standard will be overstated. We did not conduct this analysis incremental to controls applied as part of previous NAAQS analyses (e.g., O_3, NO_x, or SO_2)

[1] U.S. Office of Management and Budget. Circular A-4, September 17, 2003, available at <http://www.whitehouse.gov/omb/circulars/a004/a-4.pdf>.

because the data and modeling on which these previous analyses were based are now considered outdated and are not compatible with the current $PM_{2.5}$ NAAQS analysis.

1.3.3 The Need for National Ambient Air Quality Standards

OMB Circular A-4 indicates that one of the reasons a regulation such as the NAAQS may be issued is to address existing "externalities." An externality occurs when parties to a transaction do not bear its full consequences. An environmental problem, such as pollution generated from production of a good, which imposes health costs on those who neither produce nor consume it, is a classic case of an externality. In the presence of externalities, a free market does not ensure an efficient allocation of resources. Setting and implementing primary and secondary air quality standards is one way the government can address an externality and increase air overall public health and welfare.

1.3.4 Illustrative Nature of the Analysis

This NAAQS RIA is an illustrative analysis that provides useful insights into a limited number of emissions control scenarios that States might implement to achieve revised NAAQS. Because States are ultimately responsible for implementing strategies to meet any revised standard, the control scenarios in this RIA are necessarily hypothetical in nature. Important uncertainties and limitations are documented in the relevant portions of the analysis.

Because the illustrative goals of this RIA are somewhat different from other EPA analyses of national rules or the implementation plans States develop, the distinctions are worth brief mention. This RIA does not assess the regulatory impact of an EPA-prescribed national rule, nor does it attempt to model the specific actions that any State would take to implement a revised standard. This analysis attempts to estimate the costs and human and welfare benefits of cost-effective implementation strategies that might be undertaken to achieve national attainment of new standards. These hypothetical strategies represent a scenario where States use one set of cost-effective controls to attain a revised NAAQS. Because States—not the EPA—will implement any revised NAAQS, they will ultimately determine appropriate emissions control scenarios. SIPs would likely vary from the EPA's estimates due to differences in the data and assumptions that States use to develop these plans.

The illustrative attainment scenarios presented in this RIA were constructed with the understanding that there are inherent uncertainties in projecting emissions and controls.

1.4 Overview and Design of the Final RIA

The RIA evaluates the costs and benefits of hypothetical national strategies to attain the revised annual standard of 12 µg/m^3 and two alternative annual PM standards, incremental to attainment of the existing 15/35 µg/m^3 standards.

The EPA is retaining the current primary and secondary 24-hour PM_{10} standards, which are both set at a level of 150 µg/m^3, not to be exceeded more than once per year on average over 3 years (U.S. EPA, 1997). Because the benefit-cost analysis of the alternative PM_{10} standards was conducted when the standard was promulgated in 1997, this RIA does not repeat that analysis here.

1.4.1 Important Updates and Differences Between the PM NAAQS Proposal RIA and the Final RIA

There have been several major improvements in the analytical components the EPA used to estimate benefits and costs between the proposal RIA (June 2012) and this RIA accompanying the final PM NAAQS. Important updates to our emissions, air quality modeling and ambient data, air quality ratios, population projections, and currency-year valuation resulted in an improved analytical base for our analysis for the final rule. Based on the complexity and magnitude of the updates and improvements made between the PM RIA proposal and final RIA, it would not be appropriate to perform a simple direct comparison of results. Therefore, each analysis stands alone and must be evaluated independently as such.

Below is a list summarizing some of the analytical changes between the proposal RIA and the final RIA. Between the proposal RIA and the final RIA, we developed the control strategies based on an improved modeling platform and updated the approach in designing the control strategies. The improved modeling platform updated the current and projected air quality levels for each area across the nation and the updated approach allowed for more effective emissions reductions for each area in both attaining the current annual and daily standards for our analytical baseline and in attaining the revised annual standard. For example, in the proposal RIA we controlled precursor emissions of NO_x and SO_2 over a broader region and these emissions reductions were not as effective in reducing design value for each area as the direct $PM_{2.5}$ emissions reductions targeted in the final RIA. These analytical improvements resulted in different estimates of costs and benefits between the proposal RIA and the final RIA which we have more confidence in reflecting the approaches that states will pursue to attain the current and revised standards.

- ***New modeling platform***—In the modeling platform for the final rule we included key updates to the current ambient data that generally show improved air quality when compared to modeling for the proposed rule, although daily design values (DVs) for some areas in California increased due largely to wildfires in 2008. To address the increases in these areas we adjusted the ambient data for these atypical events.

- ***Future air quality with "on-the-books" controls***—We also project $PM_{2.5}$ air quality levels between 2007 and the 2020 base case which were generally lower in the final RIA compared to the proposal RIA largely because the starting values for the ambient data were lower (i.e., final RIA air quality projections showed more improvements than proposal RIA).

- ***Analytical baseline with attainment of 15/35***—For the final RIA, the EPA's approach to attaining the existing standards of 15/35 $\mu g/m^3$ was improved with "air quality adjustment ratios" that were based on more focused sensitivity model runs for (i) specific areas like California counties; (ii) influential sources like residential wood combustion; and (iii) specific PM emission species like directly emitted $PM_{2.5}$. In the proposal RIA, we conducted the analysis using more general air quality ratios that reflected multiple sources such as point and area and precursor PM emissions like NO_x and SO_2. As a result, in the final RIA, the daily standard of 35 $\mu g/m^3$ was attained more effectively and had less impact on annual DVs because of episodic, direct $PM_{2.5}$ reductions, while in the proposal RIA, the daily standard was attained less effectively because we pursued year-around NO_x and SO_2 reductions that necessitated more emissions reductions and had more impact on the annual DVs.

- ***Incremental air quality changes—***In the final RIA, the use of improved air quality adjustment ratios resulted in more incremental air quality improvement needed to attain the annual standard of 12 $\mu g/m^3$ in California. Thus, these larger incremental air quality improvements needed to attain the revised standard resulted in higher estimated health benefits in the final rule compared to the proposal RIA. As described above, this is because the daily standard of 35 $\mu g/m^3$ was attained more effectively and had less impact on annual DVs across the counties in California. In addition, the focus on direct $PM_{2.5}$ emissions reductions allowed for more effective controls to attain the revised standard with fewer $PM_{2.5}$ emissions reductions thereby resulting in similar fixed costs estimates.[2]

- ***Benefits analysis***—The EPA incorporated several updates that affected the core benefits estimates in this RIA. Specifically, the EPA incorporated the most recent follow-up to the Harvard Six Cities cohort study (Lepeule et al., 2012), which decreases the high end of the monetized benefits range by 13%. The EPA also updated the demographic data projections to reflect the 2010 Census, which

[2] The hybrid methodology cost estimates increased between the proposal RIA and the final RIA largely because a large amount of emissions reductions were needed from one county with a low amount of known controls.

increased the monetized benefits by 4% percent for the revised standard. Additional epidemiology studies for hospitalizations and emergency department visits and updated survival rates for non-fatal heart attacks did not affect the rounded benefits estimates.

- ***Cost estimates***—In the final RIA, the EPA presents a range of costs using both the fixed and hybrid methodologies to estimate the costs associated with unknown controls.

- ***Inflation***—The EPA updated the currency year in this RIA to use 2010 dollars, which increased both the costs and the monetized benefits by approximately 8% since the proposal.

1.4.2 Existing and Revised PM Air Quality Standards

Two primary $PM_{2.5}$ standards provide public health protection from effects associated with fine particle exposures: the annual standard and the 24-hour standard. The existing annual standard is set at a level of 15.0 μg/m^3, based on the 3-year average of the annual arithmetic mean of $PM_{2.5}$ concentrations. The existing 24-hour standard is set at a level of 35 μg/m^3, based on the 3-year average of the 98th percentile of 24-hour $PM_{2.5}$ concentrations. In this RIA, the existing primary $PM_{2.5}$ standards, including both the annual standard and 24-hour standard, are denoted as 15/35 μg/m^3. In this current PM NAAQS review, the EPA has revised the level of the primary annual $PM_{2.5}$ standard to 12 μg/m^3 in conjunction with retaining the level of the 24-hour standard at 35 μg/m^3.

Currently, the existing secondary (welfare-based) $PM_{2.5}$ standards are identical in all respects to the primary standards. In this PM NAAQS review, the EPA is retaining the current suite of secondary standards for 24-hour and annual $PM_{2.5}$. Thus, while the new primary annual standard will be revised to 12 μg/m^3, the secondary annual standard will remain at 15 μg/m^3. Non-visibility welfare effects are addressed by this suite of secondary standards, and PM-related visibility impairment is addressed by the secondary 24-hour $PM_{2.5}$ standard, which EPA is leaving unchanged at 35 μg/m^3. The secondary standards will thus remain at 15/35 μg/m^3.

With regard to the primary and secondary standards for particles less than or equal to 10 μm in diameter (PM_{10}), the EPA is retaining the current primary and secondary 24-hour PM_{10} standards. Both standards are the same. The current primary and secondary 24-hour standards are set at a level of 150 μg/m^3, not to be exceeded more than once per year on average over 3 years (U.S. EPA, 1997).

1.4.2.1 Graphical Overview of the RIA Analysis

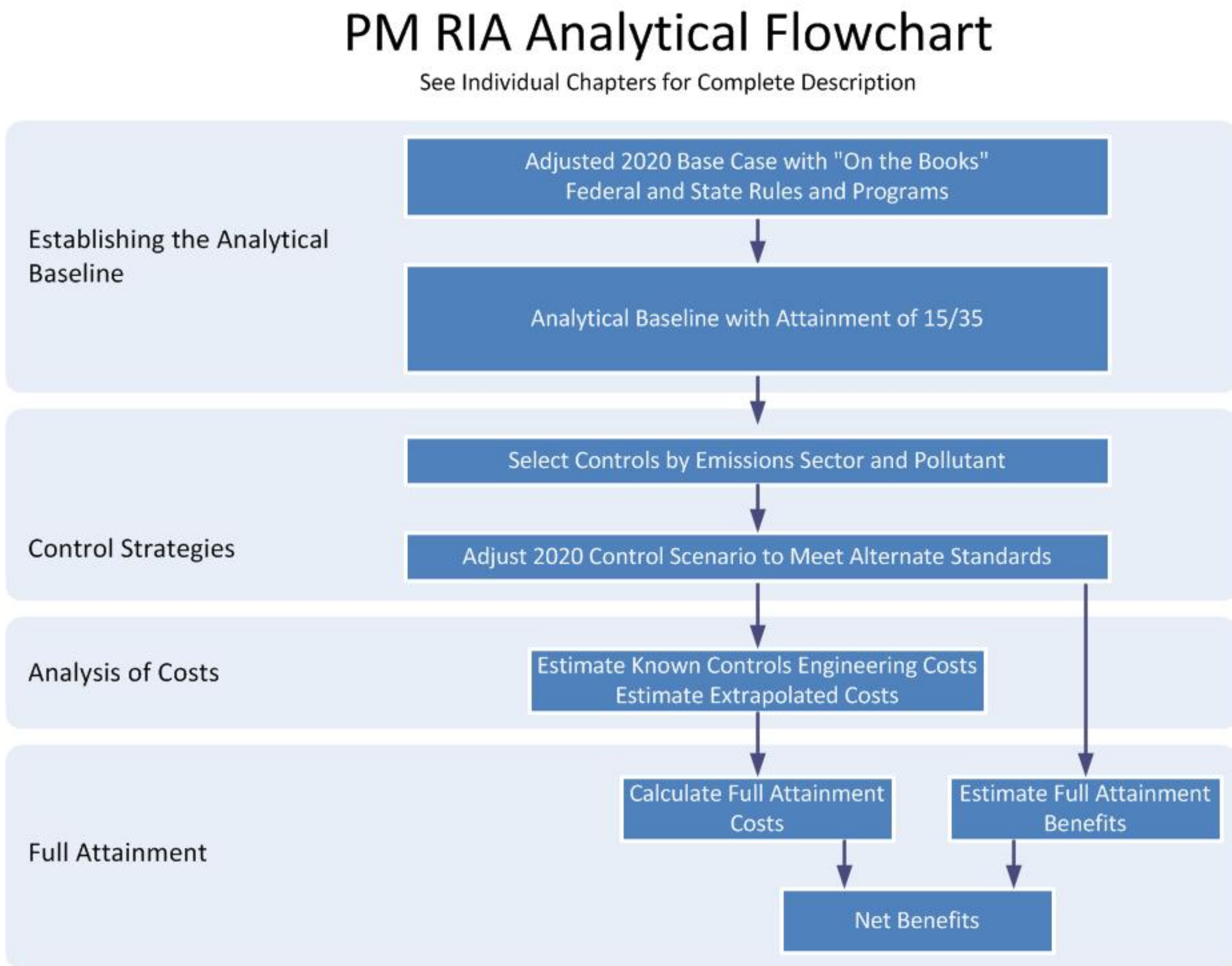


Figure 1-1. PM RIA Analytical Flow Diagram

1.4.2.2 Establishing the Analytical Baseline

The RIA is intended to evaluate the costs and benefits of reaching attainment with alternative $PM_{2.5}$ standards. In order to develop and evaluate control strategies for attaining a more stringent primary standard, it is important to account for Federal and state rules and programs currently underway, as well as to reflect attainment of the current annual and daily standards of 15/35 µg/m^3. Estimating the 2020 levels after attainment of the current standards of 15/35 µg/m^3 then allows us to estimate the incremental costs and benefits of attaining any alternative primary standard. EPA anticipates attainment with 15/35 µg/m^3 by 2020 in all but two areas in California, not expected to attain the current standards until 2025.

The analytical baseline includes reductions already achieved as a result of national regulations and reductions expected prior to 2020 from recently promulgated national regulations, referred to as the base case. Reductions achieved as a result of State and local agency regulations and voluntary programs are reflected to the extent that they are represented in emission inventory information submitted to the EPA by State and local agencies. Below is a list of some of the major national rules reflected in the base case. Refer to Chapter 3, Section 3.2.1.4 for a more detailed discussion of the rules reflected in the 2020 base case emissions inventory.

- Light-Duty Vehicle Tier 2 Rule (U.S. EPA, 1999)
- Heavy Duty Diesel Rule (U.S. EPA, 2000)
- Clean Air Nonroad Diesel Rule (U.S. EPA, 2004)
- Regional Haze Regulations and Guidelines for Best Available Retrofit Technology Determinations (U.S. EPA, 2005b)
- NO_x Emission Standard for New Commercial Aircraft Engines (U.S. EPA, 2005a)
- Emissions Standards for Locomotives and Marine Compression-Ignition Engines (U.S. EPA, 2008a)
- Control of Emissions for Nonroad Spark Ignition Engines and Equipment (U.S. EPA, 2008b)
- C3 Oceangoing Vessels (U.S. EPA, 2010a)
- Boiler MACT (U.S. EPA, 2011d)
- Hospital/Medical/Infectious Waste Incinerators: New Source Performance Standards and Emission Guidelines: Final Rule Amendments (U.S. EPA, 2009a)
- Reciprocating Internal Combustion Engines (RICE) NESHAPs (U.S. EPA, 2010b)
- Mercury and Air Toxics Standards (U.S. EPA, 2011b)
- Cross-State Air Pollution Rule (CSAPR) (U.S. EPA, 2011a)[3]

The analytical baseline for this analysis does not assume emissions controls that might be implemented to meet the other NAAQS for O_3, NO_x, or SO_2. To the extent that some of the

[3] See Chapter 3, Section 3.2.1.4 for a discussion of the role CSAPR plays in the $PM_{2.5}$ RIA and the reasons we believe CSAPR remains an appropriate proxy for this analysis.

estimated emissions reductions needed to meet the revised annual PM standard are also needed to meet the current standards for O_3, NO_x, or SO_2, the costs and benefits of meeting the revised PM annual standard will be overstated. We did not conduct this analysis incremental to controls applied as part of previous NAAQS analyses (e.g., O_3, NO_x, or SO_2) because the data and modeling on which these previous analyses were based are now considered outdated and are not compatible with the current $PM_{2.5}$ NAAQS analysis. In addition, all control strategies analyzed in all NAAQS RIAs are hypothetical. The analysis presented here is just one scenario that States may employ but does not prescribe how attainment must be achieved.

Most areas of the United States will be required to demonstrate attainment with the new standard by 2020. As a result, for these areas, the correct baseline for estimating the incremental emissions reductions that would be needed to attain the more protective standard is a baseline with emissions projected to 2020 and adjusted to reflect the additional emissions reductions that would be needed to attain the current 15/35 standards. For two areas in Southern California (South Coast and San Joaquin), the degree of projected non-attainment with the revised annual standard 12 $\mu g/m^3$ is high enough that those counties are not expected to be able to demonstrate attainment of the new standard by 2020. Instead, those two areas are likely to qualify for an extension of their attainment date of up to 5 years. If the areas are granted an attainment date extension, they will have until 2025 to demonstrate attainment of the revised annual standard of 12 $\mu g/m^3$. As a result, for these two areas, the correct baseline for estimating the incremental emissions reductions that would be needed to attain the more protective standard is a baseline with emissions projected to 2025 adjusted to reflect the additional emissions reductions that would be needed to attain the current 15/35 $\mu g/m^3$standards.

This difference in attainment year is important because between 2020 and 2025 emissions from mobile sources in California are expected to be reduced because of continued fleet turnover from older, higher emitting vehicles to newer, lower emitting vehicles. These reductions in emissions will occur as a result of previous State rules for which costs and benefits have already been counted and thus will not be costs and benefits attributable to meeting the revised annual standard of 12 $\mu g/m^3$. For California, the provided future-year 2020 and 2025 emissions included most on-the-books regulations such as those for low sulfur fuel, idling of heavy-duty vehicles, chip reflash, public fleets, trash trucks, drayage trucks, and heavy duty trucks and buses. See Chapter 3, Section 3.2.1.4 for further details on California emission inventories.

For the purposes of this analysis, we have constructed an analytical baseline that reflects attainment of 15/35 μg/m^3 in 2020. This analytical baseline is modified in the South Coast Air Quality Management District and the San Joaquin Valley Air Pollution Control District to reflect an “adjustment” for the reductions in NO_x emissions that those areas are expected to see between 2020 and 2025. These reductions in NO_x are not attributable to attainment of the current or revised PM standards but rather reflect the impacts of other programs. These NO_x emissions changes will affect baseline PM concentrations but will not affect costs or benefits of attaining the revised annual or the alternative annual standards.

To provide the most reasonable and reliable estimates of costs and benefits of full attainment for the nation, we construct an analytical baseline for estimating the costs and benefits of attaining the revised annual standard of 12 μg/m^3, 13 μg/m^3, and 11 μg/m^3 with the following characteristics: (1) reflects on-the-books regulations as implemented through 2020 plus additional emissions reductions needed to meet the 15/35 μg/m^3 standard levels, and (2) additional mobile source emissions reductions expected to occur between 2020 and 2025 for California’s South Coast and San Joaquin areas, which are likely to not demonstrate attainment until 2025. Essentially, we are modifying the baseline in those two areas to reflect an “adjustment” for the reductions in NO_x emissions that those areas are expected to see between 2020 and 2025. This allows us to generate costs and benefits of full attainment without overstating the costs and benefits in those two areas, which would occur if we forced costly emissions reductions in 2020 in areas that would not have to occur until 2025 and that will be offset because of the expected reductions in mobile source emissions due to other programs. See Chapter 3, Section 3.2.1.4 for details on emission inventories from California.

Benefits for all areas are estimated using 2020 population data for consistency, recognizing that full attainment costs and benefits will not actually be realized until 2025 for a portion of the costs and benefits. The 2020 estimates of full attainment costs and benefits will be an underestimate of benefits in 2025 because of population growth and changes in the age distribution of the population between 2020 and 2025.

1.4.3 Health and Welfare Co-Benefits Analysis Approach

The EPA estimated impacts on human health (e.g., mortality and morbidity effects) under full attainment of the three alternative annual $PM_{2.5}$ standards. We considered an array of health impacts attributable to changes in $PM_{2.5}$ exposure and estimated these benefits using the BenMAP model (Abt Associates, 2012), which has been used in many recent RIAs (e.g., U.S. EPA, 2006, 2011a, 2011b), and *The Benefits and Costs of the Clean Air Act 1990 to 2020* (U.S.

EPA, 2011c). The monetized benefits estimated in the core analysis include avoided premature deaths (derived from effect coefficients in tow cohort studies [Krewski et al. (2009) and Lepeule et al. (2012)] for adults and one for infants [Woodruff et al. (1997)]) as well as avoided morbidity effects for 10 non-fatal endpoints ranging in severity from lower respiratory symptoms to heart attacks. As noted above, because California is the only state that needs additional air quality improvement beyond the analytical baseline after accounting for expected air quality improvements expected from recent rules, all of the benefits associated with the revised standard of 12ug/m3 occur in California.

Since the proposed rule, the EPA has incorporated an array of policy and technical updates to the benefits analysis approach applied in this RIA, including incorporation of the most recent follow-up to the Harvard Six Cities cohort study (Lepeule et al., 2012), more recent demographic data projections, additional hospitalization and emergency department visit studies, inflation adjustment to 2010 dollars, and an expanded uncertainty assessment. Each of these updates is fully described in the health benefits chapter (Chapter 5) and summarized below in section ES.5. Compared with the proposal benefits, the estimated benefits for the revised standard are about double due to a combination of updates to the analytic baseline

Even though the primary standards are designed to protect against adverse effects to human health, the emission reductions will have welfare co-benefits in addition to the direct human health benefits. The term *welfare co-benefits* covers both environmental and societal benefits of reducing pollution, such as reductions in visibility impairment, materials damage, and ecosystem damage. Despite our attempts to quantify and monetize as many of the benefits as possible, the welfare co-benefits associated with meeting the alternative standards are not quantified or monetized in this analysis. Unquantified health benefits are discussed in Chapter 5, and unquantified welfare co-benefits are discussed in Chapter 6.

It is important to note that estimates of the health benefits from reduced $PM_{2.5}$ exposure reported here contain uncertainties, which are described in detail in Chapter 5 and Appendix 5b. Below are two key assumptions in the benefits analysis:

1. We assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This is an important assumption, because $PM_{2.5}$ varies considerably in composition across sources, but the scientific evidence is not yet sufficient to allow differentiation of effect estimates by particle type. The *Integrated Science Assessment for Particulate Matter* (PM ISA), which was twice reviewed by CASAC, concluded that "many constituents of $PM_{2.5}$ can be linked with multiple health effects, and the evidence is not yet sufficient to allow

differentiation of those constituents or sources that are more closely related to specific outcomes" (U.S. EPA, 2009). These uncertainties are likely to be magnified in the current analysis to the extent that the emissions controls are less diverse when focusing on one small region of the country rather than a broader geography with more diverse emissions sources and the application of a more diverse set of controls.

2. We assume that health impact functions based on national studies are representative for exposures and populations in California. In addition to the national risk coefficients we use as our core estimates, the EPA considered the cohort studies conducted in California specifically. Although we have not calculated the benefits results using the cohort studies conducted in California, we provided these risk coefficients to show how much the monetized benefits could have changed. Most of the California cohort studies report central effect estimates similar to the (nation-wide) all-cause mortality risk estimate we applied from Krewski et al. (2009) and Lepeule et al. (2012) albeit with wider confidence intervals. Three cohort studies conducted in California indicate statistically significant higher risks than the risk estimates we applied from Lepeule et al. (2012), and four studies showed insignificant results.

3. We assume that the health impact function for fine particles is log-linear without a threshold in this analysis. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5}$, including both areas that do not meet the fine particle standard and those areas that are in attainment, down to the lowest modeled concentrations.

In general, we are more confident in the magnitude of the risks we estimate from simulated $PM_{2.5}$ concentrations that coincide with the bulk of the observed PM concentrations in the epidemiological studies that are used to estimate the benefits. Likewise, we are less confident in the risk we estimate from simulated $PM_{2.5}$ concentrations that fall below the bulk of the observed data in these studies. As noted in the preamble to the rule, the range from the 25th to 10th percentiles of the air quality data in the epidemiology studies is a reasonable range below which we start to have appreciably less confidence in the magnitude of the associations observed in the epidemiological studies. Concentration benchmark analyses (e.g., 25th percentile, 10th percentile, one standard deviation below the mean,[4] and lowest measured level [LML]) provide some insight into the level of uncertainty in the estimated $PM_{2.5}$ mortality benefits. Most of the estimated avoided premature deaths for this rulemaking occur at or above the lowest measured $PM_{2.5}$ concentration in the two studies that are used to estimate mortality benefits. There are uncertainties inherent in identifying any particular point

[4] A range of one standard deviation around the mean represents approximately 68% of normally distributed data and below the mean falls between the 25th and 10th percentiles.

at which our confidence in reported associations becomes appreciably less, and the scientific evidence provides no clear dividing line. However, the EPA does not view these concentration benchmarks as a concentration threshold below which we would not quantify health benefits of air quality improvements. Rather, the core benefits estimates reported in this RIA (i.e., those based on Krewski et al. [2009] and Lepeule et al. [2012]) are the best measures because they reflect the full range of modeled air quality concentrations associated with the emission reduction strategies and because the current body of scientific literature indicates that a no-threshold model provides the best estimate of PM-related long-term mortality. It is important to emphasize that “less confidence” does not mean “no confidence.”

The estimated benefits reflect illustrative control measures and emission reductions to lower $PM_{2.5}$ concentrations at monitors projected to exceed the revised and alternative annual standards. The result is that air quality is expected to improve in counties that exceed these standards as well as surrounding areas that do not exceed the alternative standards. In order to make a direct comparison between the benefits and costs of the emission reduction strategies, it is appropriate to include all the benefits occurring as a result of the emission reduction strategies applied regardless of where they occur. Therefore, it is not appropriate to estimate the fraction of benefits that occur only in the counties that exceed the standards because it would omit benefits attributable to emission reduction in exceeding counties. In addition, we estimate benefits using modeled air quality data with 12 km grid cells, which is important because the grid cells are often substantially smaller than counties, and $PM_{2.5}$ concentrations can vary spatially within a county. Some grid cells in a county can be below the level of the alternative standard even though the highest monitor exceeds the alternative standard. Thus, emission reductions can lead to benefits in grid cells that are below the alternative standards within an exceeding county.

1.4.4 Cost Analysis Approach

The EPA estimated total costs under partial and full attainment of the alternative $PM_{2.5}$ standards. The engineering costs generally include the costs of purchasing, installing, and operating the referenced control technologies. The technologies and control strategies selected for analysis are illustrative of one way in which nonattainment areas could meet a revised standard. There are numerous ways to construct and evaluate potential control programs that would bring areas into attainment with alternative standards, and the EPA anticipates that State and local governments will consider programs that are best suited for local conditions.

The partial attainment cost analysis reflects the costs associated with applying known controls. Costs for full attainment include estimates for the engineering costs of the additional tons of emissions reductions that are needed beyond identified controls, referred to as extrapolated costs. By definition, no cost data currently exist for the additional emissions reductions needed beyond known controls. We employ two methodologies for estimating the costs of unidentified future controls: a fixed-cost methodology and a hybrid methodology; both approaches assume either that existing technologies can be applied in particular combinations or to specific sources that we currently can't predict or that innovative strategies and new control options make possible the emissions reductions needed for attainment by 2020. The two approaches, however, implicitly reflect different assumptions about technological progress and innovation in emissions reductions strategies. The fixed-cost methodology uses a \$15,000/ton estimate for each ton of $PM_{2.5}$ reduced, and the hybrid methodology generates a total annual cost curve for $PM_{2.5}$ for unknown future controls that might be applied in order to move toward 2020 attainment. The hybrid methodology has the advantage of incorporating information about how significant the needed reductions from unspecified control technologies are relative to the known control measures and matching that information with expected increasing per-ton cost for applying unknown controls. Employing the fixed-cost methodology, approximately 90% of total costs for attaining the revised annual standard of 12 $\mu g/m^3$ are from unspecified control technologies. Employing the hybrid methodology, approximately 97% of total costs for attaining the revised annual standard of 12 $\mu g/m^3$ are from unspecified control technologies. The EPA recognizes that the extrapolated portion of the engineering cost estimates reflects substantial uncertainty about which sectors and which technologies might become available for cost-effective application in the future.

The engineering cost estimates are limited in their scope. Our analysis focuses on the emissions reductions needed for attainment of the revised and alternative standards. Also, the amendments to the ambient air monitoring regulations will revise the network design requirements for $PM_{2.5}$ monitoring sites, resulting in moving 21 monitors to established near-road monitoring stations by January 1, 2015. The incremental cost associated with moving these 21 monitors is a one-time cost of \$28,570. Lastly, the EPA understands that some States will incur costs designing SIPs and implementing new control strategies to meet the revised standard. However, the EPA does not know what specific actions States will take to design their SIPs to meet the revised standards; therefore, we do not include estimated costs that government agencies may incur for managing the requirement, implementing these (or other) control strategies, or for offering incentives that may be necessary to encourage the implementation of specific technologies, especially for technologies that are not necessarily market driven.

1.5 Organization of this Regulatory Impact Analysis

This RIA includes the following 10 chapters:

- *Chapter 1: Introduction and Background*. This chapter introduces the purpose of the RIA.
- *Chapter 2: Defining the PM Air Quality Problem*. This chapter characterizes the nature, scope, and magnitude of the current-year $PM_{2.5}$ problem.
- *Chapter 3*: *Air Quality Modeling and Analysis*. The data, tools, and methodology used for the air quality modeling are described in this chapter, as well as the post-processing techniques used to produce a number of air quality metrics for input into the analysis of costs and benefits.
- *Chapter 4: Control Strategies*. This chapter presents the hypothetical control strategies, the geographic areas where controls were applied, and the results of the modeling that predicted $PM_{2.5}$ concentrations in 2020 after applying the control strategies.
- *Chapter 5: Health Benefits Analysis Approach and Results*. This chapter quantifies and monetizes the health benefits of the $PM_{2.5}$-related air quality improvements associated with the hypothetical control strategies.
- *Chapter 6*: *Welfare Co-Benefits of the Primary Standard*. This chapter describes the welfare effects, including changes in visibility, materials damage, ecological effects from PM deposition, ecological effects from nitrogen and sulfur emissions, vegetation effects from ozone exposure, ecological effects from mercury deposition, and climate effects.
- *Chapter 7*: *Engineering Cost Analysis*. This chapter summarizes the data sources and methodology used to estimate the engineering costs of partial and full attainment of several alternative standards.
- *Chapter 8: Comparison of Benefits and Costs*. This chapter compares estimates of the total benefits with total costs and summarizes the net benefits of several alternative standards.
- *Chapter 9*: *Statutory and Executive Order Impact Analyses*. This chapter summarizes the Statutory and Executive Order impact analyses.

- *Chapter 10*: *Qualitative Discussion of Employment Impacts of Air Quality Regulations.* This chapter provides a qualitative discussion of employment impacts of air quality regulations.

1.6 References for Chapter 1

Abt Associates, Inc. 2012. *BenMAP User's Manual Appendices*. Prepared for U.S. Environmental Protection Agency Office of Air Quality Planning and Standards. Research Triangle Park, NC. September.

Krewski, D., R.T. Burnett, M.S. Goldbert, K. Hoover, J. Siemiatycki, M. Jerrett, M. Abrahamowicz, and W.H. White. 2009 "Reanalysis of the Harvard Six Cities Study and the American Cancer Society Study of Particulate Air Pollution and Mortality." Special Report to the Health Effects Institute. Cambridge MA. July.

Lepeule, J., F. Laden, D. Dockery, and J. Schwartz. 2012. "Chronic Exposure to Fine Particles and Mortality: An Extended Follow-Up of the Harvard Six Cities Study from 1974 to 2009." *Environ Health Perspect.* In press. Available at: http://dx.doi.org/10.1289/ehp.1104660.

U.S. Environmental Protection Agency. 1997. *Regulatory Impact Analyses for the Particulate Matter and Ozone National Ambient Air Quality Standards and Proposed Regional Haze Rule*. Available at: http://www.epa.gov/ttn/oarpg/naaqsfin/ria.html.

U.S. Environmental Protection Agency (U.S. EPA). 1999. *Regulatory Impact Analysis—Control of Air Pollution from New Motor Vehicles: Tier 2 Motor Vehicle Emissions Standards and Gasoline Sulfur Control Requirements*. EPA 420-R-99-023. Available at: http://www.epa.gov/otaq/regs/ld-hwy/tier-2/documents/r99023.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2000. *Regulatory Impact Analysis: Control of Emissions of Air Pollution from Highway Heavy-Duty Engines*. EPA 420-R-00-010. Available at: http://www.epa.gov/otaq/regs/hd-hwy/2000frm/r00010.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2004. *Final Regulatory Analysis: Control of Emissions from Nonroad Diesel Engines*. EPA420-R-04-007. Available at: http://www.epa.gov/otaq/documents/nonroad-diesel/420r04007.pdf.

U.S. Environmental Protection Agency (U.S. EPA). November 17, 2005a. *Federal Register*. Vol. 70, No. 221. "Control of Air Pollution From Aircraft and Aircraft Engines; Emission Standards and Test Procedures." 40 CFR Part 87. Available at: http://www.epa.gov/fedrgstr/EPA-AIR/2005/November/Day-17/a22704.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2005b. *Regulatory Impact Analysis for the Final Clean Air Visibility Rule or the Guidelines for Best Available Retrofit Technology (BART) Determinations Under the Regional Haze Regulations*. EPA-452/R-05-004. Available at: http://www.epa.gov/oar/visibility/pdfs/bart_ria_2005_6_15.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2006. *Regulatory Impact Analysis, 2006 National Ambient Air Quality Standards for Particulate Matter, Chapter 5*. Office of Air Quality Planning and Standards, Research Triangle Park, NC. October. Available at: http://www.epa.gov/ttn/ecas/regdata/RIAs/Chapter%205—Benefits.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2008a. *Regulatory Impact Analysis: Control of Emissions of Air Pollution from Locomotive Engines and Marine Compression Ignition Engines Less than 30 Liters Per Cylinder*. Available at: http://www.regulations.gov/#!documentDetail;D=EPA-HQ-OAR-2003-0190-0938.

U.S. Environmental Protection Agency (U.S. EPA). 2008b. *Control of Emissions from Marine SI and Small SI Engines, Vessels, and Equipment. Final Regulatory Impact Analysis*. Available at: http://www.regulations.gov/#!documentDetail;D=EPA-HQ-OAR-2004-0008-0929.

U.S. Environmental Protection Agency (U.S. EPA). 2009a. *Economic Impacts of Revised MACT Standards for Hospital/Medical/Infectious Waste Incinerators*. Available at: http://www.epa.gov/ttn/atw/129/hmiwi/hmiwi_eia_090803.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2009b. *Integrated Science Assessment for Particulate Matter (Final Report)*. EPA-600-R-08-139F. National Center for Environmental Assessment—RTP Division. December. Available at: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=216546.

U.S. Environmental Protection Agency (U.S. EPA). 2010a. *Regulatory Impact Analysis: Control of Emissions of Air Pollution from Category 3 Marine Diesel Engines*. Available at: http://www.epa.gov/oms/regs/nonroad/marine/ci/420r09019.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2010b. *Regulatory Impact Analysis: Regulatory Impact Analysis (RIA) for Existing Stationary Spark Ignition (SI) RICE NESHAP.* Available at: http://www.epa.gov/ttn/atw/rice/fnl_si_rice_ria.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011a. *Regulatory Impact Analysis for the Federal Implementation Plans to Reduce Interstate Transport of Fine Particulate Matter and Ozone in 27 States; Correction of SIP Approvals for 22 States*. June. Available at: http://www.epa.gov/airtransport/pdfs/FinalRIA.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011b. *Regulatory Impact Analysis for the Final Mercury and Air Toxics Standards.* EPA-452/R-11-011. December. Available at: http://www.epa.gov/ttn/ecas/regdata/RIAs/matsriafinal.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011c. *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress*. Office of Air and Radiation, Office of Policy, Washington, DC. March. Available at: http://www.epa.gov/oar/sect812/feb11/fullreport.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011d. *Regulatory Impact Analysis: National Emission Standards for Hazardous Air Pollutants for Industrial, Commercial, and Institutional Boilers and Process Heaters.* February. Available at: http://www.epa.gov/ttnecas1/regdata/RIAs/boilersriafinal110221_psg.pdf.

U.S. Office of Management and Budget. Circular A-4, September 17, 2003. Available at http://www.whitehouse.gov/omb/circulars/a004/a-4.pdf.

Woodruff, T.J., J. Grillo, and K.C. Schoendorf. 1997. "The Relationship Between Selected of postneonatal infant mortality and particulate air pollution in the United States." *Environmental Health Perspectives* 105 (6): 608-612.

CHAPTER 2
DEFINING THE PM AIR QUALITY PROBLEM

2.1 Synopsis

This chapter characterizes the nature, scope and magnitude of the current year particulate matter (PM) problem. It includes a summary of the spatial and temporal distribution of $PM_{2.5}$ and the likely origin from direct emissions or atmospheric transformations of gaseous precursors and recent design values for $PM_{2.5}$.

2.2 Particulate Matter (PM) Properties

PM is a highly complex mixture of solid particles and liquid droplets distributed among numerous atmospheric gases which interact with solid and liquid phases. Particles range in size from those smaller than 1 nanometer (10^{-9} meter) to over 100 micrometer (µm, or 10^{-6} meter) in diameter (for reference, a typical strand of human hair is 70 µm in diameter and a grain of salt is about 100 µm). Atmospheric particles can be grouped into several classes according to their aerodynamic and physical sizes, including ultrafine particles (<0.1 µm), accumulation mode or "fine" particles (0.1 to ~3 µm), and coarse particles (>1 µm). For regulatory purposes, fine particles are measured as $PM_{2.5}$ and inhalable or thoracic coarse particles are measured as $PM_{10-2.5}$, corresponding to their size (diameter) range in micrometers and referring to total particle mass under 2.5 and between 2.5 and 10 micrometers, respectively. The EPA currently has standards that measure $PM_{2.5}$ and PM_{10}.

Particles span many sizes and shapes and consist of hundreds of different chemicals. Particles are emitted directly from sources and are also formed through atmospheric chemical reactions; the former are often referred to as "primary" particles, and the latter as "secondary" particles. Particle pollution also varies by time of year and location and is affected by several weather-related factors, such as temperature, clouds, humidity, and wind. A further layer of complexity comes from particles' ability to shift between solid/liquid and gaseous phases, which is influenced by concentration and meteorology, especially temperature.

Particles are made up of different chemical components. The major chemical components include carbonaceous materials (carbon soot and organic compounds), and inorganic compounds including, sulfate and nitrate compounds that usually include ammonium, and a mix of substances often apportioned to crustal materials such as soil and ash. As mentioned above, particulate matter includes both "primary" PM, which is directly emitted into the air, and "secondary" PM, which forms indirectly from emissions from fuel combustion and other sources. Primary PM consists of carbonaceous materials (soot and accompanying organics) and includes:

- Elemental carbon, organic carbon, and crustal material directly emitted from cars, trucks, heavy equipment, forest fires, some industrial processes and burning waste.
- Both combustion and process related fine metals and larger crustal material from unpaved roads, stone crushing, construction sites, and metallurgical operations.

Secondary PM forms in the atmosphere from gases. Some of these reactions require sunlight and/or water vapor. Secondary PM includes:

- Sulfates formed from sulfur dioxide (SO_2) emissions from power plants and industrial facilities;
- Nitrates formed from nitrogen oxide (NO_x) emissions from cars, trucks, industrial facilities, and power plants; and
- Ammonium formed from ammonia (NH_3) emissions from gas-powered vehicles and fertilizer and animal feed operations. These contribute to the formation of sulfates and nitrates that exist in the atmosphere as ammonium sulfate and ammonium nitrate.[1]
- Organic carbon (OC) formed from reactive organic gas emissions, including volatile organic compounds (VOCs), from cars, trucks, industrial facilities, forest fires, and biogenic sources such as trees.1

As described above, organic carbon has both primary and secondary components. The percentage contribution to total OC from directly emitted OC versus secondarily formed OC varies based on location. In an urban area, near direct sources of OC such as cars, trucks, and industrial sources, the percentage of primary OC may dominate, whereas, in a rural area with more biogenic sources, OC may be mostly secondarily formed. In addition, emissions from sources such as power plants and industrial facilities may have small amounts of directly emitted $PM_{2.5}$ speciated into sulfate. Figure 2-1 shows, in detail, the sources contributing to directly emitted $PM_{2.5}$ and PM_{10}, as well as PM precursors: SO_2, NO_x, NH_3, and VOC according to the 2008 NEI, version 2 (EPA, 2012) . In Figure 2-1, EGUs stands for Electric Generating Utilities.

[1] Direct NH_3 and VOC emissions are not controlled as part of the control strategy analysis. Emissions of $PM_{2.5}$, NO_x, and SO_2 are controlled in the control strategies, for a complete discussion please refer to Chapter 4.

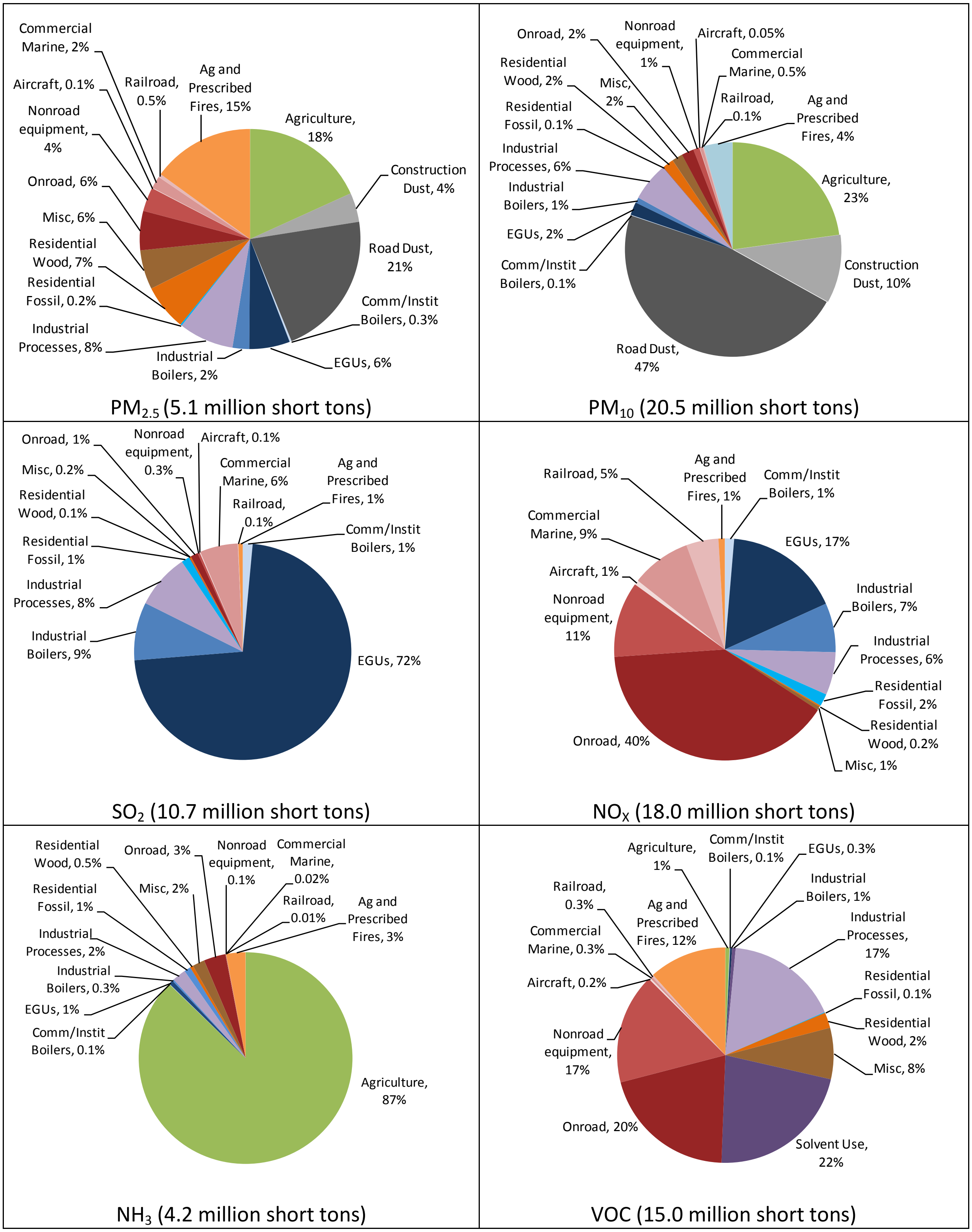


Figure 2-1. Detailed Source Categorization of Anthropogenic Emissions of Primary PM$_{2.5}$, PM$_{10}$ and Gaseous Precursor Species SO$_2$, NO$_x$, NH$_3$ and VOCs for 2008

2.2.1 $PM_{2.5}$

"Fine particles" or $PM_{2.5}$ are particles with diameters that are less than 2.5 micrometers. As discussed above, these particles are composed of both primary (derived directly from emissions) and secondary (derived from atmospheric reactions involving gaseous precursors) components.

2.2.1.1 Geographical Scale and Transport

Both local and regional sources contribute to particle pollution. Fine particles can be transported long distances by wind and weather and can be found in the air thousands of miles from where they formed. Nitrates and sulfates formed from NO_x and SO_2 are generally transported over wide areas leading to substantial background contributions in urban areas. Organic carbon, which has both a primary and secondary component, can also be transported but to a far lesser degree. In general, higher concentrations of elemental carbon and crustal matter are found closest to the sources of these emissions.

Figure 2-2 shows how much of the $PM_{2.5}$ mass can be attributed to local versus regional sources for 13 selected urban areas (EPA, 2004).[2] In each of these urban areas, monitoring sites were paired with nearby rural sites. When the average rural concentration is subtracted from the measured urban concentration, the estimated local and regional contributions become apparent. We observe a large urban excess across the U.S. for most $PM_{2.5}$ species but especially for total carbon mass with Fresno, CA having the highest observed measure. Larger urban excess of nitrates is seen in the western U.S. with Fresno, CA and Salt Lake City, UT significantly higher than all other areas. These results indicate that local sources of these pollutants are indeed contributing to the $PM_{2.5}$ air quality problem in these areas. As expected for a predominately regional pollutant, only a modest urban excess is observed for sulfates.

In the East, regional pollution contributes to more than half of total $PM_{2.5}$ concentrations. Rural background $PM_{2.5}$ concentrations are high in the East and are somewhat uniform over large geographic areas. These regional concentrations come from emission sources such as power plants, natural sources, and urban pollution and can be transported hundreds of miles and reflect to some extent the denser clustering of urban areas in the East as compared to the West. In the West, much of the measured $PM_{2.5}$ concentrations tend to be local in nature. These concentrations come from emission sources such as wood combustion and mobile sources. In general, these data indicate that reducing regional SO_2 and local sources

[2] The measured $PM_{2.5}$ concentration is not necessarily the maximum for each urban area.

of carbon in the East, and local sources of nitrate and carbon in the West will be most effective in reducing $PM_{2.5}$ concentrations.

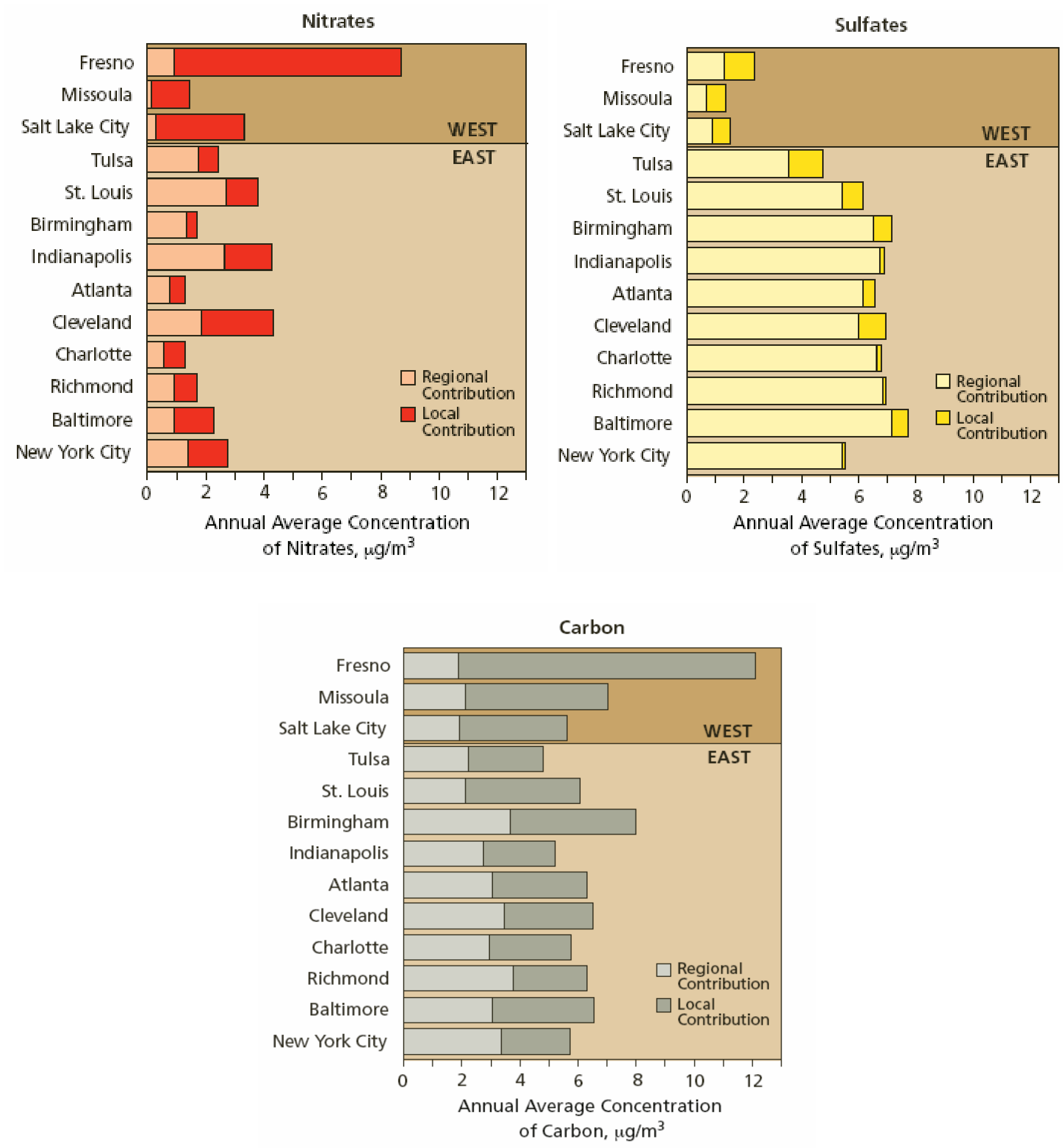


Figure 2-2. Regional and Local Contributions to Annual Average $PM_{2.5}$ by Particulate SO_4^{2-}, Nitrate and Total Carbon (i.e., organic plus EC) for Select Urban Areas Based on Paired 2000–2004 IMPROVE[a] and CSN[b] Monitoring Sites

[a] Interagency Monitoring of Protected Visual Environments (IMPROVE) http://vista.cira.colostate.edu/improve

[b] Chemical Speciation Network (CSN)

2.2.1.2 Regional and Seasonal Patterns

The chemical makeup of particles varies across the United States, as illustrated in Figure 2-3. For example, the higher regional emissions of SO_2 in the East result in higher absolute and relative amounts of sulfates as compared to the western U.S. Fine particles in southern California generally contain more nitrates than other areas of the country. Carbon is a substantial component of fine particles everywhere.

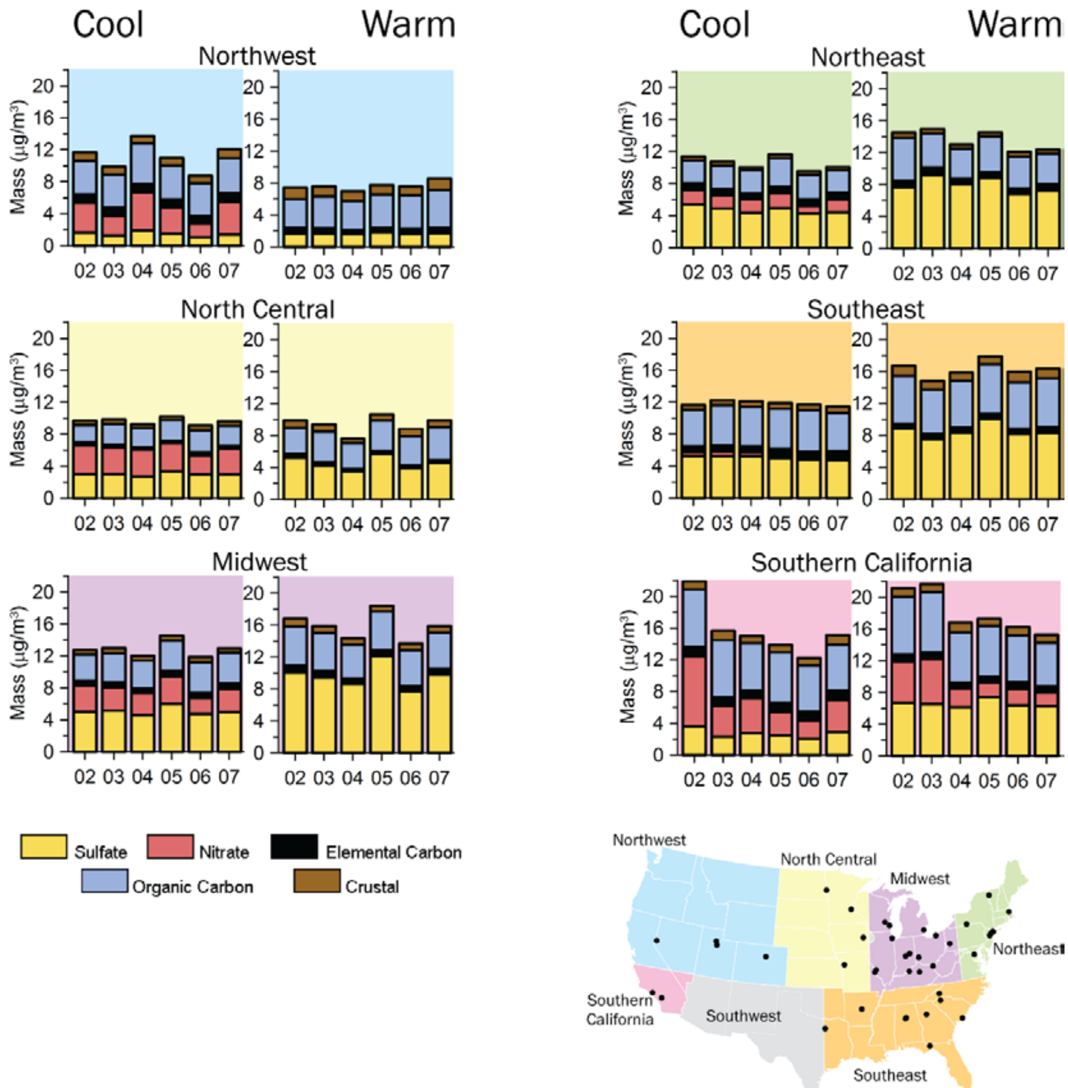


Figure 2-3. Regional and Seasonal Trends in Annual $PM_{2.5}$ Composition from 2002 to 2007 Derived Using the SANDWICH Method. Data from the 42 monitoring locations shown on the map were stratified by region and season including cool months (October–April) and warm months (May–September)

Fine particles can also have a seasonal pattern. As shown in Figure 2-3, $PM_{2.5}$ values in the eastern half of the United States are typically higher in warmer weather when meteorological conditions are more favorable for the formation and build up of sulfates from higher sulfur dioxide (SO_2) emissions from power plants in that region. Fine particle concentrations tend to be higher in the cooler calendar months in urban areas in the West, in part because fine particle nitrates and carbonaceous particles are more readily formed in cooler weather, and wood stove and fireplace use increases direct emissions of carbon.

2.2.1.3 Composition of $PM_{2.5}$ as Measured by the Federal Reference Method

The speciation measurements in the preceding analyses represented data from EPA's Speciation Trends Network, along with adjustments to reflect the fine particle mass associated with these ambient measurements. In order to more accurately predict the change in $PM_{2.5}$ design values for particular emission control scenarios, EPA characterizes the composition of $PM_{2.5}$ as measured by the Federal Reference Method (FRM). The current $PM_{2.5}$ FRM does not capture all ambient particles measured by speciation samplers as presented in the previous sections. The FRM-measured fine particle mass reflects losses of ammonium nitrate (NH_4NO_3) and other semi-volatile organic compounds (SVOCs; negative artifacts). It also includes particle-bound water (PBW) associated with hygroscopic species (positive artifacts) (Frank, 2006). Comparison of FRM and co-located speciation sampler NO_3^- values in Table 2-1 show that annual average NO_3 retention in FRM samples for six cities varies from 15% in Birmingham to 76% in Chicago, with an annual average loss of 1 $\mu g/m^3$. The volatilization is a function of temperature and relative humidity (RH), with more loss at higher temperatures and lower RH. Accordingly, nitrate is mostly retained during the cold winter days, while little may be retained during the hot summer days.

$PM_{2.5}$ FRM measurements also include water associated with hygroscopic aerosol. This is because the method derives fine particle concentrations from sampled mass equilibrated at 20–23 °C and 30–40% RH. At these conditions, the hygroscopic aerosol collected at more humid environments will retain their particle-bound water. The water content is higher for more acidic and sulfate-dominated aerosols. Combining the effects of reduced nitrate and hydrated aerosol causes the estimated nitrate and sulfate FRM mass to differ from the measured ions simply expressed as dry ammonium nitrate and ammonium sulfate. The composition of FRM mass is denoted as SANDWICH based on the Sulfate, Adjusted Nitrate Derived Water and Inferred Carbon approach from which they are derived. The $PM_{2.5}$ mass estimated from speciated measurements of fine particles is termed ReConstructed Fine Mass (RCFM). The application of SANDWICH adjustments to speciation measurements at six sites is illustrated in Table 2-1 and Figure 2-4. EPA's modeling incorporates these SANDWICH adjustments in the Model Attainment Test Software (MATS) (Abt, 2010).

Table 2-1. Annual Average FRM and CSN $PM_{2.5}$ NO_3^- and NH_4NO_3 Concentrations at Six Sites during 2003

Sampling Site Location	No. of Observations	FRM Mass	NO_3^- ($\mu g/m^3$) CSN[a]	FRM[b]	Difference (CSN – FRM)	NH_4NO_3 ($\mu g/m^3$) CSN	FRM	Percent of NH_4NO_3 in $PM_{2.5}$ FRM Mass CSN	FRM
Mayville, WI	100	9.8	2.5	1.5	1.0	3.2	1.9	33%	19%
Chicago, IL	76	14.4	2.8	2.1	0.7	3.7	2.8	25%	19%
Indianapolis, IN	92	14.8	2.5	1.3	1.3	3.2	1.6	22%	11%
Cleveland, OH	90	16.8	2.9	1.7	1.2	3.7	2.2	22%	13%
Bronx, NY	108	15.0	2.4	1.1	1.3	3.1	1.4	21%	9%
Birmingham, AL	113	17.0	1.1	0.2	0.9	1.4	0.2	8%	1%

[a] On denuded nylon-membrane filters for al sites except for Chicago, where denuded Teflon-membrane followed by nylon filters were used.

[b] On undenuded Teflon-membrane filters.

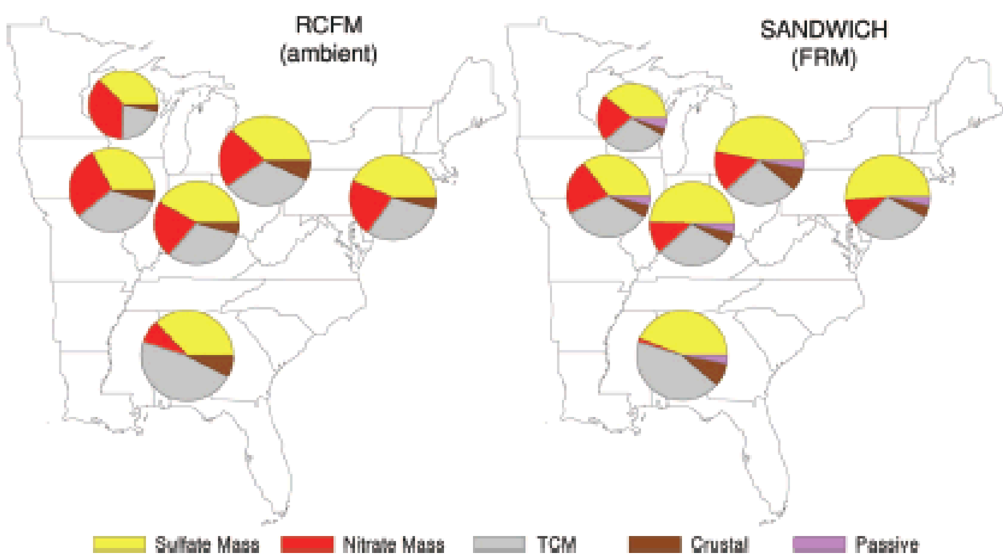


Figure 2-4. RCFM (left) versus SANDWICH (right) Pie Charts Comparing the Ambient and $PM_{2.5}$ FRM Reconstructed Mass Protocols on an Annual Average Basis[a]

[a] Estimated NH_4* and PBW for SANDWICH are included with their respective sulfate and nitrate mass slices. Circles are scaled in proportion to $PM_{2.5}$ FRM mass.

2.2.1.4 2006–2008 Design Values

Annual and 24-hour $PM_{2.5}$ design values for 2006-2008 are shown in Figures 2-5 and 2-6, respectively. These design values were calculated using 2006–2008 FRM 24-hour average $PM_{2.5}$ concentration measurements in a manner consistent with CFR Part 50.[3] For the most part, counties in the center of the U.S. have $PM_{2.5}$ design values that are above both 11 μg/m^3 for the annual standard and 35 μg/m^3 for the 24-hour standard. In the East, the counties above the current NAAQS (i.e., 15 μg/m^3 annual and 35 μg/m^3 24-hour standards) are similar. In the West, there are fewer counties above the annual level of 15 μg/m^3 than exceed the 24-hour standard of 35 μg/m^3.

[3] These years of ambient measurements are presented here since they frame the air quality model year of 2007. As discussed in Chapter 3, ambient measurement for the period 2005 through 2008 were used to construct 5-year weighted average concentrations for use as the starting point for future year projections in conjunction with air quality model predictions.

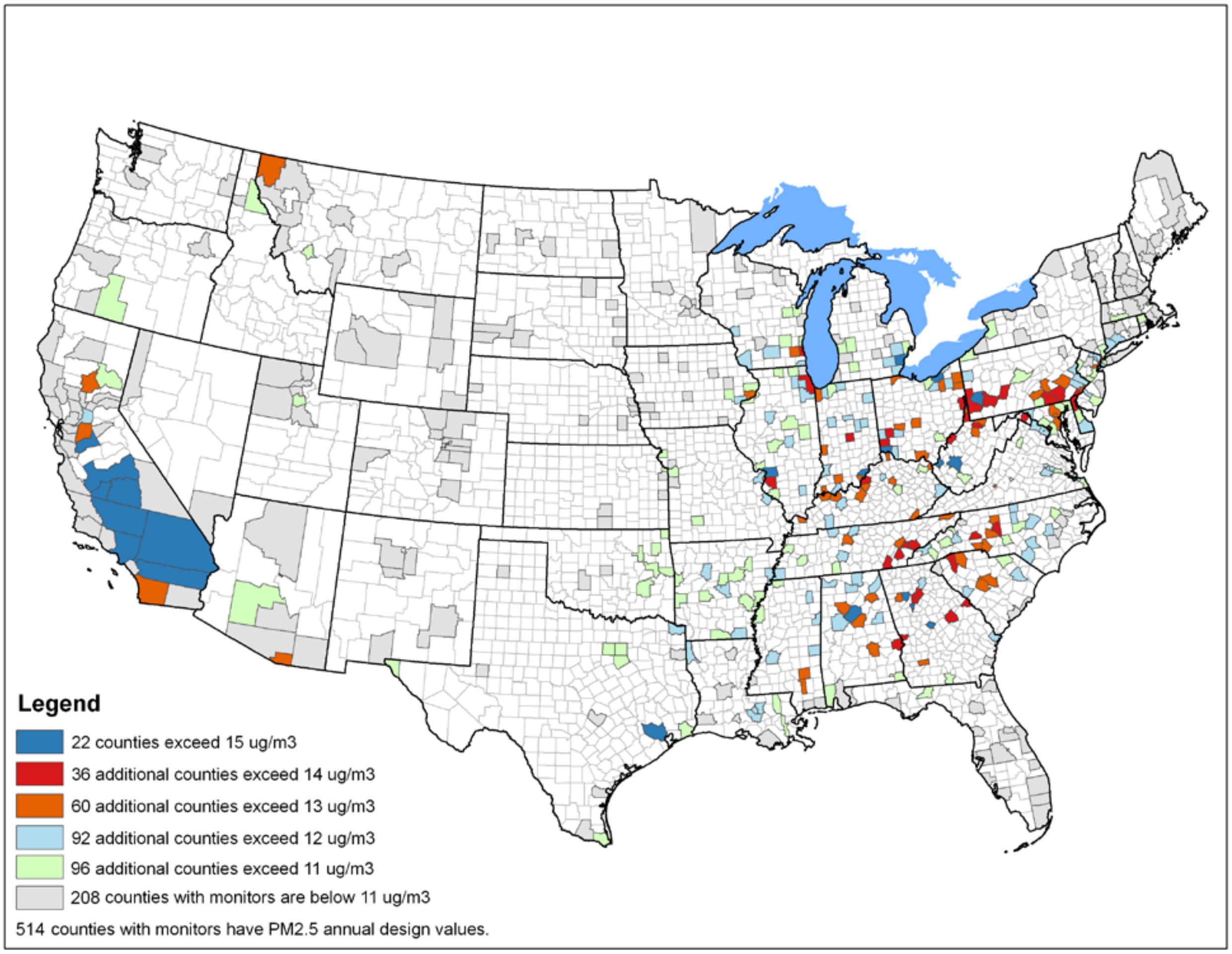


Figure 2-5. Maximum County-level $PM_{2.5}$ Annual Design Values Calculated Using 2006–2008 FRM 24-hr Average $PM_{2.5}$ Measurements.

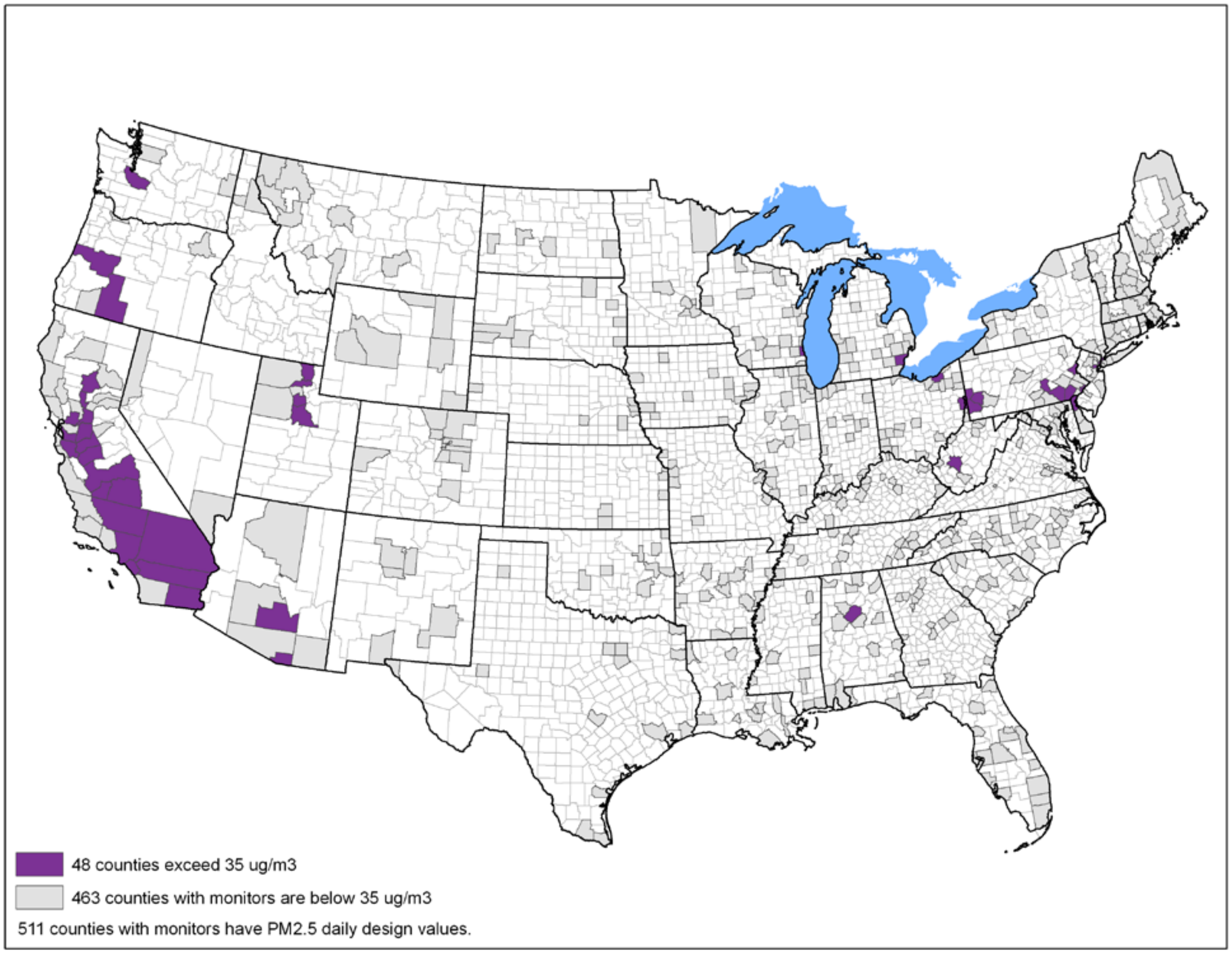


Figure 2-6. Maximum County-level $PM_{2.5}$ 24-hour Design Values Calculated Using 2006–2008 FRM 24-hr Average $PM_{2.5}$ Measurements.

2.3 References

Abt Associates, 2010. User's Guide: Modeled Attainment Test Software. http://www.epa.gov/scram001/modelingapps_mats.htm.

Frank, N.H., Retained Nitrate, Hydrated Sulfates, and Carbonaceous Mass in Federal Reference Method Fine Particulate Matter for Six Eastern U.S. Cities, J. Air & Waste Manage. Assoc. 2006, 56, 500-511.

U.S. Environmental Protection Agency (EPA). 2004. Air quality criteria for particulate matter. U.S. Environmental Protection Agency. Research Triangle Park, NC. EPA/600/P-99/002aF-bF.

U.S. Environmental Protection Agency (EPA). 2008. National Air Quality Status and Trends Through 2007. U.S. Environmental Protection Agency. Research Triangle Park. EPA-454/R-08-006.

U.S. Environmental Protection Agency (EPA). 2009. Integrated Science Assessment for Particulate Matter. U.S. Environmental Protection Agency. Research Triangle Park. EPA/600/R-08/139F.

U.S. Environmental Protection Agency (EPA). 2011. Policy Assessment for the Review of the Particulate Matter National Ambient Air Quality Standards. Office of Air Quality Planning and Standards, Research Triangle Park, NC.

U.S. Environmental Protection Agency (EPA). 2012. 2008 National Emissions Inventory, Version 2 Data and Documentation available from http://www.epa.gov/ttn/chief/net/2008inventory.html. Office of Air Quality Planning and Standards, Research Triangle Park, NC.

CHAPTER 3
AIR QUALITY MODELING AND ANALYSIS

3.1 Synopsis

In order to evaluate the health and environmental impacts of trying to reach the alterative primary standards in this final RIA, it was necessary to use models to predict concentrations in the future. The data, tools and methodology used for projecting future-year air quality are described in this chapter, as well as the post-processing techniques used to produce a number of air quality metrics for input into the analysis of costs and benefits.

3.2 Modeling $PM_{2.5}$ Levels in the Future

A national scale air quality modeling analysis was performed to estimate $PM_{2.5}$ concentrations for the annual and 24-hour primary standards for the future year of 2020.[1] Air quality ratios were then developed using model responsiveness to emissions changes based on "sensitivity" air quality modeling that was designed to determine the response of $PM_{2.5}$ concentrations to reductions in emissions of SO_2, NO_x, and directly emitted $PM_{2.5}$. The air quality ratios were used to determine the amount of emissions reductions needed to attain the revised annual standard of 12 $\mu g/m^3$ and two alternative annual standards. The emissions reductions were then used to estimate how air quality would change under each set of emissions scenarios. These data were used as inputs to the calculation of expected costs and benefits associated with the emissions and air quality changes resulting from just attaining the revised and alternative annual standards.

As described in section 3.3, air quality modeling was used in a relative sense to project future concentrations of $PM_{2.5}$. As part of this approach air quality model predictions from a base year simulation are coupled with predictions from the future case to calculate the relative change (between base year and future case) in each species component of $PM_{2.5}$. These species-specific relative response factors (RRFs) are applied to the corresponding measured concentrations to estimate future species concentrations. The future case $PM_{2.5}$ annual and daily design values are then calculated using the projected species concentrations. We used 2007 as the base year and 2020 as the future year for air quality-related analyses in this RIA. For 2020 we modeled two emissions scenarios, a 2020 base case and a 2020 control case. The 2007 and 2020 scenarios were modeled as annual model simulations. In addition to these emissions scenarios, we also performed several emissions sensitivity model runs to quantify the response

[1] In addition, we used air quality modeling to estimate light extinction in 2020 to support the analysis of the welfare benefits of this rule.

of $PM_{2.5}$ to various precursor emissions. The modeling for the 2020 base case, the 2020 control case, and sensitivity scenarios were used to inform the development of design values for the baseline which provides for attainment of the 15/35 NAAQS and the incremental emissions reductions needed to attain the revised 12 $\mu g/m^3$ annual standard and two alternative annual standards, 13 $\mu g/m^3$ and 11 $\mu g/m^3$. Details on the 2007-based air quality modeling platform, the 2007 base year and 2020 base case scenarios, and the methods and results for attaining these NAAQS levels are provided below. Information on the 2020 control case can be found in Chapter 4 of this RIA.

3.2.1 Air Quality Modeling Platform

The 2007-based Community Multi-scale Air Quality (CMAQ) modeling platform was used as the tool to project future-year air quality for 2020 and to estimate the costs and benefits for attaining the current and revised alternative NAAQS presented in this assessment. This platform provides the most recent, complete set of base year emissions information currently available for national scale modeling. In addition to the CMAQ model and the emissions data, the modeling platform includes the meteorology, and initial and boundary condition data for 2007 which are inputs to this model. The CMAQ model is a three-dimensional grid-based Eulerian air quality model designed to estimate the formation and fate of oxidant precursors, primary and secondary particulate matter concentrations and deposition over regional and urban spatial scales (e.g., over the contiguous U.S.) (Appel et al., 2008; Appel et al., 2007; Byun and Schere, 2006). Consideration of the different processes (e.g., transport and deposition) that affect primary (directly emitted) and secondary (formed by atmospheric processes) PM at the regional scale in different locations is fundamental to understanding and assessing the effects of pollution control measures that affect PM, ozone and deposition of pollutants to the surface. Because it accounts for spatial and temporal variations as well as differences in the reactivity of emissions, CMAQ is useful for evaluating the impacts of the control strategies on $PM_{2.5}$ concentrations. Version 4.7.1 of CMAQ was employed for this RIA modeling.[2] CMAQ is applied with the AERO5 aerosol module, which includes the ISORROPIA inorganic chemistry (Nenes et al., 1998) and a secondary organic aerosol module (Carlton et al., 2010). The CMAQ model is applied with sulfur and organic oxidation aqueous phase chemistry (Carlton et al., 2008) and the carbon-bond 2005 (CB05) gas-phase chemistry module (Yarwood et al., 2005).

[2] More information is available online at: www.cmaq-model.org

3.2.1.1 Air Quality Modeling Domain

Figure 3-1 shows the geographic extent of the modeling domain that was used for air quality modeling in this analysis. The domain covers the 48 contiguous states along with the southern portions of Canada and the northern portions of Mexico. This modeling domain contains 24 vertical layers with a top at about 17,600 meters, or 50 millibars (mb). A horizontal resolution of 12 x 12 km was used for modeling the 2007 base year and the 2020 base and control strategy scenarios. The model simulations produce gridded air quality concentrations on an hourly basis for the entire modeling domain.

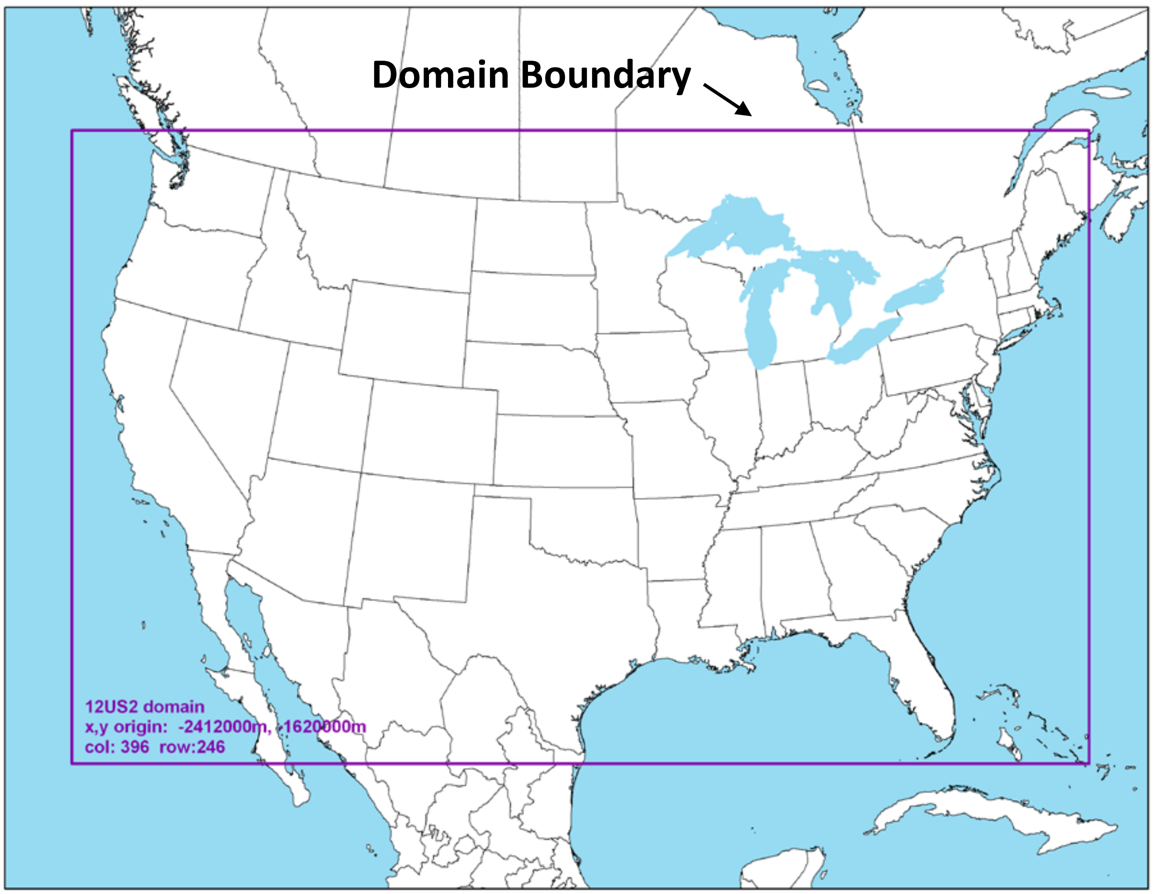


Figure 3-1. Map of the CMAQ Modeling Domain Used for PM NAAQS RIA

3.2.1.2 Air Quality Model Inputs

CMAQ requires a variety of input files that contain information pertaining to the modeling domain and simulation period. These include gridded, hourly emissions estimates and meteorological data, and initial and boundary conditions. Separate emissions inventories were prepared for the 2007 base year and the future year of 2020 base case and control strategy scenarios. All other inputs were specified for the 2007 base year model application and remained unchanged for each future-year modeling scenario.

CMAQ requires detailed emissions inventories containing temporally allocated (i.e., hourly) emissions for each grid-cell in the modeling domain for a large number of chemical species that act as primary pollutants or precursors to secondary pollutants. The annual emission inventories, described in Section 3.2.2, were preprocessed into CMAQ-ready inputs using the SMOKE emissions preprocessing system[3]. Meteorological inputs reflecting 2007 conditions across the contiguous U.S. were derived from Version 3.1 of the Weather Research Forecasting Model (WRF). These inputs included hourly-varying horizontal wind components (i.e., speed and direction), temperature, moisture, vertical diffusion rates, and rainfall rates for each grid cell in each vertical layer. Details of the annual 2007 meteorological model simulation and evaluation are provided in a separate technical support document (EPA, 2011a).

The lateral boundary and initial species concentrations are provided by a three-dimensional global atmospheric chemistry model, the GEOS-CHEM model version 8-02-03 (Yantosca, 2004)[4]. The global GEOS-CHEM model simulates atmospheric chemical and physical processes driven by assimilated meteorological observations from the NASA's Goddard Earth Observing System (GEOS). This model was run for 2007 with a grid resolution of 2.0 degrees x 2.5 degrees (latitude-longitude) and 47 vertical layers. The predictions were used to provide one-way dynamic boundary conditions at three-hour intervals and an initial concentration field for the CMAQ simulations. A GEOS-Chem evaluation was conducted for the purpose of validating the 2007 GEOS-Chem simulation for predicting selected measurements relevant to their use as boundary conditions for CMAQ. This evaluation included reproducing GEOS-Chem evaluation plots reported in the literature for previous versions of the model (Lam, 2010).

3.2.1.3 Air Quality Model Evaluation

An operational model performance evaluation for $PM_{2.5}$ and its related speciated components (e.g., sulfate, nitrate, elemental carbon, organic carbon) was performed to estimate the ability of the CMAQ modeling system to replicate 2007 measured concentrations[5]. This evaluation principally comprises statistical assessments of model predictions versus observations paired in time and space depending on the sampling period of measured data. Details on the evaluation methodology and the calculation of performance statistics are

---

[3] More information is available online at: www.smoke-model.org.

[4] More information is available online at: http://www-as.harvard.edu/chemistry/trop/geos.

[5] This operational evaluation for CMAQ included statistical and graphical comparisons of model predictions for select $PM_{2.5}$ component species to the corresponding measured data from monitoring sites in the Continuous Speciation Network (CSN), the Interagency Monitoring of PROtected Visual Environments (IMPROVE) network, and the Clean Air Status and Trends Network (CASTNet).

provided in the Technical Support Document: Air Quality Modeling for the Final PM NAAQS (AQMTSD, EPA, 2012a). Overall, the model performance statistics for sulfate, nitrate, organic carbon, and elemental carbon from the CMAQ 2007 simulation are within or close to the ranges found in other recent applications. These model performance results give us confidence that our applications of CMAQ using this 2007 modeling platform provide a scientifically credible approach for assessing $PM_{2.5}$ concentrations for the purposes of the RIA.

3.2.1.4 Emissions Inventory

The 2007 emissions inventory and the 2020 base case emissions inventory were developed using the 2007 Version 5.0 emissions modeling platform (documentation and data files available from http://www.epa.gov/ttn/chief/emch/index.html). The starting point for the 2007v5 platform was Version 2 of the 2008 National Emissions Inventory (http://www.epa.gov/ttn/chief/net/2008inventory.html). The 2008 NEI v2 is the most recently available NEI. The next NEI will be developed for 2011. Data collection for the 2011 NEI is ongoing through the end of 2012, with the inventory due to be published in 2013. Some data in the 2008 NEI v2 were adjusted to better represent 2007 for this analysis. For example, MOVES 2010b was used to compute onroad emissions and duplicate emissions values were removed where they were identified. For additional details, see the Technical Support Document: Preparation of Emissions Inventories for the Version 5.0, 2007 Emissions Modeling Platform (EITSD, EPA, 2012b). The 2020 base case inventory is the starting point for the baseline and control strategy modeling performed for this assessment. The above-referenced EITSD (EPA, 2012b) describes the development of the 2007 base year inventory in detail for all emissions sectors, along with the projection methodology applied to develop the 2020 base case inventory.

The 2020 EGU projected inventory represents demand growth, fuel resource availability, generating technology cost and performance, and other economic factors affecting power sector behavior. It also reflects the expected 2020 emissions effects due to environmental rules and regulations, consent decrees and settlements, plant closures, units built or with control devices updated since 2007, and forecast unit construction through the calendar year 2020. In this analysis, the projected EGU emissions include the Final Mercury and Air Toxics (MATS) rule announced on December 21, 2011 and the Final Cross-State Air Pollution Rule (CSAPR) issued on July 6, 2011.

On August 21, 2012, the D.C. Circuit Court of Appeals issued an opinion vacating CSAPR. In its decision, the Court also instructed EPA to “continue administering CAIR [the 2005 Clean

Air Interstate Rule] pending the promulgation of a valid replacement." In the interim, the EPA and the states are continuing to implement CAIR to address regional transport of air pollution, as directed by the Court. The EPA has filed a petition for rehearing of the Court's decision on CSAPR. In light of the Court's instructions, the EPA believes that it is appropriate to rely on CAIR emission reductions as permanent and enforceable reductions until such a time as the EPA issues a replacement transport rule.

Because of the similarity in emissions reductions associated with CSAPR and CAIR[6], and the inclusion of MATS in the RIA baseline, EPA has determined that it remains appropriate that CSAPR continue to be used in the RIA baseline as a proxy for representing the emission reductions required by CAIR for the purposes of the rulemaking's modeling projections for 2020.

Regarding the impact of MATS on this determination, the MATS emission rate standard for hydrogen chloride (HCl) is expected to result in a substantial amount of new pollution controls (scrubbers and dry sorbent injection) and upgrading of existing scrubbers that will also significantly reduce SO2 emissions from power plants. MATS implementation is projected to reduce nationwide SO2 emissions from power plants to a level more than 40 percent lower than the SO2 emissions projected under CSAPR without MATS in place (EPA-HQ-OAR-2009-0234-20131).

In addition to these conclusions, the EGU baseline used in modeling the PM NAAQS was based on EIA's AEO 2010 and represents a conservative approach to emission projections, given that more recent trends in power sector economics suggest a likelihood of lower future EGU emissions. This is supported by the results of a sensitivity analysis conducted using the electricity demand forecast from EIA's AEO 2012 that shows slightly lower EGU emissions. It is reasonable to expect that recent reductions in gas prices and increases in coal prices would yield yet lower estimations of future EGU emissions in the context of this rule's analysis. The details of this analysis can be found in the memo titled AEO 2012 Demand Sensitivity, which is available in the docket.

The EGU emissions were developed using the Integrated Planning Model (IPM) Version 4.10 Final MATS and are documented in detail at http://www.epa.gov/airmarkt/progsregs/epa-ipm/toxics.html. IPM is a multiregional, dynamic, deterministic linear programming model of the U.S. electric power sector. Note that for this analysis, no further EGU control measures

[6] U.S. EPA, Cross-State Air Pollution Rule Presentation, December 15, 2011, available at http://www.epa.gov/airtransport/pdfs/CSAPRPresentation.pdf, p. 15.

were selected for illustrating attainment of the current and proposed alternative standard levels discussed in Chapter 4. Thus, the EGU emissions are unchanged between the future-year base-case and the control strategies.

Table 3-1 provides a comprehensive list of all the control programs, growth assumptions, and facility and unit closures information in the future year base case. The future-year base non-EGU stationary source emissions inventory includes all enforceable national rules and programs including the Reciprocating Internal Combustion Engines (RICE) and cement manufacturing National Emissions Standards for Hazardous Air Pollutants (NESHAPs) and Boiler Maximum Achievable Control Technology (MACT) reconsideration reductions. Many state and local control programs are also applied where those programs were finalized and enough details were available to apply reductions to the 2007 emissions data.

The 2007 and 2020 onroad mobile source emissions were developed using emissions factors derived from the MOtor Vehicle Emission Simulator (MOVES)[7] Version 2010b. The emissions were computed by using the Sparse Matrix Operator Kernel Emissions system (SMOKE) to combine the county-, vehicle type-, and temperature-specific emission factors and vehicle miles traveled and vehicle population activity data while taking into account hourly gridded temperature data. For California we received onroad emissions directly from the California Air Resources Board (CARB) in July 2012 for 2007, 2020, and 2025. These emissions were based on the latest available data and models from their SIP development process. We allocated the California onroad emissions down to the hourly, grid-cell, and CMAQ model-species level using ratios derived from the MOVES-based emissions data output from SMOKE.

The MOVES-based 2020 onroad emissions account for changes in activity data and the impact of on-the-books national rules including: the Light-Duty Vehicle Tier 2 Rule, the Heavy Duty Diesel Rule, the Mobile Source Air Toxics Rule, the Renewable Fuel Standard, the Light Duty Green House Gas/Corporate Average Fuel Efficiency (CAFE) standards for 2012-2016, and the Heavy-Duty Vehicle Greenhouse Gas Rule. The emissions do not account for the 2017 and Later Model Year Light-Duty Vehicle Greenhouse Gas Emissions and Corporate Average Fuel Economy Standards; Final Rule (LD GHG), issued October 15, 2012. The LD GHG rule was not included in this analysis because the rule was not signed at the time the modeling was performed, and it is expected to have little impact on particulate matter emissions. The RIA for the LD GHG (EPA, 2012c) shows that in 2030 counties are showing decreases in PM 2.5 design values of up to 0.16 $\mu g/m^3$. The modeling indicates that the majority of the modeled counties

[7]More information is available online at: http://www.epa.gov/otaq/models/moves/index.htm.

will experience small changes of between 0.05 μg/m^3 and -0.05 μg/m^3 in their annual PM2.5 design values due to the vehicle standards. The impacts of the rule in 2020 should be even less than the 2030 impacts. The MOVES-based 2020 emissions include state rules related to the adoption of LEV standards, inspection and maintenance programs, Stage II refueling controls, and local fuel restrictions. For California, the provided future year 2020 and 2025 emissions included most on-the-books regulations such as those for low sulfur fuel, idling of heavy-duty vehicles, chip reflash, public fleets, trash trucks, drayage trucks, and heavy duty trucks and buses. The zero emission vehicle program prior to adoption of Advanced Clean Cars is included but has a very small impact. The California emissions do not reflect the impacts of the GHG/Smartway regulation, Advanced Clean Cars, nor the low carbon fuel standard because it is assumed that there is no impact on criteria pollutants.

Table 3-1 provides details on the national rules included to develop all categories of mobile source emissions. The nonroad mobile 2020 base emissions, including railroads and commercial marine vessel emissions also include all national control programs. These control programs include the Locomotive-Marine Engine rule, the Nonroad Spark Ignition rule and the Class 3 commercial marine vessel "ECA-IMO" program. The nonroad, locomotive, and class 1 and 2 commercial marine emissions used for California were obtained from CARB, and include nonroad rules reflected in the December 2010 Rulemaking Inventory (http://www.arb.ca.gov/regact/2010/offroadlsi10/offroadisor.pdf), those in the March 2011 Rule Inventory, the Off-Road Construction Rule Inventory for "In-Use Diesel", cargo handling equipment rules in place as of 2011 (see http://www.arb.ca.gov/ports/cargo/cargo.htm), rules through 2011 related to Transportation Refrigeration Units, the Spark-Ignition Marine Engine and Boat Regulations adopted July 24, 2008 for pleasure craft, and the 2007 and 2010 regulations to reduce emissions from commercial harbor craft. For ocean-going vessels, the data represents the 2005 voluntary Vessel Speed Reduction (VSR) within 20 nautical miles, the 2007 and 2008 auxiliary engine rules, the 40 nautical mile VSR program, the 2009 Low Sulfur Fuel regulation, the 2009-2018 cold ironing regulation, the use of 1% sulfur fuel in the ECA zone, the 2012-2015 Tier 2 NOx controls, the 2016 0.1% sulfur fuel regulation in ECA zone, and the 2016 IMO Tier 3 NOx controls. Control and growth-related assumptions in the 2020 base case are described in more detail in the EITSD.

All modeled 2007 and 2020 scenarios use the same year 2006 Canada emissions data. Note that 2006 is the latest year for which Canada provided data, and no accompanying future-year projected inventories were provided in a form suitable for this study. For Mexico, different emissions were used for 2008 and 2018 as described in the Development of Mexico National

Emissions Inventory Projections for 2008, 2012, and 2030 (ERG, 2009) and the associated technical memorandum titled Mexico 2018 Emissions Projections for Point, Area, On-Road Motor Vehicle and Nonroad Mobile Sources (ERG, 2009). All base year and projected emissions inventories are available on the EPA's Emissions Modeling Clearinghouse website at http://www.epa.gov/ttn/chief/emch/index.html.

Table 3-1(a). Control Strategies and Growth Assumptions for Creating 2020 Base Case Emissions Inventories from the 2007 Base Case for Non-EGU Point Sources

Control Strategies and/or Growth Assumptions (Grouped by Affected Pollutants or Standard and Approach)	Pollutants Affected
Non-EGU Point (ptnonipm) Controls and Growth Assumptions	
Boat Manufacturing MACT rule, national, VOC: national applied by SCC	VOC
Consent decrees on companies (based on information from the Office of Enforcement and Compliance Assurance—OECA) apportioned to plants owned/operated by the companies	VOC, CO, NO_x, PM, SO_2
Refinery Consent Decrees: plant/SCC controls	NO_x, PM, SO_2
Commercial/Institutional/Hospital/Medical/Infectious Waste Incinerator Regulations	NO_X, PM, SO_2
NESHAP: Portland Cement (09/09/10)—plant level based on **Industrial Sector Integrated Solutions (ISIS)** policy emissions in 2013. The ISIS results are from the ISIS-Cement model runs for the NESHAP and NSPS analysis of July 28, 2010 and include closures.	Hg, NO_X, SO_2, PM, HCl
New York ozone SIP controls	VOC, NO_X, HAP VOC
Additional plant and unit closures provided by state, regional, and the EPA agencies and additional consent decrees. Includes updates from CSAPR comments.	All
Reciprocating Internal Combustion Engines (RICE) NESHAP with reconsideration	NO_X, CO, PM, SO_2
Ethanol plants that account for increased ethanol production due to RFS2 mandate	All
State fuel sulfur content rules for fuel oil—as of July, 2012, effective only in Maine, Massachusetts, New Jersey, New York and Vermont.	SO_2
Emission reductions resulting from controls put on specific boiler units (not due to MACT) after 2005, identified through analysis of the control data gathered from the Information Collection Request (ICR) from the Industrial/Commercial/Institutional Boiler NESHAP.	NO_X, SO_2, HCl
Emissions reductions resulting from Boiler MACT controls to specific boiler units	NO_X, CO, PM, SO_2, VOC, HCl
Plant and unit closures resulting from state submissions and industry and web postings effective prior to January 2012	All
Aircraft growth via Itinerant (ITN) operations at airports to 2020	All
Livestock Emissions Growth from year 2008 to year 2020 (some farms in the point inventory)	NH_3, PM
Upstream adjustments to year 2020 for refineries and gasoline distribution via the Energy Information and Security Act/Renewable Fuel Standards 2 (EISA/RFS2) impacts	All

Table 3-1(b). Control Strategies and Growth Assumptions for Creating 2020 Base Case Emissions Inventories from the 2007 Base Case for Nonpoint and Onroad Mobile Sources

Control Strategies and/or Growth Assumptions (Grouped by Affected Pollutants or Standard and Approach Used to Apply to the Inventory)	Pollutants Affected
Nonpoint (nonpt sector) Controls and Growth Assumptions	
Residential Wood Combustion Growth and Change-outs from year 2008 to 2020	All
State fuel sulfur content rules for fuel oil—as of July, 2012, effective only in Maine, Massachusetts, New Jersey, New York and Vermont.	SO_2
Reciprocating Internal Combustion Engines (RICE) NESHAP with reconsideration	NO_X, CO, PM, SO_2
New York, Connecticut, and Virginia ozone SIP controls	VOC
Livestock Emissions Growth from year 2008 to year 2020 (some farms in the point inventory)	NH_3, PM
Upstream adjustments to year 2020 for refineries and gasoline distribution via the Energy Information and Security Act/Renewable Fuel Standards 2 (EISA/RFS2) impacts	All
Portable Fuel Container Mobile Source Air Toxics Rule 2 (MSAT2) inventory growth and control from year 2007 to 2020	VOC
Texas oil and gas projections to year 2020	VOC, SO_2, NO_X, CO, PM
Onroad Mobile Controls (list includes all key mobile control strategies but is not exhaustive)	
National Onroad Rules:	All
Tier 2 Rule: Signature date February 2000	
2007 Onroad Heavy-Duty Rule: February 2009	
Final Mobile Source Air Toxics Rule (MSAT2): February 2007	
Renewable Fuel Standard: March 2010	
Light-Duty Greenhouse Gas Emissions Standards and Corporate Average Fuel Efficiency Standards: May 2010	
Heavy (and Medium)-Duty Greenhouse Gas Emissions Standards and Fuel Efficiency Standards: August 2011	
Corporate Average Fuel Economy standards for 2008–2011	
Local Onroad Programs:	VOC
National Low Emission Vehicle Program (NLEV): March 1998	
Ozone Transport Commission (OTC) LEV Program: January 1995	

Table 3-1(c). Control Strategies and Growth Assumptions for Creating 2020 Base Case Emissions Inventories from the 2007 Base Case for Nonroad Mobile Sources

Control Strategies and/or Growth Assumptions (Grouped by Affected Pollutants or Standard and Approach Used to Apply to the Inventory)	Pollutants Affected
Nonroad Mobile Controls (list includes all key mobile control strategies but is not exhaustive) (continued)	
National Nonroad Controls:	All
Clean Air Nonroad Diesel Final Rule—Tier 4: June 2004	
Control of Emissions from Nonroad Large-Spark Ignition Engines and Recreational Engines (Marine and Land Based): “Pentathalon Rule”: November 2002	
Clean Bus USA Program: October 2007	
Control of Emissions of Air Pollution from Locomotives and Marine Compression-Ignition Engines Less than 30 Liters per Cylinder: October 2008	
Locomotive and marine rule (May 6, 2008)	
Marine SI rule (October 4, 1996)	
Nonroad large SI and recreational engine rule (November 8, 2002)	
Nonroad SI rule (October 8, 2008)	
Phase 1 nonroad SI rule (July 3, 1995)	
Tier 1 nonroad diesel rule (June 17, 2004)	
Locomotives:	All
Energy Information Administration (EIA) fuel consumption projections for freight rail	
Clean Air Nonroad Diesel Final Rule—Tier 4: June 2004	
Locomotive Emissions Final Rulemaking, December 17, 1997	
Locomotive rule: April 16, 2008	
Control of Emissions of Air Pollution from Locomotives and Marine: May 2008	
Commercial Marine:	All
Category 3 marine diesel engines Clean Air Act and International Maritime Organization standards (April 30, 2010)—*also includes CSAPR comments.*	
EIA fuel consumption projections for diesel-fueled vessels	
Clean Air Nonroad Diesel Final Rule—Tier 4	
Emissions Standards for Commercial Marine Diesel Engines, December 29, 1999	
Locomotive and marine rule (May 6, 2008)	
Tier 1 Marine Diesel Engines, February 28, 2003	

3.3 $PM_{2.5}$ Modeling Results and Analyses

The air quality modeling results were used in the RIA to estimate future $PM_{2.5}$ concentrations for the 2020 base case and 2020 control case as well as to calculate the air quality ratios that were used in determining the emissions reductions to attain the existing standards of 15/35, the revised annual standard of 12 μg/m^3 and the two alternative annual standards. These data are then used to estimate the costs and benefits of attaining these existing and revised NAAQS levels. Consistent with EPA guidance (EPA, 2007 and EPA, 2011b), the air quality modeling results are applied in a relative sense to estimate 2020 future design values for $PM_{2.5}$ for the 2020 base case and 2020 control case. Air quality response ratios are calculated and used to estimate the tons of emissions reductions needed beyond the 2020 control case needed to show attainment of the existing, revised, and alternative NAAQS levels. Based on the tons of emissions needed in each county, design values are calculated for attaining the revised and alternative annual standard levels for input into the benefits assessment.

The flow diagram shown in Figure 3-2 summarizes our approach for calculating future-year design values for meeting the existing standards, the revised annual standard, and alternative annual standard levels. Table 3-2 describes the specific air quality modeling simulations that informed this approach. The 2020 base case simulation (Box 1) was performed to estimate which monitors would exceed the current and alternative standard levels in 2020 based on emissions reductions expected from existing (i.e., "on-the-books") state and federal control programs. The 2020 control case simulation (Box 3) was performed to estimate the impact of emission reductions from additional controls beyond those of the 2020 base case in areas with design values above the revised and alternative standard levels. As discussed below, the 2020 base case and 2020 control case design values were adjusted to reflect $PM_{2.5}$ reductions expected from the implementation of existing burn ban programs in certain counties and to remove the effects of atypical events such as wildfires and fireworks displays(Boxes 2 and 4). To calculate future-year design values at the different standard levels, and the associated emissions reductions, these 2020 base and control case design values were adjusted downward using air quality response ratios, which give the change $PM_{2.5}$ design value (μg/m^3) per change in emissions by species (Boxes 5 through 9).

The air quality response ratios (hereafter referred to as air quality ratios) used to adjust the 2020 cases to meet the standard levels were calculated based on results of several sensitivity simulations. The sensitivity simulations, as described in Table 3-2, were defined to isolate the changes in the $(NH_4)_2SO_4$, NH_4NO_3 and direct $PM_{2.5}$ associated with changes in emissions of SO_2, NO_x and direct $PM_{2.5}$, respectively. These $PM_{2.5}$ component species were selected for reduction to meet the standard levels because they dominate the mass of $PM_{2.5}$ in the areas of concern in the 2020 cases. The sensitivity simulation referred to as "2020 NO_x_$PM_{2.5}$" was used in calculating the air quality ratios associated with changes in NO_x and direct $PM_{2.5}$ emissions. This simulation was based on anthropogenic NO_x and direct $PM_{2.5}$ emission reductions from non-EGU sources of 25% and 50%, respectively, relative to the 2020 base case. The sensitivity simulation referred to as "2020 SO_2_RWC" was used in calculating the air quality ratios associated with changes in SO_2 emissions. This simulation was based on anthropogenic SO_2 and residential wood combustion emissions reductions from non-EGU sources of 25% and 100%, respectively, relative to the 2020 base case[8]. In the sensitivity runs, emissions reductions for direct $PM_{2.5}$ were generally applied in counties with monitors with annual design values above 11 $\mu g/m^3$ level in the 2020 base case, while emission reductions for NO_x and SO_2 were generally applied in those counties as well as their adjacent counties. This approach reflects the local impacts of direct $PM_{2.5}$ emissions on air quality and the broader geographic impacts on $PM_{2.5}$ of SO_2 and NO_x emissions reductions.

The development of the air quality response ratios used in the process of adjusting the air quality modeling results to meet the current and alternative standard levels is described in Appendix 3.A.1.1. The remainder of this section describes the procedures and the results from the 2020 base case modeling and the development of the adjusted 2020 base case (Box 1 and Box 2, respectively in Figure 3-2) and the identification of the emissions reductions estimated to be needed to attain the 15/35 standard and annual standards of 13, 12, and 11 $\mu g/m^3$ (Boxes 4 through 9 in Figure 3-2).

[8] The results of this sensitivity run were also used in the method to quantify the impacts on design values of existing burn ban programs, as described in Section 3.3.1.1.

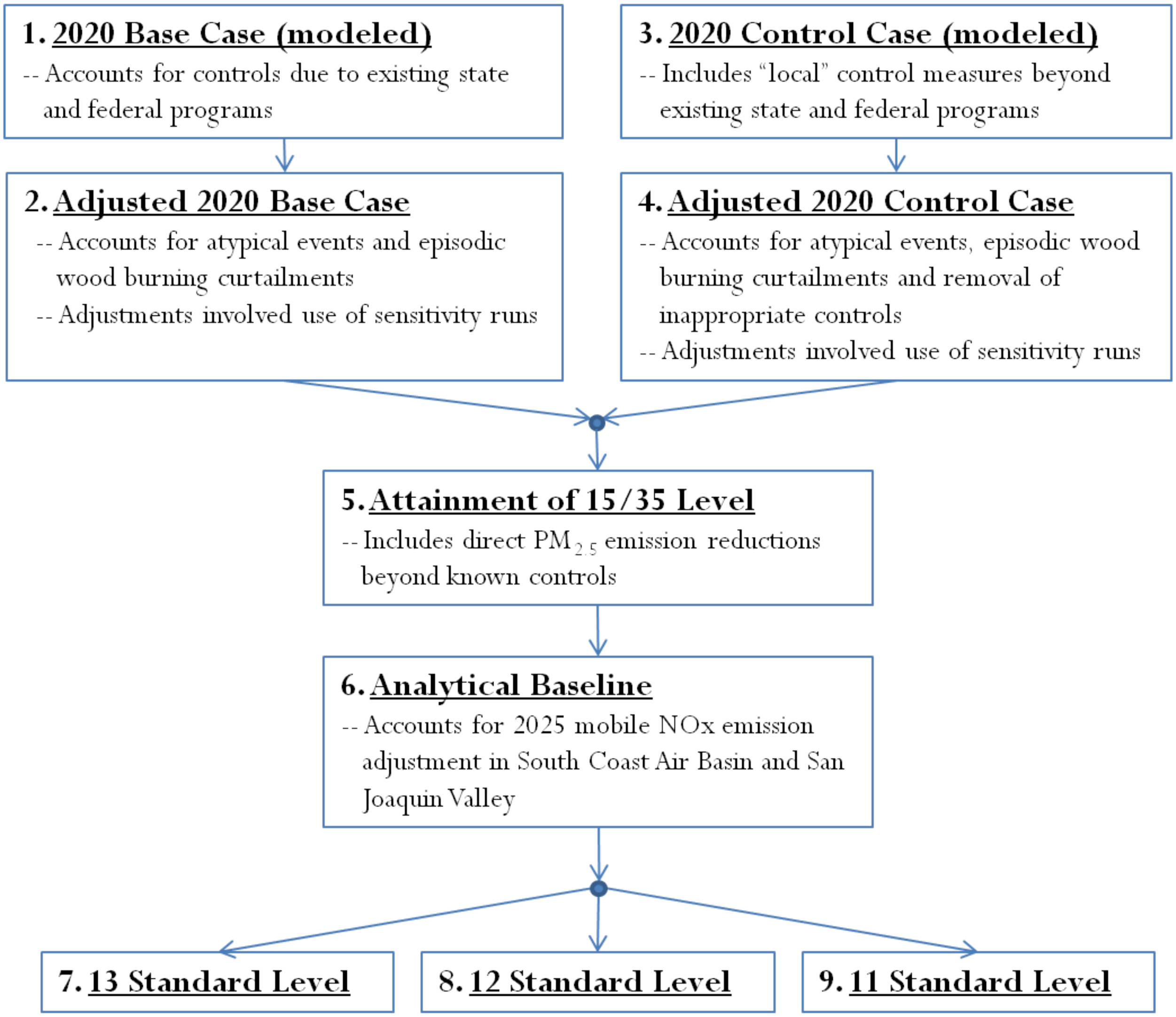


Figure 3-2. Flow Diagram of Process Used to Determine Future-Year Design Values and Associated Emission Reductions for Meeting the Current, Revised and Alternative Standard Levels

Table 3-2. Air Quality Model Simulations Used in this Regulatory Impact Analysis

Simulation	Description	Purpose
2020 base case	Simulation of 2020 that accounts for expected controls due to existing state and federal programs.	Provides estimate of future-year design values based on existing controls
2020 control case	Simulation of 2020 that includes emissions controls beyond the controls of the 2020 base case in areas with design values above the alternative standard levels in the 2020 base case.	Provides impact of additional known controls on design values in target areas; provides basis for meeting the existing, revised and alternative standard levels with emission controls beyond known controls
2020 NO_x_$PM_{2.5}$ sensitivity	Simulation of 2020 where anthropogenic NO_x and $PM_{2.5}$ emissions are decreased by 25% and 50%, respectively, relative to the 2020 base case in selected counties.	Used in estimating the response of air quality to changes in emissions of NO_x and direct $PM_{2.5}$
2020 SO_2_RWC sensitivity	Simulation of 2020 where anthropogenic SO2 and residential wood combustion emissions are decreased by 25% and 100%, respectively, relative to the 2020 base case in selected counties.	Used in estimating the response of air quality to changes in emissions of SO_2 and residential wood combustion
2020 SJV sensitivity	Nine simulations of January 2020. Each simulation has emission reductions relative to the 2020 base case in a one- or two-county group in California's Central Valley. The emission reductions in each county group are the same as those in the 2020 NO_x_$PM_{2.5}$ sensitivity case.	This series of simulations is used to estimate the contributions of emissions from counties in the California's Central Valley on air quality in other counties in the Central Valley

3.3.1 Calculating Future-year Design Values for 2020 Base and Control Cases

To predict the impact of the control strategies on future-year attainment, the air quality model results are used in a relative sense by estimating future-year $PM_{2.5}$ relative response factors (RRFs). RRFs are ratios that are calculated from the modeled changes in $PM_{2.5}$ species concentrations between the base year (2007) and future-year (2020 base case and 2020 control case) air quality modeling results. RRFs are calculated for each $PM_{2.5}$ component (i.e. sulfates, nitrates, organic carbon, etc.). Future-year estimates of the $PM_{2.5}$ annual and 24-hour standard design values at monitor locations are then calculated by applying the species-specific RRFs to ambient $PM_{2.5}$ concentrations from the Federal Reference Method (FRM) Network, which are disaggregated into species concentrations through processing and interpolation of $PM_{2.5}$ species data from the CSN and IMPROVE monitoring networks.

To more easily apply this methodology, EPA has created software, called Modeled Attainment Test Software (MATS) (Abt, 2012) to calculate future-year $PM_{2.5}$ annual and 24-hour standard design values. For this RIA, design values are projected from ambient Federal Reference Method measurements during the period 2005-2009[9] coupled with $PM_{2.5}$ species data from IMPROVE and CSN sites for the 2006–2008 time period. In addition to calculating projected future-year annual and 24-hour standard design values, MATS provides the amounts of sulfate, nitrate, ammonium, elemental carbon, organic carbon and crustal matter that comprise the annual and 24-hour standard design values for each site. These data are useful for understanding the PM species contributing to high $PM_{2.5}$ concentrations which is informative for designing control strategies to reduce the future-year design values to the proposed standard levels.

In order to derive 2020 design values for the purposes of the RIA, we made two additional adjustments to the design value calculations at those monitoring sites that 1) had observed ambient data in the base year period that reflects atypical events or highly variable events that are difficult to predict in the future year, and 2) would be affected by existing local episodic residential wood burning curtailment programs (e.g. "burn ban" programs) that we were not able to simulate in the 2020 base case and control case air quality modeling. These adjustments are described below.

3.3.1.1 Future-year Design Values Adjustments for Episodic Residential Wood Curtailment Programs

A number of Western nonattainment areas have existing rules in place that require the curtailment of residential wood burning (from fireplaces and woodstoves) on an episodic basis. The burning curtailment programs ("burn bans") are implemented at the local level based on local air quality forecasts of high $PM_{2.5}$ days. The burn ban programs vary by area, but are similar in many ways. They generally have "stage 1" (lower concentration $PM_{2.5}$ days) and "stage 2" (higher concentration $PM_{2.5}$ days) level "burn ban" days with mandatory compliance on stage 2 days. The forecast trigger level also varies by area. When the daily $PM_{2.5}$ NAAQS was lowered to 35 $\mu g/m^3$ in 2006 most areas implemented a trigger level at or below 35 $\mu g/m^3$ for a mandatory burn ban[10]. There are also a number of exemptions in each area for residents who use firewood as their sole source of heat. These programs have been strengthened in the last few years to become mandatory and also to address the 35 $\mu g/m^3$ NAAQS. Since all of the

[9] The 2005 -2007 period includes design values 2005-2007, 2006-2008, and 2007-2009.
[10] Some areas previously (before 2007) had voluntary burn ban programs with relatively high trigger levels based on the 1997 daily $PM_{2.5}$ NAAQs (65 $\mu g/m^3$).

identified areas have implemented or significantly strengthened their burn ban programs since 2007, we are assuming little or no reductions from a burn ban program in our 2007 base case and large reductions (on an episodic basis) in the 2020 future year cases.

Due to the complexity of accounting for "burn bans" on specific days in the future year modeling, we were not able to simulate the effects of "burn bans" in the 2020 base case modeling. In this regard, the 2020 model-based design value were adjusted to reflect the expected effects on design values of the episodic residential wood burning curtailment programs. Using the best available information, we estimated the impacts of episodic residential wood burning programs as a post-modeling adjustment to the 2020 base case. For this analysis, episodic residential wood burning adjustments were made for the areas identified in Table 3-3[11]:

Table 3-3. Nonattainment Areas Where Episodic Residential Wood Burning Curtailment was Applied

Nonattainment Areas Where Episodic Residential Wood Burning Adjustments Were Applied	State
Chico	CA
Los Angeles- South Coast Air Basin	CA
Sacramento	CA
San Francisco Bay Area	CA
San Joaquin Valley	CA
Yuba City-Marysville	CA
Klamath Falls	OR
Oakridge	OR
Provo	UT
Salt Lake City	UT
Seattle-Tacoma	WA

We applied two slightly different methodologies employed to adjust the annual average and daily average design values for burn bans in the selected areas. In both cases, the adjustments were based on a modeling sensitivity run that zeroed-out all emissions from the residential wood combustion category on all days of the year. Since the vast majority of residential wood combustion emissions impacts are from primary $PM_{2.5}$ emissions, we calculated the total change in primary organic carbon, elemental carbon, and crustal $PM_{2.5}$

[11] These areas were all predicted to violate the daily NAAQS in the 2020 base case and are known to have mandatory episodic curtailment programs. Adjustments were not applied to areas that solely violated the annual NAAQS or did not have an existing curtailment program. The specific counties in which episodic residential wood burning curtailment programs were applied are listed in Table 3-4.

species between the base and zero-out cases to estimate the impact on $PM_{2.5}$ from residential wood combustion controls.[12]

Since the zero-out model run reduced all residential wood combustion emissions, we had to scale the results of the sensitivity run to provide a realistic estimate of emissions reduction from a burn ban program. To quantify the compliance rate of wood burning curtailment programs we relied upon information from the Sacramento and South Coast Air Quality Management Districts. The South Coast Air Quality Management District (SCAQMD, 2012) estimated a 75% rule effectiveness for their curtailment rule and the Sacramento Metropolitan Air Quality Management District (SCMAQMD, 2009) estimated a 70% reduction in residential wood combustion emissions on burn ban days in their area. Based on this information we assumed a 70% reduction in residential wood combustion emissions on episodic burn ban days in all areas with a mandatory burn ban program. This implies a relatively high level of compliance, but recognizes that the program will provide less than a 100% reduction due to non-compliance and exemptions from the rule.

For the annual NAAQS, we assumed that the burn ban programs provide reductions in $PM_{2.5}$ concentrations that are commensurate with the reduction in primary $PM_{2.5}$ emissions[13]. We also assumed that burn bans are applicable on certain days in the 1st and 4th quarters of the year (i.e., during the residential wood combustion season). The number of days for which we applied the burn ban was based on the observed fraction[14] of measured days above 35 $\mu g/m^3$ in the 1st and 4th quarters in the 2005-2009 base period in each affected county. For multi-county areas, it was assumed that the burn ban control program would be applied by county (i.e. there may be a forecasted burn ban in only a portion of a large nonattainment area). The number of burn ban days applied per year by county is provided in Table 3-4[15].

---

[12] The sensitivity run also included SO2 emissions reductions. The SO2 reductions have no impact on the organic carbon, elemental carbon, and crustal primary $PM_{2.5}$ species concentrations.

[13] Since all of the adjustments are for primary $PM_{2.5}$, it is assumed that emissions reductions and the change in concentration are linear (i.e. a 50% reduction in residential wood combustion $PM_{2.5}$ emissions leads to a 50% reduction in the primary $PM_{2.5}$ concentrations from residential wood combustion.)

[14] FRM monitoring sites operate on different schedules (1 in 3 day, 1 in 6 day, or every day). The calculation was based on the fraction of exceedence days during the 1st and 4th quarters. This proportionality approach normalizes the number of high days between monitoring sites and allows a percentage of days to be applied to the modeled days (which include all days of the year).

[15] The number of burn ban days was based on the monitoring site in the county with the maximum percentage of exceedence days (days > 35 $\mu g/m^3$).

Table 3-4. Estimated Number of Burn Ban Days by County Based on 2005-2009 FRM Data

State	Nonattainment Area	County	Total Number of Burn Ban Days in 1st plus 4th Quarters
California	Chico	Butte	18
California	Los Angeles- South Coast Air Basin	Los Angeles	16
California	Los Angeles- South Coast Air Basin	Riverside	20
California	Los Angeles- South Coast Air Basin	San Bernardino	16
California	Sacramento	Sacramento	20
California	San Francisco Bay Area	Alameda	4
California	San Francisco Bay Area	Santa Clara	8
California	San Francisco Bay Area	Solano	8
California	San Joaquin Valley	Fresno	42
California	San Joaquin Valley	Kern	48
California	San Joaquin Valley	Kings	40
California	San Joaquin Valley	Merced	30
California	San Joaquin Valley	San Joaquin	20
California	San Joaquin Valley	Stanislaus	30
California	San Joaquin Valley	Tulare	38
California	Yuba City-Marysville	Sutter	4
Oregon	Klamath Falls	Klamath	16
Oregon	Oakridge	Lane	20
Utah	Salt Lake City	Salt Lake	16
Utah	Provo	Utah	10
Washington	Seattle-Tacoma	Pierce	16

The 2020 base case model output files were modified to replace the base case modeled concentrations with the burn ban day concentrations on the identified number of days per year (from Table 3-4) at each monitoring site in the 21 counties. The burn ban adjustment was applied to an equal number of high days per quarter in the 1st and 4th quarters (i.e., half of the burn ban days were applied to the high modeled days in the 1st quarter and half to the high modeled days in the 4th quarter). This approach provided burn ban RRFs for the 1st and 4th quarters. The modified 2020 base case predictions were re-run through the MATS tool to calculate adjusted annual average design values which account for the episodic residential wood burning curtailment programs.

A similarly representative burn ban RRF was calculated to adjust the daily design values to account for episodic residential wood burning curtailment programs. Due to the nature of the future year daily design value calculations, the methodology differed slightly from the annual average design value calculations. The daily design value modeled RRFs were calculated from the change in modeled $PM_{2.5}$ species on the 10% highest modeled $PM_{2.5}$ days in each quarter (i.e., the 9 highest modeled days per quarter). In this approach we assume that a burn ban will apply to all high $PM_{2.5}$ days (days > 35 $\mu g/m^3$) in the 1^{st} and 4^{th} quarters at each site. Therefore, we performed the calculation by applying the 70% burn ban adjustment on the 10% highest modeled days in the 1^{st} and 4^{th} quarters. The revised model data were re-run through the MATS tool to calculate an adjusted 2020 base case daily design values which account for the episodic residential wood burning curtailment programs. The impact on the 2020 base case annual design values (where the burn ban adjustments were applied) ranged from 0.03 to 0.68 $\mu g/m^3$. The impact on the 2020 base case daily design values ranged from 0.1 to 13.1 $\mu g/m^3$.The procedures for calculating 2020 control case design values that reflect the effects of the burn ban programs are described in Section 3.3.3. Additional details on the procedures for treating burn ban programs are provided in the AQMTSD.

3.3.1.2 Future-year Design Values Adjustments for Atypical or Unpredictable Events

Concentrations of $PM_{2.5}$ at a number of monitoring sites may be influenced by atypical or unpredictable events such as wildfires or fireworks. In the base year 2005-2009 FRM data, all design value calculations at all sites reflect adjustments to data that EPA officially determined have met the criteria for exclusion under the Exceptional Events Rule (EPA, 2007) during that base year period. However, under a future year scenario it is possible that some atypical events would qualify as exceptional events even though they did not qualify in the base 2005-2009 period. This is due to the nature of the “but for” test in the Exceptional Events Rule. The rule states that exceptional events cannot be removed from the design value calculations unless the monitor would not violate the NAAQS, “but for” the exceptional events. There are a number of sites that are above the current daily $PM_{2.5}$ NAAQS in the 2005-2009 period and would also **continue to violate the current NAAQS** even if certain atypical event days were removed. Therefore, those days cannot be removed from the official design value calculations for 2005-2009 period because they do not meet the “but for” test. However, in the future year 2020 projections, we assume for analysis purposes that the impact of certain atypical or highly variable events would meet the “but for” test. This is a reasonable assumption because at a certain point in our modeling analysis the design value at each violating site is reduced to a value that is slightly above the NAAQS level such that the site would attain the NAAQS “but for” the atypical events days.

The identification of atypical event data that could affect future design value calculations could involve an extensive data analysis exercise. It would be difficult to identify every potentially important past event for each monitoring site in the country and completely characterize the exact nature of those days as part of this RIA. Therefore, we limited the analysis to a small group of monitoring sites where a few atypical event days may have an important impact on the future year design value calculations. In our analysis of potentially important atypical events we included only monitoring sites that have 24-hour design values predicted to violate the 35 ug/m3 daily NAAQS in the 2020 base case. At these sites we examined the concentrations on days with daily average measurements > 35 μg/m^3. There were several categories of potentially important atypical event days that we identified:

1. Wildfires—Summer days with high concentrations at sites in the West which normally do not exceed the NAAQS in the summer[16] [62 site-days];

2. Fireworks—High $PM_{2.5}$ concentrations predominantly on July 4th or 5th [37 site-days];

3. Other unusual high data—Other site-days with very high measured $PM_{2.5}$ concentrations that were much higher than concentrations on the same days at surrounding sites [2 site-days].

Based on this assessment, we identified 101 site-days in the above categories at 25 monitoring sites (23 of them in California) in the period 2005-2009[17,18]. In all of the subsequent future year design value calculations (for both the annual and daily NAAQS), the design values have been adjusted to reflect the removal of these days. The impact on the 2005-2009 annual design values at these 25 sites ranged from 0.08 to 0.97 μg/m^3. The impact on the daily design values ranged from 0 to 12.9 μg/m^3.

We recalculated the 2020 base case and 2020 control case annual and daily $PM_{2.5}$ design values to reflect the removal of potential future atypical event days from the starting point 2005-2009 ambient measured data. Additional details on the methodology for adjusting the future year design values for the purposes of this analysis are provided in the AQMTSD.

[16] The vast majority of the wildfires days occurred during a well documented summer 2008 wildfire period in Central California.
[17] These were site days that were not already identified and removed from the ambient data as EPA-concurred exceptional events.
[18] The adjustments are made to the base year design values for the sole purpose of projecting ambient data to the future year (2020). It is not appropriate to adjust the base year 2005-2009 data for the purpose of examining current or past attainment of the NAAQS.

3.3.2 *Calculating Future-year Design Values for Meeting the Existing Standards, the Revised Annual Standard, and Alternative Annual Standard Levels*

The air quality ratios were used in the process of adjusting the air quality modeling results to meet the current and alternative standard levels. The 2020 base case modeling and the development of the adjusted 2020 base case (Box 1 and Box 2, Figure 3-2) were described above. The procedures for determining the emissions reductions estimated to be needed to attain the 15/35 standard and annual standards of 13, 12, and 11 $\mu g/m^3$ are identified in boxes 4 through 9 in Figure 3-2. These procedures and the results are described below.

Adjusted 2020 Control Case (Box 4). *Adjust design values of 2020 control case to account for episodic wood burning curtailments and to account for atypical events and inappropriate emissions controls.* The impact of atypical events on design values was removed from 2020 control case design values by removing these days from the ambient data used in the future-year design value calculations in the MATS tool, as described in section 3.3.1.2, above. To account for the impacts of wood burning curtailments in the 2020 control case, we started with the fractional change (i.e., RRF) in speciated design values between the 2020 base case and the 2020 control case (both cases without the effects of wood burning curtailment programs). We then applied these species-specific RRFs to adjust the corresponding speciated design values in the 2020 base case that reflects the application of wood burning curtailments[19].

Attainment of the 15/35 Level (Box 5). *Estimate future-year design values and emission reductions beyond the adjusted 2020 control case to meet the existing standard level (15/35).* For monitors with design values greater than 15/35 in the adjusted 2020 control case (Box 4, Figure 3-2), additional direct $PM_{2.5}$ emission reductions were applied to meet this level. The additional direct $PM_{2.5}$ emission reduction amounts were estimated using air quality ratios. The direct $PM_{2.5}$ emissions reductions needed to attain the 15/35 standard were also applied to reduce $PM_{2.5}$ design values at all attaining monitoring sites in the same county as the nonattainment monitor. For example, the highest 24-hr design value in San Bernardino County in the adjusted 2020 control case was 36.4 $\mu g/m^3$ at monitor 60719004. Additional emissions reductions of 585 tons of direct $PM_{2.5}$ were estimated to be required for this monitor to meet

[19] We also had to adjust the 2020 modeled control case design values in certain counties to remove the impacts from a subset of control measures. These control measures were deemed to be inappropriate for the purposes of the 2020 control case after the 2020 base case air quality modeling was completed. To remove the impact of these inappropriate emissions controls, the design values were adjusted either based on the air quality ratios or on the change in the design value between the 2020 base case and 2020 control case scaled by the fraction of the total emission reduction associated with the inappropriate controls.

the 24-hr standard level[20] as follows: (36.4 − 35.4) / 1.710 × 1000 = 585 tons, where 1.710 is the 24-hr direct $PM_{2.5}$ air quality ratio for the monitor 60719004 (Table 3.A-3). The 585 tons of direct $PM_{2.5}$ emissions reductions in this county were estimated to reduce the highest annual design value in San Bernardino at monitor 60710025 from 13.41 to 12.99 µg/m^3 as follows: 13.41 – (585 × 0.710 / 1000) = 12.99 µg/m^3, where 0.710 is the annual direct $PM_{2.5}$ air quality ratio for the 60710025 monitor (Table 3.A-3). The direct $PM_{2.5}$ emission reduction amounts beyond the adjusted 2020 control case that are necessary to meet the current standard level for individual counties are listed in Table 3-5.

Table 3-5. Tons of Direct $PM_{2.5}$ Emission Reductions beyond the Adjusted 2020 Control Case to Meet the Current Standard Level for Counties that Exceed the Revised or Alternative Annual Standard Levels in the Adjusted 2020 Base Case

FIPS Code	State Name	County Name	Direct $PM_{2.5}$ Emissions (tons)
6019	California	Fresno	497
6025	California	Imperial	288
6029	California	Kern	1,496
6031/6107	California	Kings/Tulare	610
6071	California	San Bernardino	585
6099	California	Stanislaus	346
42003	Pennsylvania	Allegheny	764

Emissions were controlled in certain counties in the 2020 control case that exceeded the alternative annual standard of 11 µg/m^3 but that did not exceed the existing standard level. These emissions controls are relevant for meeting the 11 µg/m^3 level (Box 9) but are not relevant for meeting the existing standard level. Therefore annual design values in the 15/35 case are set to those of the adjusted 2020 base case for monitors in the following counties: Jefferson, AL; Shoshone, ID; Cook, IL; Madison, IL; Klamath, OR; Lake, IN; Scott, IA; Wayne, MI; Milwaukee, WI; and Harris, TX.[21]

[20] A 24-hour design value of 35.4 µg/m^3 is the highest value that meets the 24-hour standard.

[21] Arrows point from Box 2 and Box 4 to Box 5 in Figure 3-2 because information from both the adjusted 2020 base case and the adjusted 2020 control case was used in developing the set of design values that correspond to attainment of the existing standard.

Analytical Baseline (Box 6). *Create analytical baseline for meeting alternative standards that accounts for 2025 mobile NOx emission adjustment in San Joaquin Valley and South Coast Air Basin.* The goal of this RIA is to provide the best estimates of the costs and benefits of an illustrative attainment strategy to just meet the revised annual 12 $\mu g/m^3$ standards, as well as two alternative annual standards of 13 $\mu g/m^3$ and 11 $\mu g/m^3$, incremental to just meeting the current standards of 15/35, and reflecting emissions projected to reflect the impact of economic growth and implementation of state and federal emissions controls. Most areas of the U.S. will be required to demonstrate attainment with the new standards by 2020. As a result, for these areas, the correct baseline for estimating the incremental emissions reductions that would be needed to attain the more protective standards is a baseline with emissions projected to 2020 and adjusted to reflect the additional emissions reductions that would be needed to attain the current 15/35 standards. For two areas in Southern California (South Coast and San Joaquin), the degree of projected non-attainment with the revised annual standard of 12 $\mu g/m^3$ is high enough that those counties are not expected to be able to demonstrate attainment with the new standard by 2020. Instead, those two areas are likely to qualify for an (up to) five year extension of their attainment date. If the areas are granted an attainment date extension, they will have until 2025 to demonstrate attainment with the revised annual standard. As a result, for these two areas, the correct baseline for estimating the incremental emissions reductions that would be needed to attain the more protective standards is a baseline with emissions projected to 2025 adjusted to reflect the additional emissions reductions that would be needed to attain the current 15/35 standards. This difference in attainment year is important because between 2020 and 2025, emissions from mobile sources in California are expected to be reduced due to continued fleet turn over from older, higher emitting vehicles to newer, lower emitting vehicles. These reductions in emissions will occur as a result of previous state rules for which costs and benefits have already been counted, and thus will not be costs and benefits attributable to meeting the revised annual standard.

Modeling of two separate years is time prohibitive, and would result in two separate years of benefits and costs which would not provide a complete picture of the nationwide costs and benefits of just meeting the new standards in either 2020 or 2025 because of differences in the baselines between the two years. To provide the most reasonable and reliable estimates of costs and benefits of full attainment for the nation, we are constructing an analytical baseline for estimating the costs and benefits of attaining the revised standard of 12 $\mu g/m^3$ and alternative annual standards of 13 $\mu g/m^3$ and 11 $\mu g/m^3$ with the following characteristics. The analytical baseline was developed by applying a mobile NOx emission adjustment to design values at levels attaining 15/35. This approach allows us to generate costs and benefits of full attainment without overstating the costs and benefits in those two areas, which would occur if we forced costly emissions reductions in 2020 in areas that would not have to be incurred until 2025, and which will be offset because of the expected reductions in mobile source emissions due to other programs.[22]

The emissions adjustment is equal to 27,467 tons of NOx emissions reductions in the South Coast Air Basin and 14,410 tons in the San Joaquin Valley. Annual design values for the 15/35 baseline were adjusted to account for these emissions reductions using the air quality ratios listed in Table 3.A-1. Incremental costs and benefits of the revised and alternative standards are assessed relative to this set of analytic baseline design values. Annual design values and exceedance categories are provided for the analytic baseline in Table 3-6 and Figure 3-3 for counties with at least one monitor that exceeds a level.[23]

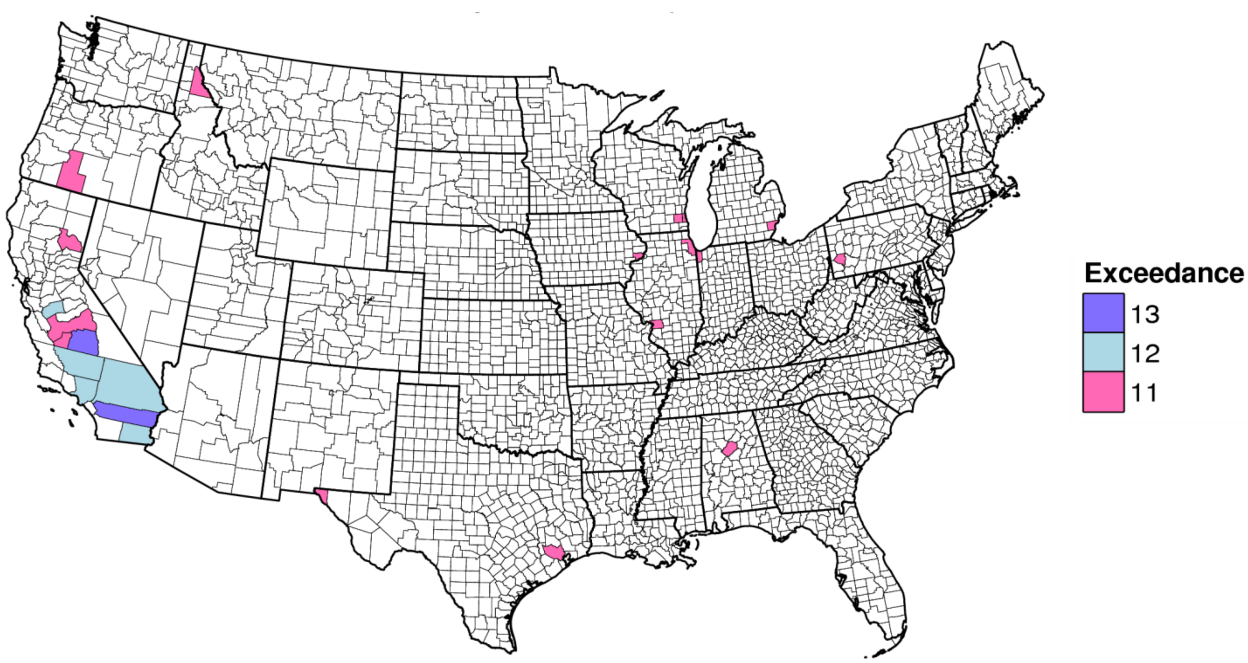


Figure 3-3. Counties that Exceed the Revised and/or Alternative Annual Standard Levels of 13, 12 and 11 $\mu g/m^3$ in the Analytical Baseline

[22] Benefits for all areas are estimated using 2020 population data for consistency, recognizing that full attainment costs and benefits will not actually be realized until 2025 for a portion of the costs and benefits. The 2020 estimates of full attainment costs and benefits will be an underestimate of benefits in 2025 because of population growth and changes in the age distribution of the population between 2020 and 2025.

[23] There were two counties (Lincoln County, MT and Santa Cruz County, AZ) that exceeded alternative standard levels in the 2020 base case for which we used a weight-of-evidence approach to determine how they would attain these levels, as described in Section 3.3.5.

Table 3-6. Annual Design Values and Exceedance Category for the Highest County Monitor in the Analytical Baseline for Counties with at Least one Monitor Above the Revised and/or Alternative Standard Levels

FIPS Code	Monitor ID	State Name	County Name	Annual DV	13/35	12/35	11/35
6065	60658005	California	Riverside	14.58	x	x	x
6107	61072002	California	Tulare	13.23	x	x	x
6029	60290016	California	Kern	12.7		x	x
6071	60710025	California	San Bernardino	12.64		x	x
6025	60250005	California	Imperial	12.57		x	x
6037	60371002	California	Los Angeles	12.34		x	x
6047	60472510	California	Merced	12.12		x	x
55079	550790059	Wisconsin	Milwaukee	12.02			x
6031	60310004	California	Kings	11.79			x
17119	171191007	Illinois	Madison	11.7			x
6019	60190008	California	Fresno	11.61			x
26163	261630033	Michigan	Wayne	11.58			x
1073	10730023	Alabama	Jefferson	11.56			x
17031	170316005	Illinois	Cook	11.52			x
16079	160790017	Idaho	Shoshone	11.52			x
19163	191630019	Iowa	Scott	11.51			x
48201	482011035	Texas	Harris	11.43			x
48141	481410044	Texas	El Paso	11.39			x
41035	410350004	Oregon	Klamath	11.3			x
55133	551330027	Wisconsin	Waukesha	11.22			x
18089	180891003	Indiana	Lake	11.17			x
6063	60631009	California	Plumas	11.15			x
42003	420030064	Pennsylvania	Allegheny	11.12			x

13 Standard Level (Box 7). *Estimate future-year design values and emission reductions beyond the analytical baseline to meet the alternative annual standard level of 13* μg/m^3*.* Annual $PM_{2.5}$ design values at monitors in Tulare and Riverside Counties in California exceeded the alternative standard level of 13 μg/m^3 in the analytical baseline (Table 3-6 and Figure 3-3). The additional direct $PM_{2.5}$ emission reductions required for these counties to meet this standard level were estimated using air quality ratios. For example, the highest annual design value in Riverside County in the analytical baseline case was 14.58 μg/m^3. Emission reductions of 626 tons of direct $PM_{2.5}$ were estimated to be required for this monitor to meet the annual standard level of 13.04 μg/m^3 as follows: (14.58 − 13.04) / 2.459 × 1000 = 626 tons, where 2.459 is the annual direct $PM_{2.5}$ air quality ratio for monitor 60658005 (Table 3.A-3). The emissions reductions by county to attain a 13 µg/m^3 standard are provided in Table 3.7. These reductions were applied to lower the annual $PM_{2.5}$ design values at all sites in the given county.[24]

Table 3-7. Tons of Direct $PM_{2.5}$ Emission Reductions Beyond the Analytical Baseline to Meet the 13 µg/m^3 Level

FIPS Code	State Name	County Name	Direct $PM_{2.5}$ Emissions Reductions (tons)
6065	California	Riverside	626
6107	California	Tulare	101

12 Standard Level (Box 8). *Estimate future-year design values and emission reductions beyond the analytical baseline to meet the revised annual standard level of 12* µg/m^3 *.* Annual $PM_{2.5}$ design values at monitors in the following 7 counties in California exceeded the revised standard level of 12 µg/m^3 in the analytical baseline (Table 3-6 and Figure 3-3): Los Angeles, Riverside, San Bernardino, Kern, Tulare, Merced, and Imperial. The additional direct $PM_{2.5}$ emission reductions required for these counties to meet the standard level of 12 µg/m^3 were estimated using air quality ratios. For example, the highest annual design value in Riverside County in the analytical baseline case was 14.58 µg/m^3. Emission reductions of 1,033 tons of direct $PM_{2.5}$ were estimated to be required for this monitor to meet the annual standard level of 12.04 µg/m^3 as follows: $(14.58 - 12.04) / 2.459 \times 1000 = 1033$ tons, where 2.459 is the annual direct $PM_{2.5}$ air quality ratio for monitor 60658005 (Table 3.A-3). The emissions reductions by county to attain a 12 µg/m^3 standard are provided in Table 3.8. These reductions were applied to lower the annual $PM_{2.5}$ design values at all sites in the given county[25].

[24] Emissions reductions needed in Tulare County were also applied to reduce the annual $PM_{2.5}$ design value at the monitor in Kings county, which is combined with Tulare in our analysis, as discussed in Appendix 3.A.1.1.
[25] For Kings and Tulare Counties, the maximum of the emission reductions required for the individual counties was applied to monitors in both counties using the air quality ratios since these counties are combined in our analysis, as discussed in Appendix 3.A.1.1.

Table 3-8. Tons of Direct $PM_{2.5}$ Emission Reductions Beyond the Analytical Baseline to Meet the 12 $\mu g/m^3$ Level[a]

FIPS Code	State Name	County Name	Direct $PM_{2.5}$ Emissions Reductions (tons)
6037	California	Los Angeles	743
6065	California	Riverside	1,033
6025	California	Imperial	294
6029	California	Kern	418
6107	California	Tulare	635
6047	California	Merced	19
6071	California	San Bernardino	844

[a] See Appendix Chapter 7.A for additional details on known and unknown emissions reductions and costs, by county, for 12 $\mu g/m^3$.

11 Standard Level (Box 9). *Estimate future-year design values and emission reductions beyond the analytical baseline to meet the alternative annual standard level of 11* $\mu g/m^3$ *.* Annual $PM_{2.5}$ design values at monitors in 23 counties exceeded the alternative standard level of 11 $\mu g/m^3$ in the analytical baseline (Table 3-6 and Figure 3-3). As discussed above, annual design values in the analytical baseline do not reflect the emission controls of the 2020 control case for counties with monitors that did not exceed the current standard level in the 2020 base case. To estimate the emission reductions beyond the known controls needed to meet the alternative standard level of 11 $\mu g/m^3$ in these counties, we started with annual design values for the adjusted 2020 control case (Box 4 of Figure 3-2). The additional direct $PM_{2.5}$ emission reductions required for these counties to meet the alternative standard level were then estimated using air quality ratios. For example, the annual design value at the high monitor in Jefferson, AL was 11.56 $\mu g/m^3$ in the adjusted 2020 base case and 11.11 $\mu g/m^3$ in the adjusted 2020 control case. The additional direct $PM_{2.5}$ emission reductions needed beyond the emission reductions of the 2020 control case for this monitor to meet the 11 $\mu g/m^3$ level were estimated using air quality ratios as follows: $(11.11 - 11.04) / 0.561 \times 1000 = 125$ tons, where 0.561 is the direct $PM_{2.5}$ air quality ratio for monitor 10730023. Annual $PM_{2.5}$ design values associated with emission reductions estimated in this way in (Table 3-9) were calculated for the counties with exceedance monitors.

Table 3-9. Tons of Direct $PM_{2.5}$ Emission Reductions Beyond the Analytical Baseline to Meet the Alternative Standard 11 $\mu g/m^3$ Level[a]

FIPS Code	State Name	County Name	Tons of Direct $PM_{2.5}$
6037	California	Los Angeles	3,222
6065	California	Riverside	1,440
1073	Alabama	Jefferson	125
6019	California	Fresno	325
6025	California	Imperial	850
6029	California	Kern	1,051
6031/6107	California	Kings/Tulare	1,168
6071	California	San Bernardino	2,252
6047	California	Merced	255
6063	California	Plumas	44
17031	Illinois	Cook	427
17119	Illinois	Madison	1,687
18089	Indiana	Lake	0
16079	Idaho	Shoshone	61
41035	Oregon	Klamath	25
42003	Pennsylvania	Allegheny	154
19163	Iowa	Scott	188
26163	Michigan	Wayne	870
55079	Wisconsin	Milwaukee	455
55133	Wisconsin	Waukesha	55
48141	Texas	El Paso	158
48201	Texas	Harris	123

[a]For the following counties, the emission reductions listed are relative to the adjusted 2020 control case design values rather than the analytical baseline: Jefferson, AL; Shoshone, ID; Cook, IL; Madison, IL; Klamath, OR; Lake, IN; Scott, IA; Wayne, MI; Milwaukee, WI; and Harris, TX.

3.3.3 Estimating Changes in Annual Average $PM_{2.5}$ for Benefits Inputs

The calculation of health benefits for the revised annual standard of 12 $\mu g/m^3$ and the two alternative annual standards uses spatial surfaces of gridded annual average $PM_{2.5}$ concentrations for the analytical baseline and spatial surface reflecting attainment of each different standard. The spatial surface for each case covers the U.S. portion of the air quality modeling domain. To create the spatial field for the analytical baseline we started with a spatial surface for the 2020 control case reflecting the removal of atypical events. The 2020 control case spatial surface was adjusted using the projected annual design values for the analytical baseline to create the spatial surface for the baseline. The spatial surface for the 2020 control case was also adjusted to reflect attainment of the different standards using the annual design values for each standard. Details of this process are described below.

The spatial surface for the 2020 control case (with removal of potential future atypical events) was developed using MATS by calculating species-specific RRFs at every grid cell within the modeling domain for the 2020 control case and applying these RRFs to ambient data that have been interpolated to cover all grid cells in the modeling domain. The basic spatial interpolation technique, called Voronoi Neighbor Averaging (VNA), was applied for annual design values for the 2020 control case and each standard to create spatial fields of annual $PM_{2.5}$ for each of these cases. As part of this technique, VNA uses the inverse distance squared weighted average of the annual design values at monitoring sites that are nearest to the center of each model grid cell. We then calculate the ratio of annual $PM_{2.5}$ for each standard level to annual $PM_{2.5}$ for the 2020 control case for each grid cell in the VNA fields. These gridded ratios are then multiplied by the gridded annual concentrations from the MATS outputs for the 2020 control case. That is, a spatial surface was calculated by adjusting the 2020 control case using a multiplicative factor calculated as the ratio of the gridded design values for attainment of each standard to the gridded design values of the 2020 control case where the design value gridded spatial fields are based on the nearest neighbor monitor locations (weighted by distance). This approach is shown mathematically in the equation below.

$$\textit{Adjusted } AQ_{ij} = \frac{\textit{VNA Interpolated } AQ_{ij} \textit{ from Alternative}}{\textit{VNA Interpolated } AQ_{ij} \textit{ from 2020}} \textit{ x MATS } AQ_{ij}$$

where *ij* refers to column *i* and row *j* of the modeling domain. This approach aims to estimate the change in population exposure associated with attaining an alternate NAAQS, relying on data from the existing monitoring network and the inverse distance squared variant of the VNA interpolation method to adjust the MATS gridded concentrations such that each area attains the standard alternatives. Using the VNA spatial averaging technique, the annual average $PM_{2.5}$ spatial surfaces are smoothed to minimize sharp gradients in $PM_{2.5}$ concentrations in the spatial fields due to changes in the monitor concentrations[26]. Because the VNA approach interpolates

[26] For the purposes of estimating benefits, this smoothed surface was then clipped to grid cells within 50 km of monitors whose design values were changed as a result of the standard level.

monitor values, it is most reliable in areas with a denser monitoring network. In areas with a sparser monitoring network, there is less observed monitoring data to support the VNA interpolation and we have less confidence in the air quality values further away from the location of monitoring sites. To the extent that any bias in the interpolated values is present, the ratio of the interpolated values should be relatively insensitive to this bias and the adjusted air quality values should be unaffected.

3.3.4 Limitations of Using Adjusted Air Quality Data

Due to time constraints, design values and $PM_{2.5}$ surfaces at the analytical baseline level and the alternative standard levels were based on adjusted fields derived from the modeled 2020 base case and 2020 control case, rather than directly on air quality simulation results. While a credible technical basis exists for the adjustment procedures used in this analysis, there are important limitations to the approaches used to estimate the response of air quality to emissions changes. For instance, air quality ratios are calculated with results from a limited number of CMAQ sensitivity runs and are based on the assumption that the monitor design values would decrease with additional emissions reductions of SO_2, NO_x and direct $PM_{2.5}$ similar to how the CMAQ sensitivity runs predicted changes in air quality concentrations. The uncertainty of this assumption will increase with increasing emissions reductions needed to estimate attainment. In addition, the model response to emissions changes are analyzed at a county-level or within a small group of counties, and we assume that air quality concentrations at a monitor will decrease linearly with emissions reductions in a county (e.g., direct $PM_{2.5}$ emission reductions) or a group of counties (e.g., SO_2 and NO_x emissions reductions). Because of the more local influence of changes in directly emitted $PM_{2.5}$ emissions on air quality, it is also particularly difficult for the air quality ratio approach to estimate well how the design value at a monitor in a county would respond to changes in direct $PM_{2.5}$ emissions in a county without knowing the location of the source (e.g., extrapolated emissions reductions) relative to the location of the monitor.

The exact impact of using this methodology to estimate the emissions reductions needed for attainment and the associated effect on the cost and benefits is uncertain and may vary from monitor-to-monitor. We do not believe that this methodology tends towards any general trend and does not always result in either an underestimation or overestimation of the costs and benefits of attaining the alternative standards.

3.3.5 Weight-of-Evidence Approach for Lincoln County, MT and Santa Cruz, NM

There were two counties that exceeded alternative standard levels in the 2020 base case for which we used a weight-of-evidence approach to determine how they would attain these levels. These counties are Lincoln County, MT and Santa Cruz County, AZ.

Lincoln County's $PM_{2.5}$ air quality problem is dominated by residential wood combustion emissions of $PM_{2.5}$, and the County has few additional emissions sources to control. The Lincoln County monitor is situated in the City of Libby in a valley that is subject to wintertime temperature inversions (Figure 3-4). These temperature inversions, which suppress air mixing and dilution of $PM_{2.5}$, combined with resident's reliance on wood burning for home heating can produce poor $PM_{2.5}$ air quality. However, since 2005, Libby has successfully implemented a woodstove change-out program that has resulted in consistent improvements in $PM_{2.5}$ air quality in recent years (Figure 3-5). The success of this program and the downward trend in annual design values at the Libby monitor suggests that Libby will meet the revised and alternative standard levels in 2020. Since residential wood combustion emissions in Libby and the emission reductions due to the wood-stove change-out program are not fully captured in our emission inventory, our modeled estimates of future-year design values are not reliable at this site. However, our weight-of-evidence considerations suggest that Lincoln County would likely attain the alternative standard levels in 2020 based on on-the-books control programs.

Figure 3-4. City of Libby in Lincoln County, Montana (Image taken from Google Earth)

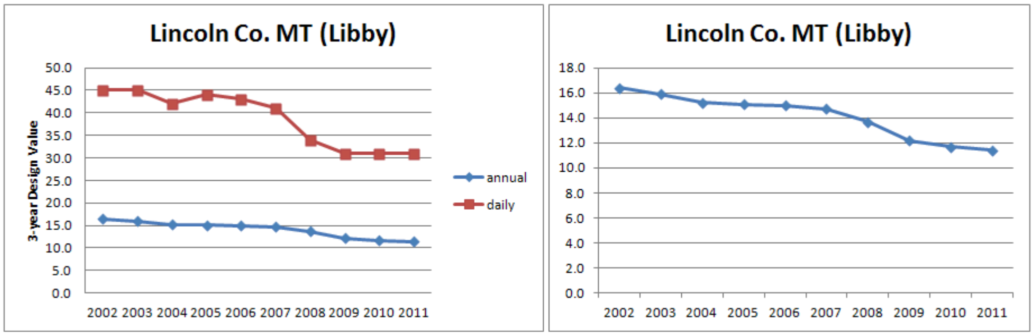


Figure 3-5. Three-year Annual and 24-hr Design Values for the Monitor in Libby, MT

Santa Cruz, AZ had a 24-hr design value of 29.7 $\mu g/m^3$ and an annual design value of 12.65 $\mu g/m^3$ in the 2020 base case. However, Santa Cruz has few local emissions sources and therefore relatively low emissions available for control. Total emissions of SO_2, NOx and direct $PM_{2.5}$ were 65, 688, and 542 tons, respectively, in Santa Cruz County in the 2020 base case. Total emissions of SO_2, NOx and direct $PM_{2.5}$ for the Mexican State of Sonora, which borders Santa Cruz, were much greater: 100,089, 53,518 and 27,641 tons, respectively. The lack of substantial local controllable emissions in Santa Cruz and the large impact of emissions from Sonora, Mexico on air quality in Santa Cruz suggest that emissions from Mexico make meeting the alternative standards for this county impractical in our analysis. Cross-border impacts of Mexican emissions on Santa Cruz County have been recognized previously. On September 25, 2012, in a Federal Register Notice, EPA Region IX approved a State Implementation Plan (SIP) revision submitted by the Arizona Department of Environmental Quality. As indicated in the Notice, EPA Region IX reviewed three years of air quality data from Arizona and determined that the Nogales nonattainment area in Santa Cruz County is attaining the National Ambient Air Quality Standard for PM_{10}, but for international emissions sources in Nogales, Sonora, Mexico. Our weight-of-evidence considerations suggest that Santa Cruz would likely not require emissions reductions in addition to those of on on-the-books control programs to attain the alternative standard levels.

3.3.6 Estimating Changes in Visibility for Analyzing Welfare Benefits

Changes in visibility were calculated in order to assess both recreational and residential visibility welfare benefits. The visibility calculations for the welfare benefits assessment are based on annual average light extinction (bext) values, converted to units of visual range (km). Since we are interested in providing visibility estimates throughout the US, we utilize gridded, speciated $PM_{2.5}$ data that is produced by MATS (Abt, 2012) along with future-year design values for the annual NAAQS.

The gridded species data used to calculate the visibility values is somewhat different than the gridded data used to calculate health benefits. The gridded $PM_{2.5}$ data used as input to BenMAP for health benefits is based on adjusted species data using the SANDWICH technique (Frank, 2006). The $PM_{2.5}$ species data is adjusted to match the nature of the $PM_{2.5}$ FRM filter data that is used as the basis for determining attainment of the $PM_{2.5}$ NAAQS. For example, in the spatial fields used in BenMAP, the nitrate data has been adjusted to account for volatilization, a particle bound water component is added to the sulfate and nitrate concentrations, and the organic carbon is calculated as the difference between the measured FRM $PM_{2.5}$ mass and the sum of the rest of the $PM_{2.5}$ species. For visibility calculations, we use the "raw" $PM_{2.5}$ species data, as measured by IMPROVE and CSN monitors. Equation 3.1 shows the "old" IMPROVE equation which is used to calculate visibility in Mm^{-1}. Note that the coarse PM component of the "old" IMPROVE equation was excluded here because this term is not used in calculating visibility spatial fields.

$$b\text{ext} = 3 \text{ x } f(\text{RH})\text{x } [Sulfate] + 3 \text{ x } f(\text{RH})\text{x } [Nitrate] + 4 \text{ x } [Organic\ Mass]$$
$$+ 10 \text{ x } [Elemental\ Carbon] + 1 \text{ x } [Fine\ Soil] + 10$$

The mass concentrations of the components indicated in brackets are in units of $\mu g/m^3$, and *f*(RH) is the unitless water growth term that depends on relative humidity. The final term in the equation is known as the Rayleigh scattering term and accounts for light scattering by the natural gases in unpolluted air. Since IMPROVE does not include ammonium ion monitoring, the assumption is made that all sulfate is fully neutralized ammonium sulfate and all nitrate is assumed to be ammonium nitrate.

The visibility values are calculated from observed concentrations for each of the PM species for each calendar quarter. Using the "old" IMPROVE equation (without the coarse mass component), and with quarterly averaged climatological average relative humidity [f(RH)] values, we calculate a quarterly average light extinction (bext) value from the IMPROVE and CSN data for the 2006-2008 base period which has been interpolated to the CMAQ grid using gradient adjusted spatial fields (eVNA). The observed sulfate and nitrate concentrations are assumed to be fully neutralized by ammonium and the organic carbon is multiplied by 1.4 to derive organic carbon mass. The interpolated gridded 2006-2008 ambient data is projected to 2020 using modeled RRFs. CMAQ derived quarterly average RRFs for sulfate, nitrate, elemental carbon, organic carbon, and crustal components are multiplied by the gridded light extinction components to get future year quarterly average visibility. The four quarterly average total light extinction values (for each grid cell) are then averaged together to get annual average visibility. The procedure was repeated for both the 2020 base case and 2020 control case scenarios.

The gridded field of 2020 base case and control case annual average visibility is used to calculate residential visibility benefits in the following manner. The visibility data at Class I areas is extracted from the gridded data to calculate recreational visibility benefits. The Class I area visibility is based on the visibility calculated at the grid cell which contains the centroid of each of the 149 Class I areas in the continental United States.

3.4 References

Abt Associates, 2010. User's Guide: Modeled Attainment Test Software. http://www.epa.gov/scram001/modelingapps_mats.htm

Appel, K.W., Bhave, P.V., Gilliland, A.B., Sarwar, G., Roselle, S.J., 2008. Evaluation of the community multiscale air quality (CMAQ) model version 4.5: Sensitivities impacting model performance; Part II—particulate matter. Atmospheric Environment 42, 6057-6066.

Appel, K.W., Gilliland, A.B., Sarwar, G., Gilliam, R.C., 2007. Evaluation of the Community Multiscale Air Quality (CMAQ) model version 4.5: Sensitivities impacting model performance Part I—Ozone. Atmospheric Environment 41, 9603-9615.

Byun, D.W., and K.L. Schere. 2006. "Review of the Governing Equations, Computational Algorithms, and Other Components of the Models-3 Community Multiscale Air Quality (CMAQ) Modeling System." J. Applied Mechanics Reviews 59(2):51-77.

Carlton, A.G., Bhave, P.V., Napelenok, S.L., Edney, E.D., Sarwar, G., Pinder, R.W., Pouliot, G.A., Houyoux, M., 2010. Model Representation of Secondary Organic Aerosol in CMAQv4.7. Environmental Science & Technology 44, 8553-8560.

Carlton, A.G., Turpin, B.J., Altieri, K.E., Seitzinger, S.P., Mathur, R., Roselle, S.J., Weber, R.J., 2008. CMAQ Model Performance Enhanced When In-Cloud Secondary Organic Aerosol is Included: Comparisons of Organic Carbon Predictions with Measurements. Environmental Science & Technology 42, 8798-8802.

ERG. 2009. Development of Mexico National Emissions Inventory Projections for 2008, 2012, AND 2030.

ERG. 2009. Technical memorandum: "Mexico 2018 Emissions Projections for Point, Area, On-Road Motor Vehicle and Nonroad Mobile Sources."

IMPROVE, (2006), "Spatial and Seasonal Patterns and Temporal Variability of Haze and its Constituents in the United States: Report IV." Cooperative Institute for Research in the

Atmosphere, ISSN: 0737-5352-74. Also available at: http://vista.cira.colostate.edu/improve/Publications/Reports/2006/PDF/IMPROVE_Report_IV.pdf

Lam, Y-F., Fu, J.S., Jacob, D.J., Jang, C., Dolwick, P., 2010. 2006-2008 GEOS-Chem for CMAQ Initial and Boundary Conditions. 9th Annual CMAS Conference, October 11-13, 2010, Chapel Hill, NC.

Nenes, A., Pandis, S.N., Pilinis, C., 1998. ISORROPIA: A new thermodynamic equilibrium model for multiphase multicomponent inorganic aerosols. Aquatic Geochemistry 4, 123-152.

South Coast Air Quality Management District, 2012. Revised Draft 2012 Air Quality Management Plan. Also available at: http://www.aqmd.gov/aqmp/2012aqmp/RevisedDraft/RevisedDraft2012AQMP-Main-clean.pdf

Sacramento Metropolitan Air quality Management District, 2009. Staff Report: Rule 421, Mandatory Episodic Curtailment of Wood and Other Solid Fuel Burning. Also available at: http://www.airquality.org/notices/Rules2009/200907Rule421WorkshopsStaffReport.pdf

U.S. Environmental Protection Agency (EPA). 2007. Guidance on the Use of Models and Other Analyses for Demonstrating Attainment of Air Quality Goals for Ozone, $PM_{2.5}$, and Regional Haze. Office of Air Quality Planning and Standards, Research Triangle Park, NC.

U.S. Environmental Protection Agency (EPA). 2011a. Meteorological Model Performance for Annual 2007 Simulations. Office of Air Quality Planning and Standards, Research Triangle Park, NC. EPA-454/R-11-007.

U.S. Environmental Protection Agency (EPA). 2011b. Update to the 24 Hour PM2.5 NAAQS Modeled Attainment Test. Office of Air Quality Planning and Standards, Research Triangle Park, NC. Also available at: http://www.epa.gov/ttn/scram/guidance/guide/Update_to_the_24-hour_PM25_Modeled_Attainment_Test.pdf

U.S. Environmental Protection Agency (EPA). 2012a. Technical Support Document: Air Quality Modeling for the Final PM NAAQS RIA. Office of Air Quality Planning and Standards, Research Triangle Park, NC. November 2012.

U.S. Environmental Projections Agency (EPA). 2012b. Technical Support Document: Preparation of Emissions Inventories for the Version 5, 2007-based Platform. Office of Air Quality Planning and Standards, Research Triangle Park, NC. November, 2012.

Yantosca, B., 2004. GEOS-CHEMv7-01-02 User's Guide, Atmospheric Chemistry Modeling Group, Harvard University, Cambridge, MA, October 15, 2004.

Yarwood, G., Rao, S., Yocke, M., Whitten, G.Z., 2005. Updates to the Carbon Bond chemical mechanism: CB05. Final Report to the US EPA, RT-0400675, December 8, 2005.

APPENDIX 3.A

ADDITIONAL AIR QUALITY MODELING INFORMATION

3.A.1 Air Quality Modeling and Analysis

This appendix provides supplemental information for the air quality modeling analysis in Chapter 3.

3.A.1.1 Development of Air Quality Response Ratios

The air quality response ratios (hereafter referred to as air quality ratios) used to adjust the 2020 cases to meet the standard levels were calculated based on results of several sensitivity simulations. The sensitivity simulations, as described in Table 3-4, were defined to isolate the changes in the $(NH_4)_2SO_4$, NH_4NO_3 and direct $PM_{2.5}$ associated with changes in emissions of SO_2, NO_x and direct $PM_{2.5}$, respectively. These $PM_{2.5}$ component species were selected for reduction to meet the standard levels because they dominate the mass of $PM_{2.5}$ in the areas of concern in the 2020 cases. The sensitivity simulation referred to as "2020 NO_x_$PM_{2.5}$" was used in calculating the air quality ratios associated with changes in NO_x and direct $PM_{2.5}$ emissions. This simulation was based on anthropogenic NO_x and direct $PM_{2.5}$ emission reductions from non-EGU sources of 25% and 50%, respectively, relative to the 2020 base case. The sensitivity simulation referred to as "2020 SO_2_RWC" was used in calculating the air quality ratios associated with changes in SO_2 emissions. This simulation was based on anthropogenic SO_2 and residential wood combustion emissions reductions from non-EGU sources of 25% and 100%, respectively, relative to the 2020 base case.[1] In the sensitivity runs, emissions reductions for direct $PM_{2.5}$ were generally applied in counties with monitors with annual design values above 11 μg/m^3 level in the 2020 base case, while emission reductions for NO_x and SO_2 were generally applied in those counties as well as their adjacent counties. This approach reflects the local impacts of direct $PM_{2.5}$ emissions on air quality and the broader geographic impacts on $PM_{2.5}$ of SO_2 and NO_x emissions reductions.

In calculating air quality ratios, a "county group" associated with each monitor was defined for estimating the change in emissions associated with a given change in the design value at the monitor. For the development of direct $PM_{2.5}$ air quality ratios, the county group included just the county containing the monitor because of the relatively local nature of the impacts of direct $PM_{2.5}$ emissions on ambient $PM_{2.5}$ concentrations. For the development of NO_x and SO_2 air quality ratios, the county group was generally defined as the county containing

[1] The results of this sensitivity run was also used in the method to quantify the impacts on design values of existing burn ban programs, as described in Section 3.3.1.1.

the nonattainment monitor plus the adjacent counties (i.e., counties that border the county with the nonattainment monitor). This multi-county group approach was used for NOx and SO_2 in view of the more widespread impacts on $(NH_4)_2SO_4$, NH_4NO_3 of local emissions reductions of NOx and SO_2 compared to direct $PM_{2.5.}$ Note that this same general approach was used in the design of the 2020 sensitivity simulations and in the 2020 control case (see Chapter 4). However, there were exceptions to this approach in certain areas in California where meteorological conditions affect the relationships between emissions and pollutant concentrations on a broader geographic scale within the South Coast Air Basin and within the San Joaquin Valley Air Basin. In the South Coast Air Basin, the county group for NO_x and SO_2 emission reductions was defined to include all counties in the air basin (i.e., Orange, Los Angeles, Riverside, and San Bernardino). For counties in the San Joaquin Valley Air Basin, the total NO_x emission change that contributed to $PM_{2.5}$ changes at a monitor in a given county was estimated using the weighted contribution of emissions changes in area counties as derived from the 2020 SJV simulations (see Appendix 3.A.1.2 for details).

In adjusting design values of the 2020 control case to meet different standard levels, Kings County and Tulare County in California were considered as a single area.[2] These counties share an east-west border and experience similar air quality due to their relative positions in the San Joaquin Valley. Also, direct $PM_{2.5}$ emissions are much smaller in Kings than in Tulare, and the Kings County monitor is close to the Tulare border (Figure 3.A-1) such that Tulare emissions have a large impact on design values in Kings County.

[2] To group these counties into a single area, the emission reductions needed for the Kings and Tulare monitors to meet the standard individually was first determined. Then the maximum of the individual emission reductions was selected and was used to adjust the design values at monitors in both counties using the air quality ratios.

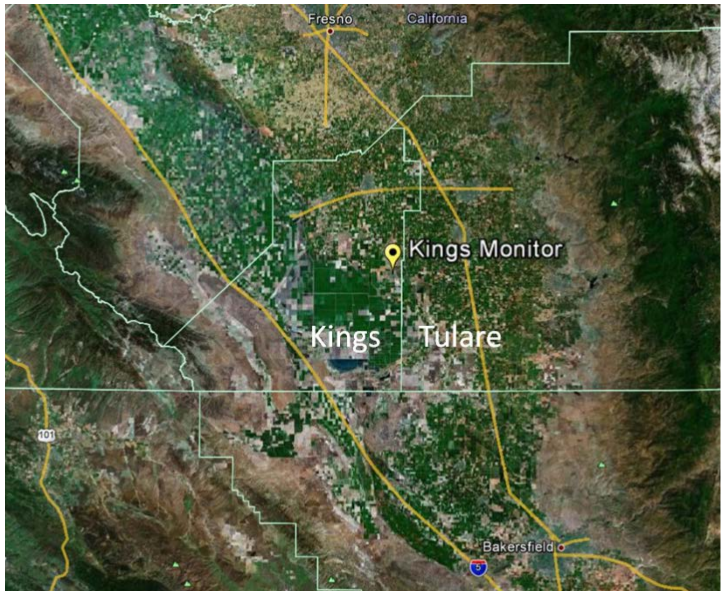


Figure 3.A-1. Location of Kings County Monitor Relative to Tulare County Border.

Air quality ratios for emissions of direct $PM_{2.5}$, SO_2 and NO_x were calculated using information from the sensitivity simulations on the response of air quality at monitors to emission changes within the county groups. Below are the steps we followed in calculating the air quality ratios:

Step 1: *Calculate the fractional change in speciated annual and 24-hr design values for the 2020 sensitivity cases relative to the 2020 base case.* Speciated annual and quarterly 24-hr RRFs were calculated for the 2020 NO_x_$PM_{2.5}$ and 2020 SO_2_RWC sensitivity simulations relative to the 2020 base case using MATS (Abt, 2010) for configurations where the 2020 base case was used as the reference case and the 2020 sensitivity cases were used as the control cases. The fractional change in the direct $PM_{2.5}$, $(NH_4)_2SO_4$ and NH_4NO_3[3] components of the design value for the 2020 sensitivity cases relative to the 2020 base case was then calculated as (RRF-1)[4] for a given monitoring site.

Step 2: *Calculate the fractional change in emissions in the relevant county group for the 2020 sensitivity cases relative to the 2020 base case.* The fractional changes in emissions of

[3] The $(NH_4)_2SO_4$ and NH_4NO_3 components are computed using the SO_4, NO_3, NH_4 and water fraction from MATS as described in EPA guidance (EPA, 2007). The direct $PM_{2.5}$ design value component is computed by summing the elemental carbon, organic carbon and crustal portions of the design value.

[4] For daily air quality ratios, a representative RRF was calculated as a weighted average of the quarterly 24-hr RRFs, where the weighting factors were the fractions of high 24-hr concentration days that occurred in the quarter in the 2020 control case.

direct $PM_{2.5}$[5], SO_2 and NO_x between the 2020 base case and 2020 sensitivity cases were determined for the county group relevant to a given monitor. County emission groups for NO_x and SO_2 for the monitors considered are listed in Tables 3.A-1 and 3.A-2.

Step 3: *Calculate the ratio of fractional change in speciated design value to fractional change in emissions for the sensitivity cases.* The ratio of the fractional change in speciated design values (Step 1) to fractional change in county group emissions (Step 2) was calculated. Specifically, we calculated the fractional change in the direct $PM_{2.5}$, $(NH_4)_2SO_4$ and NH_4NO_3 components of the annual and daily standard design values per fractional change in direct $PM_{2.5}$, SO_2 and NO_x emissions, respectively, in the county group between the 2020 sensitivity cases and the 2020 base case.

Step 4: *Calculate the ratio of the speciated design values to emissions for the 2020 control case.* Using air quality and emission data from the 2020 control case, we calculated the ratio of direct $PM_{2.5}$, $(NH_4)_2SO_4$ and NH_4NO_3 to the emissions of direct $PM_{2.5}$, SO_2 and NO_x, respectively, in the relevant county group for the 2020 control case.

Step 5: *Calculate air quality ratios using results of Steps 3 and 4.* Air quality ratios were calculated by multiplying the ratios from Step 3 by the ratios from Step 4 for each 2020 sensitivity case, individually. The overall calculation of air quality ratios for $PM_{2.5}$ component specie i and emission specie j is given by equation 3-1, where DV_i indicates the $PM_{2.5}$ component design value.

$$\text{Air Quality Ratio}=\left(\frac{RRF_i-1}{\Delta Emission_j/Emission_j}\right)_{SensitivityCase}\left(\frac{DV_i}{Emission_j}\right)_{ControlCase}\times 1000 \qquad (3.A.1)$$

Air quality ratios give an estimate of how $PM_{2.5}$ design value components ($\mu g/m^3$) would change if 1000 tons of direct $PM_{2.5}$, SO_2 and/or NO_x emissions were reduced in the county group in which the monitor is located. Annual air quality ratios that relate changes in the NH_4NO_3 component of the design value to changes in NO_x emissions are listed in Table 3.A-1 for counties in the South Coast Air Basin and San Joaquin Valley of California that received a mobile NO_x emission adjustment equal to the change in mobile NO_x emissions from the year 2020 to 2025. Annual and daily air quality ratios that relate changes in the $(NH_4)_2SO_4$ component of the design value to changes in SO_2 emissions are listed in Table 3.A-2 for monitors in counties where air quality ratios were used in adjusting daily design values for remove the impact of

[5] Direct $PM_{2.5}$ emissions are computed as the sum of emissions of elemental carbon, primary organic carbon, and unspeciated $PM_{2.5}$ mass.

inappropriate SO_2 controls. Annual and daily air quality ratios that relate changes in the direct $PM_{2.5}$ component of the design value to changes in direct $PM_{2.5}$ emissions are listed in Table 3.A-3 for monitors in counties with at least one monitor with an annual design value above 11 μg/m^3 in the 2020 base case

Table 3.A-1. Annual NO_x Air Quality Ratios for Monitors in California Counties that Received a 2025 Mobile NO_x Emission Adjustment

Monitor ID	FIPS Code	State Name	County Name	Annual NO_x Air Quality Ratio (μg/m^3 Change in NO_3 per 1000 tons NO_x)	County Emission Group
60190008	6019	California	Fresno	0.047	Weighted contributions from Kern, Kings/Tulare, Fresno/Madera, Merced, Stanislaus, San Joaquin, Alameda, and Sacramento
60195001	6019	California	Fresno	0.046	Weighted contributions from Kern, Kings/Tulare, Fresno/Madera, Merced, Stanislaus, San Joaquin, Alameda, and Sacramento
60195025	6019	California	Fresno	0.046	Weighted contributions from Kern, Kings/Tulare, Fresno/Madera, Merced, Stanislaus, San Joaquin, Alameda, and Sacramento
60290010	6029	California	Kern	0.043	Weighted contributions from Kern, Kings/Tulare, Fresno/Madera, Merced, Stanislaus, San Joaquin, Alameda, and Sacramento
60290014	6029	California	Kern	0.042	Weighted contributions from Kern, Kings/Tulare, Fresno/Madera, Merced, Stanislaus, San Joaquin, Alameda, and Sacramento
60290016	6029	California	Kern	0.042	Weighted contributions from Kern, Kings/Tulare, Fresno/Madera, Merced, Stanislaus, San Joaquin, Alameda, and Sacramento

(continued)

Table 3.A-1. Annual NO_x Air Quality Ratios for Monitors in California Counties that Received a 2025 Mobile NO_x Emission Adjustment (continued)

Monitor ID	FIPS Code	State Name	County Name	Annual NO_x Air Quality Ratio (μg/m^3 Change in NO_3 per 1000 tons NO_x)	County Emission Group
60310004	6031	California	Kings	0.049	Weighted contributions from Kern, Kings/Tulare, Fresno/Madera, Merced, Stanislaus, San Joaquin, Alameda, and Sacramento
60370002	6037	California	Los Angeles	0.007	Los Angeles, Orange, Riverside, San Bernardino
60371002	6037	California	Los Angeles	0.006	Los Angeles, Orange, Riverside, San Bernardino
60371103	6037	California	Los Angeles	0.006	Los Angeles, Orange, Riverside, San Bernardino
60371201	6037	California	Los Angeles	0.003	Los Angeles, Orange, Riverside, San Bernardino
60371301	6037	California	Los Angeles	0.005	Los Angeles, Orange, Riverside, San Bernardino
60371602	6037	California	Los Angeles	0.007	Los Angeles, Orange, Riverside, San Bernardino
60372005	6037	California	Los Angeles	0.005	Los Angeles, Orange, Riverside, San Bernardino
60374002	6037	California	Los Angeles	0.004	Los Angeles, Orange, Riverside, San Bernardino
60374004	6037	California	Los Angeles	0.004	Los Angeles, Orange, Riverside, San Bernardino
60379033	6037	California	Los Angeles	0.003	Los Angeles, Orange, Riverside, San Bernardino
60472510	6047	California	Merced	0.029	Los Angeles, Orange, Riverside, San Bernardino
60590007	6059	California	Orange	0.006	Los Angeles, Orange, Riverside, San Bernardino
60592022	6059	California	Orange	0.005	Los Angeles, Orange, Riverside, San Bernardino
60651003	6065	California	Riverside	0.012	Los Angeles, Orange, Riverside, San Bernardino
60652002	6065	California	Riverside	0.000	Los Angeles, Orange, Riverside, San Bernardino

(continued)

Table 3.A-1. Annual NO_x Air Quality Ratios for Monitors in California Counties that Received a 2025 Mobile NO_x Emission Adjustment (continued)

Monitor ID	FIPS Code	State Name	County Name	Annual NO_x Air Quality Ratio ($\mu g/m^3$ Change in NO_3 per 1000 tons NO_x)	County Emission Group
60655001	6065	California	Riverside	0.000	Los Angeles, Orange, Riverside, San Bernardino
60658001	6065	California	Riverside	0.012	Los Angeles, Orange, Riverside, San Bernardino
60658005	6065	California	Riverside	0.011	Los Angeles, Orange, Riverside, San Bernardino
60710025	6071	California	San Bernardino	0.009	Los Angeles, Orange, Riverside, San Bernardino
60710306	6071	California	San Bernardino	0.004	Los Angeles, Orange, Riverside, San Bernardino
60712002	6071	California	San Bernardino	0.011	Los Angeles, Orange, Riverside, San Bernardino
60718001	6071	California	San Bernardino	0.000	Los Angeles, Orange, Riverside, San Bernardino
60719004	6071	California	San Bernardino	0.009	Los Angeles, Orange, Riverside, San Bernardino
60771002	6077	California	San Joaquin	0.018	Weighted contributions from Kern, Kings/Tulare, Fresno/Madera, Merced, Stanislaus, San Joaquin, Alameda, and Sacramento
60990005	6099	California	Stanislaus	0.041	Weighted contributions from Kern, Kings/Tulare, Fresno/Madera, Merced, Stanislaus, San Joaquin, Alameda, and Sacramento
61072002	6107	California	Tulare	0.055	Weighted contributions from Kern, Kings/Tulare, Fresno/Madera, Merced, Stanislaus, San Joaquin, Alameda, and Sacramento

Table 3.A-2. Annual and Daily SO_2 Air Quality Ratios for Monitors in Counties where Ratios were Used in Adjusting Daily Design Values to Remove the Impact of SO_2 Controls

Monitor ID	FIPS Code	State Name	County Name	Annual SO_2 Air Quality Ratio (μg/m^3 Change in SO_4 per 1000 tons SO_2)	Daily SO_2 Air Quality Ratio (μg/m^3 Change in SO_4 per 1000 tons SO_2)	County Emission Group
60990005	6099	California	Stanislaus	0.123	0.468	Stanislaus, San Joaquin, Merced
420030008	42003	Pennsylvania	Allegheny	0.026	0.084	Allegheny, Armstrong, Beaver, Butler, Washington, Westmoreland
420030064	42003	Pennsylvania	Allegheny	0.028	0.151	Allegheny, Armstrong, Beaver, Butler, Washington, Westmoreland
420030067	42003	Pennsylvania	Allegheny	0.017	0.050	Allegheny, Armstrong, Beaver, Butler, Washington, Westmoreland
420030095	42003	Pennsylvania	Allegheny	0.021	0.066	Allegheny, Armstrong, Beaver, Butler, Washington, Westmoreland
420031008	42003	Pennsylvania	Allegheny	0.019	0.033	Allegheny, Armstrong, Beaver, Butler, Washington, Westmoreland
420031301	42003	Pennsylvania	Allegheny	0.027	0.062	Allegheny, Armstrong, Beaver, Butler, Washington, Westmoreland
420033007	42003	Pennsylvania	Allegheny	0.021	0.068	Allegheny, Armstrong, Beaver, Butler, Washington, Westmoreland

Table 3.A-3. Annual and Daily Direct $PM_{2.5}$ Air Quality Ratios for Monitors in Counties with at Least One Monitor having an Annual Design Value Above 11 $\mu g/m^3$ in the 2020 Base Case

Monitor ID	FIPS Code	State Name	County Name	Annual $PM_{2.5}$ Air Quality Ratio ($\mu g/m^3$ Change in Direct $PM_{2.5}$ per 1000 tons $PM_{2.5}$)	Daily $PM_{2.5}$ Air Quality Ratio ($\mu g/m^3$ Change in Direct $PM_{2.5}$ per 1000 tons $PM_{2.5}$)
10730023	1073	Alabama	Jefferson	0.561	N/A
10731005	1073	Alabama	Jefferson	0.257	N/A
10731009	1073	Alabama	Jefferson	0.107	N/A
10731010	1073	Alabama	Jefferson	0.221	N/A
10732003	1073	Alabama	Jefferson	0.602	N/A
10732006	1073	Alabama	Jefferson	0.383	N/A
10735002	1073	Alabama	Jefferson	0.257	N/A
10735003	1073	Alabama	Jefferson	0.195	N/A
60190008	6019	California	Fresno	1.751	5.714
60195001	6019	California	Fresno	1.534	4.825
60195025	6019	California	Fresno	1.717	4.921
60250005	6025	California	Imperial	1.801	6.594
60250007	6025	California	Imperial	1.523	5.309
60251003	6025	California	Imperial	1.612	5.270
60290010	6029	California	Kern	1.341	4.344
60290014	6029	California	Kern	1.531	4.475
60290016	6029	California	Kern	1.579	4.892
60310004	6031	California	Kings	1.277	4.919
60370002	6037	California	Los Angeles	0.367	N/A
60371002	6037	California	Los Angeles	0.404	N/A
60371103	6037	California	Los Angeles	0.404	N/A
60371201	6037	California	Los Angeles	0.279	N/A
60371301	6037	California	Los Angeles	0.419	N/A
60371602	6037	California	Los Angeles	0.401	N/A
60372005	6037	California	Los Angeles	0.322	N/A
60374002	6037	California	Los Angeles	0.325	N/A
60374004	6037	California	Los Angeles	0.299	N/A
60379033	6037	California	Los Angeles	0.119	N/A

(continued)

Table 3.A-3. Annual and Daily Direct $PM_{2.5}$ Air Quality Ratios for Monitors in Counties with at Least One Monitor having an Annual Design Value Above 11 $\mu g/m^3$ in the 2020 Base Case (continued)

Monitor ID	FIPS Code	State Name	County Name	Annual $PM_{2.5}$ Air Quality Ratio ($\mu g/m^3$ Change in Direct $PM_{2.5}$ per 1000 tons $PM_{2.5}$)	Daily $PM_{2.5}$ Air Quality Ratio ($\mu g/m^3$ Change in Direct $PM_{2.5}$ per 1000 tons $PM_{2.5}$)
60472510	6047	California	Merced	4.233	17.925
60631006	6063	California	Plumas	2.428	N/A
60631009	6063	California	Plumas	2.518	N/A
60651003	6065	California	Riverside	1.620	3.223
60652002	6065	California	Riverside	0.930	2.463
60655001	6065	California	Riverside	0.797	1.885
60658001	6065	California	Riverside	2.089	3.627
60658005	6065	California	Riverside	2.459	5.039
60710025	6071	California	San Bernardino	0.710	1.423
60710306	6071	California	San Bernardino	0.305	0.439
60712002	6071	California	San Bernardino	0.619	1.180
60718001	6071	California	San Bernardino	0.353	1.674
60719004	6071	California	San Bernardino	0.606	1.710
60771002	6077	California	San Joaquin	1.789	8.486
60990005	6099	California	Stanislaus	2.449	8.955
61072002	6107	California	Tulare	1.875	4.222
160790017	16079	Idaho	Shoshone	7.675	N/A
170310022	17031	Illinois	Cook	0.330	N/A
170310050	17031	Illinois	Cook	0.298	N/A
170310052	17031	Illinois	Cook	0.356	N/A
170310057	17031	Illinois	Cook	0.324	N/A
170310076	17031	Illinois	Cook	0.281	N/A
170312001	17031	Illinois	Cook	0.256	N/A
170313301	17031	Illinois	Cook	0.307	N/A

(continued)

Table 3.A-3. Annual and Daily Direct $PM_{2.5}$ Air Quality Ratios for Monitors in Counties with at Least One Monitor having an Annual Design Value Above 11 $\mu g/m^3$ in the 2020 Base Case (continued)

Monitor ID	FIPS Code	State Name	County Name	Annual $PM_{2.5}$ Air Quality Ratio ($\mu g/m^3$ Change in Direct $PM_{2.5}$ per 1000 tons $PM_{2.5}$)	Daily $PM_{2.5}$ Air Quality Ratio ($\mu g/m^3$ Change in Direct $PM_{2.5}$ per 1000 tons $PM_{2.5}$)
170314007	17031	Illinois	Cook	0.200	N/A
170314201	17031	Illinois	Cook	0.205	N/A
170316005	17031	Illinois	Cook	0.374	N/A
171191007	17119	Illinois	Madison	0.332	N/A
171192009	17119	Illinois	Madison	0.443	N/A
171193007	17119	Illinois	Madison	0.417	N/A
180890006	18089	Indiana	Lake	0.419	N/A
180890027	18089	Indiana	Lake	0.320	N/A
180890031	18089	Indiana	Lake	0.367	N/A
180891003	18089	Indiana	Lake	0.401	N/A
180892004	18089	Indiana	Lake	0.404	N/A
180892010	18089	Indiana	Lake	0.397	N/A
191630015	19163	Iowa	Scott	1.106	N/A
191630018	19163	Iowa	Scott	1.051	N/A
191630019	19163	Iowa	Scott	1.492	N/A
261630001	26163	Michigan	Wayne	0.404	N/A
261630015	26163	Michigan	Wayne	0.502	N/A
261630016	26163	Michigan	Wayne	0.423	N/A
261630019	26163	Michigan	Wayne	0.335	N/A
261630025	26163	Michigan	Wayne	0.241	N/A
261630033	26163	Michigan	Wayne	0.483	N/A
261630036	26163	Michigan	Wayne	0.336	N/A
261630038	26163	Michigan	Wayne	0.381	N/A
261630039	26163	Michigan	Wayne	0.406	N/A
410350004	41035	Oregon	Klamath	3.994	N/A
420030008	42003	Pennsylvania	Allegheny	0.358	1.463
420030064	42003	Pennsylvania	Allegheny	0.519	4.060
420030067	42003	Pennsylvania	Allegheny	0.222	0.657

(continued)

Table 3.A-3. Annual and Daily Direct $PM_{2.5}$ Air Quality Ratios for Monitors in Counties with at Least One Monitor having an Annual Design Value Above 11 $\mu g/m^3$ in the 2020 Base Case (continued)

Monitor ID	FIPS Code	State Name	County Name	Annual $PM_{2.5}$ Air Quality Ratio $\mu g/m^3$ Change in Direct $PM_{2.5}$ per 1000 tons $PM_{2.5}$)	Daily $PM_{2.5}$ Air Quality Ratio ($\mu g/m^3$ Change in Direct $PM_{2.5}$ per 1000 tons $PM_{2.5}$)
420030095	42003	Pennsylvania	Allegheny	0.263	0.931
420031008	42003	Pennsylvania	Allegheny	0.172	0.645
420031301	42003	Pennsylvania	Allegheny	0.405	1.409
420033007	42003	Pennsylvania	Allegheny	0.397	1.752
481410037	48141	Texas	El Paso	1.608	N/A
481410044	48141	Texas	El Paso	2.209	N/A
482010058	48201	Texas	Harris	0.188	N/A
482011035	48201	Texas	Harris	0.408	N/A
550790010	55079	Wisconsin	Milwaukee	1.566	N/A
550790026	55079	Wisconsin	Milwaukee	1.602	N/A
550790043	55079	Wisconsin	Milwaukee	1.674	N/A
550790059	55079	Wisconsin	Milwaukee	1.869	N/A
550790099	55079	Wisconsin	Milwaukee	1.689	N/A
551330027	55133	Wisconsin	Waukesha	3.297	N/A

3.A.1.2 Estimating Area NO_x Emission Contributions to NH_4NO_3 $PM_{2.5}$ in the San Joaquin Valley

We conducted 9 air quality model simulations for January 2020 for a domain centered on California (Figure 3.A-2) that is a subset of the continental U.S. domain used for the 2020 base case modeling. One of the simulations reflected the 2020 base case emission scenario, and the other 8 simulations had NO_x and direct $PM_{2.5}$ emissions reductions relative to the 2020 base case in a one- or two-county group that matched the emissions reductions in that group in the 2020 NO_X_$PM_{2.5}$ sensitivity run. The purpose of these simulations was to estimate the impact of NO_x emission reductions in a given area of California's Central Valley on NH_4NO_3 $PM_{2.5}$ in other areas of the valley. The month of January was selected for this analysis because high NH_4NO_3 $PM_{2.5}$ episodes occur during winter months in the Central Valley.

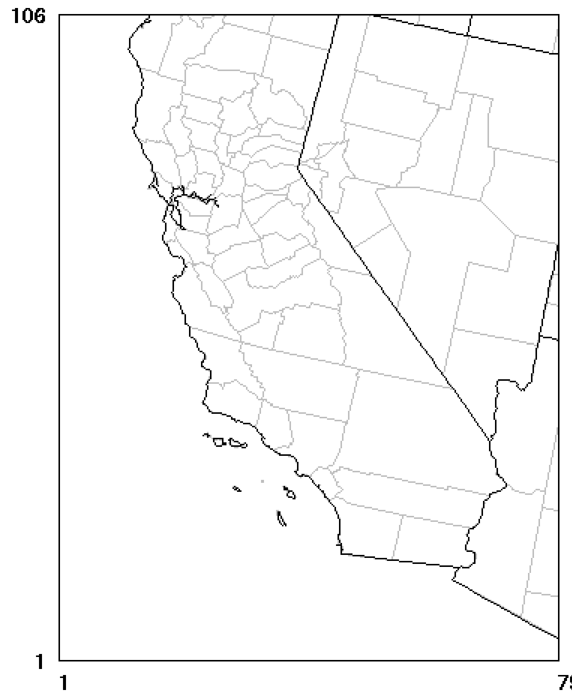


Figure 3.A-2. California Modeling Domain for 12-km Simulations

Equation 3.A.2 was used to estimate the fractional contribution of NO_x emissions from one area of California's Central Valley on another (i.e., $W_{igrp,jgrp}$).

$$W_{igrp,jgrp} = \frac{\Delta C_{NO3,igrp}/\Delta Emiss_{NOx,jgrp}}{max(\Delta C_{NO3,igrp}/\Delta Emiss_{NOx,jgrp})} \quad (3.A.2)$$

where $\Delta C_{NO3,igrp}$ is the change in average nitrate $PM_{2.5}$ concentration at a given monitor in the *igrp* county group between the simulation with 2020 base case emissions and the simulation with NO_x emissions reductions in the *jgrp* county group, and $\Delta Emiss_{NOx,jgrp}$ is the change in NO_x emissions in the *jgrp* county group between the simulations with 2020 base case emissions and the simulation with NO_x emissions reductions in the *jgrp* county group. Note that Equation 3.A.2 normalizes each Δconcentration-to-Δemission ratio for a given county group (numerator) by the maximum Δconcentration-to-Δemission ratio associated with that county group (denominator). The fraction of NO_x emissions from a given county or county group that impacts $PM_{2.5}$ nitrate in another county or county group as estimated according to Equation 3.A.2 is given in Table 3.A-4.

Table 3.A-4. Contribution Weighting Factors for NO_x Emissions in Counties or County Groups in California's Central Valley as Calculated According to Equation 3.A.2

County or County Group	Weight of Kern Emissions	Weight of Kings/ Tulare Emissions	Weight of Fresno/ Madera Emissions	Weight of Merced Emissions	Weight of Stanislaus Emissions	Weight of San Joaquin Emissions	Weight of Sacramento Emissions	Weight of Alameda Emissions
Kern	1	0.94	0.6	0.47	0.4	0.35	0.29	0.07
Kings/Tulare	0.11	1	0.89	0.56	0.4	0.31	0.24	0.05
Fresno/ Madera	0.06	0.41	1	0.65	0.48	0.38	0.35	0.04
Merced	0.02	0.07	0.23	1	0.92	0.51	0.4	0.04
Stanislaus	0.01	0.03	0.05	0.37	1	0.56	0.39	0.06
San Joaquin	0.02	0.04	0.08	0.39	0.7	0.66	1	0.09
Sacramento	0.01	0.03	0.04	0.12	0.22	0.25	1	0.01
Alameda	0.04	0.14	0.13	0.24	0.59	1	0.66	0.03

CHAPTER 4
CONTROL STRATEGIES

4.1 Synopsis

As discussed in previous chapters, the EPA is revising the $PM_{2.5}$annual standard to a level of 12 µg/$m^3$and is retaining the 24-hour standard of 35 µg/m^3. Pursuant to Executive Order 12866 and 13563 as well as the guidelines of OMB Circular A-4.1, the EPA assessed the incremental costs of hypothetical control strategies to attain the revised standard. EPA also estimated the incremental costs of attaining a less stringent alternative annual standard of 13 µg/m^3 and a more stringent alternative annual standard of 11 µg/m^3. This chapter documents the emission control measures we applied to simulate attainment with the revised and alternative $PM_{2.5}$ annual standards.

The EPA analyzed the impact that additional emissions controls across numerous sectors would have on predicted ambient $PM_{2.5}$ concentrations incremental to an analytical baseline, which includes the current $PM_{2.5}$ standard of 15/35 µg/m^3 as well as other major rules such as MATS. Thus, the analysis for the revised and alternative standards focuses specifically on incremental improvements beyond the current standard and other existing major rules, and uses control options that might be available to states for application by 2020. The hypothetical control strategies presented in this RIA represent illustrative options for emissions reductions that achieve national attainment of the revised standard as well as the alternative standards. The hypothetical control strategies are not recommendations for how the revised $PM_{2.5}$ standard should be implemented, and states will make all final decisions regarding implementation strategies for the revised NAAQS.

The traditional analytical approach to a NAAQS RIA is to perform air quality modeling for:

- base case projections, then
- baseline attainment strategy for the current standard, then
- revised and alternative control strategies incremental to the baseline strategy

Each subsequent model run would build on what was learned from the previous run. Because of the short timeframe for this analysis, we were limited to performing air quality modeling for the base case in parallel with air quality modeling for a single control scenario, along with several limited sensitivity runs (see Chapter 3 for a detailed description of the air quality modeling runs performed for this analysis). The following steps were taken by the EPA to analyze the impacts and costs of the control scenario incremental to attainment of the current standard of 15/35 μg/m^3 and beyond other existing major rules:

1. Identify geographic areas in the U.S. likely to exceed the revised or alternative standards in the year 2020 using the base case projections.

2. Develop a hypothetical control scenario for these areas and generate a control case 2020 emissions inventory. These control measures will serve as the basis for the "known" controls in this analysis.

3. Perform air quality modeling to assess the air quality impacts of the hypothetical control scenario (as mentioned, the analyses were performed in parallel with base case air quality modeling).

4. Adjust results to remove controls deemed as inappropriate for application to a specific source (e.g., controls likely to already be in place or controls estimated to have a very high cost but little impact on emissions).

5. Calculate the portion of the hypothetical control scenario control measures that are attributed to meeting the current standard of 15/35 μg/m^3. These are the known "analytical baseline" reductions. Estimate any additional emission reductions beyond the known controls that are needed to meet the current standard (15/35 μg/m^3). Costs of known controls incremental to (i.e., over and above) the analytical baseline reductions are attributed to the costs of meeting the revised and alternative standards.

6. Estimate the additional emission reductions incremental to the analytical baseline and beyond the known controls that are needed to meet the revised and/or alternative standards. These are referred to as emission reductions needed beyond known controls (i.e., extrapolated tons).

7. Calculate the total costs of reductions from emission reductions from known controls and emission reductions needed beyond known controls (extrapolated costs) incremental to the analytical baseline.

This chapter discusses the steps listed above that were key to conducting the control strategy analysis for year 2020 for the revised and alternative standards.

4.2 $PM_{2.5}$ Control Strategy Analysis

4.2.1 Identify Geographic Areas

The first step in the analysis was to identify the geographic areas likely to exceed the revised and/or alternative standards in 2020 for the base case. For a detailed description of the process used to identify these geographic areas, see Chapter 3. For the revised standard, there were seven counties projected to exceed the standard, all in California. For the hypothetical control scenario for the revised standard we also identified and applied control measures in one county adjacent to one of the projected exceedance counties in order to reduce transport emissions. For a description of the areas included in the final analysis of the revised and alternative standards, see Figures 4-1 through 4-4.

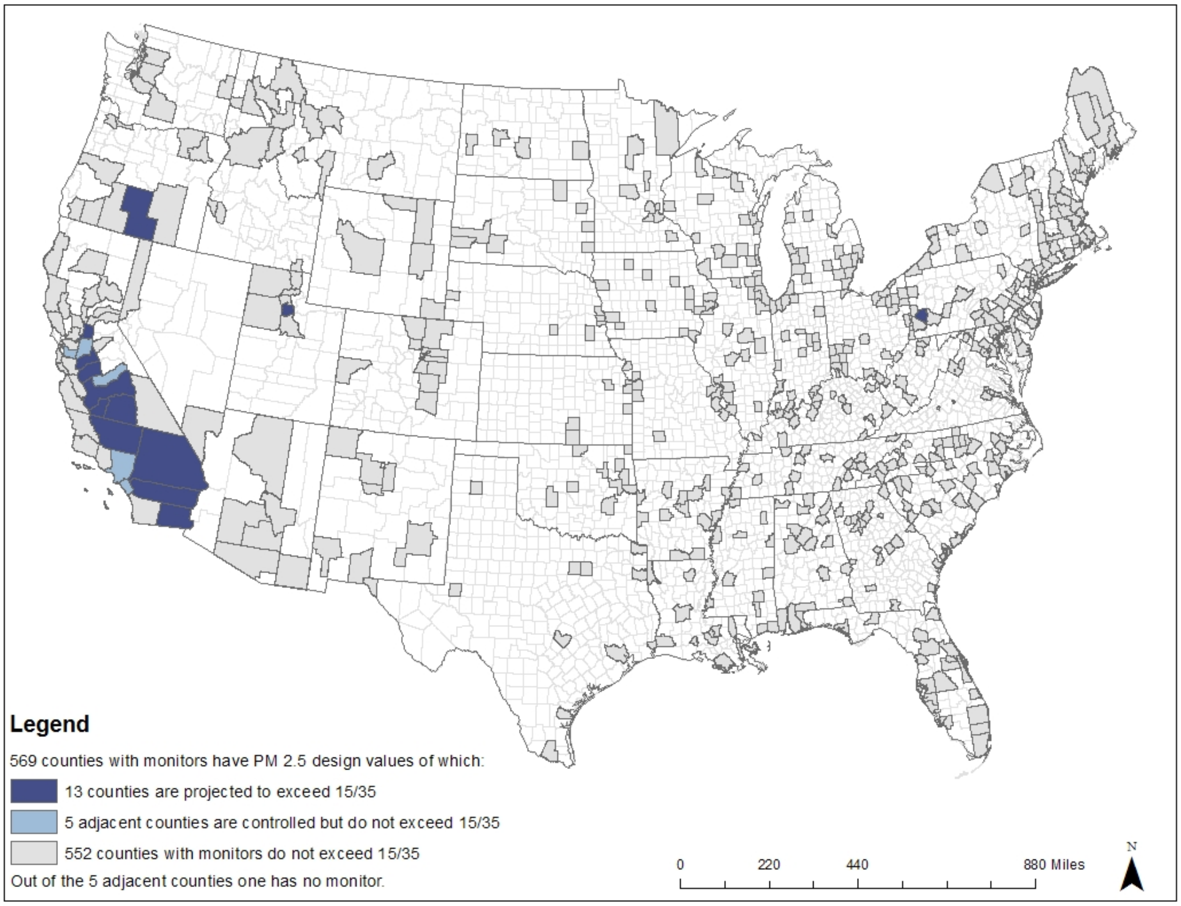


Figure 4-1. Counties Projected to Exceed the Current $PM_{2.5}$ Standard (15/35) in 2020

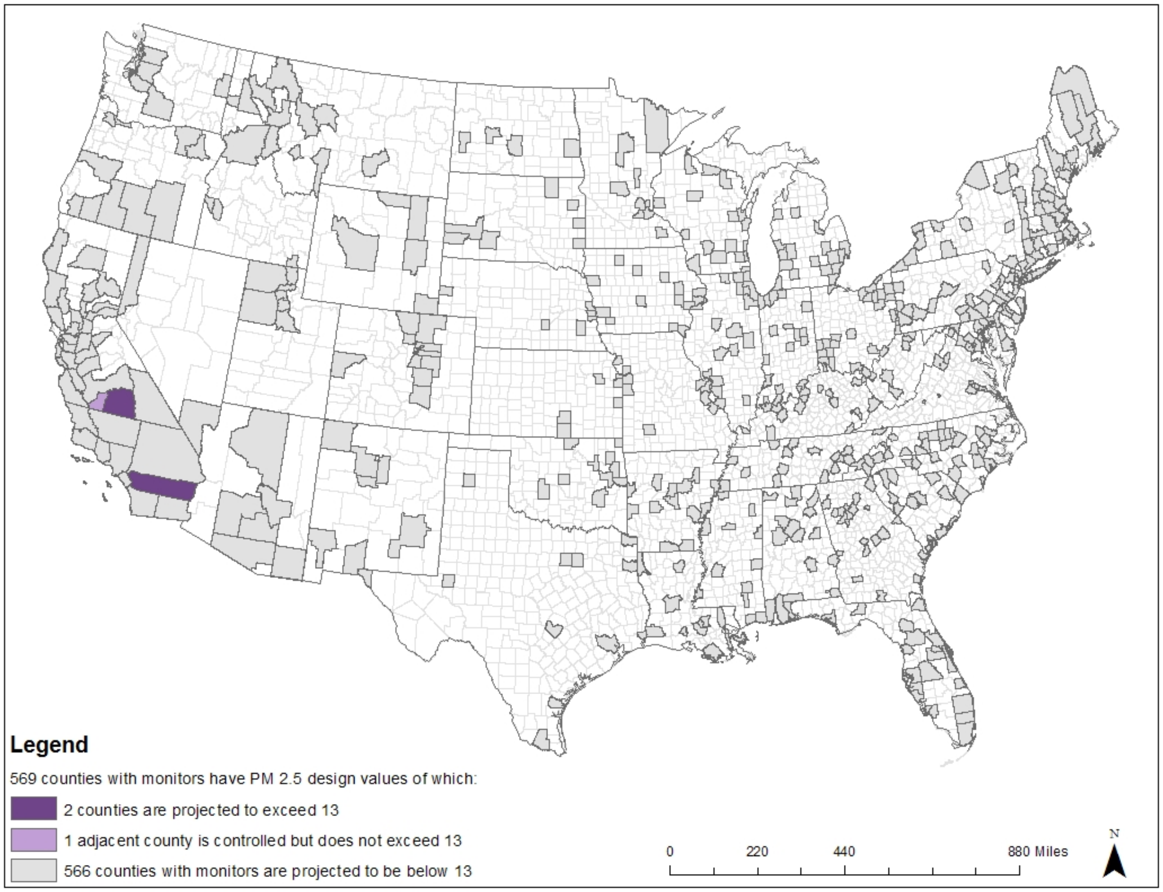


Figure 4-2. Counties Projected to Exceed the 13 ug/m^3 Alternative Standard in the Analytical Baseline

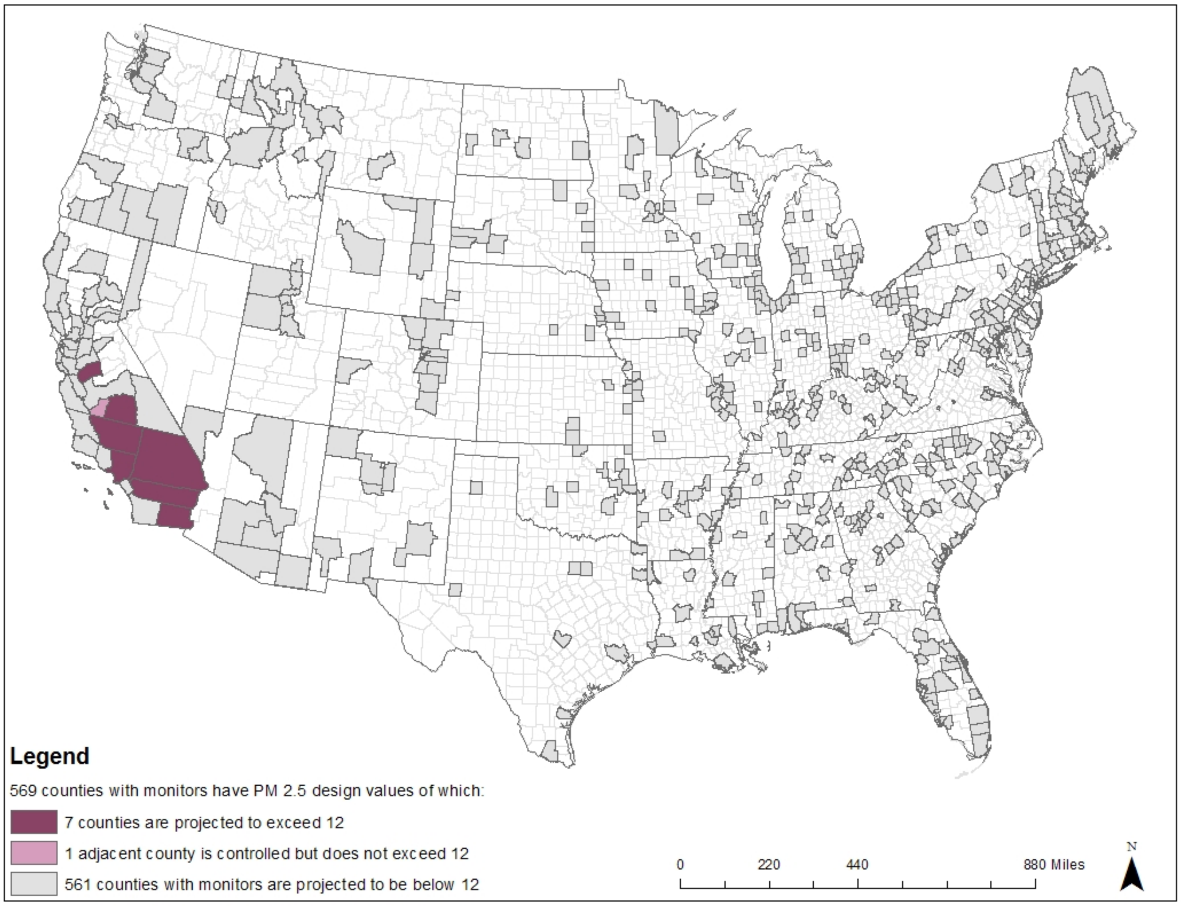


Figure 4-3. Counties Projected to Exceed the 12 ug/m^3 Revised Standard in the Analytical Baseline

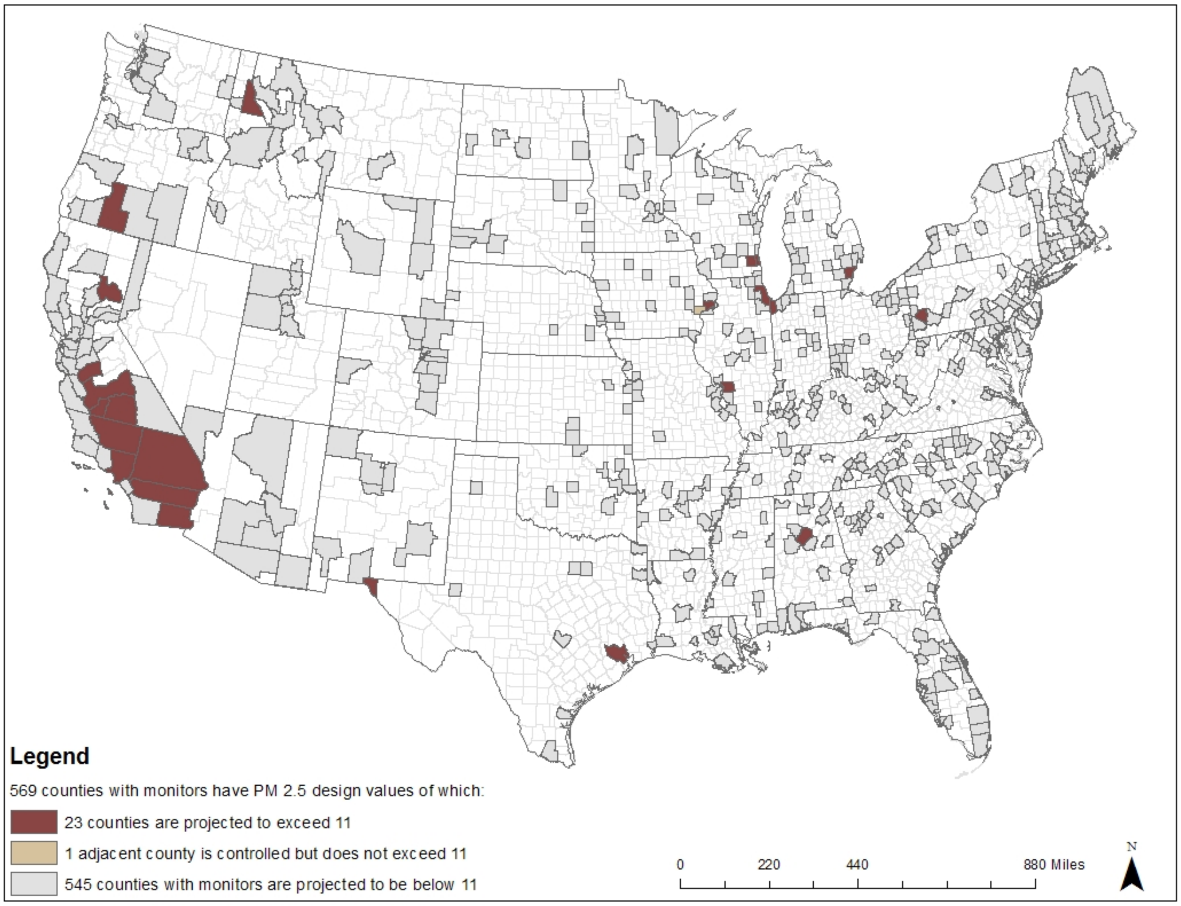


Figure 4-4. Counties Projected to Exceed the 11 ug/m^3 Alternative Standard in the Analytical Baseline

4.2.2 Developing the Control Scenario

The U.S. EPA used monitoring and emissions inventory information, including monitoring speciation, to identify the pollutants that were the primary contributors to $PM_{2.5}$ exceedances at the subject monitors. This allowed us to select a set of control measures tailored to the conditions for each area.

Non-EGU point, nonpoint (area), and onroad mobile control measures were applied for the control strategy for demonstrating attainment of the current standard (15/35 µg/m^3). Non-EGU point and nonpoint control measures were applied for the revised and alternative standards control strategies. These controls were identified using the U.S. EPA's Control

Strategy Tool[1] (CoST). These controls are summarized in Appendix 4.A. Additional control measures were not applied to electric generating units (EGUs) due to the extensive nature of controls resulting from the inclusion of MATS.

Nonpoint and onroad mobile source emissions data are generated at the county level, and therefore controls for these emissions sectors were applied at the county level. Non-EGU point source controls are applied to individual point sources. Nonpoint source controls were applied to NO_x, SO_2, and $PM_{2.5}$. The analysis for non-EGUs applied NO_x, SO_2, and $PM_{2.5}$ controls to the following source categories: industrial boilers, commercial and institutional boilers, sulfuric acid plants (both stand alone and at other facilities such as copper and lead smelters), primary metal plants (iron and steel mills, lead smelters), mineral products (primarily cement kilns), and petroleum refineries. Among the control measures applied were: wet FGD scrubbers and spray dryer absorbers (SDA) for SO_2 reductions, fabric filters for $PM_{2.5}$ reductions, and SCR and low NO_x burners for NO_x. Table 4-1 lists the major controls applied to each sector.

Table 4-1. Controls Applied in the Revised and Alternative Standard Analysis

Sector/Pollutant	Control Measure	15/35	13	12	11
Non-EGU Point					
$PM_{2.5}$	Diesel Particulate Filter	X		X	X
	Dry Electrostatic Precipitator (ESP)				X
	Fabric Filters	X		X	X
	Venturi Scrubber				X
	Wet Electrostatic Precipitator (ESP)				
SO_2	Coal Washing				X
	Flue Gas Desulfurization (FGD)				X
	Spray Dry Absorber				
	Sulfur Recovery and/or Tail Gas Treatment				
	Wet FGDs	X			X
NO_x	Biosolid Injection	X			
	Low NO_x Burner (LNB)	X			
	Low NO_x Burner (LNB) + Selective Catalytic Reduction (SCR)	X			
	Non-Selective Catalytic Reduction (NSCR)	X			
	OXY-Firing	X			

[1] See http://www.epa.gov/ttn/ecas/cost.htm for a description of CoST.

Sector/Pollutant	Control Measure	15/35	13	12	11
	SCR + Steam Injection	X			
	Selective Catalytic Reduction (SCR)	X			
	Selective Non-Catalytic Reduction (SNCR)	X			
Non-Point (Area)					
$PM_{2.5}$	ESPs for Commercial Cooking	X	X	X	X
	Low NO_x Burners for Residential Natural Gas	X			
	Substitute chipping for open burning	X		X	X
	Substitute landfilling for open burning	X			
SO_2	Chemical Additives to Waste	X			X
	Fuel Switching for Stationary Source Fuel Combustion	X			X
	Low Sulfur Home Heating Fuel				X
NO_x	Low NO_x Burners for Residential Natural Gas	X			
	Water heater + Low NO_x Burner Space Heaters	X			
Onroad Mobile					
NO_x	Elimination of Long Duration Truck Idling (diesel trucks)	X			
	Continuous Inspection and Maintenance (gasoline cars)	X			

*Note that control measures indicated in the table for 13, 12, and 11 µg/m^3 levels are incremental to control measures indicated for the 15/35 µg/m^3 analysis.

To more accurately depict available controls, the EPA employed a decision rule in which controls were not applied to any non-EGU or area sources with 50 tons/year of emissions or less. Furthermore, controls were not applied to sources unless at least 5 tons/year of emission reductions were achieved. This decision rule is the same rule we employed for sources in the previous $PM_{2.5}$ NAAQS RIA completed in 2006. The reason for applying this decision rule is based on a finding that most point sources with emissions of this level or less had controls already in place. This decision rule helps fill gaps in information regarding existing controls on non-EGU sources. An additional constraint was applied in the control strategy selection that concerned cost of control. No control measures were applied that cost greater than $20,000/ton of emission reduction. We did not include known controls at an annual cost of more than $20,000 per ton because either (i) the remaining emissions sources were relatively small sources, or we believe they are already controlled, or (ii) the equations in CoST were not applicable to these remaining emissions sources. Note that there were potential controls available at an annual cost of more than $20,000 per ton for ten of the geographic areas included in the analysis. The application of these control strategies results in some, but not all, geographic areas reaching attainment for the alternative $PM_{2.5}$ standards.

As stated above, because of the tight time constraints for this analysis, we performed air quality modeling for a single control scenario and then used the results of sensitivity analyses to identify the subset of controls and associated emission reductions in the control scenario that were needed to meet the current baseline. We then used a similar approach to determine the subset of additional controls and emissions reductions incremental to those applied in the analytical baseline that were needed to meet the revised and alternative standards.

4.2.3 Identify Known Controls Needed to Meet the Analytical Baseline

The analytical baseline includes reductions already achieved as a result of national regulations, reductions expected prior to 2020 from recently promulgated national regulations, adjustments for expected emission reductions in two areas (South Coast and San Joaquin Valley, CA) not expected to reach attainment until 2025, and reductions from additional controls which the U.S. EPA estimates need to be included to attain the current standard (15/35). Reductions achieved as a result of state and local agency regulations and voluntary programs are reflected to the extent that they are represented in emission inventory information submitted to the U.S. EPA by state and local agencies. Two steps were used to develop the analytical baseline. First, the reductions expected in national $PM_{2.5}$ concentrations from national rules promulgated prior to this analysis were considered (referred to as the base case). Below is a list of some of the major national rules reflected in the base case. Refer to Chapter 3, Section 3.2.1.4 for a more detailed discussion of the rules reflected in the 2020 base case emissions inventory.

- Light-Duty Vehicle Tier 2 Rule (U.S. EPA, 1999)
- Heavy Duty Diesel Rule (U.S. EPA, 2000)
- Clean Air Nonroad Diesel Rule (U.S. EPA, 2004)
- Regional Haze Regulations and Guidelines for Best Available Retrofit Technology Determinations (U.S. EPA, 2005b)
- NO_x Emission Standard for New Commercial Aircraft Engines (U.S. EPA, 2005)
- Emissions Standards for Locomotives and Marine Compression-Ignition Engines (U.S. EPA, 2008)
- Control of Emissions for Nonroad Spark Ignition Engines and Equipment (U.S. EPA, 2008)
- C3 Oceangoing Vessels (U.S. EPA, 2010)

- Hospital/Medical/Infectious Waste Incinerators: New Source Performance Standards and Emission Guidelines: Final Rule Amendments (U.S. EPA, 2009)
- Reciprocating Internal Combustion Engines (RICE) NESHAPs (U.S. EPA, 2010)
- Mercury and Air Toxics Standards (U.S. EPA, 2011)
- Cross-State Air Pollution Rule (U.S. EPA, 2011)[2]

Most areas of the U.S. will be required to demonstrate attainment with the new standards by 2020. As a result, for these areas, the correct baseline for estimating the incremental emissions reductions that would be needed to attain the more protective standards is a baseline with emissions projected to 2020 and adjusted to reflect the additional emissions reductions that would be needed to attain the current 15/35 $\mu g/m^3$ standard. For two areas in Southern California (South Coast and San Joaquin), the degree of projected non-attainment with the revised annual standard of 12 $\mu g/m^3$ is high enough that those counties are not expected to have to demonstrate attainment with the new standards by 2020. Instead, those two areas will likely have until 2025 to demonstrate attainment with the revised annual standard of 12 $\mu g/m^3$. As a result, for these two areas, the correct baseline for estimating the incremental emissions reductions that would be needed to attain the more protective standards is a baseline with emissions projected to 2025 adjusted to reflect the additional emissions reductions that would be needed to attain the current 15/35 $\mu g/m^3$ standard. The following paragraphs describe the steps we followed for this analysis.

This difference in attainment year is important because between 2020 and 2025, emissions from mobile sources in California are expected to be reduced due to continued fleet turn over from older, higher emitting vehicles to newer, lower emitting vehicles. These reductions in emissions will occur as a result of previous EPA and California rules for which costs (and benefits) have already been counted and thus will not be attributed to meeting the revised or alternative standards[3].

Modeling of two separate years is time prohibitive and would result in two separate years of benefits and costs which would not provide a complete picture of the nationwide costs and benefits of just meeting the new standards in either 2020 or 2025 because of differences in the baselines between the two years. To provide the most reasonable and reliable estimates of

[2] See Chapter 3, Section 3.2.1.4 for a discussion of the role CSAPR plays in the $PM_{2.5}$ RIA.
[3] See Chapter 3, Section 3.2.1.4 for a listing of the EPA and California rules reflected in the 15/35 analysis.

costs and benefits of full attainment for the nation, we constructed an analytical 2020 baseline for estimating the costs and benefits of attaining the selected annual standard of 12 µg/m^3 and alternative standards of 13 µg/m^3 and 11 µg/m^3 with the following characteristics: (1) reflects "on the books" regulations as implemented through 2020 for all nonattainment counties, and (2) mobile source emissions reductions expected to occur between 2020 and 2025 for California's South Coast and San Joaquin areas, which are likely to not demonstrate attainment until 2025. Essentially, we are adjusting emissions for the two areas in California to reflect future emissions reductions that they will achieve prior to their 2025 attainment date. This allows us to generate costs and benefits of full attainment without overstating the costs and benefits in those two areas which would occur if forced to apply costly emissions reductions in 2020 in areas that would not have to be incurred until 2025.

The 2020 analytical baseline for this analysis presents one scenario of future year air quality based upon specific control measures, promulgated federal rules such as MATS, years of air quality monitoring and emissions data. This analysis presents one illustrative strategy relying on the identified federal measures and other strategies that states may employ. States may ultimately employ other strategies and/or other federal rules may be adopted that would also help in achieving attainment. The number of counties that will be part of the designations process may be different than the number of counties projected to exceed as part of this analysis.

A map of the country is presented in Figure 4-1 which shows the counties projected to exceed the current $PM_{2.5}$ standard of 15/35 µg/m^3 in the year 2020. Control measures were identified in the control scenario run that were needed for these counties in the analytical baseline analysis to meet the current $PM_{2.5}$ standard. In addition, control measures were applied to five California counties adjacent to exceedance counties in order to address transport coming from these adjacent counties.

The additional known controls included in the analytical baseline to simulate attainment with the current $PM_{2.5}$ NAAQS are listed in Table 4-1; details regarding the individual controls are provided in Appendix 4.A. Controls were applied to directly emitted $PM_{2.5}$ and the $PM_{2.5}$ precursors of NO_x and SO_2 given that nitrate, sulfate, and primary $PM_{2.5}$ species usually dominate measured $PM_{2.5}$ based on speciation data measured at the Chemical Speciation Network (CSN) sites. Control measures that directly reduced emissions of $PM_{2.5}$ in proximity to the exceeding monitors were determined to be most effective at bringing areas into attainment, with NO_x and SO_2 controls supplementing the $PM_{2.5}$ controls depending upon the monitor speciation data. $PM_{2.5}$ control measures were applied in the county containing the exceeding monitor for the non-EGU point and area source emissions. If additional emissions control was needed, SO_2 and NO_x control measures were applied within the county exceeding and in a small number of cases, an adjacent county or counties.

For the analytical baseline, there were several geographic areas that did not reach attainment with known controls. For these geographic areas, we estimated the additional emission reductions needed beyond identified known controls for $PM_{2.5}$ to attain the standard.

4.2.4 Identify Known Controls Needed to Meet the Revised and Alternative Standards

After identifying the known controls in the control scenario that were needed to meet the analytical baseline, additional known controls needed to meet the revised and alternative standards were identified. The EPA used air quality modeling results to determine whether the control scenario was sufficient to meet the revised and alternative standards for each geographic area. Where the control scenario modeling resulted in design value reductions below the level needed for the revised or alternative standards for specific geographic areas, county-specific ratios of air quality response to emission reductions were used to determine the subset of controls that were needed to attain the standard. Where it was determined that the control scenario was not sufficient in attaining the standard, these same response factors were used to calculate the amount of additional emission reductions beyond known controls needed to meet the standard. For the revised and alternative control strategy analysis, known controls for two sectors were used: non-EGU point and area sources. Onroad mobile source controls were not used in the revised and alternative standards analysis because they were applied previously in the analytical baseline analysis where they were deemed to be most cost effective.

In the revised and alternative standards analysis, $PM_{2.5}$ controls were sought first because they were generally more cost-effective. If it was determined that additional controls were needed, SO_2 and NO_x control measures were selected depending on the chemistry of each specific geographic area.

It should be noted that while $PM_{2.5}$ controls were applied only within the counties with monitors projected to exceed the alternative standard being analyzed, SO_2 and NO_x controls were applied in the exceeding county as well as a small number of adjacent counties because of the transport of NO_x and SO_2 across counties. Table 4-2 shows the number of exceeding counties and the number of adjacent counties to which controls were applied for the revised and alternative standards. Table 4-3 shows the emission reductions from known controls for the revised and alternative standards analyzed.

Table 4-2. Number of Counties with Exceedances and Number of Additional Counties Where Reductions Were Applied

Revised/Alternative Standard	Number of Counties with Exceedances	Number of Additional Counties Where Reductions Were Applied
13	2	1
12	7	1
11	23	1

Table 4-3. Emission Reductions from Known Controls for the Revised and Alternative Standards[a]

Emission Reductions in 2020 (annual tons/year)				
Revised/ Alternative Standard	Region	$PM_{2.5}$	SO_2	NO_x
13	East	—	—	—
	West	—	—	—
	CA	53	—	—
	Total	**53**	**—**	**—**
12	East	—	—	—
	West	—	—	—
	CA	803	—	—
	Total	**803**	**—**	**—**
11	East	3,400	21,000	9
	West	75	43	—
	CA	930	—	—
	Total	**4,400**	**21,000**	**9**

[a] Estimates are rounded to two significant figures.

4.2.5 Identify Emission Reductions Beyond Known Controls Needed to Meet the Revised and Alternative Standards

There were several areas where known controls did not achieve enough emission reductions to attain the revised or alternative standards in 2020. To complete the analysis, the EPA then estimated the additional emission reductions beyond known controls needed to reach attainment. For information on the methodology used to develop those estimates, see Chapter

3, Section 3.3.2. Table 4-4 shows the emission reductions needed from unknown controls for the alternative standards analyzed.

Table 4-4. Emission Reductions Needed Beyond Known Controls for the Revised and Alternative Standards[a]

Emission Reductions in 2020 (annual tons/year)				
Revised/ Alternative Standard	**Region**	**$PM_{2.5}$**	**SO_2**	**NO_x**
13	East	—	—	—
	West	—	—	—
	CA	674	—	—
	Total	**674**	**—**	**—**
12	East	—	—	—
	West	—	—	—
	CA	3,190	—	—
	Total	**3,190**	**—**	**—**
11	East	4,800	—	—
	West	86	—	—
	CA	9,700	—	—
	Total	**15,000**	**—**	**—**

[a] Estimates are rounded to two significant figures.

4.3 Limitations and Uncertainties

The EPA's analysis is based on its best judgment for various input assumptions that are uncertain. As a general matter, the Agency selects the best available information from engineering studies of air pollution controls and has set up what it believes is the most reasonable modeling framework for analyzing the cost, emissions changes, and other impacts of regulatory controls. However, the estimates of emissions reductions associated with our control strategies above are subject to important limitations and uncertainties. We outline, and qualitatively assess the impact of, those limitations and uncertainties that are most significant.

A number of limitations and uncertainties are associated with the analysis of emission control measures are listed in Table 4-5.

Table 4-5. Summary of Qualitative Uncertainty for Elements of Control Strategies

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Costs[a]	Degree of Confidence in Our Analytical Approach[b]	Ability to Assess Uncertainty[c]
Uncertainties Associated with PM Concentration Changes				
Projections of future levels of emissions and emissions reductions necessary to achieve the NAAQS	Both[d]	Medium	Medium	Tier 1
Responsiveness of air quality model to changes in precursor emissions from control scenarios	Both	Medium-high	Medium	Tier 1
Air quality model chemistry, particularly for formation of ambient nitrate concentrations	Both	Medium	High	Tier 1
Post-processing of air quality modeled concentrations to estimate future-year $PM_{2.5}$design value and spatial fields of $PM_{2.5}$ concentrations	Both	High	High	Tier 1
Uncertainties Associated with Control Strategy Development				
Control Technology Data ▪ Technologies applied may not reflect most current emerging devices that may be available in future years ▪ Control efficiency data is dependent upon equipment being well maintained. ▪ Area source controls assume a constant estimate of emission reductions, despite variability in extent and scale of application.	Both	Medium-high	High	Tier 2
Control Strategy Development ▪ States may develop different control strategies than the ones illustrated ▪ Lack of data on analytical baseline controls from current SIPs ▪ Timing of control strategies may be different than envisioned in RIA ▪ Controls are applied within the county with the exceeding monitor. In some cases, additional known controls are also applied in adjacent contributing counties. ▪ Emissions growth and control from new sources locating in these analysis areas is not included.	Both	Medium-high	Medium-high	Tier 0

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Costs[a]	Degree of Confidence in Our Analytical Approach[b]	Ability to Assess Uncertainty[c]
Technological Change ▪ Emission reductions do not reflect potential effects of technological change that may be available in future years ▪ Effects of "learning by doing" are no accounted for in the emission reduction estimates	Likely over-estimate	Medium-high	Low	Tier 0
Emission Reductions from Unidentified Controls ▪ emission control cut points for each pollutant	Both	High	Low	Tier 1

[a] Magnitude of Impact

High—If error could influence the total costs by more than 25%

Medium—If error could influence the total costs by 5%–25%

Low—If error could influence the total costs by less than 5%

[b] Degree of Confidence in Our Analytic Approach

High—The current evidence is plentiful and strongly supports the selected approach

Medium—Some evidence exists to support the selected approach, but data gaps are present

Low—Limited data exists to support the selected approach

[c] Ability to Assess Uncertainty (using WHO Uncertainty Framework)

Tier 0—Screening level, generic qualitative characterization

Tier 1—Scenario-specific qualitative characterization

Tier 2—Scenario-specific sensitivity analysis

Tier 3—Scenario-specific probabilistic assessment of individual and combined uncertainty

[d] Future expected emissions are difficult to predict because they depend on many independent factors. Emission inventories are aggregated from many spatially and technically diverse sources of emissions, so simplifying assumptions are necessary to make estimating the future tractable.

4.4 References

U.S. Environmental Protection Agency (U.S. EPA). 1999. Control of Air Pollution from New Motor Vehicles: Tier 2 Motor Vehicle Emissions Standards and Gasoline Sulfur Control Requirements. Office of Transportation and Air Quality. Available at http://www.epa.gov/tier2/frm/fr-t2reg.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2000. Heavy-Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements. Office of Transportation and Air Quality. Available at http://www.epa.gov/otaq/highway-diesel/index.htm.

U.S. Environmental Protection Agency (U.S. EPA). 2004. Control of Emissions of Air Pollution from Nonroad Diesel Engines and Fuel. Office of Transportation and Air Quality. Available at http://www.regulations.gov/search/Regs/contentStreamer?objectId=09000064800be203&disposition=attachment&contentType=pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2005. New Emission Standards for New Commercial Aircraft Engines. Office of Transportation and Air Quality. Available at http://www.epa.gov/oms/regs/nonroad/aviation/420f05015.htm.

U.S. Environmental Protection Agency (U.S. EPA). 2005b. Regional Haze Regulations and Guidelines for Best Available Retrofit Technology Determinations. Office of Air Quality Planning and Standards. Available at http://www.epa.gov/fedrgstr/EPA-AIR/2005/July/Day-06/a12526.pdf and http://www.epa.gov/visibility/fs_2005_6_15.html.

U.S. Environmental Protection Agency (U.S. EPA). 2005. Clean Air Interstate Rule. Office of Atmospheric Programs, Washington, D.C. Available at http://edocket.access.gpo.gov/2005/pdf/05-5723.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2007. Guidance on the Use of Models and other Analyses for Demonstrating Attainment of Air Quality Goals for Ozone, $PM_{2.5}$, and Regional Haze. Office of Air Quality Planning and Standards, Research Triangle Park, NC. Available at http://www.epa.gov/ttn/scram/guidance/guide/final-03-pm-rh-guidance.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2008. Control of Emissions for Nonroad Spark Ignition Engines and Equipment. Office of Transportation and Air Quality. Available at http://www.epa.gov/otaq/regs/nonroad/marinesi-equipld/bondfrm.pdf.

U.S. Environmental Protection Agency (U.S. EPA).2008. Emissions Standards for Locomotives and Marine Compression-Ignition Engines. Office of Transportation and Air Quality. Available at http://www.epa.gov/nonroad/420f08004.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2009. Standards of Performance for New Stationary Sources and Emission Guidelines for Existing Sources: Hospital/Medical/Infectious Waste Incinerators. Office of Air Quality Planning and Standards, Research Triangle Park, NC. Available at http://www.epa.gov/ttn/atw/129/hmiwi/fr06oc09.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2010. Control of Emissions from New Marine Compression-Ignition Engines at or Above 30 Liters per Cylinder**.** Office of Transportation and Air Quality. Available at http://www.regulations.gov/search/Regs/contentStreamer?objectId=0900006480ae43a6&disposition=attachment&contentType=pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011. Control Strategy Tool (CoST) Documentation Report. Office of Air Quality Planning and Standards, Research Triangle Park, NC. Available at http://www.epa.gov/ttnecas1/cost.htm.

U.S. Environmental Protection Agency (U.S. EPA). 2011. Cross-State Air Pollution Rule. Office of Atmospheric Programs, Washington, D.C. Available at http://www.gpo.gov/fdsys/pkg/FR-2011-08-08/pdf/2011-17600.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011. Mercury and Air Toxics Standards. Office of Atmospheric Programs, Washington, D.C. Available at www.gpo.gov/fdsys/pkg/FR-2012-02-16/pdf/2012-806.pdf.

APPENDIX 4.A
ADDITIONAL CONTROL STRATEGY INFORMATION

4.A.1 Control Measures for Stationary Sources

This appendix describes measures that were employed in this analysis to illustrate a hypothetical scenario for controlling emissions of PM and precursors from non-EGU point and nonpoint (area) source categories to attain the baseline or to attain the revised or alternative standards for $PM_{2.5}$. Most of the control measures available are add-on technologies but some other technologies and practices that are not add-on in nature can reduce emissions of PM and PM precursors.

4.A.1.1 PM Emissions Control Technologies[1]

This section summarizes control measures focused on reduction of $PM_{2.5}$ from non-EGU point and nonpoint sources. However, it should be noted that PM_{10} will also be reduced by these measures. The amount of PM_{10} reduction varies by the fraction of PM_{10} in the inlet stream to the control measure and the specific design of the measure.

4.A.1.1.1 PM Control Measures for Non-EGU Point Sources

Most control measures for non-EGU point sources are add-on technologies. These technologies include: fabric filters (baghouses), ESPs, and wet PM scrubbers. Fabric filters collect particles with sizes ranging from below 1 micrometer to several hundred micrometers in diameter at efficiencies in excess of 99%, and this device is used where high-efficiency particle collection is required. A fabric filter unit consists of one or more isolated compartments containing rows of fabric bags in the form of round, flat, or shaped tubes, or pleated cartridges. Particle-laden gas passes up (usually) along the surface of the bags then radially through the fabric. Particles are retained on the upstream face of the bags, and the cleaned gas stream is vented to the atmosphere. The filter is operated cyclically, alternating between relatively long periods of filtering and short periods of cleaning. Dust that accumulates on the bags is removed from the fabric surface when cleaning and deposited in a hopper for subsequent disposal.

ESPs use electrical forces to move particles out of a flowing gas stream and onto collector plates. The particles are given an electrical charge by forcing them to pass through a corona, a region in which gaseous ions flow. The electrical field that forces the charged particles to the plates comes from electrodes maintained at high voltage in the center of the flow lane.

[1] The descriptions of add-on technologies throughout this section are taken from the EPA Air Pollution Control Cost Manual, Sixth Edition. This is found on the Internet at http://epa.gov/ttn/catc/products.html#cccinfo.

Once particles are on the collector plates, they must be removed without re-entraining them into the gas stream. This is usually accomplished by rapping the plates mechanically which loosens the collected particles from the collector plates, allowing the particles to slide down into a hopper from which they are evacuated. This removal of collected particles is typical of a "dry" ESP. A "wet" ESP operates by having a water flow applied intermittently or continuously to wash away the collected particles for disposal. The advantage of wet ESPs is that there are no problems with rapping re-entrainment or with "back coronas" (unintended injection of positively charged ions which reduces the charge on particles and lowers the collection efficiency). The disadvantage is that the collected slurry must be handled more carefully than a dry product, adding to the expense of disposal. ESPs capture particles with sizes ranging from below 1 micrometer to several hundred micrometers in diameter at efficiencies from 95 to up to 99% and higher.

Wet PM scrubbers remove PM and acid gases from waste gas streams of stationary point sources. The pollutants are removed primarily through the impaction, diffusion, interception and/or absorption of the pollutant onto droplets of liquid. The liquid containing the pollutant is then collected for disposal. Collection efficiencies for wet scrubbers vary by scrubber type, and with the PM size distribution of the waste gas stream. In general, collection efficiency decreases as the PM size decreases. Collection efficiencies range from in excess of 99% for venturi scrubbers to 40% to 60% for simple spray towers. Wet scrubbers are generally smaller and more compact than fabric filters or ESPs, and have lower capital cost and comparable operation and maintenance (O&M) costs. Wet scrubbers, however, operate with a higher pressure drop than either fabric filters or ESPs, thus leading to higher energy costs. In addition, they are limited to lower waste gas flow rates and operating temperatures than fabric filters or ESPs, and also generate sludge that requires additional treatment or disposal. This RIA only applies wet scrubbers to fluid catalytic cracking units (FCCUs) at petroleum refineries.

In addition, we also examined additional add-on control measures specifically for steel mills. Virtually all steel mills have some type of PM control measure, but there is additional equipment that in many cases could be installed to further reduce emissions. Capture hoods that route PM emissions from a blast furnace casthouse to a fabric filter can provide 80% to 90% additional emission reductions from a steel mill. Other capture and control systems at blast oxygen furnaces (BOFs) can also provide 80% to 90% additional reductions.

Table 4.A-1 lists some of these technologies. For more information on these technologies, refer to the EPA Air Pollution Control Cost Manual.[2]

Table 4.A-1. Example PM Control Measures for Non-EGU Point Source Categories

Control Measure	Sector(s) to which Control Measure Can Apply	Control Efficiency (percent)	Average Annualized Cost/Ton
Fabric Filters[a]	Industrial Boilers, Iron and Steel Mills, Pulp and Paper Mills	98 to 99.9	\$2,000–\$100,000
ESPs—wet or dry[a]	Industrial Boilers, Iron and Steel Mills, Pulp and Paper Mills	95 to 99.9	\$1,000–\$20,000
Wet Scrubbers	Industrial Boilers, Iron and Steel Mills	40 to 99	\$750–\$2,800
Secondary Capture and Control Systems—Capture Hoods for Blast Oxygen Furnaces	Coke Ovens	80 to 90	\$5,000
CEM Upgrade and Increased Monitoring Frequency	Non-EGUs with an ESP	5 to 7	\$600–\$5,000

[a] CoST contains equations to estimate capital and annualized costs for ESP and FF installation and operation. The average annualized cost/ton estimates presented here for these control measures are outputs from our modeling, not inputs. They also reflect applications of control where there is no PM control measure currently operating except if the control measure is an upgrade (e.g., ESP upgrades).

4.A.1.1.2 PM Control Measures for Nonpoint Sources

Specific controls exist for a number of stationary nonpoint sources. Nonpoint source PM controls at stationary sources include:

- catalytic oxidizers on conveyorized charbroilers at restaurants (up to 80% reduction of PM), and
- diesel particulate filters, applied to existing diesel-fueled compression-ignition (C-I) engines (up to a 90reduction in fine PM).

Diesel particulate filters are being applied to new C-I engines as part of a NSPS that was implemented beginning in 2006.

[2] Please refer to Footnote 1.

Table 4.A-2. Example PM Control Measures for Nonpoint Sources[a]

Control Measures	Sectors to which These Control Measures Can Apply	Control Efficiency (percent)	Average Annualized Cost/ton
Catalytic oxidizers for conveyorized charbroilers	Restaurants	83	$1,300
Replace open burning of wood waste with chipping for landfill disposal	Residential waste sources	Near 100	$3,500

[a] The estimates for these control measures reflect applications of control where there is no PM nonpoint source control measure currently operating. Also, the control efficiency is for total PM, and thus accounts for PM_{10} and $PM_{2.5}$. Data for these measures is available in the CoST Control Measures Documentation Report at http://www.epa.gov/ttn/ecas/models/CoST_CMDB_Document_2010-06-09.pdf.

4.A.1.2 SO_2 Control Measures

4.A.1.2.1 SO_2 Control Measures for Non-EGU Point Sources

The SO_2 emission control measures used in this analysis are similar to those used in the $PM_{2.5}$ RIA prepared about four years ago. Flue gas desulfurization (FGD) scrubbers can achieve 95–98% control of SO_2 for Non-EGU point sources and for utility boilers. Spray dryer absorbers (SDA) are another commonly employed technology, and SDA can achieve up to 90% or more control of SO_2. For specific source categories, other types of control technologies are available that are more specific to the sources controlled. Table 4.A-3 lists some of these technologies. For more information on these technologies, please refer to the CoST control measures documentation report.[3]

[3] For a complete description of the control technologies used in CoST, please refer to the report at http://www.epa.gov/ttn/ecas/models/CoST_CMDB_Document_2010-06-09.pdf.

Table 4.A-3. Example SO_2 Control Measures for Non-EGU Point[a]

Control Measure	Sectors to Which These Control Measures Can Be Applied	Control Efficiency (percent)	Average Annualized Cost/Ton (2006$)
Wet and Dry FGD scrubbers and SDA	ICI boilers—all fuel types, kraft pulp mills, Mineral Products (e.g., Portland cement plants (all fuel types), primary metal plants, petroleum refineries	95—FGD scrubbers, 90—for SDA	\$800–\$8,000—FGD \$900–\$7,000—SDA
Increase percentage sulfur conversion to meet sulfuric acid NSPS (99.7% reduction)	Sulfur recovery plants	75–95	\$4,000
Sulfur recovery and/or tail gas treatment	Sulfuric Acid Plants	95–98	\$1,000–\$4,000
Cesium promoted catalyst	Sulfuric Acid Plants with Double-Absorption process	50%	\$1,000

[a] Sources: CoST control measures documentation report, May 2008, NESCAUM Report on Applicability of NO_x, SO_2, and PM Control Measures to Industrial Boilers, November 2008 available at http://www.nescaum.org/documents/ici-boilers-20081118-final.pdf, and Comprehensive Industry Document on Sulphuric Acid Plant, Govt. of India Central Pollution Control Board, May 2007. The estimates for these control measures reflect applications of control where there is no SO_2 control measure currently operating except for the Cesium promoted catalyst.

4.A.1.2.2 SO_2 Control Technology for Nonpoint Sources

Fuel switching from high to low-sulfur fuels is the predominant control measure available for SO_2 nonpoint sources. For home heating oil users, our analyses included switching from a high-sulfur oil (approximately 2,500 parts per million (ppm) sulfur content) to a low-sulfur oil (approximately 500 ppm sulfur). A similar control measure is available for oil-fired industrial boilers. For more information on these measures, please refer to the CoST control measures documentation report.[4]

4.A.1.3 NO_x Emissions Control Measures

4.A.1.3.1 NO_x Control Measures for Non-EGU Point Sources

This section describes available measures for controlling emissions of NO_x from non-EGU point sources. In general, low-NO_x burners (LNB) are often applied as a control technology for industrial boilers and for some other non-EGU sources because of their wide applicability and cost-effectiveness. While all controls presented in this analysis are considered generally

[4] Please refer to Footnote 3.

technically feasible for each class of sources, source-specific cases may exist where a control technology is in fact not technically feasible.

Several types of NO_x control technologies exist for non-EGU sources: selective catalytic reduction (SCR), selective noncatalytic reduction (SNCR), natural gas reburn (NGR), coal reburn, and low-NO_x burners. The two control measures chosen most often were LNB and SCR because of their breadth of application. In some cases, LNB accompanied by flue gas recirculation (FGR) is applicable, such as when fuel-borne NO_x emissions are expected to be of greater importance than thermal NO_x emissions. When circumstances suggest that combustion controls are not feasible as a control technology (e.g., sintering processes, coke oven batteries, sulfur recovery plants), SNCR or SCR may be an appropriate choice. Finally, SCR can be applied along with a combustion control such as LNB with overfire air (OFA) to further reduce NO_x emissions. All of these control measures are available for application on industrial boilers.

Besides industrial boilers, other non-EGU source categories covered in this RIA include petroleum refineries, kraft pulp mills, cement kilns, stationary internal combustion engines, glass manufacturing, combustion turbines, and incinerators. NO_x control measures available for petroleum refineries, particularly process heaters at these plants, include LNB, SNCR, FGR, and SCR along with combinations of these technologies. NO_x control measures available for kraft pulp mills include those available to industrial boilers, namely LNB, SCR, SNCR, along with water injection (WI). NO_x control measures available for cement kilns include those available to industrial boilers, namely LNB, SCR, and SNCR. In addition, mid-kiln firing (MKF), ammonia-based SNCR, and biosolids injection can be used on cement kilns where appropriate. Non-selective catalytic reduction (NSCR) can be used on stationary internal combustion engines. OXY-Firing, a technique to modify combustion at glass manufacturing plants, can be used to reduce NO_x emissions at such plants. LNB, SCR, and SCR combined with steam injection (SI) are available measures for combustion turbines. Finally, SNCR is an available control technology at incinerators. Table 4.A-4 lists typical examples of the control measures available for these categories. For more information on these measures, please refer to the CoST control measures documentation report.[5]

[5] Please refer to Footnote 3.

Table 4.A-4. Example NO_x Control Measures for Non-EGU Source Categories[a]

Control Measures	Sectors to Which These Control Measures Apply	Control Efficiency (percent)	Average Annualized Cost/ton
LNB	Industrial boilers—all fuel types, Petroleum refineries, Cement manufacturing, Pulp and Paper mills	25 to 50%	$200 to $1,000
LNB + FGR	Petroleum refineries	55	$4,000
SNCR (urea-based or not)	Industrial boilers—all fuel types, Petroleum refineries, Cement manufacturing, pulp and paper mills, incinerators	45 to 75	$1,000 to $2,000
SCR	Industrial boilers—all fuel types, Petroleum refineries, Cement manufacturing, pulp and paper mills, Combustion turbines	80 to 90	$2,000 to 7,000
OXY-Firing	Glass manufacturing	85	$2,500 to 6,000
NSCR	Stationary internal combustion engines	90	500
MKF	Cement manufacturing—dry	25	−$460 to 720
Biosolids Injection	Cement manufacturing—dry	23	$300
SCR + SI	Industrial boilers—all fuel types	95	$2,700

[a] Source: CoST control measures documentation report (June 2010). Note: a negative sign indicates a cost savings from application of a control measure. The estimates for these control measures reflect applications of control where there is no NO_x control measure currently operating except for post-combustion controls such as SCR and SNCR. For these measures, the costs presume that a NO_x combustion control (such as LNB) is already operating on the unit to which the SCR or SNCR is applied.

CHAPTER 5
HUMAN HEALTH BENEFITS ANALYSIS APPROACH AND RESULTS

5.1 Synopsis

This chapter presents the estimated human health benefits for the revised National Ambient Air Quality Standards (NAAQS) for particulate matter (PM). In this chapter, we quantify the health-related benefits of the fine particulate matter ($PM_{2.5}$)-related air quality improvements resulting from the illustrative emission control scenarios that reduce emissions of directly emitted particles and precursor pollutants including SO_2 and NO_x to reach alternative $PM_{2.5}$ NAAQS levels in 2020.[1]

These benefits are relative to an analytical baseline reflecting nationwide attainment of the current primary $PM_{2.5}$ standards (i.e., annual standard of 15 µg/m^3 and 24-hour standard of 35 µg/m^3) that includes promulgated national regulations and illustrative emission controls to simulate attainment with 15/35 as well as a NO_x emission adjustment to reflect expected reductions in mobile NO_x emissions between 2020 and 2025. We project $PM_{2.5}$ levels in certain areas that would exceed the revised annual standard of 12 µg/m^3 as well as alternative annual standards of 13 and 11 µg/m^3 after simulated attainment with 15/35 in the analytical baseline. Table 5-1 summarizes the total monetized benefits of the revised and alternative $PM_{2.5}$ standards in 2020. These estimates reflect the sum of the economic value of estimated $PM_{2.5}$ mortality impacts identified and the value of all morbidity impacts.

The estimated benefits for the revised and alternative standards are in addition to the substantial benefits estimated for several recent implementation rules (U.S. EPA, 2009a, 2011c, 2011d, 2011e). Rules such as the Mercury and Air Toxics Standard (MATS) and other emission reductions will have substantially reduced ambient $PM_{2.5}$ concentrations by 2020 in the East, such that no additional controls would be needed in the East for the revised annual standard of 12 µg/m^3. Thus, all of the estimated benefits occur in California because this is the only state that needs additional air quality improvement beyond the analytical baseline after accounting for air quality improvements from recent rules. Because of the national focus of many of the inputs to the benefits model, benefits estimated for any particular location have greater uncertainty than benefits occurring nationally. Compared with the proposal benefits, the estimated benefits for the revised standard are about double due to a combination of updates

[1] The estimates in this chapter reflect incremental emissions reductions from an analytical baseline that gives an adjustment to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

to the analytic baseline (See Chapter 3). We believe that all of these updates improve our estimate of benefits for the revised annual primary standard.

Table 5-1. Estimated Monetized Benefits of the of Revised and Alternative Annual $PM_{2.5}$ Standards in 2020 Incremental to the Analytical Baseline (billions of 2010$)[a]

Benefits Estimate	13 µg/m^3	12 µg/m^3	11 µg/m^3
Economic value of avoided $PM_{2.5}$-related morbidities and premature deaths using $PM_{2.5}$ mortality estimate from Krewski et al. (2009)			
3% discount rate	$1.3 + B	$4.0 +B	$13 + B
7% discount rate	$1.2 + B	$3.6 +B	$12 + B
Economic value of avoided $PM_{2.5}$-related morbidities and premature deaths using $PM_{2.5}$ mortality estimate from Lepeule et al. (2012)			
3% discount rate	$2.9 + B	$9.1 +B	$29 + B
7% discount rate	$2.6 + B	$8.2 +B	$26 + B

[a] Rounded to two significant figures. Avoided premature deaths account for over 98% of monetized benefits here, which are discounted over the SAB-recommended 20-year segmented lag. It was not all possible to quantify all benefits due to data limitations in this analysis. "B" is the sum of all unquantified health benefits and welfare co-benefits.

As we describe in detail below, we estimate PM-related health impacts using concentration-response relationships drawn from the epidemiological literature. In general, we are more confident in the magnitude of the risks we estimate from simulated $PM_{2.5}$ concentrations that coincide with the bulk of the observed PM concentrations in the epidemiological studies that are used in the benefits estimates. Likewise, we are less confident in the risk we estimate from simulated $PM_{2.5}$ concentrations that fall below the bulk of the observed data in these studies. As noted in the preamble to the rule, the range from the 25th to 10th percentiles of the air quality data in the epidemiology studies is a reasonable range below which we start to have appreciably less confidence in the magnitude of the associations observed in the epidemiological studies. Most of the estimated avoided premature deaths occur at or above the lowest measured $PM_{2.5}$ concentration in the two studies that are used to estimate mortality benefits.

In addition to $PM_{2.5}$ benefits, implementation of emissions controls to reach some of the alternative $PM_{2.5}$ standards would reduce other ambient pollutants, such as SO_2, NO_2, and ozone. However, because the method used in this analysis to simulate attainment does not account for changes in ambient concentrations of other pollutants, we were not able to quantify the co-benefits of reduced exposure to other pollutants. In addition, due to data and methodology limitations, we were unable to estimate additional health benefits associated with exposure to $PM_{2.5}$ or the additional co-benefits from improvements in welfare effects (i.e., non-health effects) associated with emission reductions to attain the primary standard, such as visibility. We describe the unquantified health benefits in this chapter and the unquantified welfare co-benefits in Chapter 6.

5.2 Overview

This chapter contains a subset of the estimated health benefits of the revised and alternative $PM_{2.5}$ standards in 2020 that the EPA was able to quantify, given the available resources and methods. The analysis in this chapter aims to characterize the benefits of the air quality changes resulting from the implementation of new PM standards by answering two key questions:

1. What are the health effects of changes in ambient particulate matter ($PM_{2.5}$) resulting from reductions in directly emitted $PM_{2.5}$ and precursors due to the attainment of a new $PM_{2.5}$ standard?
2. What is the economic value of these effects?

In this analysis, we consider an array of health impacts attributable to changes in $PM_{2.5}$. The *Integrated Science Assessment for Particulate Matter* ("PM ISA")(U.S. EPA, 2009b) identifies the human health effects associated with ambient particles, which include premature mortality and a variety of morbidity effects associated with acute and chronic exposures. Table 5-2 summarizes human health categories contained within the core benefits estimate as well as those categories that were unquantified due to limited data or resources. It is important to emphasize that the list of unquantified benefit categories is not exhaustive, nor is quantification of each effect complete. To identify human health benefits that we could quantify with confidence we selected endpoints that were classified as causal or likely causal in the PM ISA, and we excluded effects not identified as having at least a causal, likely causal, or suggestive relationship with the affected pollutants in the most recent comprehensive scientific assessment, such as an ISA. This decision does not imply that additional relationships between these and other human health and environmental co-benefits and the affected pollutants do not exist. Due to this decision criterion, we excluded some effects that were identified in previous lists of unquantified benefits in other RIAs (e.g., UVb exposure).

The benefits analysis in this chapter relies on an array of data inputs—including air quality modeling, health impact functions and valuation estimates among others—which are themselves subject to uncertainty and may also in turn contribute to the overall uncertainty in this analysis. We employ several techniques to characterize this uncertainty, which are described in detail in Section 5.4.

Table 5-2. Human Health Effects of Pollutants Potentially Affected by Strategies to Attain the Primary $PM_{2.5}$ Standards

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Improved Human Health				
Reduced incidence of premature mortality from exposure to $PM_{2.5}$	Adult premature mortality based on cohort study estimates and expert elicitation estimates (age >25 or age >30)	✓	✓	Section 5.6
	Infant mortality (age <1)	✓	✓	Section 5.6
	Non-fatal heart attacks (age > 18)	✓	✓	Section 5.6
	Hospital admissions—respiratory (all ages)	✓	✓	Section 5.6
	Hospital admissions—cardiovascular (age >20)	✓	✓	Section 5.6
	Emergency department visits for asthma (all ages)	✓	✓	Section 5.6
	Acute bronchitis (age 8–12)	✓	✓	Section 5.6
	Lower respiratory symptoms (age 7–14)	✓	✓	Section 5.6
	Upper respiratory symptoms (asthmatics age 9–11)	✓	✓	Section 5.6
	Asthma exacerbation (asthmatics age 6–18)	✓	✓	Section 5.6
	Lost work days (age 18–65)	✓	✓	Section 5.6
	Minor restricted-activity days (age 18–65)	✓	✓	Section 5.6
	Chronic Bronchitis (age >26)	—[a]	—[a]	Section 5.6
	Emergency department visits for cardiovascular effects (all ages)	—[a]	—[a]	Section 5.6
	Strokes and cerebrovascular disease (age 50–79)	—[a]	—[a]	Section 5.6
	Other cardiovascular effects (e.g., other ages)	—	—	PM ISA[b]
	Other respiratory effects (e.g., pulmonary function, non-asthma ER visits, non-bronchitis chronic diseases, other ages and populations)	—	—	PM ISA[b]
	Reproductive and developmental effects (e.g., low birth weight, pre-term births, etc.)	—	—	PM ISA[b,c]
	Cancer, mutagenicity, and genotoxicity effects	—	—	PM ISA[b,c]

(continued)

Table 5-2. Human Health Effects of Pollutants Potentially Affected by Strategies to Attain the Primary $PM_{2.5}$ Standards (continued)

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Reduced incidence of mortality from exposure to ozone	Premature mortality based on short-term study estimates (all ages)	—	—	Ozone ISA[d]
	Premature mortality based on long-term study estimates (age 30–99)	—	—	Ozone ISA[d]
	Hospital admissions—respiratory causes (age > 65)	—	—	Ozone ISA[d]
	Hospital admissions—respiratory causes (age <2)	—	—	Ozone ISA[d]
	Emergency department visits for asthma (all ages)	—	—	Ozone ISA[d]
	Minor restricted-activity days (age 18–65)	—	—	Ozone ISA[d]
	School absence days (age 5–17)	—	—	Ozone ISA[d]
	Decreased outdoor worker productivity (age 18–65)	—	—	Ozone ISA[d]
	Other respiratory effects (e.g., premature aging of lungs)	—	—	Ozone ISA[b]
	Cardiovascular and nervous system effects	—	—	Ozone ISA[c]
	Reproductive and developmental effects	—	—	Ozone ISA[c]
Reduced incidence of morbidity from exposure to NO_2	Asthma hospital admissions (all ages)	—	—	NO_2 ISA[d]
	Chronic lung disease hospital admissions (age > 65)	—	—	NO_2 ISA[d]
	Respiratory emergency department visits (all ages)	—	—	NO_2 ISA[d]
	Asthma exacerbation (asthmatics age 4–18)	—	—	NO_2 ISA[d]
	Acute respiratory symptoms (age 7–14)	—	—	NO_2 ISA[d]
	Premature mortality	—	—	NO_2 ISA[b,c]
	Other respiratory effects (e.g., airway hyperresponsiveness and inflammation, lung function, other ages and populations)	—	—	NO_2 ISA[b,c]

(continued)

Table 5-2. Human Health Effects of Pollutants Potentially Affected by Strategies to Attain the Primary $PM_{2.5}$ Standards (continued)

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Reduced incidence of morbidity from exposure to SO_2	Respiratory hospital admissions (age > 65)	—	—	SO_2 ISA[d]
	Asthma emergency department visits (all ages)	—	—	SO_2 ISA[d]
	Asthma exacerbation (asthmatics age 4–12)	—	—	SO_2 ISA[d]
	Acute respiratory symptoms (age 7–14)	—	—	SO_2 ISA[d]
	Premature mortality	—	—	SO_2 ISA[b,c]
	Other respiratory effects (e.g., airway hyperresponsiveness and inflammation, lung function, other ages and populations)	—	—	SO_2 ISA[b,c]
Reduced incidence of morbidity from exposure to methylmercury (through role of sulfate in methylation)	Neurologic effects—IQ loss	—	—	IRIS; NRC, 2000[d]
	Other neurologic effects (e.g., developmental delays, memory, behavior)	—	—	IRIS; NRC, 2000[b]
	Cardiovascular effects	—	—	IRIS; NRC, 2000[b,c]
	Genotoxic, immunologic, and other toxic effects	—	—	IRIS; NRC, 2000[b,c]

[a] We quantify these benefits in a sensitivity analysis, but not in the core analysis.

[b] We assess these benefits qualitatively because we do not have sufficient confidence in available data or methods.

[c] We assess these benefits qualitatively because current evidence is only suggestive of causality or there are other significant concerns over the strength of the association.

[d] We assess these benefits qualitatively due to time and resource limitations for this analysis.

As described in Chapter 1 of this RIA, there are important differences worth noting in the design and analytical objectives of NAAQS RIAs compared to RIAs for implementation rules, such as the recent MATS rule (U.S. EPA, 2011d). The NAAQS RIAs illustrate the potential costs and benefits of attaining a revised air quality standard nationwide based on an array of emission reduction strategies for different sources including known and unknown controls, incremental to implementation of existing regulations and controls needed to attain the current standards. In short, NAAQS RIAs hypothesize, but do not predict, the emission reduction strategies that States may choose to enact when implementing a revised NAAQS. The setting of a NAAQS does not directly result in costs or benefits, and as such, NAAQS RIAs are merely illustrative and the estimated costs and benefits are not intended to be added to the costs and benefits of other regulations that result in specific costs of control and emission reductions. By contrast, the emission reductions from implementation rules are generally for specific, well-characterized sources, such as the recent MATS rule addressing emissions from coal and oil-fired electricity generating units (U.S. EPA, 2011d). In general, the EPA is more confident in the magnitude and location of the emission reductions for implementation rules. As such, emission reductions achieved under promulgated implementation rules such as MATS have been reflected in the baseline of this NAAQS analysis. Subsequent implementation rules will be reflected in the baseline for the next PM NAAQS review. For this reason, the benefits estimated provided in this RIA and all other NAAQS RIAs should not be added to the benefits estimated for implementation rules.

5.3 Updated Methodology Presented in this RIA

The benefits analysis presented in this chapter incorporates an array of policy and technical changes that the Agency has adopted since the previous review of the $PM_{2.5}$ standards in 2006, and since publication of the proposal RIA for this rulemaking. Below we note the aspects of this analysis that differ from the proposal RIA (U.S. EPA, 2012a):

1. *Incorporation of the newest Harvard Six Cities mortality study*. In 2012, Lepeule et al. published an extended analysis of the Six Cities cohort. Compared to the study it replaces (Laden et al., 2006), this new analysis follows the cohort for a longer time and includes more years of $PM_{2.5}$ monitoring data. The all-cause $PM_{2.5}$ mortality risk coefficient drawn from Lepeule et al. (2012) is roughly similar to the Laden et al. (2006) risk coefficient applied in the EPA's recent analyses of long-term $PM_{2.5}$ mortality and has narrower confidence intervals.

2. *Updated demographic data*. We updated the population demographic data in BenMAP to reflect the 2010 Census and future projections based on economic forecasting models developed by Woods and Poole, Inc. (Woods and Poole, 2012). These data replace the earlier demographic projection data from Woods and Poole (2007).

3. *Incorporation of new morbidity studies*. Since the publication of the PM ISA (U.S. EPA, 2009) the epidemiological literature has produced several new studies examining the association between short-term $PM_{2.5}$ exposure and respiratory and cardiovascular hospitalizations, respiratory and cardiovascular emergency department visits, and stroke. Upon careful evaluation of this new literature identified in the *Provisional Assessment of Recent Studies on Health Effects of Particulate Matter Exposure* ("Provisional Assessment")(U.S. EPA, 2012b) we added several new studies to our health impact assessment.

4. *Updated the survival rates for non-fatal acute myocardial infarctions*. Based on recent data from Agency for Healthcare Research and Quality's Healthcare Utilization Project National Inpatient Sample database (AHRQ, 2009), we identified death rates for adults hospitalized with acute myocardial infarction stratified by age. These rates replace the survival rates from Rosamond et al. (1999).

5. *Expanded uncertainty assessment*. We clarified the comprehensive assessment of the various uncertain parameters and assumptions within the benefits analysis and expanded the evaluation of air quality benchmarks (previously the LML analysis).

Although the list above identifies the major changes implemented since the proposal RIA, the EPA has also updated several additional components of the benefits analysis since the 2006 PM NAAQS RIA (U.S. EPA, 2006a). In the Portland Cement NESHAP proposal RIA (U.S. EPA, 2009a), the Agency no longer assumed a concentration threshold in the concentration-response function for $PM_{2.5}$-related health effects and began using the benefits derived from the two major cohort studies of $PM_{2.5}$ and mortality as the core benefits estimates, while still including a range of sensitivity estimates based on the EPA's $PM_{2.5}$ mortality expert elicitation. In the NO_2 NAAQS proposal RIA (U.S. EPA, 2009a), we revised the estimate used for the value-of-a-statistical life to be consistent with Agency guidance. In the proposed CSAPR (previously the "Transport Rule") (U.S. EPA, 2010g), we incorporated the "lowest measured level" assessment to help characterize uncertainty in estimates of benefits of reductions in $PM_{2.5}$ at lower baseline concentrations of $PM_{2.5}$. In the final CSAPR (U.S. EPA, 2011c), we updated the baseline incidence rates for hospital admissions and emergency department visits and asthma prevalence rates. We direct the reader to each of these RIAs for more information on these changes. In the proposal RIA for this NAAQS review (U.S. EPA, 2012a), we updated the American Cancer Society cohort study to Krewski et al. (2009), updated health endpoints in the core and sensitivity analyses, incorporated new morbidity studies, updated the median wage data in the cost-of-illness studies, and expanded the uncertainty assessment.

5.4 Human Health Benefits Analysis Methods

We follow a "damage-function" approach in calculating total benefits of the modeled changes in environmental quality.[2] This approach estimates changes in individual health endpoints (specific effects that can be associated with changes in air quality) and assigns values to those changes assuming independence of the values for those individual endpoints. Total benefits are calculated simply as the sum of the values for all non-overlapping health

[2] The damage function approach is a more comprehensive method of estimating total benefits than the hedonic price approach applied to housing prices, which requires homebuyers to be knowledgeable of the full magnitude of health risks associated with their home purchase. See discussion of hedonic studies in Chapter 6.

endpoints. The “damage-function” approach is the standard method for assessing costs and benefits of environmental quality programs and has been used in several recent published analyses (Levy et al., 2009; Fann et al., 2012a; Tagaris et al., 2009).

To assess economic value in a damage-function framework, the changes in environmental quality must be translated into effects on people or on the things that people value. In some cases, the changes in environmental quality can be directly valued, as is the case for changes in visibility. In other cases, such as for changes in ozone and PM, a impact analysis must first be conducted to convert air quality changes into effects that can be assigned dollar values. For the purposes of this RIA, the health impacts analysis (HIA) is limited to those health effects that are directly linked to ambient levels of air pollution and specifically to those linked to $PM_{2.5}$.

We note at the outset that the EPA rarely has the time or resources to perform extensive new research to measure directly either the health outcomes or their values for regulatory analyses. Thus, similar to Kunzli et al. (2000) and other, more recent health impact analyses, our estimates are based on the best available methods of benefits transfer. Benefits transfer is the science and art of adapting primary research from similar contexts to obtain the most accurate measure of benefits for the environmental quality change under analysis. Adjustments are made for the level of environmental quality change, the socio-demographic and economic characteristics of the affected population, and other factors to improve the accuracy and robustness of benefits estimates.

5.4.1 Health Impact Assessment

The health impact assessment (HIA) quantifies the changes in the incidence of adverse health impacts resulting from changes in human exposure to $PM_{2.5}$ and ozone air quality. HIAs are a well-established approach for estimating the retrospective or prospective change in adverse health impacts expected to result from population-level changes in exposure to pollutants (Levy et al., 2009). PC-based tools such as the environmental *Ben*efits *M*apping and *A*nalysis *P*rogram (BenMAP) can systematize health impact analyses by applying a database of key input parameters, including health impact functions and population projections—provided that key input data are available, including air quality estimates and risk coefficients (Abt Associates, 2010). Analysts have applied the HIA approach to estimate human health impacts resulting from hypothetical changes in pollutant levels (Hubbell et al., 2005; Tagaris et al., 2009; Fann et al., 2012a). The EPA and others have relied upon this method to predict future changes in health impacts expected to result from the implementation of regulations affecting air quality (e.g., U.S. EPA, 2011d). For this assessment, the HIA is limited to those health effects that are directly linked to ambient $PM_{2.5}$ concentrations. There may be other indirect health impacts associated with implementing emissions controls, such as occupational health exposures.

The HIA approach used in this analysis involves three basic steps: (1) utilizing projections of $PM_{2.5}$ air quality[3] and estimating the change in the spatial distribution of the ambient air quality; (2) determining the subsequent change in population-level exposure; (3) calculating health impacts by applying concentration-response relationships drawn from the epidemiological literature (Hubbell et al., 2009) to this change in population exposure.

A typical health impact function might look as follows:

$$\Delta y = 1 - \left(e^{\beta \cdot \Delta x}\right) y_o \cdot Pop \tag{5.1}$$

where y_0 is the baseline incidence rate for the health endpoint being quantified (for example, a health impact function quantifying changes in mortality would use the baseline, or background, mortality rate for the given population of interest); *Pop* is the population affected by the change in air quality; Δx is the change in air quality; and β is the effect coefficient drawn from the epidemiological study. Figure 5-1 provides a simplified overview of this approach.

[3] Projections of ambient $PM_{2.5}$ concentrations for this analysis were generated using the Community Multiscale Air Quality model (CMAQ). See Chapter 3 of this RIA for more information on the air quality modeling.

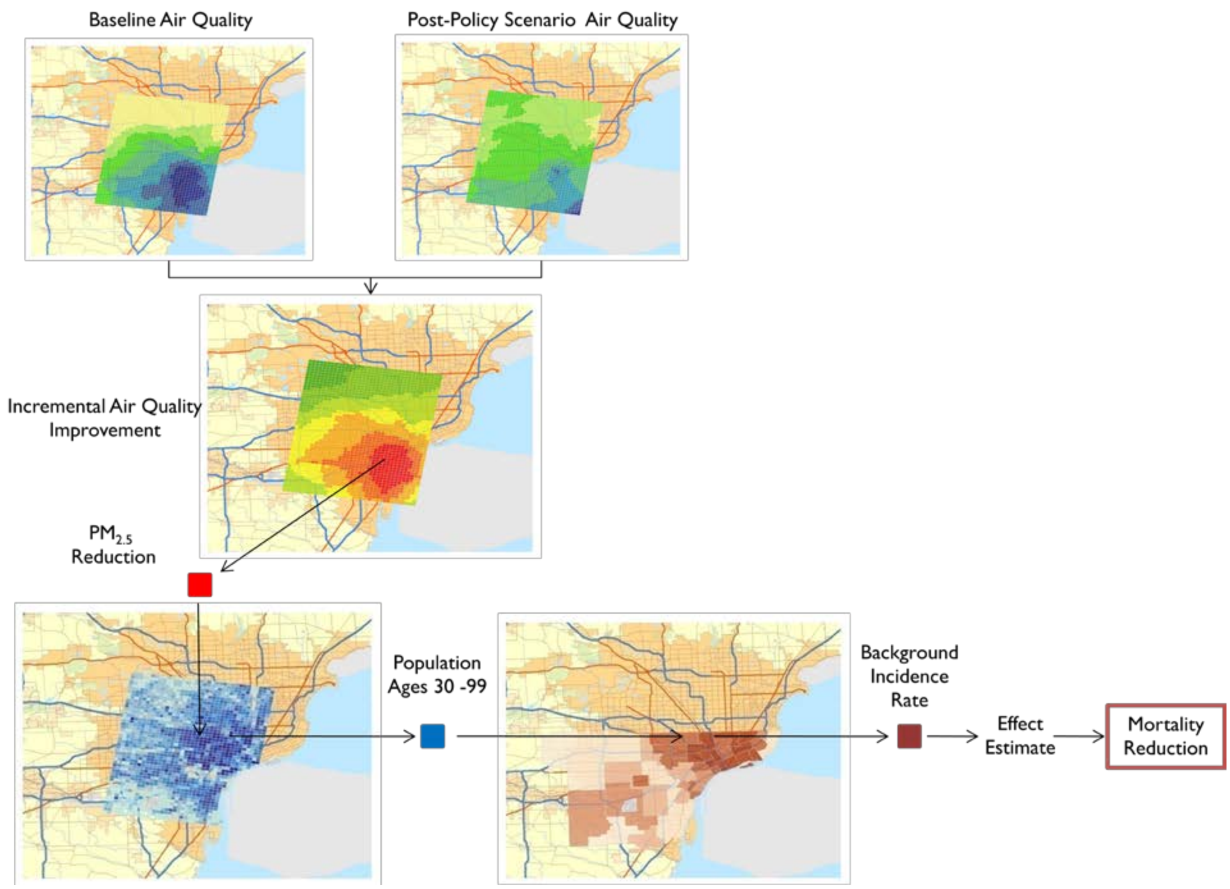


Figure 5-1. Illustration of BenMAP Approach

5.4.2 Economic Valuation of Health Impacts

After quantifying the change in adverse health impacts, the final step is to estimate the economic value of these avoided impacts. The appropriate economic value for a change in a health effect depends on whether the health effect is viewed *ex ante* (before the effect has occurred) or *ex post* (after the effect has occurred). Reductions in ambient concentrations of air pollution generally lower the risk of future adverse health effects by a small amount for a large population. The appropriate economic measure is therefore *ex ante* willingness to pay (WTP) for changes in risk. Epidemiological studies generally provide estimates of the relative risks of a particular health effect for a given increment of air pollution (often per 10 $\mu g/m^3$ for $PM_{2.5}$). These relative risks can be used to develop risk coefficients that relate a unit reduction in $PM_{2.5}$ to changes in the incidence of a health effect. In order to value these changes in incidence, WTP for changes in risk need to be converted into WTP per statistical incidence. This measure is calculated by dividing individual WTP for a risk reduction by the related observed change in risk. For example, suppose a measure is able to reduce the risk of premature mortality from 2 in 10,000 to 1 in 10,000 (a reduction of 1 in 10,000). If individual WTP for this risk reduction is

$100, then the WTP for an avoided statistical premature mortality amounts to $1 million ($100/0.0001 change in risk). Using this approach, the size of the affected population is automatically taken into account by the number of incidences predicted by epidemiological studies applied to the relevant population. The same type of calculation can produce values for statistical incidences of other health endpoints.

For some health effects, such as hospital admissions, WTP estimates are generally not available. In these cases, we use the cost of treating or mitigating the effect. For example, for the valuation of hospital admissions we use the avoided medical costs as an estimate of the value of avoiding the health effects causing the admission. These cost-of-illness (COI) estimates generally (although not necessarily in every case) understate the true value of reductions in risk of a health effect. They tend to reflect the direct expenditures related to treatment but not the value of avoided pain and suffering from the health effect.

We use the BenMAP model version 4.0.52 (Abt Associates, 2010) to estimate the health impacts and monetized health benefits for the proposed standard. Figure 5-2 shows the data inputs and outputs for the BenMAP model.

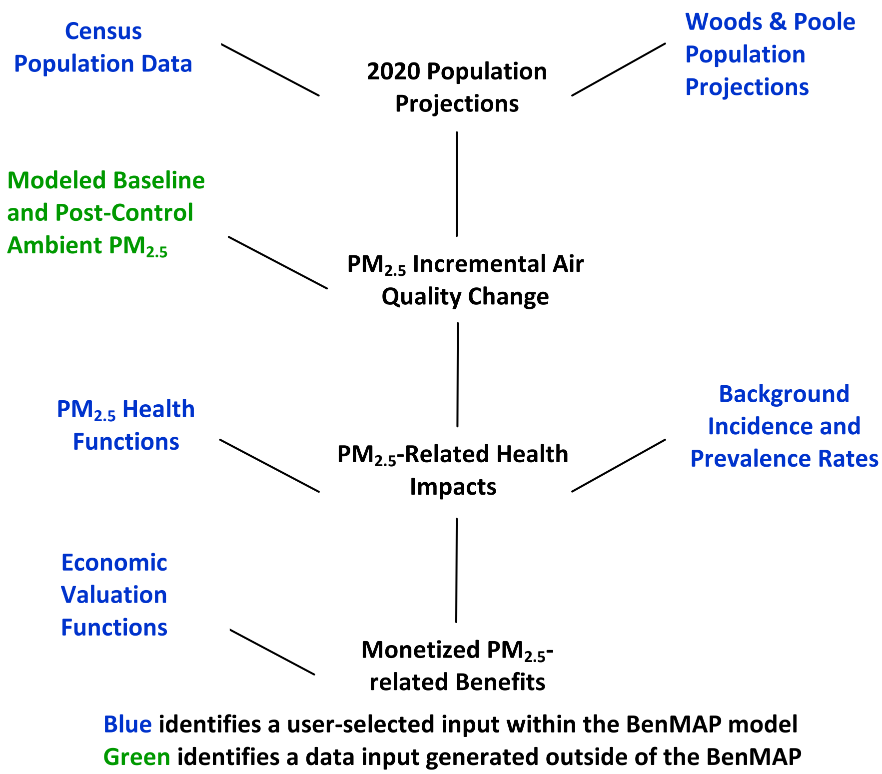


Figure 5-2. Data Inputs and Outputs for the BenMAP Model

5.5 Uncertainty Characterization

In any complex analysis using estimated parameters and inputs from numerous models, there are likely to be many sources of uncertainty. This analysis is no exception. As outlined both in this and preceding chapters, this analysis includes many data sources as inputs, including emission inventories, air quality data from models (with their associated parameters and inputs), population data, population estimates, health effect estimates from epidemiology studies, economic data for monetizing benefits, and assumptions regarding the future state of the world (i.e., regulations, technology, and human behavior). Each of these inputs may be uncertain and would affect the benefits estimate. When the uncertainties from each stage of the analysis are compounded, even small uncertainties can have large effects on the total quantified benefits.

After reviewing the EPA's approach, the National Research Council (NRC) (2002, 2008), which is part of the National Academies of Science, concluded that the EPA's general methodology for calculating the benefits of reducing air pollution is reasonable and informative in spite of inherent uncertainties. The NRC also highlighted the need to conduct rigorous quantitative analyses of uncertainty and to present benefits estimates to decision makers in ways that foster an appropriate appreciation of their inherent uncertainty. Since the publication of these reports, the EPA has continued work to improve the characterization of uncertainty in both health incidence and benefits estimates. In response to these recommendations, we have expanded our previous analyses to incorporate additional quantitative and qualitative characterizations of uncertainty. Although we have not yet been able to make as much progress towards a full, probabilistic uncertainty assessment as envisioned by the NAS as we had hoped, we have added a number of additional quantitative and qualitative analyses to highlight the impact that uncertain assumptions may have on the benefits estimates. In addition, for some inputs into the benefits analysis, such as the air quality data, it is difficult to address uncertainty probabilistically due to the complexity of the underlying air quality models and emission inputs. Therefore, we decline to make up alternative assumptions simply for the purpose of probabilistic uncertainty characterization when there is no scientific literature to support alternate assumptions.

To characterize uncertainty and variability, we follow an approach that combines elements from two recent analyses by the EPA (U.S. EPA, 2010b; 2011a), and uses a tiered approach developed by the World Health Organization (WHO) for characterizing uncertainty (WHO, 2008). We present this tiered assessment as well as an assessment of the potential impact and magnitude of each aspect of uncertainty In Appendix 5c. Data limitations prevent us from treating each source of uncertainty quantitatively and from reaching a full-probabilistic simulation of our results, but we were able to consider the influence of uncertainty in the risk coefficients and economic valuation functions by incorporating six quantitative analyses described in more detail below:

1. A Monte Carlo assessment that accounts for random sampling error and between study variability in the epidemiological and economic valuation studies;
2. A concentration benchmark assessment that characterizes the distribution of avoided $PM_{2.5}$-related deaths relative to specific concentrations in the long-term epidemiological studies used to estimate $PM_{2.5}$-related mortality;
3. The quantification of PM-related mortality using alternative $PM_{2.5}$ mortality effect estimates drawn from two long-term cohort studies and an expert elicitation;
4. Sensitivity analyses of several aspects of PM-related benefits;
5. Distributional analyses of $PM_{2.5}$-related benefits by location, race, income, and education; and
6. An analysis of the influence of various parameters on total monetized benefits.

5.5.1 *Monte Carlo Assessment*

Similar to other recent RIAs, we used Monte Carlo methods for characterizing random sampling error associated with the concentration response functions from epidemiological studies and random effects modeling to characterize both sampling error and variability across the economic valuation functions. The Monte Carlo simulation in the BenMAP software randomly samples from a distribution of incidence and valuation estimates to characterize the effects of uncertainty on output variables. Specifically, we used Monte Carlo methods to generate confidence intervals around the estimated health impact and monetized benefits. The reported standard errors in the epidemiological studies determined the distributions for individual effect estimates for endpoints estimated using a single study. For endpoints estimated using a pooled estimate of multiple studies, the confidence intervals reflect both the standard errors and the variance across studies. The confidence intervals around the monetized benefits incorporate the epidemiology standard errors as well as the distribution of the valuation function. These confidence intervals do not reflect other sources of uncertainty inherent within the estimates, such as baseline incidence rates, populations exposed and transferability of the effect estimate to diverse locations. As a result, the reported confidence intervals and range of estimates give an incomplete picture about the overall uncertainty in the benefits estimates.

5.5.2 Concentration Benchmark Analysis for $PM_{2.5}$

We include a concentration benchmark assessment, which identifies the baseline (i.e., pre-rule) annual mean $PM_{2.5}$ levels at which populations are exposed and specific concentrations in the two long-term cohort studies we use to quantify mortality impacts. This analysis characterizes avoided $PM_{2.5}$-related deaths relative to the 10^{th} and 25^{th} percentiles of the air quality data used the Krewski et al. (2009) study as well as the lowest measured level (LML) of the Krewski et al. (2009) and Lepeule et al. (20) studies.

5.5.3 Alternative Concentration-Response Functions for $PM_{2.5}$–Related Mortality

We assign the greatest economic value to the reduction in $PM_{2.5}$ related mortality risk. Therefore, it is particularly important to attempt to characterize the uncertainties associated with reductions in premature mortality. To better understand the concentration-response relationship between $PM_{2.5}$ exposure and premature mortality, the EPA conducted an expert elicitation in 2006 (Roman et al., 2008; IEc, 2006).[4] In general, the results of the expert elicitation support the conclusion that the benefits of $PM_{2.5}$ control are very likely to be substantial.

Alternative concentration-response functions are useful for assessing uncertainty beyond random statistical error, including uncertainty in the functional form of the model or alternative study design. Thus, we include the expert elicitation results as well as standard errors approaches to provide insights into the likelihood of different outcomes and about the state of knowledge regarding the benefits estimates. In this analysis, we present the results derived from the expert elicitation as indicative of the uncertainty associated with a major component of the health impact functions, and we provide the independent estimates derived from each of the twelve experts to better characterize the degree of variability in the expert responses.

In previous RIAs, the EPA presented benefits estimates using concentration response functions derived from the $PM_{2.5}$ Expert Elicitation (Roman et al., 2008) as a range from the lowest expert value (Expert K) to the highest expert value (Expert E). However, this approach did not indicate the agency's judgment on what the best estimate of $PM_{2.5}$ benefits may be, and the EPA's independent Science Advisory Board (SAB) recommended refinements to the way EPA presented the results of the elicitation (U.S. EPA-SAB, 2008). Therefore, we began to

[4] Expert elicitation is a formal, highly structured and well documented process whereby expert judgments, usually of multiple experts, are obtained (Ayyub, 2002).

present the cohort-based studies (Krewski et al., 2009; Laden et al., 2006)[5] as our core estimates in the proposal RIA for the Portland Cement NESHAP (U.S. EPA, 2009a). Using alternate relationships between $PM_{2.5}$ and premature mortality supplied by experts, higher and lower benefits estimates are plausible, but most of the expert-based estimates of the mean $PM_{2.5}$ effect on mortality fall between the two epidemiology-based estimates (Roman et al., 2008). In addition to these studies, we have included a discussion or other recent multi-state cohort studies conducted in North America, but we have not estimated benefits using the effect coefficients from these studies. Please note that the benefits estimates results presented are not the direct results from the studies or expert elicitation; rather, the estimates are based in part on the effect coefficients provided in those studies or by experts. In addition, the experts provided distributions around their mean $PM_{2.5}$ effect estimates, which provides more information regarding the overall range of uncertainty, and this overall range is larger than the range of the mean effect estimates from each of the experts.

Even these multiple characterizations with confidence intervals omit the contribution to overall uncertainty from uncertainty in air quality changes, baseline incidence rates, and populations exposed. Furthermore, the approach presented here does not yet include methods for addressing correlation between input parameters and the identification of reasonable upper and lower bounds for input distributions characterizing uncertainty in additional model elements. As a result, the reported confidence intervals and range of estimates give an incomplete picture about the overall uncertainty in the estimates. This information should be interpreted within the context of the larger uncertainty surrounding the entire analysis.

5.5.4 Sensitivity Analyses

For some aspects of uncertainty, we have sufficient data to conduct sensitivity analyses. In this analysis, we performed five such analyses for the revised standard level. In particular, we:

1. Assessed the sensitivity of the economic value of reductions in the risk of $PM_{2.5}$-related death according to differing assumptions regarding the lag between $PM_{2.5}$ exposure and premature death. The timing of such premature deaths affects the magnitude of the discounted $PM_{2.5}$-related mortality benefits. In this sensitivity assessment, we consider 6 alternative cessation lags.

2. Characterized the sensitivity of the economic value of the health endpoints valued using willingness-to-pay estimates to a higher and a lower assumption regarding

[5] We have since updated the the Harvard Six Cities cohort study from Laden et al. (2006) to use the most recent follow-up publication of this cohort (Lepeule et al, 2012).

income elasticity. As we discuss below, economic theory argues that individual willingness to pay increases as personal income grows. The relationship between growth in personal income and willingness-to-pay to reduce mortality and morbidity risk is characterized by the income growth factor.

3. Summarized the avoided cases of certain health endpoints for which we either lacked an appropriate economic value (cardiovascular hospital admissions and stroke) or in which we no longer had sufficient confidence to retain in our primary benefits estimate (chronic bronchitis).

4. Assessed the sensitivity of the benefits results to the new population data from the 2010 Census.

5. Assessed the sensitivity of the benefits results to an analysis year of 2025.

5.5.5 Distributional Assessment

In an Appendix to the proposal RIA, we characterized the distribution of $PM_{2.5}$-related benefits based on the geographic distribution of race and education in areas where the illustrative emission reduction strategies would reduce $PM_{2.5}$ concentrations. In that assessment, we aimed to answer two key questions:

1. What was the estimated distribution of $PM_{2.5}$-related mortality risk based on the race and education characteristics of the population living within areas projected to exceed alternative combinations of the proposed primary $PM_{2.5}$ standards?

2. How would air quality improvements within these counties change the distribution of risk among populations of different races and educational attainment?[6]

That assessment was generally consistent with the distributional assessments performed in support of MATS (U.S. EPA, 2011c), with one key difference. The environmental justice analyses accompanying the MATS RIA applied CMAQ-modeled $PM_{2.5}$ predictions that represent the change in air quality after the implementation of each rule. By contrast, the proposal RIA aimed to illustrate the potential benefits and costs of attaining alternative primary $PM_{2.5}$ standards; the states will ultimately implement attainment strategies, which may differ greatly from the least-cost strategy the EPA modeled here. Alternative emission reduction strategies—particularly those that maximize benefits to human health and provide a more equitable distribution of risk—are also available to the states, though not modeled here (Fann

[6] In this analysis we assess the change in risk among populations of different race and educational attainment. As we discuss further in the methodology, we consider this last variable because of the availability of education-modified $PM_{2.5}$ mortality risk estimates.

et al., 2012b). Due in part to time constraints, the EPA did not perform such an analysis for the final RIA, and instead refers readers to the Appendix noted above.

5.5.6 Influence Analysis—Quantitative Assessment of Uncertainty

In the past few years, the EPA has initiated several projects to improve the characterization of uncertainty for benefits analysis. In particular, the EPA recently completed the first phase of a quantitative uncertainty analysis of benefits, hereafter referred to as the "Influence Analysis" (Mansfield et al., 2009). The Influence Analysis diagramed the uncertain components of each step within the benefits analysis process, identified plausible ranges for a sensitivity analysis, and assessed the sensitivity to total benefits to changes in each component. Although this analysis does not quite fulfill the goal of a full probabilistic assessment, it accomplished the necessary first steps and identified the challenges to accomplishing that goal. Below are some of the preliminary observations from the first phase of the project.

- The components that contribute the most to uncertainty of the monetized benefits and mortality incidence (in order of importance) are the value-of-a-statistical-life (VSL), the concentration-response (C-R) function for mortality, and change in $PM_{2.5}$ concentration.
- The components that contribute the least to uncertainty of the monetized benefits and mortality incidence are population, morbidity valuation, and income elasticity.
- The choice of a C-R function for mortality affects the mortality incidence and monetized benefits more than other sources of uncertainty within each C-R function.
- Alternative cessation lag structures for mortality have a moderate effect on the monetized benefits.
- Because the health impact function is essentially linear, the key components show the same sensitivity across all mortality C-R functions even if the midpoints differ significantly from one expert to another.

5.5.7 Qualitative Assessment of Uncertainty and Other Analysis Limitations

Although we strive to incorporate as many quantitative assessments of uncertainty as possible, there are several aspects we are only able to address qualitatively. These aspects are important factors to consider when evaluating the relative benefits of the emission reduction strategies for the revised and alternative annual standards:

The total monetized benefits presented in this chapter are based on our interpretation of the best available scientific literature and methods and supported by the EPA's independent SAB (Health Effects Subcommittee) (SAB-HES) (U.S. EPA- SAB, 2010a) and the National Academies of Science (NAS) (NRC, 2002). The benefits estimates are subject to a number of assumptions and uncertainties. For example, the key assumptions underlying the estimates for premature mortality, which account for over 98% of the total monetized benefits in this analysis, include the following:

1. We assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This is an important assumption, because $PM_{2.5}$ varies considerably in composition across sources, but the scientific evidence is not yet sufficient to allow differentiation of effect estimates by particle type. The PM ISA, which was twice reviewed by Clean Air Scientific Advisory Committee (SAB-CASAC), concluded that "many constituents of $PM_{2.5}$ can be linked with multiple health effects, and the evidence is not yet sufficient to allow differentiation of those constituents or sources that are more closely related to specific outcomes" (U.S. EPA, 2009b). These uncertainties are likely to be magnified in the current analysis to the extent that the emissions controls are less diverse when focusing on one small region of the country rather than a broader geography with more diverse emissions sources and the application of a more diverse set of controls.

2. We assume that health impact functions based on national studies are representative for exposures and populations in California. In addition to the national risk coefficients we use as our core estimates, the EPA considered the cohort studies conducted in California specifically. Although we have not calculated the benefits results using the cohort studies conducted in California, we provided these risk coefficients to show how much the monetized benefits could have changed. Most of the California cohort studies report central effect estimates similar to the (nation-wide) all-cause mortality risk estimate we applied from Krewski et al. (2009) and Lepeule et al. (2012) albeit with wider confidence intervals. Three cohort studies conducted in California indicate statistically significant higher risks than the risk estimates we applied from Lepeule et al. (2012), and four studies showed insignificant results.

3. We assume that the health impact function for fine particles is log-linear without a threshold in this analysis. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5}$, including both areas that do not meet the fine particle standard and those areas that are in attainment, down to the lowest modeled concentrations.

4. We assume that there is a "cessation" lag between the change in PM exposures and the total realization of changes in mortality effects. Specifically, we assume that

some of the incidences of premature mortality related to $PM_{2.5}$ exposures occur in a distributed fashion over the 20 years following exposure based on the advice of the SAB-HES (U.S. EPA-SAB, 2004c), which affects the valuation of mortality benefits at different discount rates.

5. To characterize the uncertainty in the relationship between $PM_{2.5}$ and premature mortality (which account for over 98% of total monetized benefits in this analysis), we include a set of twelve estimates based on results of the expert elicitation study in addition to our core estimates. Even these multiple characterizations omit the uncertainty in air quality estimates, baseline incidence rates, populations exposed and transferability of the effect estimate to diverse locations. As a result, the reported confidence intervals and range of estimates give an incomplete picture about the overall uncertainty in the $PM_{2.5}$ estimates. This information should be interpreted within the context of the larger uncertainty surrounding the entire analysis.

As previously described, we strive to monetize as many of the benefits anticipated from the revised and alternative standards as possible given data and resource limitations, but the monetized benefits estimated in this RIA inevitably only reflect a portion of the benefits. Specifically, only certain benefits attributable to the health impacts associated with exposure to ambient fine particles have been monetized in this analysis. Data and methodological limitations prevented the EPA from quantifying or monetizing the benefits from several important health benefit categories from emission reduction strategies to reach the revised annual standard in this RIA, including potential co-benefits from reducing ozone exposure, NO_2 exposure, SO_2 exposure, and methylmercury exposure (see section 5.6.5 for more information). If we could fully monetize all of the benefit categories, the total monetized benefits would exceed the costs by an even greater margin than we currently estimate.

To more fully address all these uncertainties including those we cannot quantify, we apply a four-tiered approach using the WHO uncertainty framework (WHO, 2008), which provides a means for systematically linking the characterization of uncertainty to the sophistication of the underlying risk assessment. The EPA has applied similar approaches in analyses (U.S. EPA, 2010b, 2011a). Using this framework, we summarize the key uncertainties in the health benefits analysis, including our assessment of the direction of potential bias, magnitude of impact on the monetized benefits, degree of confidence in our analytical approach, and our ability to assess the source of uncertainty. More information on this approach and the uncertainty characterization are available in Appendix 5B.

5.6 Benefits Analysis Data Inputs

In Figure 5-2, we summarized the key data inputs to the health impact and economic valuation estimate. Below we summarize the data sources for each of these inputs, including demographic projections, incidence and prevalence rates, effect coefficients, and economic valuation. We indicate where we have updated key data inputs since the benefits analysis conducted for the MATS RIA (U.S. EPA, 2011d) and the proposal RIA (U.S. EPA, 2012a).

5.6.1 Demographic Data

Quantified and monetized human health impacts depend on the demographic characteristics of the population, including age, location, and income. We use population projections based on economic forecasting models developed by Woods and Poole, Inc. (Woods and Poole, 2012). The Woods and Poole (WP) database contains county-level projections of population by age, sex, and race out to 2040, relative to a baseline using the 2010 Census data; the proposal RIA incorporated WP projections relative to a baseline using 2000 Census data. An analysis exploring the sensitivity of population and health impact estimates to this update can be found in Appendix 5.A. Projections in each county are determined simultaneously with every other county in the United States to take into account patterns of economic growth and migration. The sum of growth in county-level populations is constrained to equal a previously determined national population growth, based on Bureau of Census estimates (Hollman et al., 2000). According to WP, linking county-level growth projections together and constraining to a national-level total growth avoids potential errors introduced by forecasting each county independently. County projections are developed in a four-stage process:

- First, national-level variables such as income, employment, and populations are forecasted.
- Second, employment projections are made for 179 economic areas defined by the Bureau of Economic Analysis (U.S. BEA, 2004), using an "export-base" approach, which relies on linking industrial-sector production of non-locally consumed production items, such as outputs from mining, agriculture, and manufacturing with the national economy. The export-based approach requires estimation of demand equations or calculation of historical growth rates for output and employment by sector.
- Third, population is projected for each economic area based on net migration rates derived from employment opportunities and following a cohort-component method based on fertility and mortality in each area.

- Fourth, employment and population projections are repeated for counties, using the economic region totals as bounds. The age, sex, and race distributions for each region or county are determined by aging the population by single year of age by sex and race for each year through 2040 based on historical rates of mortality, fertility, and migration.

5.6.2 Baseline Incidence and Prevalence Estimates

Epidemiological studies of the association between pollution levels and adverse health effects generally provide a direct estimate of the relationship of air quality changes to the *relative risk* of a health effect, rather than estimating the absolute number of avoided cases. For example, a typical result might be that a 10 $\mu g/m^3$ decrease in daily $PM_{2.5}$ levels might be associated with a decrease in hospital admissions of 3%. The baseline incidence of the health effect is necessary to convert this relative change into a number of cases. A baseline incidence rate is the estimate of the number of cases of the health effect per year in the assessment location, as it corresponds to baseline pollutant levels in that location. To derive the total baseline incidence per year, this rate must be multiplied by the corresponding population number. For example, if the baseline incidence rate is the number of cases per year per million people, that number must be multiplied by the millions of people in the total population.

Table 5-3 summarizes the sources of baseline incidence rates and provides average incidence rates for the endpoints included in the analysis. For both baseline incidence and prevalence data, we used age-specific rates where available. We applied concentration-response functions to individual age groups and then summed over the relevant age range to provide an estimate of total population benefits. In most cases, we used a single national incidence rate, due to a lack of more spatially disaggregated data. Whenever possible, the national rates used are national averages, because these data are most applicable to a national assessment of benefits. For some studies, however, the only available incidence information comes from the studies themselves; in these cases, incidence in the study population is assumed to represent typical incidence at the national level. County, state and regional incidence rates are available for hospital admissions, and county-level data are available for premature mortality.

We projected mortality rates such that future mortality rates are consistent with our projections of population growth (Abt Associates, 2012). To perform this calculation, we began first with an average of 2004–2006 cause-specific mortality rates. Using Census Bureau projected national-level annual mortality rates stratified by age range, we projected these mortality rates to 2050 in 5-year increments (Abt Associates, 2012; U.S. Bureau of the Census 2002).

The baseline incidence rates for hospital admissions and emergency department visits reflect the revised rates first applied in the CSAPR RIA (U.S. EPA, 2011c). In addition, we have revised the baseline incidence rates for acute myocardial infarction. These revised rates are more recent (AHRQ, 2007), which provides a better representation of the rates at which populations of different ages, and in different locations, visit the hospital and emergency department for air pollution-related illnesses. Also, the new baseline incidence rates are more spatially refined. For many locations within the U.S., these data are resolved at the county- or state-level, providing a better characterization of the geographic distribution of hospital and emergency department visits than the previous national rates. Lastly, these rates reflect unscheduled hospital admissions only, which represents a conservative assumption that most air pollution-related visits are likely to be unscheduled. If air pollution-related hospital admissions are scheduled, this assumption would underestimate these benefits.

For the set of endpoints affecting the asthmatic population, in addition to baseline incidence rates, prevalence rates of asthma in the population are needed to define the applicable population. Table 5-4 lists the prevalence rates used to determine the applicable population for asthma symptoms. Note that these reflect current asthma prevalence and assume no change in prevalence rates in future years. We updated these rates in the CSAPR RIA (U.S. EPA, 2011c).

Table 5-3. Baseline Incidence Rates and Population Prevalence Rates for Use in Impact Functions, General Population

Endpoint	Parameter	Rates	
		Value	Source
Mortality	Daily or annual mortality rate projected to 2020[a]	Age-, cause-, and county-specific rate	CDC WONDER (2004–2006) U.S. Census bureau, 2000
Hospitalizations	Daily hospitalization rate	Age-, region-, state-, county- and cause-specific rate	2007 HCUP data files[b]
ER Visits	Daily ER visit rate for asthma and cardiovascular events	Age-, region-, state-, county- and cause-specific rate	2007 HCUP data files[b]
Cerebrovascular events	Incidence of new cerebrovascular events among populations 50–79	0.001575I	Table 3 of Miller et al. (2007)
Chronic Bronchitis[c]	Annual prevalence rate per person • Aged 18–44 • Aged 45–64 • Aged 65 and older	 • 0.0315 • 0.0549 • 0.0563	American Lung Association (2010a, Table 4).
	Annual incidence rate per person	0.00378	Abbey et al. (1993, Table 3)
Nonfatal Myocardial Infarction (heart attacks)	Daily nonfatal myocardial infarction incidence rate per person, 18+	Age-, region-, state-, and county-specific rate	2007 HCUP data files;[b] adjusted by 0.93 for probability of surviving after 28 days (Rosamond et al., 1999)
Asthma Exacerbations	Incidence among asthmatic African-American children • daily wheeze • daily cough • daily shortness of breath	 • 0.173 • 0.145 • 0.074	Ostro et al. (2001)
Acute Bronchitis	Annual bronchitis incidence rate, children	0.043	American Lung Association (2002c, Table 11)
Lower Respiratory Symptoms	Daily lower respiratory symptom incidence among children[d]	0.0012	Schwartz et al. (1994, Table 2)
Upper Respiratory Symptoms	Daily upper respiratory symptom incidence among asthmatic children	0.3419	Pope et al. (1991, Table 2)

(continued)

Table 5-3. Baseline Incidence Rates and Population Prevalence Rates for Use in Impact Functions, General Population (continued)

Endpoint	Parameter	Rates	
		Value	Source
Work Loss Days	Daily WLD incidence rate per person (18–65) • Aged 18–24 • Aged 25–44 • Aged 45–64	 • 0.00540 • 0.00678 • 0.00492	1996 HIS (Adams, Hendershot, and Marano, 1999, Table 41); U.S. Census Bureau (2000)
School Loss Days	Rate per person per year, assuming 180 school days per year	9.9	National Center for Education Statistics (1996) and 1996 HIS (Adams et al., 1999, Table 47);
Minor Restricted-Activity Days	Daily MRAD incidence rate per person	0.02137	Ostro and Rothschild (1989, p. 243)

[a] Mortality rates are only available at 5-year increments.

[b] Healthcare Cost and Utilization Program (HCUP) database contains individual level, state and regional-level hospital and emergency department discharges for a variety of ICD codes (AHRQ, 2007).

[c] Assessed in sensitivity analysis only. The rate numbers may be slightly different from those in Table 4 because we received more current estimates from ALA.

[d] Lower respiratory symptoms are defined as two or more of the following: cough, chest pain, phlegm, and wheeze.

Table 5-4. Asthma Prevalence Rates

Population Group	Asthma Prevalence Rates	
	Value	Source
All Ages	0.0780	American Lung Association (2010b, Table 7)
< 18	0.0941	
5–17	0.1070	
18–44	0.0719	
45–64	0.0745	
65+	0.0716	
African American, 5–17	0.1776	American Lung Association (2010b, Table 9)
African American, <18	0.1553	American Lung Association[a]

[a] Calculated by ALA for U.S. EPA, based on NHIS data (CDC, 2008).

5.6.3 *Effect Coefficients*

The first step in selecting effect coefficients is to identify the health endpoints to be quantified. We base our selection of health endpoints on consistency with the EPA's Integrated Science Assessments (which replace previous Criteria Documents), with input and advice from the SAB-HES, a scientific review panel specifically established to provide advice on the use of the scientific literature in developing benefits analyses for the EPA's Report to Congress on *The Benefits and Costs of the Clean Air Act 1990 to 2020* (U.S. EPA, 2011a). In addition, we have included more recent epidemiology studies from the PM ISA (U.S. EPA, 2009b) and *the* Provisional Assessment (U.S. EPA, 2012b).[7] In general, we follow a weight of evidence approach, based on the biological plausibility of effects, availability of concentration-response functions from well conducted peer-reviewed epidemiological studies, cohesiveness of results across studies, and a focus on endpoints reflecting public health impacts (like hospital admissions) rather than physiological responses (such as changes in clinical measures like Forced Expiratory Volume [FEV1]).

There are several types of data that can support the determination of types and magnitude of health effects associated with air pollution exposures. These sources of data include toxicological studies (including animal and cellular studies), human clinical trials, and observational epidemiology studies. All of these data sources provide important contributions to the weight of evidence surrounding a particular health impact. However, only epidemiology studies provide direct concentration-response relationships that can be used to evaluate population-level impacts of reductions in ambient pollution levels in a health impact assessment.

For the data-derived estimates, we relied on the published scientific literature to ascertain the relationship between $PM_{2.5}$ and adverse human health effects. We evaluated epidemiological studies using the selection criteria summarized in Table 5-5. These criteria include consideration of whether the study was peer-reviewed, the match between the pollutant studied and the pollutant of interest, the study design and location, and characteristics of the study population, among other considerations. In general, the use of concentration-response functions from more than a single study can provide a more representative distribution of the effect estimate. However, there are often differences between studies examining the same endpoint, making it

[7] The peer-reviewed studies in the *Provisional Assessment* have not yet undergone external review by the SAB. The new studies from the PM ISA and *Provisional Assessment* for health endpoints not previously quantified in EPA's RIAs are presented in a sensitivity analysis in Appendix 5B, but these new endpoints have not been incorporated into the core benefits analysis.

difficult to pool the results in a consistent manner. For example, studies may examine different pollutants or different age groups. For this reason, we consider very carefully the set of studies available examining each endpoint and select a consistent subset that provides a good balance of population coverage and match with the pollutant of interest. In many cases, either because of a lack of multiple studies, consistency problems, or clear superiority in the quality or comprehensiveness of one study over others, a single published study is selected as the basis of the effect estimate.

When several effect estimates for a pollutant and a given health endpoint have been selected, they are quantitatively combined or pooled to derive a more robust estimate of the relationship. The BenMAP Manual Technical Appendices provides details of the procedures used to combine multiple impact functions (Abt Associates, 2012). In general, we used fixed or random effects models to pool estimates from different single city studies of the same endpoint. Fixed effects pooling simply weights each study's estimate by the inverse variance, giving more weight to studies with greater statistical power (lower variance). Random effects pooling accounts for both within-study variance and between-study variability, due, for example, to differences in population susceptibility. We used the fixed effects model as our null hypothesis and then determined whether the data suggest that we should reject this null hypothesis, in which case we would use the random effects model. [8] Pooled impact functions are used to estimate hospital admissions and asthma exacerbations. When combining evidence across multi-city studies (e.g., cardiovascular hospital admission studies), we use equal weights pooling. The effect estimates drawn from each multi-city study are themselves pooled across a large number of urban areas. For this reason, we elected to give each study an equal weight rather than weighting by the inverse of the variance reported in each study. For more details on methods used to pool incidence estimates, see the BenMAP Manual Appendices (Abt Associates, 2012).

Effect estimates selected for a given health endpoint were applied consistently across all locations nationwide. This applies to both impact functions defined by a single effect estimate and those defined by a pooling of multiple effect estimates. Although the effect estimate may, in fact, vary from one location to another (e.g., because of differences in population susceptibilities or differences in the composition of PM), location-specific effect estimates are generally not available.

[8] EPA recently changed the algorithm BenMAP uses to calculate study variance, which is used in the pooling process. Prior versions of the model calculated population variance, while the version used here calculated sample variance. This change did not affect the selection of random or fixed effects for the pooled incidence estimates between the proposal and final RIA.

Table 5-5. Criteria Used When Selecting C-R Functions

Consideration	Comments
Peer-Reviewed Research	Peer-reviewed research is preferred to research that has not undergone the peer-review process.
Study Type	Among studies that consider chronic exposure (e.g., over a year or longer), prospective cohort studies are preferred over ecological studies because they control for important individual-level confounding variables that cannot be controlled for in ecological studies.
Study Period	Studies examining a relatively longer period of time (and therefore having more data) are preferred, because they have greater statistical power to detect effects. Studies that are more recent are also preferred because of possible changes in pollution mixes, medical care, and lifestyle over time. However, when there are only a few studies available, studies from all years will be included.
Population Attributes	The most technically appropriate measures of benefits would be based on impact functions that cover the entire sensitive population but allow for heterogeneity across age or other relevant demographic factors. In the absence of effect estimates specific to age, sex, preexisting condition status, or other relevant factors, it may be appropriate to select effect estimates that cover the broadest population to match with the desired outcome of the analysis, which is total national-level health impacts. When available, multi-city studies are preferred to single city studies because they provide a more generalizable representation of the concentration-response function.
Study Size	Studies examining a relatively large sample are preferred because they generally have more power to detect small magnitude effects. A large sample can be obtained in several ways, including through a large population or through repeated observations on a smaller population (e.g., through a symptom diary recorded for a panel of asthmatic children).
Study Location	U.S. studies are more desirable than non-U.S. studies because of potential differences in pollution characteristics, exposure patterns, medical care system, population behavior, and lifestyle. National estimates are most appropriate when benefits are nationally distributed; the impact of regional differences may be important when benefits only accrue to a single area.
Pollutants Included in Model	When modeling the effects of ozone and PM (or other pollutant combinations) jointly, it is important to use properly specified impact functions that include both pollutants. Using single-pollutant models in cases where both pollutants are expected to affect a health outcome can lead to double-counting when pollutants are correlated.
Measure of PM	For this analysis, impact functions based on $PM_{2.5}$ are preferred to PM_{10} because of the focus on reducing emissions of $PM_{2.5}$ precursors, and because air quality modeling was conducted for this size fraction of PM. Where $PM_{2.5}$ functions are not available, PM_{10} functions are used as surrogates, recognizing that there will be potential downward (upward) biases if the fine fraction of PM_{10} is more (less) toxic than the coarse fraction.
Economically Valuable Health Effects	Some health effects, such as forced expiratory volume and other technical measurements of lung function, are difficult to value in monetary terms. These health effects are not quantified in this analysis.
Non-overlapping Endpoints	Although the benefits associated with each individual health endpoint may be analyzed separately, care must be exercised in selecting health endpoints to include in the overall benefits analysis because of the possibility of double-counting of benefits.

The specific studies from which effect estimates for the core analysis are drawn are included in Table 5-6. We highlight in blue those studies that have been added since the benefits analysis conducted for the MATS RIA (U.S. EPA, 2011d), and we highlight those studies in red that have been added since the proposal RIA (U.S. EPA, 2012a). In all cases where effect estimates are drawn directly from epidemiological studies, standard errors are used as a partial representation of the uncertainty in the size of the effect estimate. Table 5-7 summarizes those health endpoints and studies we have included as in sensitivity analyses.

Table 5-6. Health Endpoints and Epidemiological Studies Used to Quantify Health Impacts in the Core Analysis [a]

Endpoint	Study	Study Population	Risk Estimate (95th Percentile Confidence Interval)[a]
Premature Mortality			
Premature mortality—cohort study, all-cause	Krewski et al. (2009) Lepeule et al. (2012)	Premature mortality—cohort study, all-cause	Krewski et al. (2009) Lepeule et al. (2012)
Premature mortality, total exposures	$PM_{2.5}$ Expert Elicitation (Roman et al., 2008)	Premature mortality, total exposures	$PM_{2.5}$ Expert Elicitation (Roman et al., 2008)
Premature mortality—all-cause	Woodruff et al. (1997)	Premature mortality—all-cause	Woodruff et al. (1997)
Chronic Illness			
Nonfatal heart attacks	Peters et al. (2001) *Pooled estimate:* Pope et al. (2006) Sullivan et al. (2005)	Nonfatal heart attacks	Peters et al. (2001) *Pooled estimate:* Pope et al. (2006) Sullivan et al. (2005)
Nonfatal heart attacks (cont'd)	Zanobetti et al. (2009) Zanobetti and Schwartz (2006)	Nonfatal heart attacks (cont'd)	Zanobetti et al. (2009) Zanobetti and Schwartz (2006)
Hospital Admissions			
Respiratory	Zanobetti et al. (2009)—ICD 460-519 (All respiratory) Kloog et al. (2012)—ICD 460-519 (All Respiratory Moolgavkar (2000)—ICD 490–496 (Chronic lung disease) Babin et al. (2007)—ICD 493 (asthma) Sheppard (2003)—ICD 493 (asthma)	Respiratory	Zanobetti et al. (2009)—ICD 460-519 (All respiratory) Kloog et al. (2012)—ICD 460-519 (All Respiratory Moolgavkar (2000)—ICD 490–496 (Chronic lung disease) Babin et al. (2007)—ICD 493 (asthma) Sheppard (2003)—ICD 493 (asthma)

(continued)

Table 5-6. Health Endpoints and Epidemiological Studies Used to Quantify Health Impacts in the Core Analysis [a] (continued)

Endpoint	Study	Study Population	Risk Estimate (95th Percentile Confidence Interval)[a]
Cardiovascular	*Pooled estimate:*	>64 years	
	Zanobetti et al. (2009)—ICD 390-459 (all cardiovascular)		β=0.00189 (0.000283)
	Peng et al. (2009)—ICD 426-427; 428; 430-438; 410-414; 429; 440-449 (Cardio-, cerebro- and peripheral vascular disease)		β=0.00068 (0.000214)
	Peng et al. (2008)—ICD 426-427; 428; 430-438; 410-414; 429; 440-449 (Cardio-, cerebro- and peripheral vascular disease)		β=0.00071 (0.00013)
	Bell et al. (2008)—ICD 426-427; 428; 430-438; 410-414; 429; 440-449 (Cardio-, cerebro- and peripheral vascular disease)		β=0.0008 (0.000107)
	Moolgavkar (2000)—ICD 390–429 (all cardiovascular)	20–64 years	RR=1.04 (t statistic: 4.1) per 10 μg/m^3
Asthma-related ER visits	*Pooled estimate:* Mar et al. (2010)	All ages	RR = 1.04 (1.01–1.07) per 7 μg/m^3
	Slaughter et al. (2005)		RR = 1.03 (0.98–1.09) per 10 μg/m^3
	Glad et al. (2012)		β=0.00392 (0.002843)
Other Health Endpoints			
Acute bronchitis	Dockery et al. (1996)	8–12 years	OR = 1.50 (0.91–2.47) per 14.9 μg/m^3
Asthma exacerbations	*Pooled estimate:* Ostro et al. (2001) (cough, wheeze and shortness of breath) [b]	6–18 years [b]	OR = 1.03 (0.98–1.07) OR = 1.06 (1.01–1.11) OR = 1.08 (1.00–1.17) per 30 μg/m^3
	Mar et al. (2004) (cough, shortness of breath)		RR = 1.21 (1–1.47) per RR = 1.13 (0.86–1.48) per 10 μg/m^3
Work loss days	Ostro (1987)	18–65 years	β=0.0046 (0.00036)
Acute respiratory symptoms	Ostro and Rothschild (1989) (Minor restricted activity days)	18–65 years	β=0.00220 (0.000658)
Upper respiratory symptoms	Pope et al. (1991)	Asthmatics, 9–11 years	1.003 (1–1.006) per 10 μg/m^3
Lower respiratory symptoms	Schwartz and Neas (2000)	7–14 years	OR = 1.11 (1.58–1.58) per 15 μg/m^3

[a] Studies highlighted in blue represent updates incorporated since the RIA for MATS (U.S. EPA, 2011d). Studies highlighted in red represent updates incorporated since the proposal RIA (U.S. EPA, 2012a).

[b] The original study populations were 8 to 13 for the Ostro et al. (2001) study and 7 to 12 for the Mar et al. (2004) study. Based on advice from the SAB-HES, we extended the applied population to 6-18, reflecting the common biological basis for the effect in children in the broader age group. See: U.S. EPA-SAB (2004) and NRC (2002).

Table 5-7. Health Endpoints and Epidemiological Studies Used to Quantify Health Impacts in the Sensitivity Analysis[a]

Endpoint	Study	Study Population
Chronic Illness		
Chronic bronchitis	Abbey et al. (1995)	>26 years
Stroke	Miller et al. (2007)	50–79 years
Hospital Admissions		
Cardiovascular ED Visits	Metzger et al. (2004)	0–99
	Tolbert et al. (2007)	0–99
	Mathes et al. (2011)	40–99

[a] Studies highlighted in blue represent updates incorporated since the RIA for MATS (U.S. EPA, 2011d). Studies highlighted in red represent updated incorporated since the proposal RIA (U.S. EPA, 2012a).

5.6.3.1 $PM_{2.5}$ Premature Mortality Effect Coefficients

Core Mortality Effect Coefficients for Adults. A substantial body of published scientific literature documents the association between elevated $PM_{2.5}$ concentrations and increased premature mortality (U.S. EPA, 2009b). This body of literature reflects thousands of epidemiology, toxicology, and clinical studies. The PM ISA completed as part of the this review of the PM standards, which was twice reviewed by the SAB-CASAC (U.S. EPA-SAB, 2009b, 2009c), concluded that there is a causal relationship between mortality and both long-term and short-term exposure to $PM_{2.5}$ based on the entire body of scientific evidence (U.S. EPA, 2009b). The size of the mortality effect estimates from epidemiological studies, the serious nature of the effect itself, and the high monetary value ascribed to prolonging life make mortality risk reduction the most significant health endpoint quantified in this analysis.

Researchers have found statistically significant associations between $PM_{2.5}$ and premature mortality using different types of study designs. Time-series methods have been used to relate short-term (often day-to-day) changes in $PM_{2.5}$ concentrations and changes in daily mortality rates up to several days after a period of elevated $PM_{2.5}$ concentrations. Cohort methods have been used to examine the potential relationship between community-level $PM_{2.5}$ exposures over multiple years (i.e., long-term exposures) and community-level annual mortality rates that have been adjusted for individual level risk factors. When choosing between using short-term studies or cohort studies for estimating mortality benefits, cohort analyses are thought to capture more of the public health impact of exposure to air pollution over time because they account for the effects of long-term exposures as well as some fraction of short-term exposures (Kunzli et al., 2001; NRC, 2002). The NRC stated that “it is essential to use the cohort studies in benefits analysis to capture all important effects from air pollution exposure”

(NRC, 2002, p. 108). The NRC further notes that "the overall effect estimates may be a combination of effects from long-term exposure plus some fraction from short-term exposure. The amount of overlap is unknown" (NRC, 2002, p. 108-9). To avoid double counting, we focus on applying the risk coefficients from the long-term cohort studies in estimating the mortality impacts of reductions in $PM_{2.5}$.

Over the last two decades, several studies using "prospective cohort" designs have been published that are consistent with the earlier body of literature. Two prospective cohort studies, often referred to as the Harvard "Six Cities Study" (Dockery et al., 1993; Laden et al., 2006; Lepeule et al., 2012) and the "American Cancer Society" or "ACS study" (Pope et al., 1995; Pope et al., 2002; Pope et al., 2004; Krewski et al., 2009), provide the most extensive analyses of ambient $PM_{2.5}$ concentrations and mortality. These studies have found consistent relationships between fine particle indicators and premature mortality across multiple locations in the United States. The credibility of these two studies is further enhanced by the fact that the initial published studies (Pope et al., 1995; Dockery et al., 1993) were subject to extensive reexamination and reanalysis by an independent team of scientific experts commissioned by the Health Effect Institute (HEI) and by a Special Panel of the HEI Health Review Committee (Krewski et al., 2000). Publication of studies confirming and extending the findings of the 1993 Six Cities Study and the 1995 ACS study using more recent air quality and a longer follow-up period for the ACS cohort provides additional validation of the findings of these original studies (Pope et al., 2002, 2004; Laden et al., 2006; Krewski et al., 2009; Lepeule et al., 2012). The SAB-HES also supported using these two cohorts for analyses of the benefits of PM reductions, and concluded, "the selection of these cohort studies as the underlying basis for PM mortality benefit estimates to be a good choice. These are widely cited, well studied and extensively reviewed data sets" (U.S. EPA-SAB, 2010a). As both the ACS and Six Cities studies have inherent strengths and weaknesses, we present benefits estimates using relative risk estimates from the most recent extended reanalysis of these cohorts (Krewski et al., 2009; Lepeule et al., 2012). Presenting results using both ACS and Six Cities is consistent with other recent RIAs (e.g., U.S. EPA, 2006a, 2010c, 2011c, 2011d). The PM ISA concludes that the ACS and Six Cities cohorts provide the strongest evidence of the association between long-term $PM_{2.5}$ exposure and premature mortality with support from a number of additional cohort studies (described below).

The extended analyses of the ACS cohort data (Krewski et al., 2009) provides additional refinements to the analysis of PM-related mortality by (a) extending the follow-up period by 2 years to the year 2000, for a total of 18 years; (b) incorporating almost double the number of urban areas (c) addressing confounding by spatial autocorrelation by incorporating ecological, or community-level, co-variates; (d) performing an extensive spatial analysis using land use regression modeling in two large urban areas. These enhancements make this analysis well-suited for the assessment of mortality risk from long-term $PM_{2.5}$ exposures for the EPA's benefits analyses.

In 2009, the SAB-HES again reviewed the choice of mortality risk coefficients for benefits analysis, concluding that "[t]he Krewski et al. (2009) findings, while informative, have not yet undergone the same degree of peer review as have the aforementioned studies. Thus, the SAB-HES recommends that EPA not use the Krewski et al. (2009) findings for generating the Primary Estimate" (U.S. EPA-SAB, 2010a). Since this time, the Krewski et al. (2009) has undergone additional peer review, which we believe strengthens the support for including this study in this RIA. For example, the PM ISA (U.S. EPA, 2009b) included this study among the key mortality studies. In addition, the risk assessment supporting the PM NAAQS (U.S. EPA, 2010b) utilized risk coefficients drawn from the Krewski et al. (2009) study, the most recent reanalysis of the ACS cohort data. The risk assessment cited a number of advantages that informed the selection of the Krewski et al. (2009) study as the source of the core effect estimates, including the extended period of observation, the rigorous examination of model forms and effect estimates, the coverage for ecological variables, and the large dataset with over 1.2 million individuals and 156 MSAs (U.S. EPA, 2010b). The SAB-CASAC also provided extensive peer review of the risk assessment and supported the use of effect estimates from this study (U.S. EPA-SAB, 2009a, 2010b, c).

Consistent with the *Quantitative Health Risk Assessment for Particulate Matter* (U.S. EPA, 2010b) which was reviewed by the SAB-CASAC (U.S. EPA-SAB, 2009), we use the all-cause mortality risk estimate based on the random-effects Cox proportional hazard model that incorporates 44 individual and 7 ecological covariates (RR=1.06, 95% confidence intervals 1.04–1.08 per 10μg/m^3 increase in $PM_{2.5}$). The relative risk estimate (1.06 per 10μg/m^3 increase in $PM_{2.5}$) is identical to the risk estimate drawn from the earlier Pope et al. (2002) study, though the confidence interval around the Krewski et al. (2009) risk estimate is tighter.

In the most recent Six Cities study, which was published after the last SAB-HES review, Lepeule et al. (2012) evaluated the sensitivity of previous Six Cities results to model specifications, lower exposures, and averaging time using eleven additional years of cohort follow-up that incorporated recent lower exposures. The authors found significant associations between $PM_{2.5}$ exposure and increased risk of all-cause, cardiovascular and lung cancer mortality. The authors also concluded that the concentration-response relationship was linear down to $PM_{2.5}$ concentrations of 8 μg/m^3, and that mortality rate ratios for $PM_{2.5}$ fluctuated over time, but without clear trends, despite a substantial drop in the sulfate fraction. We use the all-cause mortality risk estimate based on a Cox proportional hazard model that incorporates 3 individual covariates. (RR=1.14, 95% confidence intervals 1.07–1.22 per 10 μg/m^3 increase in $PM_{2.5}$). The relative risk estimate is slightly smaller than the risk estimate drawn from Laden et al. (2006), with relatively smaller confidence intervals.

Implicit in the calculation of $PM_{2.5}$-related premature mortality impacts are several key assumptions, which are described in further detail later in this chapter. First, we assume that there is a "cessation" lag in time between the reduction in PM exposure and the full reduction in mortality risk that affects the timing (and thus discounted monetary valuation) of the resulting premature deaths (see section 5.6.6.1). Second, following conclusions of the PM ISA, we assume that all fine particles are equally potent in causing premature mortality (see section 5.7.2). Third, following conclusions of the PM ISA, we assume that the health impact function for fine particles is linear within the range of ambient concentrations affected by these standards (see section 5.7.4).

Alternate Mortality Effect Coefficients for Adults. In addition to the ACS and Six Cities cohorts, several recent cohort studies conducted in North America provide evidence for the relationship between long-term exposure to $PM_{2.5}$ and the risk of premature death. Many of these additional cohort studies are described in the PM ISA (U.S. EPA, 2009) and the Provisional Assessment (U.S. EPA, 2012b) (and thus not summarized here).[9,10] Table 5-8 provides the effect estimates from each of these cohort studies for all-cause, cardiovascular, cardiopulmonary, and ischemic heart disease (IHD) mortality as well as the lowest measured air quality level (LML) and mean concentration in the study.

We also draw upon the results of the 2006 expert elicitation sponsored by the EPA (Roman et al., 2008; IEc, 2006) to demonstrate the sensitivity of the benefits estimates to 12 expert-defined concentration-response functions. The $PM_{2.5}$ expert elicitation and the derivation of effect estimates from the expert elicitation results are described in detail in the

[9] It is important to note that the newer studies in the *Provisional Assessment* are published in peer-reviewed journals and meet our study selection criteria, but they have not been assessed in the context of an *Integrated Science Assessment* nor gone through review by the SAB. In addition, only the ACS and H6C cohort studies have been recommended by the SAB as appropriate for benefits analysis of national rulemakings.

[10] In this chapter, we only describe multi-state cohort studies. There are additional cohort studies that we have not included in this list, including cohort studies that focus on single cities (e.g., Gan et al., 2012) and cohort studies focusing on methods development. In Appendix 5A, we provide additional information regarding cohort studies in California, which is the only state for which we identified single state cohorts.

2006 $PM_{2.5}$ NAAQS RIA (U.S. EPA, 2006a), the elicitation summary report (IEc, 2006) and Roman et al. (2008), and so we summarize here the key attributes of this study relative to the interpretation of the estimates of PM-related mortality reported here. We describe also how the epidemiological literature has evolved since the expert elicitation was conducted in 2005 and 2006.

Table 5-8. Summary of Effect estimates from Associated With Change in Long-Term Exposure to $PM_{2.5}$ in Recent Cohort Studies in North America

				Hazard Ratios per 10 μg/m^3 Change in $PM_{2.5}$ (95[th] percentile confidence intervals)			
Study	**Cohort (age)**	**LML (μg/m^3)**	**Mean (μg/m^3)**	**All Causes**	**Cardiovascular**	**Cardiopulmonary**	**IHD**
Pope et al. (2002)	ACS (age >30)	7.5	18.2	1.06 (1.02–1.11)	1.12 (1.08–1.15)	1.09 (1.03–1.16)	N/A
Laden et al. (2006)	Six Cities (age > 25)	10	16.4	1.16 (1.07–1.26)	1.28 (1.13–1.44)	N/A	N/A
Lipfert et al. (2006)[a]	Veterans (age 39–63)	<14.1	14.3	1.15 (1.05–1.25)	N/A	N/A	N/A
Miller et al. (2007)[b]	WHI (age 50–79)	3.4	13.5	N/A	1.76 (1.25–2.47)	N/A	2.21 (1.17–4.16)
Eftim et al. (2008)	Medicare (age > 65)	6	13.6	1.21 (1.15–1.27)	N/A	N/A	N/A
Zeger et al. (2008)[c]	Medicare (age > 65)	<9.8	13.2	1.068 (1.049–1.087)	N/A	N/A	N/A
Krewski et al. (2009)[d]	ACS (age >30)	5.8	14	1.06 (1.04–1.08)	N/A	1.13 (1.10–1.16)	1.24 (1.19–1.29)
Puett et al. (2009)[b]	NHS (age 30–55)	5.8	13.9	1.26 (1.02–1.54)	N/A	N/A	2.02 (1.07–3.78)
Crouse et al. (2011)[d,e]	Canadian census	1.9	8.7	1.06 (1.01–1.10)	N/A	N/A	N/A
Puett et al. (2011)[f]	Health Professionals (age 40–75)	<14.4	17.8	0.86 (0.70–1.00)	1.02 (0.84–1.23)	N/A	N/A
Lepeule et al. (2012)[d]	Six Cities (age > 25)	8	15.9	1.14 (1.07–1.22)	1.26 (1.14–1.40)	N/A	N/A

[a] Low socio-economic status (SES) men only. Used traffic proximity as a surrogate of exposure.

[b] Women only.

[c] Reflects risks in the Eastern U.S. Risks in the Central U.S. were higher, but the authors found no association in the Western U.S.

[d] Random effects Cox model with individual and ecologic covariates.

[e] Canadian population.

[f] Men with high socioeconomic status only.

The primary goal of the 2006 study was to elicit from a sample of health experts probabilistic distributions describing uncertainty in estimates of the reduction in mortality among the adult U.S. population resulting from reductions in ambient annual average $PM_{2.5}$ levels. These distributions were obtained through a formal interview protocol using methods designed to elicit subjective expert judgments. These experts were selected through a peer-nomination process and included experts in epidemiology, toxicology, and medicine. The elicitation interview consisted of a protocol of carefully structured questions, both qualitative and quantitative, about the nature of the $PM_{2.5}$-mortality relationship designed to build twelve individual distributions for the coefficient (or slope) of the C-R function relating changes in annual average $PM_{2.5}$ exposures to annual, adult all-cause mortality. The elicitation also provided useful information regarding uncertainty characterization in the $PM_{2.5}$-mortality relationship. Specifically, during their interviews, the experts highlighted several uncertainties inherent within the epidemiology literature, such as causality, concentration thresholds, effect modification, the role of short- and long-term exposures, potential confounding, and exposure misclassification. In Appendix 5c, we evaluate each of these uncertainties in the context of this health impact assessment. For several of these uncertainties, such as causality, we are able to use the expert-derived functions to quantify the impacts of applying different assumptions. The elicitation received favorable peer review in 2006 (Mansfield and Patil, 2006).

Prior to providing a quantitative estimate of the risk of premature death associated with long-term $PM_{2.5}$ exposure, the experts answered a series of "conditioning questions." One such question asked the experts to identify which epidemiological studies they found most informative. The "ideal study attributes"[11] according to the experts included:

- Geographic representation of the entire U.S. (e.g., monitoring sites across the country)
- Collection of information on individual risk factors and residential information both at the beginning and throughout the follow-up period
- Large sample size that is representative of the general U.S. population
- Collection of genetic information from cohort members to identify and assess potential effect modifiers
- Monitoring of individual exposures (e.g., with a personal monitor)
- Collection of data on levels of several co-pollutants (not only those that are monitored for compliance purposes
- Accurate characterization of outcome (i.e., cause of death)

[11] These criteria are substantively similar to EPA's study selection criteria identified in Table 5-5 of this chapter.

- Follow-up for a long period of time, up to a lifetime
- Prospective study design

Although no single epidemiological study completely satisfies each of these criteria, the experts determined that the ACS and Six Cities cohort studies best satisfy a majority of these ideal attributes. To varying degrees the studies examining these two cohorts: are geographically representative; have collected information on individual risk factors; include a large sample size; have collected data on co-pollutants in the case of the ACS study; have accurately characterized the health outcome; include a long (and growing) follow-up period; and, are prospective in nature. The experts also noted a series of limitations in these two cohort studies. In the case of the Six Cities study (Laden et al., 2006), the experts identified the "small sample size, limited number of cities, and concerns about representativeness of the six cities for the U.S. as a whole" as weaknesses. When considering the ACS study (Pope et al., 2002), the experts indicated that the "method of recruitment for the study, which resulted in a group with higher income, more education, and a greater proportion of whites than is representative of the general U.S. population" represented a shortcoming. Several experts also argued that because the ACS study relied upon "…whatever monitors were available to the study…a single monitor represent[ed] exposure for an entire metropolitan area…whereas [the Six Cities study] often had exposures assigned at the county level." Despite these limitations, the experts considered the Pope et al. (2002) extended analysis of the ACS cohort and the Laden et al. (2006) extended analysis of the Six Cities cohort to be particularly influential in their opinions (see Exhibit 3-3 of the elicitation summary report [IEc, 2006]).

Since the completion of the EPA's expert elicitation in 2006, additional epidemiology literature has become available, including 9 new multi-state cohort studies shown in Table 5-8. This newer literature addresses some of the weaknesses identified in the prior literature. For example, in an attempt to improve its characterization of population exposure the most recent extended analysis of the ACS cohort Krewski et al. (2009) incorporates two case studies that employ more spatially resolved estimates of population exposure.

In light of the availability of this newer literature, we have updated the presentation of results in the RIA. Specifically, we focus the core analysis on results derived from the two most recent studies of the ACS and Six Cities cohorts (Krewski et al., 2009; Lepeule et al., 2012). Because the other multi-state cohorts generally have limited geography and age/gender representativeness, these limitations preclude us from using these studies in our core benefits results, and we instead present the risk coefficients from these other multi-state cohorts in

Table 5-8. In addition, we now present the full distributions of the expert-derived results in a probabilistic graphic (rather than cascading the expert-derived results throughout the results tables as done in prior RIAs). We do not combine the expert results in order to preserve the breadth and diversity of opinion on the expert panel (Roman et. al., 2006). This presentation of the expert-derived results is generally consistent with SAB advice (U.S. EPA-SAB, 2008), which recommended that the EPA emphasize that "scientific differences existed only with respect to the magnitude of the effect of $PM_{2.5}$ on mortality, not whether such an effect existed" and that the expert elicitation "supports the conclusion that the benefits of $PM_{2.5}$ control are very likely to be substantial". Although it is possible that the newer literature could revise the experts' quantitative responses if elicited again, we believe that these general conclusions are unlikely to change.

Mortality Effect Coefficients for Infants. In addition to the adult mortality studies described above, several studies show an association between PM exposure and premature mortality in children under 5 years of age.[12] The PM ISA states that less evidence is available regarding the potential impact of $PM_{2.5}$ exposure on infant mortality than on adult mortality and the results of studies in several countries include a range of findings with some finding significant associations. Specifically, the PM ISA concluded that evidence exists for a stronger effect at the post-neonatal period and for respiratory-related mortality, although this trend is not consistent across all studies. In addition, compared to avoided premature deaths estimated for adult mortality, avoided premature deaths for infants are significantly smaller because the number of infants in the population is much smaller than the number of adults and the epidemiology studies on infant mortality provide smaller risk coefficients associated with exposure to $PM_{2.5}$.

In 2004, the SAB-HES noted the release of the WHO Global Burden of Disease Study focusing on ambient air, which cites several recently published time-series studies relating daily PM exposure to mortality in children (U.S. EPA-SAB, 2004a). The SAB-HES also cites the study by Belanger et al. (2003) as corroborating findings linking PM exposure to increased respiratory inflammation and infections in children. A study by Chay and Greenstone (2003) found that reductions in TSP caused by the recession of 1981–1982 were statistically associated with reductions in infant mortality at the county level. With regard to the cohort study conducted by Woodruff et al. (1997), the SAB-HES notes several strengths of the study, including the use of a larger cohort drawn from a large number of metropolitan areas and efforts to control for a

[12] For the purposes of this analysis, we only calculate benefits for infants age 0–1, not all children under 5 years old.

variety of individual risk factors in infants (e.g., maternal educational level, maternal ethnicity, parental marital status, and maternal smoking status). Based on these findings, the SAB-HES recommended that the EPA incorporate infant mortality into the primary benefits estimate and that infant mortality be evaluated using an impact function developed from the Woodruff et al. (1997) study (U.S. EPA-SAB, 2004a).

In 2010, the SAB-HES again noted the increasing body of literature relating infant mortality and PM exposure and supported the inclusion of infant mortality in the monetized benefits (U.S. EPA-SAB, 2010a). The SAB-HES generally supported the approach of estimating infant mortality based on Woodruff et al. (1997) and noted that a more recent study by Woodruff et al. (2006) continued to find associations between $PM_{2.5}$ and infant mortality in California. The SAB-HES also noted, "when PM_{10} results are scaled to estimate $PM_{2.5}$ impacts, the results yield similar risk estimates." Consistent with the *Costs and Benefits of the Clean Air Act* (U.S. EPA, 2011a), we continue to rely on the earlier 1997 study in part due to the national–scale of the earlier study.

5.6.3.2 Nonfatal Acute Myocardial Infarctions (AMI) (Heart Attacks)

Nonfatal heart attacks have been linked with short-term exposures to $PM_{2.5}$ in the United States (Mustafić et al., 2012; Peters et al., 2001; Sullivan et al., 2005; Pope et al., 2006; Zanobetti and Schwartz, 2006; Zanobetti et al., 2009) and other countries (Poloniecki et al., 1997; Barnett et al., 2006; Peters et al., 2005). In previous health impact assessments, we have relied upon a study by Peters et al. (2001) as the basis for the impact function estimating the relationship between $PM_{2.5}$ and nonfatal heart attacks. The Peters et al. (2001) study exhibits a number of strengths. In particular, it includes a robust characterization of populations experiencing acute myocardial infarctions (AMIs). The researchers interviewed patients within 4 days of their AMI events and, for inclusion in the study, patients were required to meet a series of criteria including minimum kinase levels, an identifiable onset of pain or other symptoms and the ability to indicate the time, place and other characteristics of their AMI pain in an interview.

Since the publication of Peters et al. (2001), a number of other single and multi-city studies have appeared in the literature. These studies include Sullivan et al. (2005), which considered the risk of $PM_{2.5}$-related hospitalization for AMIs in King County, WA; Pope et al. (2006), based in Wasatch Range, UT; Zanobetti and Schwartz (2006), based in Boston, MA; and, Zanobetti et al. (2009), a multi-city study of 26 U.S. communities. Each of these single and multi-city studies, with the exception of Pope et al. (2006), measure AMIs using hospital discharge rates. Conversely, the Pope et al. (2006) study is based on a large registry with angiographically characterized patients—arguably a more precise indicator of AMI. Because the Pope et al. (2006) study reflected both myocardial infarctions and unstable angina, this produces a more comprehensive estimate of acute ischemic heart disease events than the other studies. However, unlike the Peters study (Peters et al., 2006), Pope and colleagues did not measure the time of symptom onset, and $PM_{2.5}$ data were not measured on an hourly basis.

As a means of recognizing the strengths of the Peters study while also incorporating the newer evidence found in the four single and multi-city studies, we present a range of AMI estimates. The upper end of the range is calculated using the Peters study while the lower end of the range is the result of an equal-weights pooling of these four newer studies. It is important to note that when calculating the incidence of nonfatal AMI, the fraction of fatal heart attacks is subtracted to ensure that there is no double-counting with premature mortality estimates. Specifically, we apply an adjustment factor in the concentration-response function to reflect the probability of surviving a heart attack. Based on recent data from the Agency for Healthcare Research and Quality's Healthcare Utilization Project National Inpatient Sample database (AHRQ, 2009), we identified death rates for adults hospitalized with acute myocardial infarction stratified by age (e.g., 1.852% for ages 18–44, 2.8188% for ages 45–64, and 7.4339% for ages 65+). These rates show a clear downward trend over time between 1994 and 2009 for the average adult and thus replace the 7% survival rate previously applied across all age groups from Rosamond et al. (1999).

5.6.3.3 Hospital Admissions and Emergency Department Visits

Because of the availability of detailed hospital admission and discharge records, there is an extensive body of literature examining the relationship between hospital admissions and air pollution. For this reason, we pool together the incidence estimates using several different studies for many of the hospital admission endpoints. In addition, some studies have examined the relationship between air pollution and emergency department visits. Since most emergency department visits do not result in an admission to the hospital (i.e., most people going to the emergency department are treated and return home), we treat hospital admissions and emergency department visits separately, taking account of the fraction of emergency department visits that are admitted to the hospital. Specifically, within the baseline incidence rates, we parse out the scheduled hospital visits from unscheduled ones as well as the hospital visits that originated in the emergency department.

The two main groups of hospital admissions estimated in this analysis are respiratory admissions and cardiovascular admissions. There is not much evidence linking $PM_{2.5}$ with other types of hospital admissions. Both asthma- and cardiovascular-related visits have been linked to $PM_{2.5}$ in the United States, though as we note below, we are able to assign an economic value to asthma-related events only. To estimate the effects of $PM_{2.5}$ air pollution reductions on asthma-related ER visits, we use the effect estimate from a study of children 18 and under by Mar et al. (2010), Slaughter et al. (2005), and Glad et al. (2012). The first two studies examined populations 0 to 99 in Washington State, while Glad et al. examined populations 0-99 in Pittsburgh, PA. Mar and colleagues perform their study in Tacoma, while Slaughter and colleagues base their study in Spokane. We apply random/fixed effects pooling to combine evidence across these two studies.

To estimate avoided incidences of cardiovascular hospital admissions associated with $PM_{2.5}$, we used studies by Moolgavkar (2000), Zanobetti et al. (2009), Peng et al. (2008, 2009) and Bell et al., (2008). Only Moolgavkar (2000) provided a separate effect estimate for adults 20 to 64, while the remainder estimate risk among adults over 64.[13] Total cardiovascular hospital admissions are thus the sum of the pooled estimate for adults over 65 and the single study estimate for adults 20 to 64. Cardiovascular hospital admissions include admissions for myocardial infarctions. To avoid double-counting benefits from reductions in myocardial infarctions when applying the impact function for cardiovascular hospital admissions, we first adjusted the baseline cardiovascular hospital admissions to remove admissions for myocardial infarctions. We applied equal weights pooling to the multi-city studies assessing risk among adults over 64 because these studies already incorporated pooling across the city-level estimates. One potential limitation of our approach is that while the Zanobetti et al. (2009) study assesses all cardiovascular risk, Bell et al. (2008), and Peng et al., (2008, 2009) studies estimate a subset of cardiovascular hospitalizations as well as certain cerebro- and peripheral-vascular diseases. To address the potential for the pooling of these four studies to produce a biased estimate, we match the pooled risk estimate with a baseline incidence rate that excludes cerebro- and peripheral-vascular disease. An alternative approach would be to use the Zanobetti et al. (2009) study alone, though this would prevent us from drawing upon the strengths of the three multi-city studies.

[13] Note that the Moolgavkar (2000) study has not been updated to reflect the more stringent GAM convergence criteria. However, given that no other estimates are available for this age group, we chose to use the existing study. Given the very small (<5%) difference in the effect estimates for people 65 and older with cardiovascular hospital admissions between the original and reanalyzed results, we do not expect this choice to introduce much bias. For a discussion of the GAM convergence criteria, and how it affected the size of effect coefficients reported by time series epidemiological studies using NMMAPS data, see: http://www.healtheffects.org/Pubs/st-timeseries.htm.

To estimate avoided incidences of respiratory hospital admissions associated with $PM_{2.5}$, we used a number of studies examining total respiratory hospital admissions as well as asthma and chronic lung disease. We estimated impacts among three age groups: adults over 65, adults 18 to 64 and children 0 to 17. For adults over 65, the multi-city studies by Zanobetti et al. (2009) and Kloog et al. (2012) provide effect coefficients for total respiratory hospital admissions (defined as ICD codes 460–519). We pool these two studies using equal weights. Moolgavkar et al. (2003) examines $PM_{2.5}$ and chronic lung disease hospital admissions (less asthma) in Los Angeles, CA among adults 18 to 64. For children 0 to 18, we pool two studies using random/fixed effects. The first is Babin et al. (2007) which assessed $PM_{2.5}$ and asthma hospital admissions in Washington, DC among children 1 to 18; we adjusted the age range for this study to apply to children 0 to 18. The second is Sheppard et al. (2003) which assessed $PM_{2.5}$ and asthma hospitalizations in Seattle, Washington, among children 0 to 18.

5.6.3.4 Acute Health Events and School/Work Loss Days

In addition to mortality, chronic illness, and hospital admissions, a number of acute health effects not requiring hospitalization are associated with exposure to $PM_{2.5}$. The sources for the effect estimates used to quantify these effects are described below.

Approximately 4% of U.S. children between the ages of 5 and 17 experience episodes of acute bronchitis annually (ALA, 2002). Acute bronchitis is characterized by coughing, chest discomfort, slight fever, and extreme tiredness, lasting for a number of days. According to the MedlinePlus medical encyclopedia,[14] with the exception of cough, most acute bronchitis symptoms abate within 7 to 10 days. Incidence of episodes of acute bronchitis in children between the ages of 5 and 17 were estimated using an effect estimate developed from Dockery et al. (1996). Health effects from air pollution can also result in missed days of work (either from personal symptoms or from caring for a sick family member). Days of work lost due to $PM_{2.5}$ were estimated using an effect estimate developed from Ostro (1987). Children may also be absent from school because of respiratory or other diseases caused by exposure to air pollution, but we have not quantified these effects for this rule.

We estimate three types of acute respiratory symptoms: lower respiratory symptoms, upper respiratory symptoms, and minor restricted activity days (MRAD). Incidences of lower respiratory symptoms (e.g., wheezing, deep cough) in children aged 7 to 14 were estimated using an effect estimate from Schwartz and Neas (2000). Incidences of upper respiratory symptoms in asthmatic children aged 9 to 11 are estimated using an effect estimate developed

[14] See http://www.nlm.nih.gov/medlineplus/ency/article/001087.htm, accessed April 2012.

from Pope et al. (1991). Because asthmatics have greater sensitivity to stimuli (including air pollution), children with asthma can be more susceptible to a variety of upper respiratory symptoms (e.g., runny or stuffy nose; wet cough; and burning, aching, or red eyes). Research on the effects of air pollution on upper respiratory symptoms has thus focused on effects in asthmatics.

Minor restricted activity days (MRAD) result when individuals reduce most usual daily activities and replace them with less strenuous activities or rest, yet not to the point of missing work or school. For example, a mechanic who would usually be doing physical work most of the day will instead spend the day at a desk doing paper and phone work because of difficulty breathing or chest pain. The effect of $PM_{2.5}$ on MRAD was estimated using an effect estimate derived from Ostro and Rothschild (1989).

More recently published literature examining the relationship between short-term $PM_{2.5}$ exposure and acute respiratory symptoms was available in the PM ISA (U.S. EPA, 2009), but proved to be unsuitable for use in this benefits analysis. In particular, the best available study (Patel et al., 2010) specified a population aged 13–20, which overlaps with the population in which we asses asthma exacerbation. As we describe in detail below, to avoid the chance of double-counting impacts, we do not estimate changes in acute respiratory symptoms and asthma exacerbation among populations of the same age.

For this RIA, we have followed the SAB-HES recommendations regarding asthma exacerbations in developing the core estimate (U.S. EPA-SAB, 2004a). Although certain studies of acute respiratory events characterize these impacts among only asthmatic populations, others consider the full population, including both asthmatics and non-asthmatics. For this reason, incidence estimates derived from studies focused only on asthmatics cannot be added to estimates from studies that consider the full population—to do so would double-count impacts. To prevent such double-counting, we estimated the exacerbation of asthma among children and excluded adults from the calculation. Asthma exacerbations occurring in adults are assumed to be captured in the general population endpoints such as work loss days and MRADs. Finally, note also the important distinction between the exacerbation of asthma among asthmatic populations, and the onset of asthma among populations not previously suffering from asthma; in this RIA, we quantify the exacerbation of asthma among asthmatic populations and not the onset of new cases of asthma.

To characterize asthma exacerbations in children, we selected two studies (Ostro et al., 2001; Mar et al., 2004) that followed panels of asthmatic children. Ostro et al. (2001) followed a group of 138 African-American children in Los Angeles for 13 weeks, recording daily occurrences of respiratory symptoms associated with asthma exacerbations (e.g., shortness of breath, wheeze, and cough). This study found a statistically significant association between $PM_{2.5}$, measured as a 12-hour average, and the daily prevalence of shortness of breath and wheeze endpoints. Although the association was not statistically significant for cough, the results were still positive and close to significance; consequently, we decided to include this endpoint, along with shortness of breath and wheeze, in generating incidence estimates (see below).

Mar et al. (2004) studied the effects of various size fractions of particulate matter on respiratory symptoms of adults and children with asthma, monitored over many months. The study was conducted in Spokane, Washington, a semiarid city with diverse sources of particulate matter. Data on respiratory symptoms and medication use were recorded daily by the study's subjects, while air pollution data was collected by the local air agency and Washington State University. Subjects in the study consisted of 16 adults—the majority of whom participated for over a year—and nine children, all of whom were studied for over eight months. Among the children, the authors found a strong association between cough symptoms and several metrics of particulate matter, including $PM_{2.5}$. However, the authors found no association between respiratory symptoms and PM of any metric in adults. Mar et al. therefore concluded that the discrepancy in results between children and adults was due either to the way in which air quality was monitored, or a greater sensitivity of children than adults to increased levels of PM air pollution.

We employed the following pooling approach in combining estimates generated using effect estimates from the two studies to produce a single asthma exacerbation incidence estimate. First, we used random/fixed effects pooling to combine the Ostro and Mar estimates for shortness of breath and cough. Next, we pooled the Ostro estimate of wheeze with the pooled cough and shortness of breath estimates to derive an overall estimate of asthma exacerbation.

5.6.3.5 Effect Coefficients Selected for the Sensitivity Analyses

Chronic Bronchitis. Chronic bronchitis is characterized by mucus in the lungs and a persistent wet cough for at least 3 months a year for several years in a row. Chronic bronchitis affects an estimated 5% of the U.S. population (ALA, 1999). A limited number of studies have estimated the impact of air pollution on new incidences of chronic bronchitis. Schwartz (1993) and Abbey et al. (1995) provide evidence that long-term $PM_{2.5}$ exposure gives rise to the development of chronic bronchitis in adults in the United States; these remain the two most recent studies observing a relationship between long-term exposure to $PM_{2.5}$ and the onset of chronic bronchitis in the U.S. The absence of newer studies finding a relationship between long-term $PM_{2.5}$ exposure and chronic bronchitis argues for moving this endpoint from the core benefits analysis to a sensitivity analysis. In their review of the scientific literature on chronic obstructive pulmonary disease (COPD), which includes chronic bronchitis and emphysema, the American Thoracic Society concluded that air pollution is "associated with COPD, but sufficient criteria for causation were not met" (Eisner et al., 2010).

Stroke. The PM ISA (U.S. EPA, 2009) includes several new studies that have examined the relationship between $PM_{2.5}$ exposure and cerebrovascular events (U.S. EPA, 2009). Time-series studies have generally been inconsistent with several studies showing positive associations (Dominici et al., 2006; Metzger et al., 2004; Lippman et al., 2000; Lisabeth et al., 2008). Several other studies have demonstrated null or negative associations (Anderson et al., 2001; Barnett et al., 2006; Peel et al., 2007). In general, these studies examined cerebrovascular disease as a group, though a few studies partition ischemic and hemorrhagic strokes separately (Lisabeth et al., 2008). A key limitation of these time-series studies is that they use hospital discharge rates as the diagnosis and relatively short lags (0–2 days)—this is problematic, as discharge rates are an imperfect diagnosis and strokes may occur several days before admission to the hospital.

Longer-term prospective cohort studies of $PM_{2.5}$ and stroke include Miller et al. (2007). Miller et al. (2007) estimated the change in risk among post-menopausal women enrolled in the Women's Health Initiative (U.S. EPA, 2009b). After adjusting for age, race, smoking status, educational level, household income, body-mass index, diabetes, hypertension, and hypercholesterolemia, hazard ratios were estimated for the first cardiovascular event. Because this study considers first-time cardiovascular events, a key challenge to incorporating this study into the core health impact assessment is matching the baseline incidence rate correctly, and we have approximated this information using the data in the study.

In addition, Wellenius et al. (2012) examined the association of $PM_{2.5}$ with neurologist confirmed ischemic stroke in Boston adults in a time-stratified case-crossover study. A key feature of this study is that it included the time of stroke symptom onset for most patients. Similar to the challenge with Miller et al. (2007), we do not have baseline incidence rates, and we do not have sufficient data from the study to approximate it. Three factors argue for treating this endpoint in the sensitivity analysis: (1) the epidemiological literature examining PM-related cerebrovascular events is still evolving; (2) there are special uncertainties associated with quantifying this endpoint; (3) we have not yet identified an appropriate method for estimating the economic value of this endpoint.

Cardiovascular Emergency Department Visits. A large number of recent U.S.-based studies provide support for an association between short-term increases in $PM_{2.5}$ and increased risk of ED visits for ischemic heart diseases (U.S. EPA, 2009b). Both Metzger et al. (2004) and Tolbert et al. (2007) published interim results from the *Study of Particles and Health in Atlanta* (SOPHIA), finding a relationship between $PM_{2.5}$ exposure and cardiovascular emergency department visits. These cardiovascular emergency department visits are distinct from cardiovascular hospital admissions and non-fatal heart attacks. To ensure no double-counting, we excluded ICD-9-411 (ischemic heart disease) from the baseline incidence rates for cardiovascular emergency department visits. Mathes et al. (2011) find relationships between $PM_{2.5}$ levels and cardiovascular emergency department visits in New York City. The principal challenge to incorporating these studies is the absence of readily-available economic valuation estimates for cardiovascular emergency department visits. Until we develop an approach for estimating the economic value of this endpoint, we will quantify these ED visits in a sensitivity analysis only.

5.6.4 Unquantified Human Health Benefits

The illustrative emission reduction strategies to reach the revised and alternative annual standards described in Chapter 4 would reduce emissions of directly emitted particles, as well as SO_2, and NO_x for an alternative standard for 11 µg/m^3. The extent to which down wind exposure to secondary pollutants ozone, and mercury would actually be reduced would depend on the specific control strategy that States would use to reduce $PM_{2.5}$ in a given area as well as local geographic and meteorological conditions. Although we have quantified many of the health benefits associated with reducing exposure to $PM_{2.5}$, as shown in Table 5-2, we are unable to quantify the health benefits associated with reducing the potential for ozone exposure, SO_2 exposure, NO_2 exposure or contamination of local water bodies with mercury due to the absence of air quality modeling data for these pollutants in this analysis. Although the method we applied simulated the impact of attaining the revised and alternative annual standards on ambient levels of $PM_{2.5}$, this method does not simulate how the illustrative emission reductions would affect ambient levels of ozone, SO_2, or NO_2. Furthermore, the air quality modeling conducted for this analysis did not assess mercury, so we are unable to estimate mercury deposition associated with the illustrative controls or subsequent bioaccumulation and exposure. Below we provide a qualitative description of these health benefits. In general, previous analyses have shown that the monetized value of these additional health benefits is much smaller than $PM_{2.5}$-related benefits (U.S. EPA, 2010a, 2010c, 2010d).

Reducing NO_x emissions also reduces ozone concentrations in most areas. Reducing ambient ozone concentrations is associated with significant human health benefits, including mortality and respiratory morbidity (U.S. EPA, 2008a, 2010d). Epidemiological researchers have associated ozone exposure with adverse health effects in numerous toxicological, clinical and epidemiological studies (U.S. EPA, 2006b; U.S. EPA, 2012c). When adequate data and resources are available, the EPA generally quantifies several health effects associated with exposure to ozone (e.g., U.S. EPA, 2008a, 2010d, 2011a, 2011c). These health effects include respiratory morbidity such as asthma attacks, hospital and emergency department visits, school loss days, as well as premature mortality. The scientific literature suggests that exposure to ozone is also associated with chronic respiratory damage and premature aging of the lungs, but the EPA has not quantified these effects in benefits analyses previously.

Following an extensive evaluation of health evidence from epidemiologic and laboratory studies, the *Integrated Science Assessment for Sulfur Dioxide—Health Criteria* (SO_2 ISA) concluded that there is a causal relationship between respiratory health effects and short-term exposure to SO_2 (U.S. EPA, 2008c). The immediate effect of SO_2 on the respiratory system in humans is bronchoconstriction. Asthmatics are more sensitive to the effects of SO_2 likely resulting from preexisting inflammation associated with this disease. A clear concentration-response relationship has been demonstrated in laboratory studies following exposures to SO_2 at concentrations between 20 and 100 ppb, both in terms of increasing severity of effect and percentage of asthmatics adversely affected. Based on our review of this information, we identified four short-term morbidity endpoints that the SO_2 ISA identified as a "causal relationship": asthma exacerbation, respiratory-related emergency department visits, and respiratory-related hospitalizations. The differing evidence and associated strength of the evidence for these different effects is described in detail in the SO_2 ISA. The SO_2 ISA also concluded that the relationship between short-term SO_2 exposure and premature mortality was "suggestive of a causal relationship" because it is difficult to attribute the mortality risk effects to SO_2 alone. Although the SO_2 ISA stated that studies are generally consistent in reporting a relationship between SO_2 exposure and mortality, there was a lack of robustness of the observed associations to adjustment for pollutants. We did not quantify these benefits due to data constraints.

Epidemiological researchers have associated NO_2 exposure with adverse health effects in numerous toxicological, clinical and epidemiological studies, as described in the *Integrated*

Science Assessment for Oxides of Nitrogen—Health Criteria (NO_2 ISA) (U.S. EPA, 2008b). The NO_2 ISA provides a comprehensive review of the current evidence of health and environmental effects of NO_2. The NO_2 ISA concluded that the evidence "is sufficient to infer a likely causal relationship between short-term NO_2 exposure and adverse effects on the respiratory system." These epidemiologic and experimental studies encompass a number of endpoints including emergency department visits and hospitalizations, respiratory symptoms, airway hyperresponsiveness, airway inflammation, and lung function. Effect estimates from epidemiologic studies conducted in the United States and Canada generally indicate a 2–20% increase in risks for ED visits and hospital admissions and higher risks for respiratory symptoms. The NO_2 ISA concluded that the relationship between short-term NO_2 exposure and premature mortality was "suggestive but not sufficient to infer a causal relationship" because it is difficult to attribute the mortality risk effects to NO_2 alone. Although the NO_2 ISA stated that studies consistently reported a relationship between NO_2 exposure and mortality, the effect was generally smaller than that for other pollutants such as PM. We did not quantify these benefits due to data constraints.

5.6.5 Economic Valuation Estimates

Reductions in ambient concentrations of air pollution generally lower the risk of future adverse health effects for a large population. Therefore, the appropriate economic measure is WTP for changes in risk of a health effect rather than WTP for a health effect that would occur with certainty (Freeman, 1993). Epidemiological studies generally provide estimates of the relative risks of a particular health effect that is avoided because of a reduction in air pollution. We converted those changes in risk to units of avoided statistical incidence for ease of presentation. We calculated the value of avoided statistical incidences by dividing individual WTP for a risk reduction by the related observed change in risk.

WTP estimates generally are not available for some health effects, such as hospital admissions. In these cases, we instead used the cost of treating or mitigating the effect to estimate the economic value. COI estimates generally (although not necessarily in all cases) understate the true value of reducing the risk of a health effect, because they reflect the direct expenditures related to treatment, but not the value of avoided pain and suffering (Harrington and Portney, 1987; Berger, 1987).

We provide unit values for health endpoints (along with information on the distribution of the unit value) in Table 5-9. All values are in constant year 2006 dollars, adjusted for growth in real income for WTP estimates out to 2020 using projections provided by Standard and

Poor's, which is discussed in further detail in Section 5.6.8. Economic theory argues that WTP for most goods (such as environmental protection) will increase if real income increases. Several of the valuation studies used in this analysis were conducted in the late 1980s and early 1990s, and we are in the process of reviewing the literature to update these unit values. The discussion below provides additional details on valuing specific $PM_{2.5}$-related related endpoints.

5.6.5.1 Mortality Valuation

Following the advice of the SAB's Environmental Economics Advisory Committee (SAB-EEAC), the EPA currently uses the value of statistical life (VSL) approach in calculating the core estimate of mortality benefits, because we believe this calculation provides the most reasonable single estimate of an individual's willingness to trade off money for reductions in mortality risk (U.S. EPA-SAB, 2000). The VSL approach is a summary measure for the value of small changes in mortality risk experienced by a large number of people. For a period of time (2004–2008), the Office of Air and Radiation (OAR) valued mortality risk reductions using a VSL estimate derived from a limited analysis of some of the available studies. OAR arrived at a VSL using a range of $1 million to $10 million (2000$) consistent with two meta-analyses of the wage-risk literature. The $1 million value represented the lower end of the interquartile range from the Mrozek and Taylor (2002) meta-analysis of 33 studies. The $10 million value represented the upper end of the interquartile range from the Viscusi and Aldy (2003) meta-analysis of 43 studies. The mean estimate of $5.5 million (2000$) was also consistent with the mean VSL of $5.4 million estimated in the Kochi et al. (2006) meta-analysis. However, the Agency neither changed its official guidance on the use of VSL in rule-makings nor subjected the interim estimate to a scientific peer-review process through SAB or other peer-review group.

Table 5-9. Unit Values for Economic Valuation of Health Endpoints (2010$)[a]

Health Endpoint	Central Estimate of Value Per Statistical Incidence: 1990 Income Level	Central Estimate of Value Per Statistical Incidence: 2020 Income Level	Derivation of Distributions of Estimates
Premature Mortality (Value of a Statistical Life)	$8,000,000	$9,600,000	The EPA currently recommends a central VSL of $4.8m (1990$, 1990 income) based on a Weibull distribution fitted to 26 published VSL estimates (5 contingent valuation and 21 labor market studies). The underlying studies, the distribution parameters, and other useful information are available in Appendix B of the EPA's *Guidelines for Preparing Economic Analyses* (U.S. EPA, 2010e).
Nonfatal Myocardial Infarction (heart attack)			No distributional information available. Age-specific cost-of-illness values reflect lost earnings and direct medical costs over a 5-year period following a nonfatal MI. Lost earnings estimates are based on Cropper and Krupnick (1990). Direct medical costs are based on simple average of estimates from Russell et al. (1998) and Wittels et al. (1990).
3% discount rate			Lost earnings:
Age 0–24	$98,000	$98,000	Cropper and Krupnick (1990). Present discounted value of 5 years of lost earnings in 2000$:
Age 25–44	$110,000	$110,000	age of onset: at 3% / at 7%
Age 45–54	$120,000	$120,000	25–44: $9,000 / $8,000
Age 55–64	$200,000	$200,000	45–54: $13,000 / $12,000
Age 65 and over	$98,000	$98,000	55–65: $77,000 / $69,000
7% discount rate			Direct medical expenses (2000$): An average of:
Age 0–24	$97,000	$97,000	1. Wittels et al. (1990) ($100,000—no discounting)
Age 25–44	$110,000	$110,000	2. Russell et al. (1998), 5-year period ($22,000 at 3% discount rate; $21,000at 7% discount rate)
Age 45–54	$110,000	$110,000	
Age 55–64	$190,000	$190,000	
Age 65 and over	$97,000	$97,000	

(continued)

Table 5-9. Unit Values for Economic Valuation of Health Endpoints (2010$) [a] (continued)

Health Endpoint	Central Estimate of Value Per Statistical Incidence: 2000 Income Level	Central Estimate of Value Per Statistical Incidence: 2020 Income Level	Derivation of Distributions of Estimates
Hospital Admissions			
Chronic Lung Disease (18–64)	$21,000	$21,000	No distributional information available. The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total chronic lung illnesses) reported in Agency for Healthcare Research and Quality (2007) (www.ahrq.gov).
Asthma Admissions (0–64)	$16,000	$16,000	No distributional information available. The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total asthma category illnesses) reported in Agency for Healthcare Research and Quality (2007) (www.ahrq.gov).
All Cardiovascular			No distributional information available. The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total cardiovascular category illnesses) reported in Agency for Healthcare Research and Quality (2007) (www.ahrq.gov).
Age 18–64	$42,000	$42,000	
Age 65–99	$41,000	$41,000	
All respiratory (ages 65+)	$36,000	$36,000	No distributions available. The COI point estimates (lost earnings plus direct medical costs) are based on ICD-9 code level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total respiratory category illnesses) reported in Agency for Healthcare Research and Quality, 2007 (www.ahrq.gov).
Emergency Department Visits for Asthma	$430	$430	No distributional information available. Simple average of two unit COI values (2000$): (1) $310, from Smith et al. (1997) and (2) $260, from Stanford et al. (1999).

(continued)

Table 5-9. Unit Values for Economic Valuation of Health Endpoints (2010$) [a] (continued)

Health Endpoint	Central Estimate of Value Per Statistical Incidence: 2000 Income Level	Central Estimate of Value Per Statistical Incidence: 2020 Income Level	Derivation of Distributions of Estimates
Respiratory Ailments Not Requiring Hospitalization			
Upper Respiratory Symptoms (URS)	$31	$33	Combinations of the three symptoms for which WTP estimates are available that closely match those listed by Pope et al. result in seven different "symptom clusters," each describing a "type" of URS. A dollar value was derived for each type of URS, using mid-range estimates of WTP (IEc, 1994) to avoid each symptom in the cluster and assuming additivity of WTPs. In the absence of information surrounding the frequency with which each of the seven types of URS occurs within the URS symptom complex, we assumed a uniform distribution between $9.2 and $43 (2000$).
Lower Respiratory Symptoms (LRS)	$20	$21	Combinations of the four symptoms for which WTP estimates are available that closely match those listed by Schwartz et al. result in 11 different "symptom clusters," each describing a "type" of LRS. A dollar value was derived for each type of LRS, using mid-range estimates of WTP (IEc, 1994) to avoid each symptom in the cluster and assuming additivity of WTPs. The dollar value for LRS is the average of the dollar values for the 11 different types of LRS. In the absence of information surrounding the frequency with which each of the 11 types of LRS occurs within the LRS symptom complex, we assumed a uniform distribution between $6.9 and $25 (2000$).
Asthma Exacerbations	$54	$58	Asthma exacerbations are valued at $45 per incidence, based on the mean of average WTP estimates for the four severity definitions of a "bad asthma day," described in Rowe and Chestnut (1986). This study surveyed asthmatics to estimate WTP for avoidance of a "bad asthma day," as defined by the subjects. For purposes of valuation, an asthma exacerbation is assumed to be equivalent to a day in which asthma is moderate or worse as reported in the Rowe and Chestnut (1986) study. The value is assumed have a uniform distribution between $16 and $71 (2000$).

(continued)

Table 5-9. Unit Values for Economic Valuation of Health Endpoints (2010$)[a] (continued)

Health Endpoint	Central Estimate of Value Per Statistical Incidence: 2000 Income Level	Central Estimate of Value Per Statistical Incidence: 2020 Income Level	Derivation of Distributions of Estimates
Respiratory Ailments Not Requiring Hospitalization (continued)			
Acute Bronchitis	$450	$480	Assumes a 6-day episode, with the distribution of the daily value specified as uniform with the low and high values based on those recommended for related respiratory symptoms in Neumann et al. (1994). The low daily estimate of $10 is the sum of the mid-range values recommended by IEc (1994) for two symptoms believed to be associated with acute bronchitis: coughing and chest tightness. The high daily estimate was taken to be twice the value of a minor respiratory restricted-activity day, or $110 (2000$).
Work Loss Days (WLDs)	Variable (U.S. median = $150)	Variable (U.S. median = $150)	No distribution available. Point estimate is based on county-specific median annual wages divided by 52 and then by 5—to get median daily wage. U.S. Year 2000 Census, compiled by Geolytics, Inc. (Geolytics, 2002)
Minor Restricted Activity Days (MRADs)	$64	$68	Median WTP estimate to avoid one MRAD from Tolley et al. (1986). Distribution is assumed to be triangular with a minimum of $22 and a maximum of $83, with a most likely value of $52 (2000$). Range is based on assumption that value should exceed WTP for a single mild symptom (the highest estimate for a single symptom—for eye irritation—is $16.00) and be less than that for a WLD. The triangular distribution acknowledges that the actual value is likely to be closer to the point estimate than either extreme.

[a] All estimates are rounded to two significant digits. Unrounded estimates in 2000$ are available in the Appendix J of the BenMAP user manual (Abt Associates, 2012).

During this time, the Agency continued work to update its guidance on valuing mortality risk reductions, including commissioning a report from meta-analytic experts to evaluate methodological questions raised by the EPA and the SAB on combining estimates from the various data sources. In addition, the Agency consulted several times with the SAB-EEAC on the issue. With input from the meta-analytic experts, the SAB-EEAC advised the Agency to update its guidance using specific, appropriate meta-analytic techniques to combine estimates from unique data sources and different studies, including those using different methodologies (i.e., wage-risk and stated preference) (U.S. EPA-SAB, 2007).

Until updated guidance is available, the Agency determined that a single, peer-reviewed estimate applied consistently best reflects the SAB-EEAC advice it has received. Therefore, the Agency has decided to apply the VSL that was vetted and endorsed by the SAB in the *Guidelines for Preparing Economic Analyses* (U.S. EPA, 2000)[15] while the Agency continues its efforts to update its guidance on this issue. This approach calculates a mean value across VSL estimates derived from 26 labor market and contingent valuation studies published between 1974 and 1991. The mean VSL across these studies is $4.8 million (1990$) or $6.3 million (2000$).[16] The Agency is committed to using scientifically sound, appropriately reviewed evidence in valuing mortality risk reductions and has made significant progress in responding to the SAB-EEAC's specific recommendations. In the process, the Agency has identified a number of important issues to be considered in updating its mortality risk valuation estimates. These are detailed in a white paper on "Valuing Mortality Risk Reductions in Environmental Policy," which recently underwent review by the SAB-EEAC. A meeting with the SAB on this paper was held on March 14, 2011 and formal recommendations were transmitted on July 29, 2011 (U.S. EPA-SAB, 2011). Draft guidance responding to SAB recommendations will be developed shortly.

The economics literature concerning the appropriate method for valuing reductions in premature mortality risk is still developing. The adoption of a value for the projected reduction in the risk of premature mortality is the subject of continuing discussion within the economics and public policy analysis community. The EPA strives to use the best economic science in its analyses. Given the mixed theoretical finding and empirical evidence regarding adjustments to VSL for risk and population characteristics (e.g., Smith et al., 2004; Alberini et al., 2004; Aldy and Viscusi, 2008), we use a single VSL for all reductions in mortality risk.

[15] In the updated *Guidelines for Preparing Economic Analyses* (U.S. EPA, 2010e), EPA retained the VSL endorsed by the SAB with the understanding that further updates to the mortality risk valuation guidance would be forthcoming in the near future.

[16] In this analysis, we adjust the VSL to account for a different currency year (2010$) and to account for income growth to 2020. After applying these adjustments to the $6.3 million value, the VSL is $8.9M.

Although there are several differences between the labor market studies the EPA uses to derive a VSL estimate and the $PM_{2.5}$ air pollution context addressed here, those differences in the affected populations and the nature of the risks imply both upward and downward adjustments. Table 5-10 lists some of these differences and the expected effect on the VSL estimate for air pollution-related mortality. In the absence of a comprehensive and balanced set of adjustment factors, the EPA believes it is reasonable to continue to use the $4.8 million (1990$) value adjusted for inflation and income growth over time while acknowledging the significant limitations and uncertainties in the available literature.

Table 5-10. Influence of Applied VSL Attributes on the Size of the Economic Benefits of Reductions in the Risk of Premature Death (U.S. EPA, 2006a)

Attribute	Expected Direction of Bias
Age	Uncertain, perhaps overestimate
Life Expectancy/Health Status	Uncertain, perhaps overestimate
Attitudes Toward Risk	Underestimate
Income	Uncertain
Voluntary vs. Involuntary	Uncertain, perhaps underestimate
Catastrophic vs. Protracted Death	Uncertain, perhaps underestimate

The SAB-EEAC has reviewed many potential VSL adjustments and the state of the economics literature. The SAB-EEAC advised the EPA to "continue to use a wage-risk-based VSL as its primary estimate, including appropriate sensitivity analyses to reflect the uncertainty of these estimates," and that "the only risk characteristic for which adjustments to the VSL can be made is the timing of the risk" (U.S. EPA-SAB, 2000). In developing our core estimate of the benefits of premature mortality reductions, we have followed this advice.

For premature mortality, we assume that there is a "cessation" lag between PM exposures and the total realization of changes in health effects. We assumed for this analysis that some of the incidences of premature mortality related to $PM_{2.5}$ exposures occur in a distributed fashion over the 20 years following exposure and discounted over the period between exposure and premature mortality. Although the structure of the lag is uncertain, the EPA follows the advice of the SAB-HES to assume a segmented lag structure characterized by 30% of mortality reductions in the first year, 50% over years 2 to 5, and 20% over the years 6 to 20 after the reduction in $PM_{2.5}$ (U.S. EPA-SAB, 2004c). Additional cessation lag structures are described and assessed in Appendix 5.A of this RIA. To take this into account in the valuation of reductions in premature mortality, we discount the value of premature mortality occurring in future years using rates of 3% and 7%.[17] Changes in the cessation lag assumptions do not change the total number of estimated deaths but rather the timing of those deaths. As such, the monetized benefits using a 7% discount rate are only approximately 10% less than the monetized benefits using a 3% discount rate. Further discussion of this topic appears in the EPA's *Guidelines for Preparing Economic Analyses* (U.S. EPA, 2010e).

*Uncertainties Specific to Premature Mortality Valuation***.** The economic benefits associated with reductions in the risk of premature mortality are the largest category of monetized benefits in this RIA. In addition, in prior analyses, the EPA has identified valuation of mortality-related benefits as the largest contributor to the range of uncertainty in monetized benefits (Mansfield et al., 2009).[18] Because of the uncertainty in estimates of the value of reducing premature mortality risk, it is important to adequately characterize and understand the various types of economic approaches available for valuing reductions in mortality risk. Such an assessment also requires an understanding of how alternative valuation approaches reflect that some individuals may be more susceptible to air pollution-induced mortality or reflect differences in the nature of the risk presented by air pollution relative to the risks studied in the relevant economics literature.

The health science literature on air pollution indicates that several human characteristics affect the degree to which mortality risk affects an individual. For example, some age groups appear to be more susceptible to air pollution than others (e.g., the elderly and children). Health status prior to exposure also affects susceptibility. An ideal benefits estimate of mortality risk reduction would reflect these human characteristics, in addition to an individual's WTP to improve one's own chances of survival plus WTP to improve other individuals' survival rates. The ideal measure would also take into account the specific nature of the risk reduction commodity that is provided to individuals, as well as the context in which risk is reduced. To measure this value, it is important to assess how reductions in air pollution reduce the risk of dying from the time that reductions take effect onward and how individuals

[17] The choice of a discount rate, and its associated conceptual basis, is a topic of ongoing discussion within the federal government. To comply with Circular A-4, EPA provides monetized benefits using discount rates of 3% and 7% (OMB, 2003). A 3% discount reflects reliance on a "social rate of time preference" discounting concept. A 7% rate is consistent with an "opportunity cost of capital" concept to reflect the time value of resources directed to meet regulatory requirements.

[18] This conclusion was based on an assessment of uncertainty based on statistical error in epidemiological effect estimates and economic valuation estimates. Additional sources of model error such as those examined in the $PM_{2.5}$ mortality expert elicitation (Roman et al., 2008) may result in different conclusions about the relative contribution of sources of uncertainty.

value these changes. Each individual's survival curve, or the probability of surviving beyond a given age, should shift as a result of an environmental quality improvement. For example, changing the current probability of survival for an individual also shifts future probabilities of that individual's survival. This probability shift will differ across individuals because survival curves depend on such characteristics as age, health state, and the current age to which the individual is likely to survive.

Although a survival curve approach provides a theoretically preferred method for valuing the benefits of reduced risk of premature mortality associated with reducing air pollution, the approach requires a great deal of data to implement. The economic valuation literature does not yet include good estimates of the value of this risk reduction commodity. As a result, in this study we value reductions in premature mortality risk using the VSL approach.

Other uncertainties specific to premature mortality valuation include the following:

- *Across-study variation:* There is considerable uncertainty as to whether the available literature on VSL provides adequate estimates of the VSL for risk reductions from air pollution reduction. Although there is considerable variation in the analytical designs and data used in the existing literature, the majority of the studies involve the value of risks to a middle-aged working population. Most of the studies examine differences in wages of risky occupations, using a hedonic wage approach. Certain characteristics of both the population affected and the mortality risk facing that population are believed to affect the average WTP to reduce the risk. The appropriateness of a distribution of WTP based on the current VSL literature for valuing the mortality-related benefits of reductions in air pollution concentrations therefore depends not only on the quality of the studies (i.e., how well they measure what they are trying to measure), but also on the extent to which the risks being valued are similar and the extent to which the subjects in the studies are similar to the population affected by changes in pollution concentrations.

- *Level of risk reduction:* The transferability of estimates of the VSL from the wage-risk studies to the context of the PM NAAQS analysis rests on the assumption that, within a reasonable range, WTP for reductions in mortality risk is linear in risk reduction. For example, suppose a study provides a result that the average WTP for a reduction in mortality risk of 1/100,000 is $50, but that the actual mortality risk reduction resulting from a given pollutant reduction is 1/10,000. If WTP for reductions in mortality risk is linear in risk reduction, then a WTP of $50 for a reduction of 1/100,000 implies a WTP of $500 for a risk reduction of 1/10,000 (which is 10 times the risk reduction valued in the study). Under the assumption of linearity, the estimate of the VSL does not depend on the particular amount of risk reduction being valued. This assumption has been shown to be reasonable provided the change in the risk being valued is within the range of risks evaluated in the underlying studies (Rowlatt et al., 1998).

- *Voluntariness of risks evaluated:* Although job-related mortality risks may differ in several ways from air pollution-related mortality risks, the most important difference may be that job-related risks are incurred voluntarily, or generally assumed to be, whereas air pollution-related risks are incurred involuntarily. Some evidence suggests that people will pay more to reduce involuntarily incurred risks than risks incurred voluntarily (e.g., Lichtenstein and Slovic, 2006). If this is the case, WTP estimates based on wage-risk studies may understate WTP to reduce involuntarily incurred air pollution-related mortality risks.

- *Sudden versus protracted death:* A final important difference related to the nature of the risk may be that some workplace mortality risks tend to involve sudden, catastrophic events, whereas air pollution-related risks tend to involve longer periods of disease and suffering prior to death. Some evidence suggests that WTP to avoid a risk of a protracted death involving prolonged suffering and loss of dignity and personal control is greater than the WTP to avoid a risk (of identical magnitude) of sudden death (e.g., Tsuge et al., 2005; Alberini and Scasny, 2011). To the extent that the mortality risks addressed in this assessment are associated with longer periods of illness or greater pain and suffering than are the risks addressed in the valuation literature, the WTP measurements employed in the present analysis would reflect a downward bias.

- *Self-selection and skill in avoiding risk:* Recent research (Shogren and Stamland, 2002) suggests that VSL estimates based on hedonic wage studies may overstate the average value of a risk reduction. This is based on the fact that the risk-wage trade-off revealed in hedonic studies reflects the preferences of the marginal worker (i.e., that worker who demands the highest compensation for his risk reduction for a given job). This worker must have either a higher workplace risk than the average worker in a given occupation, a lower risk tolerance than the average worker in that occupation, or both. Conversely, the marginal worker should have a higher risk tolerance than workers employed in less-risky sectors. However, the risk estimate used in hedonic studies is generally based on average risk, so the VSL may be biased, in an ambiguous direction, because the wage differential and risk measures do not match.

- *Baseline risk and age:* Recent research (Smith, Pattanayak, and Van Houtven, 2006) finds that because individuals reevaluate their baseline risk of death as they age, the marginal value of risk reductions does not decline with age as predicted by some lifetime consumption models. This research supports findings in recent stated preference studies that suggest only small reductions in the value of mortality risk reductions with increasing age (e.g., Alberini et al., 2004).

5.6.5.2 Nonfatal Myocardial Infarctions Valuation

We were not able to identify a suitable WTP value for reductions in the risk of nonfatal heart attacks. Instead, we use a COI unit value with two components: the direct medical costs and the opportunity cost (lost earnings) associated with the illness event. Because the costs associated with a myocardial infarction extend beyond the initial event itself, we consider costs incurred over several years. Using age-specific annual lost earnings estimated by Cropper and Krupnick (1990) and a 3% discount rate, we estimated a rounded present discounted value in lost earnings (in 2000$) over 5 years due to a myocardial infarction of $8,800 for someone between the ages of 25 and 44, $13,000 for someone between the ages of 45 and 54, and $75,000 for someone between the ages of 55 and 65. The rounded corresponding age-specific estimates of lost earnings (in 2000$) using a 7% discount rate are $7,900, $12,000, and $67,000, respectively. Cropper and Krupnick (1990) do not provide lost earnings estimates for populations under 25 or over 65. As such, we do not include lost earnings in the cost estimates for these age groups.

We found three possible sources in the literature of estimates of the direct medical costs of myocardial infarction, which provide significantly different values (see Table 5-11):

- Wittels et al. (1990) estimated expected total medical costs of myocardial infarction over 5 years to be $51,000 (rounded in 1986$) for people who were admitted to the hospital and survived hospitalization. (There does not appear to be any discounting used.) This estimated cost is based on a medical cost model, which incorporated therapeutic options, projected outcomes, and prices (using "knowledgeable cardiologists" as consultants). The model used medical data and medical decision algorithms to estimate the probabilities of certain events and/or medical procedures being used. The authors note that the average length of hospitalization for acute myocardial infarction has decreased over time (from an average of 12.9 days in 1980 to an average of 11 days in 1983). Wittels et al. used 10 days as the average in their study. It is unclear how much further the length of stay for myocardial infarction may have decreased from 1983 to the present. The average length of stay for ICD code 410 (myocardial infarction) in the year-2000 Agency for Healthcare Research and Quality (AHRQ) HCUP database is 5.5 days (AHRQ, 2000). However, this may include patients who died in the hospital (not included among our nonfatal myocardial infarction cases), and whose length of stay was therefore substantially shorter than it would be if they had not died.

- Eisenstein et al. (2001) estimated 10-year costs of $45,000 in rounded 1997$ (using a 3% discount rate) for myocardial infarction patients, using statistical prediction (regression) models to estimate inpatient costs. Only inpatient costs (physician fees and hospital costs) were included.

Table 5-11. Alternative Direct Medical Cost of Illness Estimates for Nonfatal Heart Attacks[a]

Study	Direct Medical Costs (2010$)	Over an x-Year Period, for x =
Wittels et al. (1990)	$160,000[b]	5
Russell et al. (1998)	$33,000[c]	5
Average (5-year) costs	$98,000	5
Eisenstein et al. (2001)	$74,000[c]	10

[a] All estimates rounded to two significant digits. Unrounded estimates in 2000$ are available in appendix J of the BenMAP user manual (Abt Associates, 2012).

[b] Wittels et al. (1990) did not appear to discount costs incurred in future years.

[c] Using a 3% discount rate. Discounted values as reported in the study.

As noted above, the estimates from these three studies are substantially different, and we have not adequately resolved the sources of differences in the estimates. Because the wage-related opportunity cost estimates from Cropper and Krupnick (1990) cover a 5-year period, we used estimates for medical costs that similarly cover a 5-year period (i.e., estimates from Wittels et al. (1990) and Russell et al. (1998). We used a simple average of the two 5-year estimates, or rounded to $85,000, and added it to the 5-year opportunity cost estimate. The resulting estimates are given in Table 5-12.

Table 5-12. Estimated Costs Over a 5-Year Period of a Nonfatal Myocardial Infarction (in 2010$)[a]

Age Group	Opportunity Cost	Medical Cost[b]	Total Cost
0–24	$0	$98,000	$98,000
25–44	$12,000[c]	$98,000	$110,000
45–54	$17,000[c]	$98,000	$120,000
55–65	$100,000[c]	$98,000	$200,000
> 65	$0	$98,000	$98,000

[a] All estimates rounded to two significant digits, so estimates may not sum across columns. Unrounded estimates in 2000$ are available in appendix J of the BenMAP user manual (Abt Associates, 2012).

[b] An average of the 5-year costs estimated by Wittels et al. (1990) and Russell et al. (1998).

[c] From Cropper and Krupnick (1990), using a 3% discount rate for illustration.

5.6.6 Hospital Admissions and Emergency Department Valuation

In the absence of estimates of societal WTP to avoid hospital visits/admissions for specific illnesses, we derive COI estimates for use in the benefits analysis. The International

Classification of Diseases (WHO, 1977) code-specific COI estimates used in this analysis consist of estimated hospital charges and the estimated opportunity cost of time spent in the hospital (based on the average length of a hospital stay for the illness). We based all estimates of hospital charges and length of stays on statistics provided by the Agency for Healthcare Research and Quality's Healthcare Utilization Project National Inpatient Sample (NIS) database (AHRQ, 2007). We estimated the opportunity cost of a day spent in the hospital as the value of the lost daily wage, regardless of whether the hospitalized individual is in the workforce. To estimate the lost daily wage, we divided the median weekly wage reported by the 2007 American Community Survey (ACS) by five and deflated the result to year 2010$ using the CPI-U "all items" (Abt Associates, 2012). The resulting national average lost daily wage is $148. The total cost-of-illness estimate for an ICD code-specific hospital stay lasting *n* days, then, was the mean hospital charge plus $148 multiplied by *n.* In general, the mean length of stay has decreased since the 2000 database used in previous version of BenMAP while the mean hospital charge has increased. We provide the rounded unit values in 2010$ for the COI functions used in this analysis in Table 5-13.

Table 5-13. Unit Values for Hospital Admissions

End Point	ICD Codes	Age Range min.	Age Range max.	Mean Hospital Charge (2010$)	Mean Length of Stay (days)	Total Cost of Illness (unit value in 2010$)
HA, Chronic Lung Disease	490–496	18	64	$19,000	3.9	$21,000
HA, Asthma	493	0	64	$14,000	3.0	$16,000
HA, All Cardiovascular	390–429	18	64	$40,000	4.1	$42,000
HA, All Cardiovascular	390–429	65	99	$37,000	4.9	$41,000
HA, All Respiratory	460–519	65	99	$31,000	6.1	$36,000

* All estimates rounded to two significant digits. Unrounded estimates in 2000$ are available in Appendix J of the BenMAP user manual (Abt Associates, 2012).

To value asthma emergency department visits, we used a simple average of two estimates from the health economics literature. The first estimate comes from Smith et al. (1997), who reported approximately 1.2 million asthma-related emergency department visits in 1987, at a total cost of $186 million (1987$). The average cost per visit that year was $155; in 2010$, that cost was $464 (using the CPI-U for medical care to adjust to 2010$). The second estimate comes from Stanford et al. (1999), who reported the cost of an average asthma-related emergency department visit at $335, based on 1996–1997 data. A simple average of the two estimates yields a unit value of $388.

5.6.7 Minor Restricted Activity Days Valuation

No studies are reported to have estimated WTP to avoid a minor restricted activity day. However, Neumann et al. (1994) derived an estimate of willingness to pay to avoid a minor *respiratory* restricted activity day, using estimates from Tolley et al. (1986) of WTP for avoiding a combination of coughing, throat congestion and sinusitis. This estimate of WTP to avoid a minor respiratory restricted activity day is $38 (1990$), or about $68 (2010$). Although Ostro and Rothschild (1989) statistically linked ozone and minor restricted activity days, it is likely that most MRADs associated with ozone and $PM_{2.5}$ exposure are, in fact, minor *respiratory* restricted activity days. For the purpose of valuing this health endpoint, we used the estimate of mean WTP to avoid a minor respiratory restricted activity day.

5.6.8 Growth in WTP Reflecting National Income Growth Over Time

Our analysis accounts for expected growth in real income over time. This is a distinct concept from inflation and currency year. Economic theory argues that WTP for most goods (such as environmental protection) will increase if real incomes increase. There is substantial empirical evidence that the income elasticity[19] of WTP for health risk reductions is positive, although there is uncertainty about its exact value. Thus, as real income increases, the WTP for environmental improvements also increases. Although many analyses assume that the income elasticity of WTP is unit elastic (i.e., a 10% higher real income level implies a 10% higher WTP to reduce risk changes), empirical evidence suggests that income elasticity is substantially less than one and thus relatively inelastic. As real income rises, the WTP value also rises but at a slower rate than real income.

The effects of real income changes on WTP estimates can influence benefits estimates in two different ways: through real (national average) income growth between the year a WTP study was conducted and the year for which benefits are estimated, and through differences in income between study populations and the affected populations at a particular time. The SAB-EEAC advised the EPA to adjust WTP for increases in real income over time but not to adjust WTP to account for cross-sectional income differences "because of the sensitivity of making such distinctions, and because of insufficient evidence available at present" (U.S. EPA-SAB, 2000). An advisory by another committee associated with the SAB, the Advisory Council on Clean Air Compliance Analysis (SAB-Council), has provided conflicting advice. While agreeing with "the general principle that the willingness to pay to reduce mortality risks is likely to increase with growth in real income" and that "[t]he same increase should be assumed for the

[19] Income elasticity is a common economic measure equal to the percentage change in WTP for a 1% change in income.

WTP for serious nonfatal health effects," they note that "given the limitations and uncertainties in the available empirical evidence, the Council does not support the use of the proposed adjustments for aggregate income growth as part of the primary analysis" (U.S. EPA-SAB, 2004b). Until these conflicting advisories have been reconciled, the EPA will continue to adjust valuation estimates to reflect income growth using the methods described below, while providing sensitivity analyses for alternative income growth adjustment factors.

Based on a review of the available income elasticity literature, we adjusted the valuation of human health benefits upward to account for projected growth in real U.S. income. Faced with a dearth of estimates of income elasticities derived from time-series studies, we applied estimates derived from cross-sectional studies in our analysis. Details of the procedure can be found in Kleckner and Neumann (1999). We note that the literature has evolved since the publication of this memo and that an array of newer studies identifying potentially suitable income elasticity estimates are available (IEc, 2012). The EPA anticipates seeking an SAB review of these studies, and its approach to adjusting WTP estimates to account for changes in personal income, in 2013. As such, these newer studies have not yet been incorporated into the benefits analysis. An abbreviated description of the procedure we used to account for WTP for real income growth between 1990 and 2020 is presented below.

Reported income elasticities suggest that the severity of a health effect is a primary determinant of the strength of the relationship between changes in real income and WTP. As such, we use different elasticity estimates to adjust the WTP for minor health effects, severe and chronic health effects, and premature mortality. Note that because of the variety of empirical sources used in deriving the income elasticities, there may appear to be inconsistencies in the magnitudes of the income elasticities relative to the severity of the effects (*a priori* one might expect that more severe outcomes would show less income elasticity of WTP). We have not imposed any additional restrictions on the empirical estimates of income elasticity. One explanation for the seeming inconsistency is the difference in timing of conditions. WTP for minor illnesses is often expressed as a short term payment to avoid a single episode. WTP for major illnesses and mortality risk reductions are based on longer term measures of payment (such as wages or annual income). Economic theory suggests that relationships become more elastic as the length of time grows, reflecting the ability to adjust spending over a longer time period (U.S. EPA, 2010e, p.A-9). Based on this theory, it would be expected that WTP for reducing long term risks would be more elastic than WTP for reducing short term risks. The relative magnitude of the income elasticity of WTP for visibility compared with those for health effects suggests that visibility is not as much of a necessity as health, thus, WTP is more elastic with respect to income. The elasticity values used to adjust estimates of benefits in 2020 are presented in Table 5-14.[20]

Table 5-14. Elasticity Values Used to Account for Projected Real Income Growth[a]

Benefit Category	Central Elasticity Estimate
Minor Health Effect	0.14
Severe and Chronic Health Effects	0.45
Premature Mortality	0.40

[a] Derivation of estimates can be found in Kleckner and Neumann (1999). COI estimates are not adjusted for income growth.

In addition to elasticity estimates, projections of real gross domestic product (GDP) and populations from 1990 to 2020 are needed to adjust benefits to reflect real per capita income growth. For consistency with the emissions and benefits modeling, we used national population estimates for the years 1990 to 1999 based on U.S. Census Bureau estimates (Hollman, Mulder, and Kallan, 2000). These population estimates are based on application of a cohort-component model applied to 1990 U.S. Census data projections (U.S. Bureau of Census, 2000). For the years between 2000 and 2020, we applied growth rates based on the U.S. Census Bureau projections to the U.S. Census estimate of national population in 2000. We used projections of real GDP provided in Kleckner and Neumann (1999) for the years 1990 to 2010.[21] We used projections of real GDP (in chained 1996 dollars) provided by Standard and Poor's (2000) for the years 2010 to 2020.[22]

Using the method outlined in Kleckner and Neumann (1999) and the population and income data described above, we calculated WTP adjustment factors for each of the elasticity estimates listed in Table 5-15. Benefits for each of the categories (minor health effects, severe and chronic health effects, premature mortality, and visibility) are adjusted by multiplying the unadjusted benefits by the appropriate adjustment factor. For premature mortality, we applied the income adjustment factor specific to the analysis year, but we do not adjust for income

[20] We expect that the WTP for improved visibility in Class 1 areas would also increase with growth in real income (see Chapter 6).

[21] U.S. Bureau of Economic Analysis, *Table 2A—Real Gross Domestic Product* (1997) and U.S. Bureau of Economic Analysis, *The Economic and Budget Outlook: An Update*, Table 4—*Economic Projections for Calendar Years 1997 Through 2007 (1997)*. Note that projections for 2007 to 2010 are based on average GDP growth rates between 1999 and 2007.

[22] In previous analyses, we used the Standard and Poor's projections of GDP directly. This led to an apparent discontinuity in the adjustment factors between 2010 and 2011. We refined the method by applying the relative growth rates for GDP derived from the Standard and Poor's projections to the 2010 projected GDP based on the Bureau of Economic Analysis projections.

growth over the 20-year cessation lag. Our approach could underestimate the benefits for the later years of the lag.

There is some uncertainty regarding the total costs of illness in the future. Specifically, the nature of medical treatment is changing, including a shift towards more outpatient treatment. Although we adjust the COI estimates for inflation, we do not have data to project COI estimates for the cost of treatment in the future or income growth over time, which leads to an inherent though unavoidable inconsistency between COI- and WTP-based estimates This approach may underpredict benefits in future years because it is likely that increases in real U.S. income would also result in increased COI (due, for example, to increases in wages paid to medical workers) and increased cost of work loss days and lost worker productivity (reflecting that if worker incomes are higher, the losses resulting from reduced worker production would also be higher). In addition, cost-of-illness estimates do not include sequelae costs or pain and suffering, the value of which would likely increase in the future. To the extent that costs would be expected to increase over time, this increase may be partially offset by advancement in medical technology that improves the effectiveness of treatment at lower costs. For these reasons, we believe that the cost-of-illness estimates in this RIA may underestimate (on net) the total economic value of avoided health impacts.

Table 5-15. Adjustment Factors Used to Account for Projected Real Income Growth[a]

Benefit Category	2020
Minor Health Effect	1.07
Severe and Chronic Health Effects	1.22
Premature Mortality	1.20

[a] Based on elasticity values reported in Table 5-3, U.S. Census population projections, and projections of real GDP per capita.

5.7 Benefits Results

5.7.1 Benefits of the Revised and Alternative Annual Primary $PM_{2.5}$ Standards

Applying the impact and valuation functions described previously in this chapter to the estimated changes in $PM_{2.5}$ yields estimates of the changes in physical damages (e.g., premature mortalities, cases of acute bronchitis and hospital admissions) and the associated monetary values for those changes. Not all known PM health effects could be quantified or monetized. The monetized value of these unquantified effects is represented by adding an unknown “B” to the aggregate total. The estimate of total monetized health benefits is thus equal to the subset of monetized PM-related health benefits plus B, the sum of the nonmonetized health benefits and welfare co-benefits; this B represents both uncertainty and a bias in this analysis, as it reflects those benefits categories that we are unable to monetize in this analysis.

Table 5-16 shows the population-weighted air quality change for the alternative standards averaged across the continental U.S. Tables 5-17 through 5-22 present the benefits results for the annual $PM_{2.5}$ standards. These benefits are relative to a 2020 analytical baseline reflecting attainment nationwide of the current primary $PM_{2.5}$ standards (i.e., 15/35) that includes promulgated national regulations and illustrative emission controls to simulate attainment with 15/35 as well as an adjustment to NO_x emissions to reflect expected reductions in mobile NO_x emissions between 2020 and 2025.[23] Figure 5-3 graphically displays the total monetized benefits of the revised annual primary standard of 12 $\mu g/m^3$, using alternative concentration-response functions at discount rates of 3% and 7%. Figure 5-4 graphically displays the cumulative distribution of total monetized benefits using the 2 epidemiology-derived and the 12 expert-derived relationships between $PM_{2.5}$ and mortality for the revised standard, which provides the full range of uncertainty within and across the expert-derived relationships.

Table 5-16. Population-Weighted Air Quality Change for the Revised and Alternative Annual Primary $PM_{2.5}$ Standards Relative to Analytical Baseline

Standard	Population-Weighted Air Quality Change
13 $\mu g/m^3$	0.014 $\mu g/m^3$
12 $\mu g/m^3$	0.043 $\mu g/m^3$
11 $\mu g/m^3$	0.207 $\mu g/m^3$

[23] The estimates in this chapter reflect incremental emissions reductions from an analytical baseline that gives “an adjustment” to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

Table 5-17. Emission Reductions in Illustrative Emission Reduction Strategies for the Revised and Alternative Annual Primary $PM_{2.5}$ Standards, by Pollutant and Region in 2020 (tons)[a]

	13 µg/m^3	**12** µg/m^3	**11** µg/m^3
Directly emitted $PM_{2.5}$			
East	0	0	8,200
West	0	0	160
CA	730	4,000	10,600
SO_2			
East	0	0	21,000
West	0	0	43
CA	0	0	0
NOx			
East	0	0	9
West	0	0	0
CA	0	0	0

[a] See Chapter 4 for more information on the illustrative emission reduction strategies. The emissions in this table reflect both known and unknown controls.

Table 5-18. Estimated number of Avoided $PM_{2.5}$ Health Impacts for the Revised and Alternative Annual Primary $PM_{2.5}$ Standards (Incremental to the Analytical Baseline)[a]

Health Effect	Revised and Alterative Annual Standards (95th percentile confidence interval) 13 µg/m³	12 µg/m³	11 µg/m³
Avoided Mortality			
Krewski et al. (2009) (adult mortality age 30+)	140 (100--190)	460 (320--590)	1,500 (1,000--1,900)
Lepeule et al. (2012) (adult mortality age 25+)	330 (180--480)	1,000 (560--1,500)	3,300 (1,800--4,800)
Woodruff et al. (1997) (infant mortality)	0 (0--1)	1 (1--2)	4 (2--6)
Avoided Morbidity			
Non-fatal heart attacks			
Peters et al. (2001) (age >18)	160 (49--260)	480 (150--800)	1,600 (480--2,600)
Pooled estimate of 4 studies (age >18)	17 (8--41)	52 (24--130)	170 (78--410)
Hospital admissions—respiratory (all ages)[b]	31 (-9--58)	110 (-30--200)	380 (-100--720)
Hospital admissions—cardiovascular (age > 18)	43 (20--76)	140 (66--240)	480 (230--0,840)
Emergency department visits for asthma (all ages)[b]	67 (-22--140)	230 (-74--470)	810 (-260--1,600)
Acute bronchitis (ages 8–12)[b]	280 (-36--580)	870 (-110--1,800)	2,700 (-350--5,500)
Lower respiratory symptoms (ages 7–14)	3,500 (1500--5500)	11,000 (4,900--17,000)	34,000 (15,000--53,000)
Upper respiratory symptoms (asthmatics ages 9–11)	5,100 (1300--8900)	16,000 (4,100--28,000)	49,000 (12,000--86,000)
Asthma exacerbation (asthmatics ages 6–18)	13,000 (270--81000)	40,000 (850--250,000)	120,000 (2,600--770,000)
Lost work days (ages 18–65)	22,000 (19000--25000)	71,000 (61,000--81,000)	230,000 (190,000--260,000)
Minor restricted-activity days (ages 18–65)	130,000 (110,000--150,000)	420,000 (350,000--490,000)	1,300,000 (1,100,000--1,600,000)

[a] All incidence estimates are rounded to whole numbers with a maximum of two significant digits. These estimates reflect incremental emissions reductions from an analytical baseline that gives "an adjustment" to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025. Additional health endpoints, such as cardiovascular emergency department visits, are only quantified in a sensitivity analysis in Table 5.A-6 because we do not yet have a valuation estimate for this endpoint.

[b] The negative estimates at the 5th percentile confidence estimates for these morbidity endpoints reflect the statistical power of the study used to calculate these health impacts. These results do not suggest that reducing air pollution results in additional health impacts.

Table 5-19. Monetized $PM_{2.5}$ Health Benefits for the Revised and Alternative Annual Primary $PM_{2.5}$ Standards (Incremental to Analytical Baseline) (Millions of 2006$, 3% discount rate)[a]

	Revised and Annual Standards (95th percentile confidence interval)		
Health Effect	**13 µg/m³**	**12 µg/m³**	**11 µg/m³**
Avoided Mortality[b]			
Krewski et al. (2009) (adult mortality age 30+)	$1,300 ($120--$3,500)	$4,000 ($370--$11,000)	$13,000 ($1,200--$35,000)
Lepeule et al. (2012) (adult mortality age 25+)	$2,900 ($250--$8,100)	$9,000 ($800--$26,000)	$29,000 ($2,600--$82,000)
Woodruff et al. (1997) (infant mortality)	$3.4 ($0.29--$10)	$11 ($0.91--$32)	$35 ($3.0--$100)
Avoided Morbidity			
Non-fatal heart attacks			
Peters et al. (2001) (age >18)	$18 ($3.0--$46)	$55 ($9.1--$140)	$180 ($31--$460)
Pooled estimate of 4 studies (age >18)	$2.0 ($0.43--$6.8)	$6.0 ($1.3--$21)	$20.0 ($4.4--$68)
Hospital admissions—respiratory (all ages)[c]	$0.86 (-$0.22--$1.6)	$3.0 (-$0.8--$5.5)	$11 (-$2.7--$20)
Hospital admissions—cardiovascular (age > 18)	$1.70 ($0.85--$2.8)	$5.3 ($2.7--$9.2)	$18 ($10--$32)
Emergency department visits for asthma (all ages)	$0.03 (-$0.0052--$0.061)	$0.10 (-$0.018--$0.21)	$0.34 (-$0.063--$0.73)
Acute bronchitis (ages 8–12)[c]	$0.13 (-$0.0060--$0.37)	$0.42 (-$0.019--$1.2)	$1.30 (-$0.059--$3.5)
Lower respiratory symptoms (ages 7–14)	$0.08 ($0.025--$0.10)	$0.24 ($0.078--$0.47)	$0.71 ($0.24--$1.4)
Upper respiratory symptoms (asthmatics ages 9–11)	$0.17 ($0.038--$0.42)	$0.54 ($0.12--$1.30)	$1.6 ($0.36--$4.0)
Asthma exacerbation (asthmatics ages 6–18)	$0.7 ($0.027--$5.2)	$2.30 ($0.085--$16)	$7.0 ($0.26--$49)
Lost work days (ages 18–65)	$3.3 ($2.90--$3.70)	$11 ($9.4--$12)	$35 ($30--$39)
Minor restricted-activity days (ages 18–65)	$8.8 ($4.70--$13)	$29 ($15--$43)	$91 ($48--$140)

[a] All estimates are rounded to two significant digits. Estimates do not include unquantified health benefits noted in Table 5-2 or Section 5.6.5 or welfare co-benefits noted in Chapter 6. These estimates reflect incremental emissions reductions from an analytical baseline that gives “an adjustment” to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

[b] The reduction in premature fatalities each year accounts for over 98% of total monetized benefits in this analysis. Mortality risk valuation assumes discounting over the SAB-recommended 20-year segmented lag structure.

[c] The negative estimates at the 5th percentile confidence estimates for this morbidity endpoint reflects the statistical power of the study used to calculate these health impacts. These results do not suggest that reducing air pollution results in additional health impacts.

Table 5-20. Monetized $PM_{2.5}$ Health Benefits for the Revised and Alternative Annual Primary $PM_{2.5}$ Standards (Incremental to Analytical Baseline) (Millions of 2006$, 7% discount rate)[a]

Health Effect	Revised and Alterative Annual Standards (95th percentile confidence interval)		
	13 µg/m³	12 µg/m³	11 µg/m³
Avoided Mortality[b]			
Krewski et al. (2009) (adult mortality age 30+)	$1,100 ($110--$3,100)	$3,600 ($330--$9,800)	$11,000 ($1,100--$31,000)
Lepeule et al. (2012) (adult mortality age 25+)	$2,600 ($230--$7,300)	$8,100 ($720--$23,000)	$26,000 ($2,300--$74,000)
Woodruff et al. (1997) (infant mortality)	$3.4 ($0.29--$100)	$11 ($0.91--$32)	$35 ($3.0--$100)
Avoided Morbidity			
Non-fatal heart attacks			
Peters et al. (2001) (age >18)	$18 ($2.8--$46)	$54 ($8.4--$140)	$180 ($28--$450)
Pooled estimate of 4 studies (age >18)	$1.9 ($0.40--$6.7)	$5.9 ($1.2--$20)	$20 ($4.1--$67)
Hospital admissions—respiratory (all ages)[c]	$0.86 (-$0.22--$1.6)	$3.0 (-$0.8--$5.5)	$11 (-$2.7--$20)
Hospital admissions—cardiovascular (age > 18)	$1.70 ($0.85--$2.8)	$5.3 ($2.7--$9.2)	$18 ($10--$32)
Emergency department visits for asthma (all ages)	$0.03 (-$0.0052--$0.061)	$0.10 (-$0.018--$0.21)	$0.34 (-$0.063--$0.73)
Acute bronchitis (ages 8–12)[c]	$0.13 (-$0.0060--$0.37)	$0.42 (-$0.019--$1.2)	$1.3 (-$0.059--$3.5)
Lower respiratory symptoms (ages 7–14)	$0.08 ($0.025--$0.10)	$0.24 ($0.078--$0.47)	$0.71 ($0.24--$1.4)
Upper respiratory symptoms (asthmatics ages 9–11)	$0.17 ($0.038--$0.42)	$0.54 ($0.12--$1.30)	$1.6 ($0.36--$4.0)
Asthma exacerbation (asthmatics ages 6–18)	$0.7 ($0.027--$5.2)	$2.30 ($0.085--$16)	$7.0 ($0.26--$49)
Lost work days (ages 18–65)	$3.3 ($2.90--$3.70)	$11 ($9.4--$12)	$35 ($30--$39)
Minor restricted-activity days (ages 18–65)	$8.8 ($4.70--$13)	$29 ($15--$43)	$91 ($48--$140)

[a] All estimates are rounded to two significant digits. Estimates do not include unquantified health benefits noted in Table 5-2 or Section 5.6.5 or welfare co-benefits noted in Chapter 6. These estimates reflect incremental emissions reductions from an analytical baseline that gives “an adjustment” to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

[b] The reduction in premature fatalities each year accounts for over 98% of total monetized benefits in this analysis. Mortality risk valuation assumes discounting over the SAB-recommended 20-year segmented lag structure.

[c] The negative estimates at the 5th percentile confidence estimates for this morbidity endpoint reflects the statistical power of the study used to calculate these health impacts. These results do not suggest that reducing air pollution results in additional health impacts.

Table 5-21. Total Estimated Monetized Benefits of the for Revised and Alternative Annual Primary $PM_{2.5}$ Standards (Incremental to the Analytical Baseline) (billions of 2006$) [a,b]

Benefits Estimate	13 µg/m³	12 µg/m³	11 µg/m
Economic value of avoided $PM_{2.5}$-related morbidities and premature deaths using $PM_{2.5}$ mortality estimate from Krewski et al. (2009)			
3% discount rate	$1.3 + B	$4.0 +B	$13 + B
7% discount rate	$1.2 + B	$3.6 +B	$12 + B
Economic value of avoided $PM_{2.5}$-related morbidities and premature deaths using $PM_{2.5}$ mortality estimate from Lepeule et al. (2012)			
3% discount rate	$2.9 + B	$9.1 +B	$29 + B
7% discount rate	$2.6 + B	$8.2 +B	$26 + B

[a] Rounded to two significant figures. The reduction in premature fatalities each year accounts for over 98% of total monetized benefits in this analysis. Mortality risk valuation assumes discounting over the SAB-recommended 20-year segmented lag structure. Not all possible benefits are quantified and monetized in this analysis. B is the sum of all unquantified health and welfare co-benefits. Data limitations prevented us from quantifying these endpoints, and as such, these benefits are inherently more uncertain than those benefits that we were able to quantify.

[b] These estimates reflect incremental emissions reductions from an analytical baseline that gives "an adjustment" to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

Table 5-22. Regional Breakdown of Monetized Benefits Results

	Revised and Alterative Annual Standards		
Region	**13 µg/m³**	**12 µg/m³**	**11 µg/m³**
East[a]	0%	0%	23%
California[b]	100%	100%	77%
Rest of West	0%	0%	<1%

[a] Includes Texas and those states to the north and east. Several recent rules such as MATS and CSAPR will have substantially reduced $PM_{2.5}$ levels by 2020 in the East, thus few additional controls would be needed to reach 12 µg/m³.

[b] For 12 and 13 µg/m³, all of the benefits occur in California because this highly populated area is where the most air quality improvement beyond the analytical baseline is needed to reach these levels. These estimates reflect incremental emissions reductions from an analytical baseline that gives "an adjustment" to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

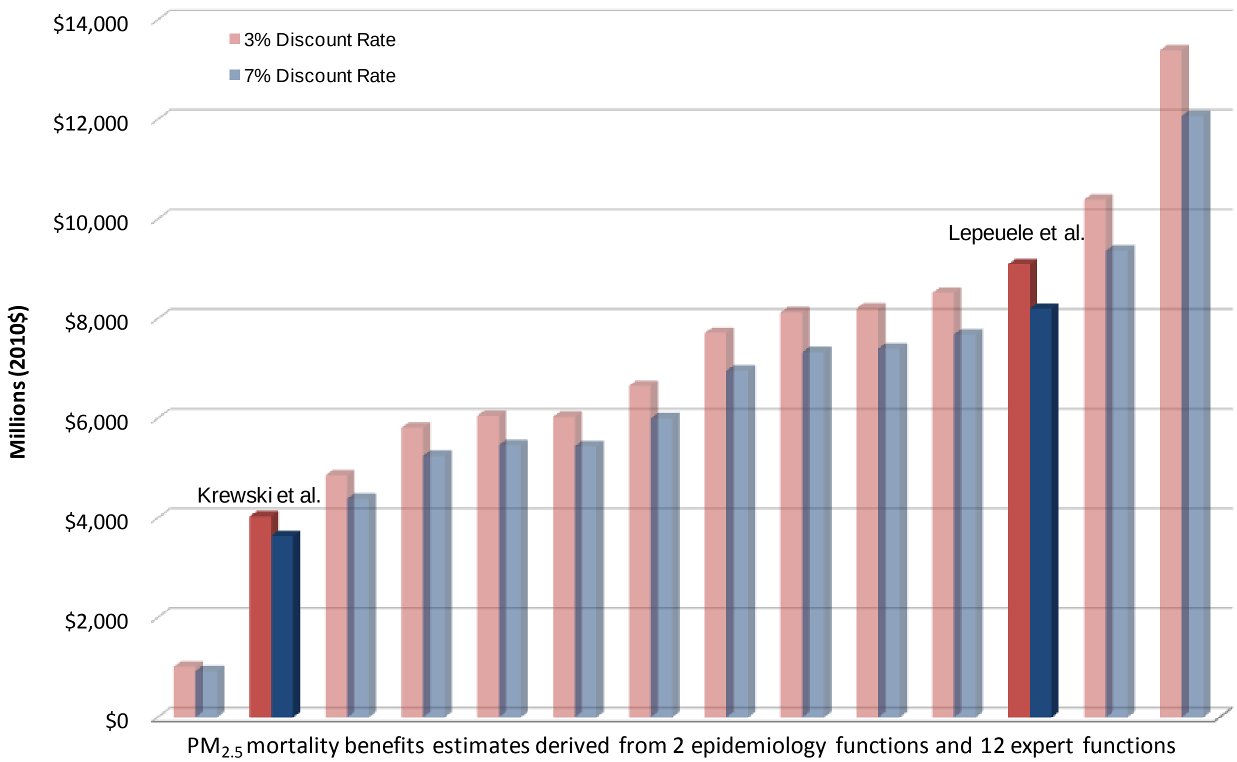


Figure 5-3. Estimated $PM_{2.5}$-Related Premature Mortalities Avoided According to Epidemiology or Expert-Derived $PM_{2.5}$ Mortality Risk Estimate for 12 µg/m^3 in 2020[a]

[a] These estimates reflect incremental emissions reductions from an analytical baseline that gives “an adjustment” to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

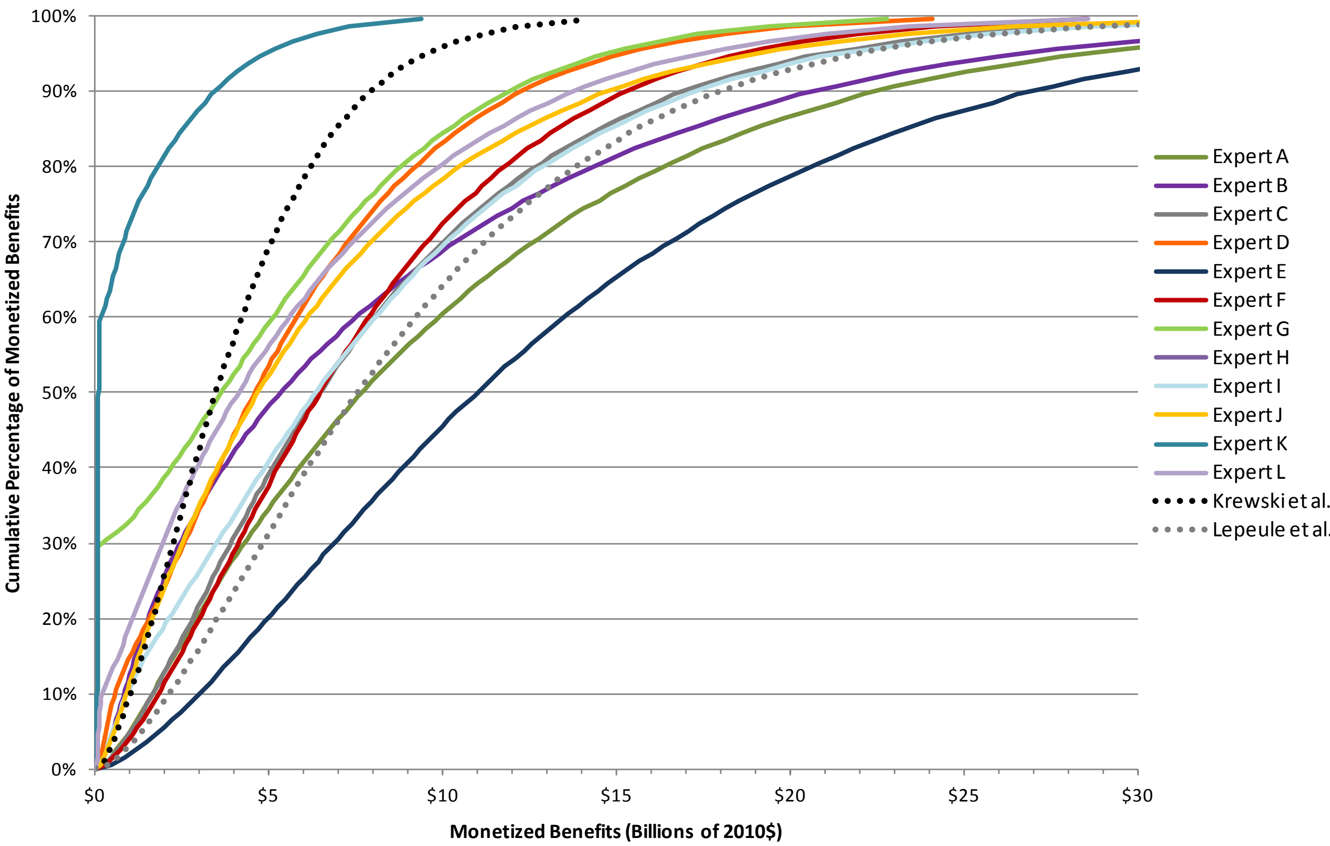


Figure 5-4. Total Monetized Benefits Using 2 Epidemiology-Derived and 12-Expert Derived Relationships Between $PM_{2.5}$ and Premature Mortality for 12 µg/m^3 in 2020[a]

[a] These estimates reflect incremental emissions reductions from an analytical baseline that gives "an adjustment" to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

5.7.2 Uncertainty in Benefits Results

Mortality benefits account for 98% of total monetized benefits, in part because we are unable to quantify most of the non-health benefits. The next largest benefit is for reductions in chronic illness (nonfatal heart attacks), although this value is more than an order of magnitude lower than for premature mortality. Hospital admissions for respiratory and cardiovascular causes, MRADs and work loss days account for the majority of the remaining benefits. The remaining categories each account for a small percentage of total benefit; however, they represent a large number of avoided incidences affecting many individuals. A comparison of the incidence table to the monetary benefits table reveals that there is not always a close correspondence between the number of incidences avoided for a given endpoint and the monetary value associated with that endpoint. For example, we estimate almost 1,000 times more work loss days would be avoided than premature mortalities, yet work loss days account for only a very small fraction of total monetized benefits. This reflects the fact that many of the less severe health effects, while more common, are valued at a lower level than the more severe health effects. Also, some effects, such as hospital admissions, are valued using a proxy measure of WTP. As such, the true value of these effects may be higher than that reported in the tables above.

$PM_{2.5}$ mortality benefits represent a substantial proportion of total monetized benefits (over 98% in this analysis), and these estimates have the following key assumptions and uncertainties.

1. We assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This is an important assumption, because $PM_{2.5}$ produced varies considerably in composition across sources, but the scientific evidence is not yet sufficient to allow differential effects estimates by particle type. The PM ISA, which was twice reviewed by SAB-CASAC, concluded that "many constituents of $PM_{2.5}$ can be linked with multiple health effects, and the evidence is not yet sufficient to allow differentiation of those constituents or sources that are more closely related to specific outcomes" (U.S. EPA, 2009b).

2. We assume that the health impact function for fine particles is log-linear without a threshold in this analysis. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5}$, including both areas that do not meet the fine particle standard and those areas that are in attainment, down to the lowest modeled concentrations.

3. We assume that there is a "cessation" lag between the change in PM exposures and the total realization of changes in mortality effects. Specifically, we assume that some of the incidences of premature mortality related to $PM_{2.5}$ exposures occur in a distributed fashion over the 20 years following exposure based on the advice of the SAB-HES (U.S. EPA-SAB, 2004c), which affects the valuation of mortality benefits at different discount rates.

Given that reductions in premature mortality dominate the size of the overall monetized benefits, more focus on uncertainty in mortality-related benefits gives us greater confidence in our uncertainty characterization surrounding total benefits.

5.7.3 Estimated Life Years Gained and Reduction in the Percentage of Deaths Attributable to $PM_{2.5}$

In their 2008 review of the EPA's approach to estimating ozone-related mortality benefits, NRC indicated, "EPA should consider placing greater emphasis on reporting decreases in age-specific death rates in the relevant population and develop models for consistent calculation of changes in life expectancy and changes in number of deaths at all ages" (NRC, 2008). In addition, NRC noted in an earlier report that "[f]rom a public-health perspective, life-years lost might be more relevant than annual number of mortality cases" (NRC, 2002). This advice is consistent with that of the SAB-HES, which agreed that "…the interpretation of mortality risk results is enhanced if estimates of lost life-years can be made" (U.S. EPA-SAB, 2004a). To address these recommendations, we use simplifying assumptions to estimate the number of life years that might be gained. We also estimate the reduction in the percentage of deaths attributed to $PM_{2.5}$ resulting from the illustrative emission reduction strategies to reach the revised annual primary standard. The EPA included similar estimates of life years gained in a previous assessment of $PM_{2.5}$ benefits (U.S. EPA, 2006a, 2011a), the latter of which was peer reviewed by the SAB-HES (U.S. EPA-SAB, 2010a).

Changes in life years and changes in life expectancy at birth are frequently conflated, thus it is important to distinguish these two very different metrics at the outset. Life expectancy varies by age. CDC defines life expectancy as the "average number of years of life remaining for persons who have attained a given age" (CDC, 2011). In other words, changes in life expectancy refer to an average change for the entire population, and refer to the future. Over the past 50 years, average life expectancy at birth in the U.S. has increased by 8.4 years (CDC, 2001). For example, life expectancy at birth was estimated in 2007 to be 77.9 years for an average person born in the U.S., but for people surviving to age 60, estimated life expectancy is 82.5 years (i.e., 4.6 years more than life expectancy at birth) (CDC, 2011). Life years, on the other hand, measure the amount of time that an individual loses if they die before the age of their life expectancy. Life years refer to individuals, and refer to the past, e.g., when the individual has already died. If a 60-year old individual dies, we estimate that this individual would lose about 22.5 years of life (i.e., the average population life expectancy for an individual of this age minus this person's age at death).

5.7.3.1 Estimated Life Years Gained

For estimating the potential life years gained by reducing exposure to $PM_{2.5}$ in the U.S. adult population, we use the same general approach as Hubbell (2006) and Fann et al. (2012a). We have not estimated the change in average life expectancy at birth in this RIA. Hubbell (2006) estimated that reducing exposure to $PM_{2.5}$ from air pollution regulations may result in an average gain of 15 years of life for those adults prematurely dying from $PM_{2.5}$ exposure. In contrast, Pope et al. (2009) estimated changes in average life expectancy at birth over a twenty year period, suggesting that reducing exposure to air pollution may increase average life expectancy at birth by approximately 7 months, which was 15% of the overall increase in life expectancy at birth from 1980 through 2000. These results are not necessarily inconsistent because they are reporting different metrics. Because life expectancy is an average of the entire population (including both those whose deaths would likely be attributed to PM exposure as well as those whose deaths would not), average life expectancy changes associated with PM exposure would be expected to always be significantly smaller than the average number of life years lost by an individual who is projected to die prematurely from PM exposure.

To estimate the potential distribution of life years gained for population subgroups defined by the age range at which their reduction in $PM_{2.5}$ exposure is modeled to occur we use standard life tables available from the CDC (2003) and the following formula:

$$Total\ Life\ Years = \sum_{i=1}^{n} LE_i \times M_i \tag{5.2}$$

where LE_i is the average remaining life expectancy for age interval i, M_i is the estimated change in number of deaths in age interval i, and n is the number of age intervals.

To get M_i (the estimate the number of avoided premature deaths attributed to changes in $PM_{2.5}$ exposure in 2020), we use a health impact function that incorporates risk coefficients estimated for the adult population in the U.S. and age-specific mortality rates. That is, we use risk coefficients that do not vary by age, but use baseline mortality rates that do. Because mortality rates for younger populations are much lower than mortality rates for older populations, most but not all, of the avoided deaths tend to be in older populations. Table 5-23 summarizes the modeled number of life years gained by reducing $PM_{2.5}$ exposure to 12 $\mu g/m^3$ in 2020. We then calculated the average number of life years gained per avoided premature mortality. Figure 5-5 shows the potential life years gained as a result of meeting a primary standard of 12 $\mu g/m^3$ in 2020, partitioned by the age when exposure reduction occurred, not necessarily age at death.

By comparing the projected age distribution of the avoided premature deaths with the age distribution of life years gained, we observed that about half of the deaths occur in populations age 75–99, but half of the life years would occur in populations younger than 65. This is because the younger populations have the potential to lose more life years per death than older populations based on changes in $PM_{2.5}$ exposure in 2020. We estimate that the average individual who would otherwise have died prematurely from PM exposure would gain 16 additional years of life. However, this approach does not account for whether or not people who are older are more likely to be susceptible to the health effects of PM or whether that susceptibility was in and of itself caused by PM exposure (for a more complete discussion of this issue, see Kunzli et al., 2001).

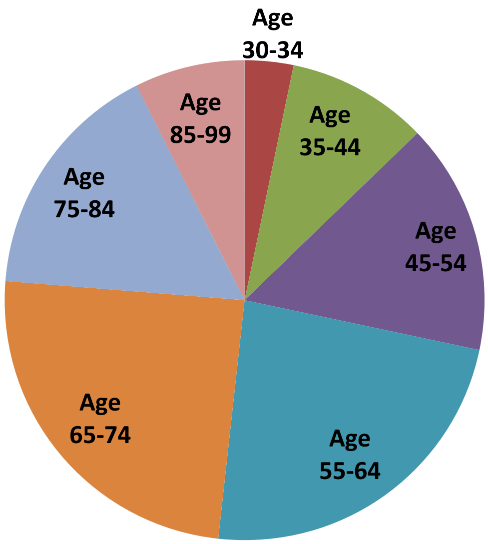
Table 5-23. Sum of Life Years Gained by Age Range Attributed to the Revised Annual Primary $PM_{2.5}$ Standard of 12 μg/m^3 in 2020[a,b]

Age Range[b]	Krewski et al. (2009) Risk Coefficient [c]	Lepeule et al. (2012) Risk Coefficient
25–29	—	610
30–34	210	470
35–44	550	1,200
45–54	1,000	2,300
55–64	1,600	3,500
65–74	1,700	3,900
75–84	1,300	2,900
85–99	560	1,300
Total life years gained	7,000	16,000
Average life years gained per individual	15.0	16.0

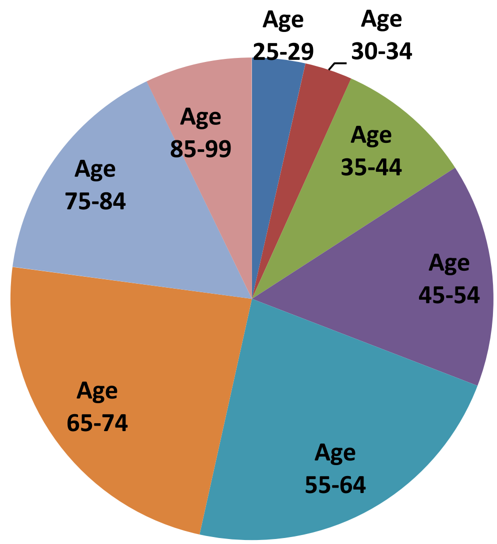
[a] Estimates rounded to two significant figures.

[b] Because we assume that there is a "cessation" lag between PM exposures and the reduction in the risk of premature death, there is uncertainty regarding the specific ages that people die relative to the change in exposure.

[c] The youngest age in the population cohort of this study is 30.



Calculated using Krewski et al. (2009) risk coefficient [b]



Calculated using Lepeule et al. (2012) risk coefficient

Figure 5-5. Estimated Life Years Gained as a Result of the Revised Annual Primary $PM_{2.5}$ Standard of 12 µg/m^3 in 2020, Partitioned by the Age When Exposure Reduction Occurred, Not Necessarily Age at Death[a]

[a] As shown in these charts, slightly more than half of the avoided premature deaths occur in populations age 75–99, but slightly more than half of the avoided life years occur in populations age <65 due to the fact that the younger populations would lose more life years per death than older populations. Results would be similar for other standard levels on a percentage basis. Because we assume that there is a "cessation" lag between PM exposures and the reduction in the risk of premature death, there is uncertainty regarding the specific ages that people die relative to the change in exposure.

[b] The youngest age in the population cohort of this study used to estimate $PM_{2.5}$ mortality incidence is 30.

5.7.3.2 Percent of PM-related Mortality Reduced

To estimate the percentage of all-cause mortality attributed to reduced $PM_{2.5}$ exposure in 2020 as a result of the illustrative emission reduction strategies, we use M_i from the equation above, dividing the number of excess deaths estimated for each alternative standard by the total number of deaths in each county. Table 5-24 shows the reduction in all-cause mortality attributed to reducing $PM_{2.5}$ exposure to the revised annual standard of 12 µg/m^3 in 2020. Figure 5-6 shows the percentage of avoided premature deaths attributed to meeting the revised primary annual standard of 12 µg/m^3 in 2020, partitioned by the age when exposure reduction occurred, not necessarily age at death.

Table 5-24. Estimated Reduction in the Percentage of All-Cause Mortality Attributed to the Revised Annual Primary $PM_{2.5}$ Standard of 12 µg/m^3 in 2020[a,b]

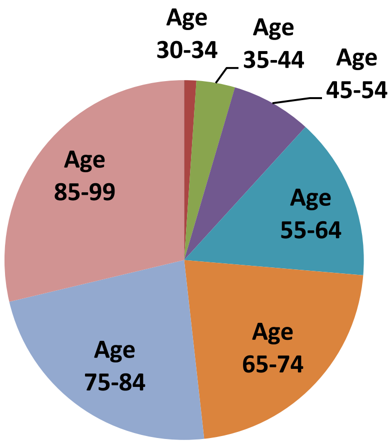
Age Range[b]	Krewski et al. (2009) Risk Coefficient[c]	Lepeule et al. (2012) Risk Coefficient
25–29	—	0.80%
30–34	0.35%	0.78%
35–44	0.32%	0.73%
45–54	0.32%	0.73%
55–64	0.33%	0.73%
65–74	0.33%	0.73%
75–84	0.32%	0.73%
85–99	0.29%	0.65%

[a] Rounded to two significant figures. Results would be similar for other standard levels on a percentage basis. These estimates reflect incremental emissions reductions from an analytical baseline that gives "an adjustment" to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

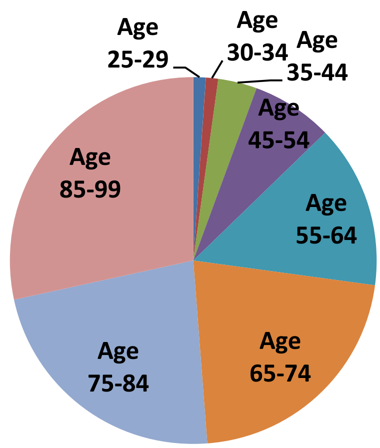
[b] Because we assume that there is a "cessation" lag between PM exposures and the reduction in the risk of premature death, there is uncertainty regarding the specific ages that people die relative to the change in exposure.

[c] The youngest age in the population cohort of this study is 30.

The relative distributions of the potential number of life years gained (Figure 5-5) and the estimated avoided mortalities (Figure 5-6) would be similar across alternative annual standards. Because reduction in PM exposure is not associated with an immediate improvement in chronic health conditions, we assume that there is a "cessation" lag between PM exposures and the reduction in the risk of premature death. There is uncertainty regarding the specific ages at which people "avoid" death relative to the change in exposure. While the structure of the lag is uncertain, some studies suggest that most of the premature deaths are avoided within the first 3 years after the change in exposure, while others are unable to identify conclusively a critical window of exposure (U.S. EPA, 2004c; Schwartz, 2008; Krewski et al. 2009). These studies did not examine whether the cessation lag was modified by either age at the time when exposure is reduced or the extent of cumulative lifetime exposure.



Calculated using Krewski et al. (2009) risk coefficient [b]



Calculated using Lepeule et al. (2012) risk coefficient.

Figure 5-6. Estimate of Avoided Premature Deaths Attributed to the Revised Annual Primary $PM_{2.5}$ Standard of 12 µg/m^3 in 2020, Partitioned by the Age When Exposure Reduction Occurred, Not Necessarily Age at Death[a]

[a] As shown in these charts, slightly more than half of the avoided premature deaths occur in populations age 75-99, but slightly more than half of the avoided life years occur in populations age <65 due to the fact that the younger populations would lose more life years per death than older populations. Results would be similar for other standard levels on a percentage basis. Because we assume that there is a "cessation" lag between PM exposures and the reduction in the risk of premature death, there is uncertainty regarding the specific ages that people die relative to the change in exposure.

[b] The youngest age in the population cohort of this study is 30.

5.7.4 Evaluation of Mortality Impacts Relative to Various Concentration Benchmarks

In this analysis, we estimate the number of avoided $PM_{2.5}$-related deaths occurring due to $PM_{2.5}$ reductions down to various $PM_{2.5}$ concentration benchmarks, including the Lowest Measured Level (LML) of each long-term $PM_{2.5}$ mortality study. This analysis is one of several sensitivities that the EPA has historically performed that characterize the uncertainty associated with the PM-mortality relationship and the economic value of reducing the risk of premature death (Roman et al., 2008; U.S. EPA, 2006a, 2011a; Mansfield, 2009).

Our review of the current body of scientific literature indicates that a log-linear no-threshold model provides the best estimate of PM-related long-term mortality. The PM ISA (U.S. EPA, 2009b), which was twice reviewed by the EPA's Clean Air Scientific Advisory Committee (U.S. EPA-SAB, 2009a, 2009b), concluded that the evidence supports the use of a no-threshold log-linear model while also recognizing potential uncertainty about the exact shape of the concentration-response function.[24] Consistent with this finding, we estimate benefits associated with the full range of $PM_{2.5}$ exposure in conjunction with sensitivity analyses to recognize the potential uncertainty at lower concentrations. Specifically, we incorporated a LML assessment, a method the EPA has employed in several recent RIAs (U.S. EPA, 2010g, 2011c, 2011d). In addition, we have incorporated an assessment using specific concentration benchmarks identified in the EPA's *Policy Assessment for Particulate Matter* (U.S. EPA, 2011b).

These two approaches summarize the distribution of avoided $PM_{2.5}$-related mortality impacts relative to baseline (i.e., pre-rule) annual mean $PM_{2.5}$ levels. The LML approach compares the percentage of avoided premature deaths estimated to occur above and below the minimum observed air quality level of each long-term cohort study we use to quantify PM. In the air quality benchmark approach, we summarize the impacts occurring at different points in the distribution of the air quality data used in these same epidemiology studies.

Our confidence in the estimated number of premature deaths avoided (but not in the existence of a causal relationship between PM and premature mortality) diminishes as we estimate these impacts in locations where $PM_{2.5}$ levels are below the LML. This interpretation is consistent with the *Policy Assessment* (U.S. EPA, 2011b) and advice from SAB-CASAC during their peer review (U.S. EPA-SAB, 2010d). As noted in the preamble to the final rule, the *Policy Assessment* (U.S. EPA, 2011b) concludes that the range from the 25th to the 10th percentile is a reasonable range of the air quality distribution below which we start to have appreciably less confidence in the magnitude of the associations observed in the epidemiological studies. In general, we are more confident in the magnitude of the risks we estimate from simulated $PM_{2.5}$ concentrations that coincide with the bulk of the observed PM concentrations in the epidemiological studies at are used to estimate the benefits. Likewise, we are less confident in the risk we estimate from simulated $PM_{2.5}$ concentrations that fall below the bulk of the observed data in these studies. However, there are uncertainties inherent in identifying any particular point at which our confidence in reported associations becomes appreciably less, and the scientific evidence provides no clear dividing line.

For these reasons, we consider the LML as well as one standard deviation below the mean[25] air quality levels when characterizing the distribution of mortality impacts. It is

---

[24] For a summary of the scientific review statements regarding the lack of a threshold in the $PM_{2.5}$-mortality relationship, see the Technical Support Document (TSD) entitled *Summary of Expert Opinions on the Existence of a Threshold in the Concentration-Response Function for $PM_{2.5}$-related Mortality* (U.S. EPA, 2010f).

[25] A range of one standard deviation around the mean represents approximately 68 percent of normally distributed data, and, below the mean falls between the 25th and 10th percentiles.

important to emphasize that “less confidence” does not mean “no confidence.” In addition, while we may have less confidence in the magnitude of the risk, we still have high confidence that $PM_{2.5}$ is causally associated with risk at those lower air quality concentrations. To clarify this concept, Figure 5-7 graphically displays the spectrum of confidence using illustrative concentration benchmarks from the major epidemiology studies cited in this chapter.

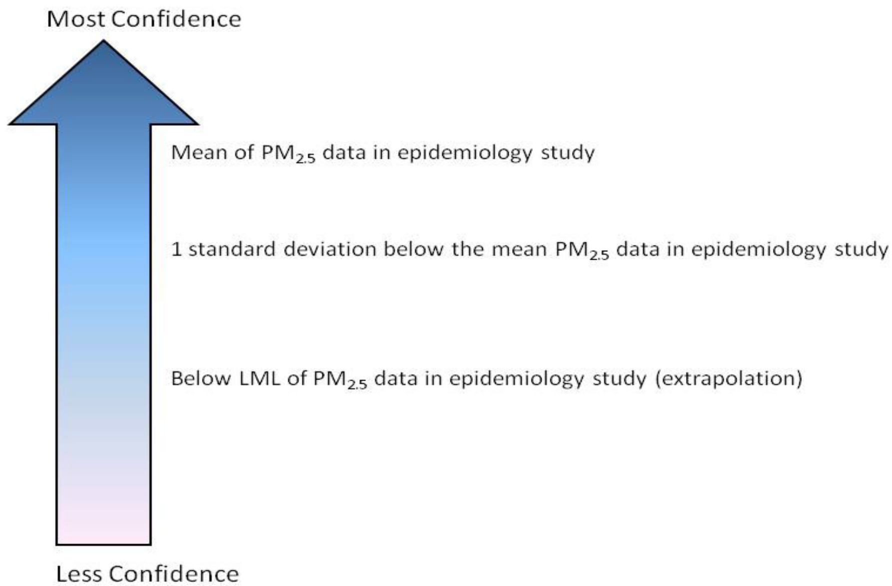


Figure 5-7. Relationship between the Size of the PM Mortality Estimates and the $PM_{2.5}$ Concentration Observed in the Epidemiology Study

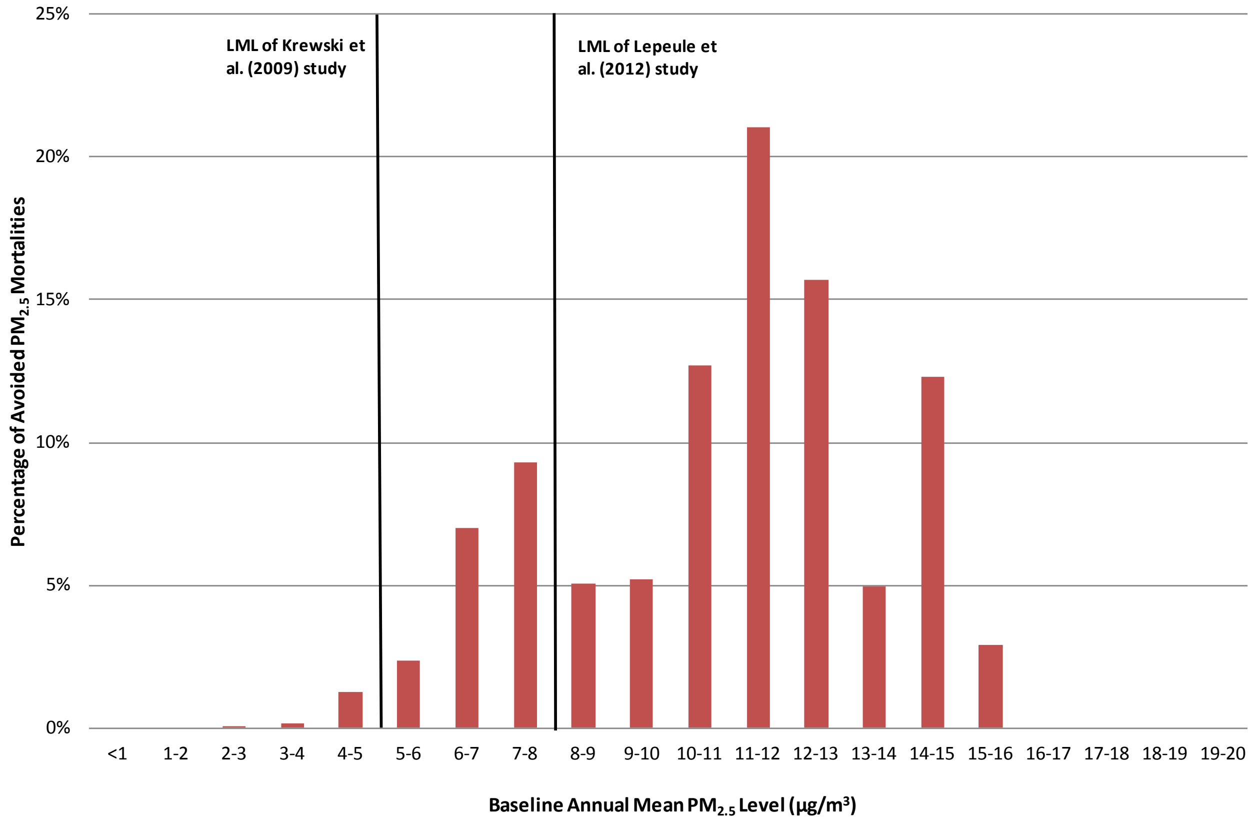
Although these types of concentration benchmark analyses (e.g., LML, one standard deviation below the mean, etc.) provide some insight into the level of uncertainty in the estimated $PM_{2.5}$ mortality benefits, the EPA does not view these concentration benchmarks as a concentration threshold below which we would not quantify health benefits of air quality improvements Rather, the core benefits estimates reported in this RIA (i.e., those based on Krewski et al. [2009] and Lepeule et al. [2012]) are the best measures because they reflect the full range of modeled air quality concentrations associated with the emission reduction strategies. In reviewing the *Policy Assessment*, SAB-CASAC confirmed that “[a]lthough there is increasing uncertainty at lower levels, there is no evidence of a threshold (i.e., a level below which there is no risk for adverse health effects)” (U.S. EPA-SAB, 2010d). In addition, in reviewing the *Costs and Benefits of the Clean Air Act* (U.S. EPA, 2011a), the SAB-HES noted that “[t]his [no-threshold] decision is supported by the data, which are quite consistent in showing effects down to the lowest measured levels. Analyses of cohorts using data from more recent years, during which time PM concentrations have fallen, continue to report strong associations with mortality” (U.S. EPA-SAB, 2010a). Therefore, the best estimate of benefits includes estimates below and above these concentration benchmarks but uncertainty is higher in the magnitude of health benefits estimated at lower concentrations, with the lowest confidence below the LML. Estimated health impacts reflecting air quality improvements below and above these concentration benchmarks are appropriately included in the total benefits estimate. In other words, our higher confidence in the estimated benefits above these concentration benchmarks should not imply an absence of confidence in the benefits estimated below these concentration benchmarks.

We estimate that most of the avoided PM-related impacts we estimate in this analysis occur among populations exposed at or above the LML of the Lepeule et al. (2012) study, while a majority of the impacts occur at or above the LML of the Krewski et al. (2009) study. We show the estimated reduction in incidence of premature mortality above and below the LML or air quality benchmarks of these studies in Tables 5-25, and we graphically display the distribution of $PM_{2.5}$-related mortality impacts for the selected standard in Figures 5-8 and 5-9.

Table 5-25. Estimated Reduction in Incidence of Adult Premature Mortality Occurring Above and Below Various Concentration Benchmarks in the Underlying Epidemiology Studies[a]

Revised and Alternative Standard Level	Epidemiology Study	Total Reduced Mortality Incidence	Allocation of Reduced Mortality Incidence			
			Below 1 Std. Dev. Below AQ Mean	At or Above 1 Std. Dev. Below AQ Mean	Below LML	At or Above LML
13 μg/m³	Krewski et al. (2009)	*140*	79 (54%)	66 (46%)	6 (4%)	140 (96%)
	Lepeule et al. (2012)	*330*	N/A	N/A	130 (38%)	200 (62%)
12 μg/m³	Krewski et al. (2009)	*460*	200 (43%)	260 (57%)	14 (3%)	440 (97%)
	Lepeule et al. (2012)	*1,000*	N/A	N/A	310 (30%)	720 (70%)
11 μg/m³	Krewski et al. (2009)	*1,500*	690 (47%)	770 (53%)	27 (2%)	1,400 (98%)
	Lepeule et al. (2012)	*3,300*	N/A	N/A	1,000 (31%)	2,300 (69%)

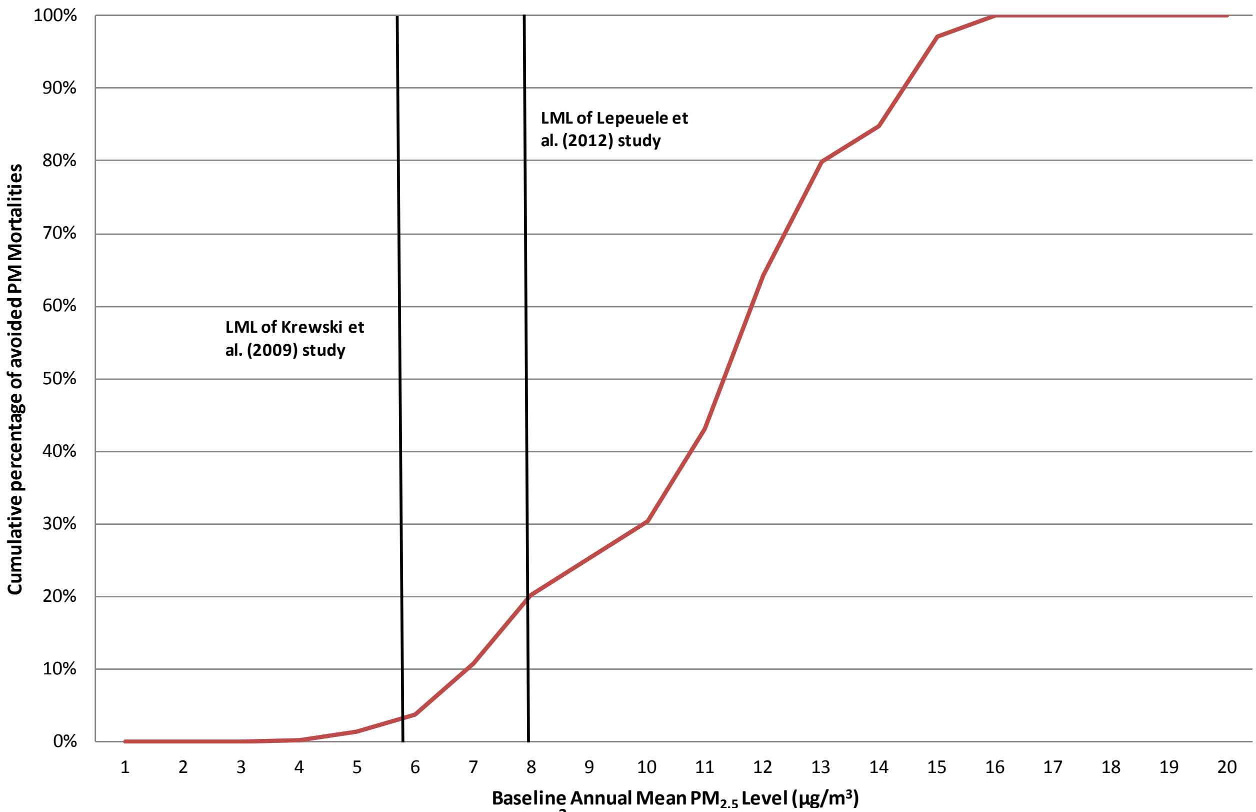
[a] Mortality incidence estimates are rounded to whole numbers and two significant digits, so estimates may not sum across columns. One standard deviation below the mean is equivalent to the middle of the range between the 10th and 25th percentile. For Krewski, the LML is 5.8 μg/m³ and one standard deviation below the mean is 11.0 μg/m³. For Lepeule et al., the LML is 8 μg/m³ and we do not have the data for one standard deviation below the mean. It is important to emphasize that although we have lower levels of confidence in levels below the LML for each study, the scientific evidence does not support the existence of a level below which health effects from exposure to $PM_{2.5}$ do not occur.



Of total $PM_{2.5}$-Related deaths avoided for 12 µg/m^3 :

97% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Krewski et al. (2009) study.
70% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Lepeule et al. (2012) study.

Figure 5-8. Number of Premature $PM_{2.5}$-related Deaths Avoided for the Revised Annual Primary $PM_{2.5}$ Standard of 12 µg/m^3 in 2020 According to the Baseline Level of $PM_{2.5}$ and the Lowest Measured Air Quality Levels of Each Mortality Study

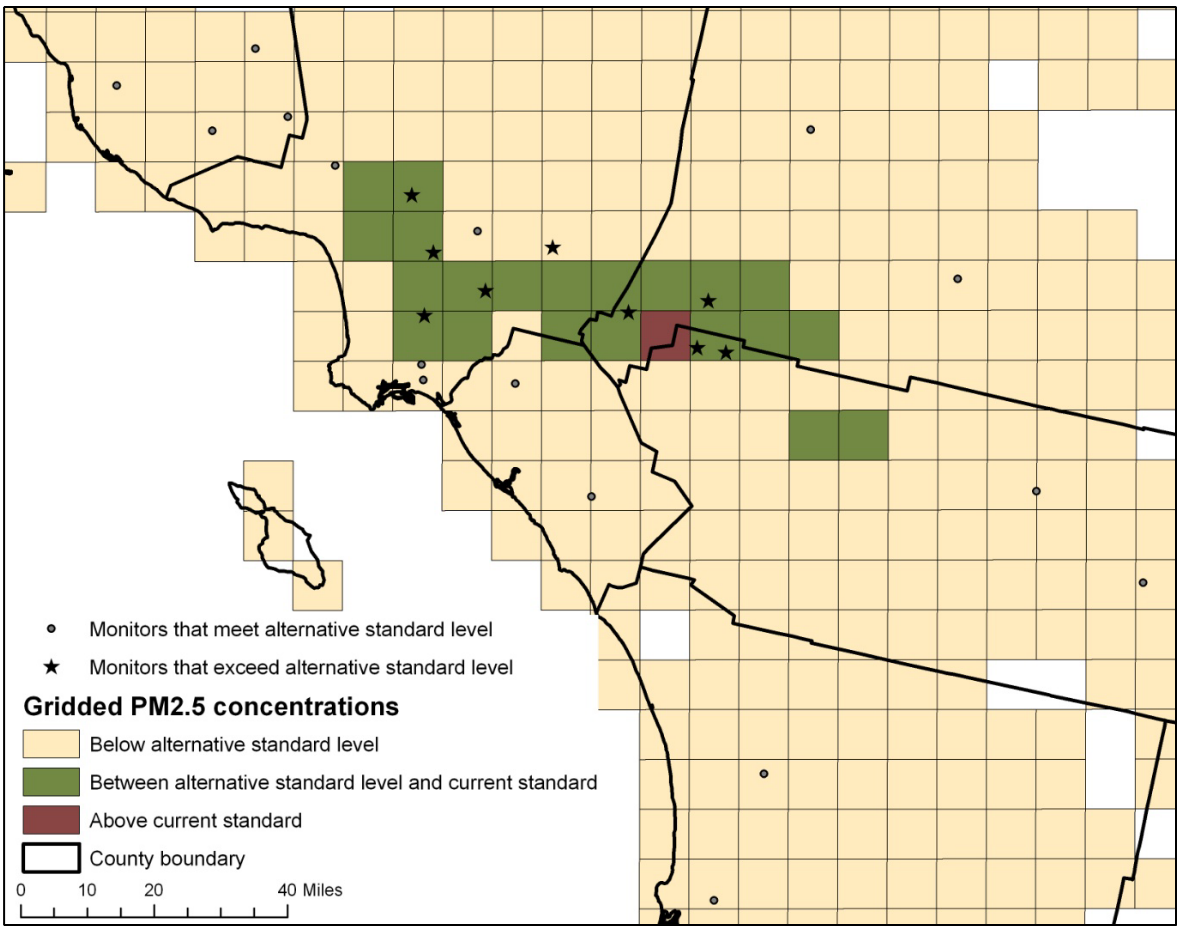


Of total $PM_{2.5}$-Related deaths avoided for 12 µg/m^3:

97% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Krewski et al. (2009) study.
70% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Lepeule et al. (2012) study.

Figure 5-9. Number of Premature $PM_{2.5}$-related Deaths Avoided for the Revised Annual Primary $PM_{2.5}$ Standard of 12 µg/m^3 in 2020 According to the Baseline Level of $PM_{2.5}$ and the Lowest Measured Air Quality Levels of Each Mortality Study

When interpreting these LML and air quality benchmarks results, it is important to understand that the avoided $PM_{2.5}$ deaths are estimated to occur from $PM_{2.5}$ reductions in the baseline air quality simulation, which assumes that 15/35 is already met. When simulating attainment with revised and alternative standards, we adjust the design value at each monitor exceeding the standard alternative to equal that standard and use an air quality interpolation technique to simulate the change in PM levels surrounding that monitor. This technique tends to simulate the greatest air quality changes nearest the monitor. We estimate benefits using modeled air quality data with 12 km grid cells, which is important because the grid cells are often substantially smaller than counties and $PM_{2.5}$ concentrations vary spatially within a county. Therefore, there may be a small number of grid cells with concentrations slightly greater than 15 µg/m^3 in the gridded baseline even though all monitors could meet an annual standard of 15 µg/m^3. In addition, some grid cells in a county can be below the level of a standard even though the highest monitor value is above that standard. Thus, emission reductions can lead to benefits in grid cells that are below a standard even within a county with a monitor that exceeds that standard. Furthermore, our approach to simulating attainment can lead to benefits in counties that are below alternative standard. Emission reduction strategies designed to reduce $PM_{2.5}$ concentrations at a given monitor will frequently improve air quality in neighboring counties. In order to make a direct comparison between the benefits and costs of these emission reduction strategies, it is appropriate to include all the benefits occurring as a result of the emission reduction strategies applied regardless of where they occur. Therefore, it is not appropriate to estimate the fraction of benefits that occur only in counties that exceed the alternative standards because it would omit benefits attributable to emission reductions in exceeding counties. Figure 5-10 provides an illustration of this concept.



As this illustration shows, because 12km grid cells are much smaller than counties and because $PM_{2.5}$ concentrations vary within a county, there can be modeled grid cells with $PM_{2.5}$ concentrations below an standard level even when the county exceeds that level. Because we model benefits using grid cells, this is a key reason why the LML graphs show benefits at levels below the alternative standards. In addition, emission reductions in an exceeding county can have benefits in a neighboring county that does not exceed.

Figure 5-10. Illustration of Relative Size of County with Exceeding Monitor and Modeled Grid Cells

While the LML of each study is important to consider when characterizing and interpreting the overall level of $PM_{2.5}$-related benefits, as discussed earlier in this chapter, the EPA believes that both of the cohort-based mortality estimates are suitable for use in air pollution health impact analyses. When estimating PM-related premature deaths avoided using risk coefficients drawn from the Lepeule et al. (2012) analysis of the Harvard Six Cities and the Krewski et al. (2009) analysis of the ACS cohorts there are innumerable other attributes that may affect the size of the reported effect estimates—including differences in population demographics, the size of the cohort, activity patterns and particle composition among others. The LML assessment presented here provides a limited representation of one key difference between the two studies.

5.7.5 Additional Sensitivity Analyses

The details of these sensitivity analyses are provided in Appendix 5.A, and summarized here. The use of an alternate lag structure would change the $PM_{2.5}$-related mortality benefits discounted at 3% discounted by between 10% and –27%; when discounted at 7%, these benefits change by between 22% and −52%. When applying higher and lower income growth adjustments, the monetary value of $PM_{2.5}$-related premature mortality changes between 33% and −14%; the value of acute endpoints changes between 8% and −4%. Using the updated population projection data, the rounded estimates of total monetized benefits increases by 4.4% for 12 $\mu g/m^3$. These sensitivity analysis results would be similar on a percentage basis for the alternative annual standards.

5.8 Discussion

The analysis in this Chapter demonstrates the potential for significant health benefits of the illustrative emission controls applied to simulate attainment with the revised annual primary $PM_{2.5}$ standard. We estimate that by 2020 the emissions reductions to reach the revised annual primary standard would have reduced the number of $PM_{2.5}$-related premature mortalities and produce substantial non-mortality benefits. This rule promises to yield significant welfare impacts as well (see Chapter 6), though the quantification of those endpoints is absent in this RIA. Even considering the quantified and unquantified uncertainties identified in this chapter, we believe that the implementing the revised annual $PM_{2.5}$ standard of 12 $\mu g/m^3$ would have substantial public health benefits that outweigh the costs (see Chapter 7).

Inherent in any complex RIA such as this one are multiple sources of uncertainty. Some of these we characterized through our quantification of statistical error in the concentration response relationships and our use of the expert elicitation-derived $PM_{2.5}$ mortality functions. Others, including the projection of atmospheric conditions and source-level emissions, the projection of baseline morbidity rates, incomes and technological development are unquantified. When evaluated within the context of these uncertainties, the health impact and monetized benefits estimates in this RIA can provide useful information regarding the public health benefits associated with the revised PM NAAQS.

There are important differences worth noting in the design and analytical objectives of NAAQS RIAs compared to RIAs for implementation rules, such as the recent MATS rule (U.S. EPA, 2011d). The NAAQS RIAs illustrate the potential costs and benefits of a revised air quality standard nationwide based on an array of emission reduction strategies for different sources, incremental to implementation of existing regulations and controls needed to attain the current standards. In short, NAAQS RIAs hypothesize, but do not predict, the emission reduction strategies that States may choose to enact when implementing a revised NAAQS. The setting of a NAAQS does not directly result in costs or benefits, and as such, NAAQS RIAs are merely illustrative and are not intended to be added to the costs and benefits of other regulations that result in specific costs of control and emission reductions. By contrast, the emission reductions from implementation rules are generally for specific, well-characterized sources, such as the recent MATS rule (U.S. EPA, 2011d). In general, the EPA is more confident in the magnitude and location of the emission reductions for implementation rules. As such, emission reductions achieved under promulgated implementation rules such as MATS have been reflected in the baseline of this NAAQS analysis. Subsequent implementation rules will be reflected in the baseline for the next PM NAAQS review. For this reason, the benefits estimated provided in this RIA and all other NAAQS RIAs should not be added to the benefits estimated for implementation rules.

In setting the NAAQS, the EPA considers that $PM_{2.5}$ concentrations vary over space and time. While the standard is designed to limit concentrations at the highest monitor in an area, it is understood that emission controls put in place to meet the standard of the highest monitor will simultaneously result in lower $PM_{2.5}$ concentrations throughout the entire area. In fact, the *Quantitative Risk and Exposure Assessment for Particulate Matter* (U.S. EPA, 2010b) shows how different standard levels would affect the entire distribution of $PM_{2.5}$ concentrations, and thus people's exposures and risk, across urban areas. For this reason, it is inappropriate to use the NAAQS level as a bright line for health effects.

The NAAQS are not set at levels that eliminate the risk of air pollution completely. Instead, the Administrator sets the NAAQS at a level requisite to protect public health with an adequate margin of safety, taking into consideration effects on susceptible populations based on the scientific literature. The risk analysis prepared in support of this PM NAAQS reported risks below these levels, while acknowledging that the confidence in those effect estimates is higher at levels closer to the standard (U.S. EPA, 2010b). While benefits occurring below the standard may be somewhat more uncertain than those occurring above the standard, the EPA considers these to be legitimate components of the total benefits estimate. Though there are greater uncertainties at lower $PM_{2.5}$ concentrations, there is no evidence of a threshold in $PM_{2.5}$-related health effects in the epidemiology literature. Given that the epidemiological literature in most cases has not provided estimates based on threshold models, there would be additional uncertainties imposed by assuming thresholds or other non-linear concentration-response functions for the purposes of benefits analysis.

The estimated benefits for the revised and alternative annual standards are in addition to the substantial benefits estimated for several recent implementation rules (U.S. EPA, 2009a, 2011c, 2011d, 2011e). Rules such as MATS and other emission reductions will have substantially reduced ambient $PM_{2.5}$ concentrations by 2020 in the East, such that no additional controls would be needed to reach 12 $\mu g/m^3$ in the East beyond the analytical baseline. These rules that have already been promulgated have tremendous combined benefits that explain why the number of avoided premature deaths associated with this NAAQS revision are smaller than were estimated in the previous PM NAAQS RIA (U.S. EPA, 2006a) for the year 2020 and even smaller than the mortality risks estimated for the current year in the PM REA (U.S. EPA, 2010b). In addition, because our analytical baseline excludes benefits associated with attaining the current annual and daily standards as well as the mobile NOx emissions anticipated by 2025, including the benefits associated with those air quality improvements would result in higher benefits than we have estimated here.

For the revised annual standard of 12 $\mu g/m^3$, all of the estimated benefits occur in California because this is the only state that needs additional air quality improvement beyond the analytical baseline after accounting for the air quality improvements from recent rules. Because all of the monetized human health benefits are projected to occur in California, we have considered the cohort studies conducted in California specifically in addition to the national risk coefficients we use as our core estimates. Although we have not calculated the benefits results using these cohort studies, we provided the risk coefficients from these California cohorts to show how much the monetized benefits could have changed if we used these cohort studies. Most of the California cohort studies report central effect estimates similar to the (nation-wide) all-cause mortality risk estimate we applied from Krewski et al.

(2009) and Lepeule et al. (2012) albeit with wider confidence intervals. Three cohort studies conducted in California indicate statistically significant higher risks than the risk estimates we applied from Lepeule et al. (2012), and four studies showed insignificant results.

5.9 References

Abbey, David E et al. 1995. “Chronic Respiratory Symptoms Associated with Estimated Long-Term Ambient Concentrations of Fine Particulates less than 2.5 Microns in aerodynamic diameter (PM2.5) and Other Air Pollutants.” *Journal of Exposure Analysis and Environmental Epidemiology* 5(2): 137-158.

Abbey, D.E., N. Nishino, W.F. McDonnell, R.J. Burchette, S.F. Knutsen, W. Lawrence Beeson, and J.X. Yang. 1999. “Long-term inhalable particles and other air pollutants related to mortality in nonsmokers.” *American Journal of Respiratory and Critical Care Medicine* 159(2):373-382.

Abt Associates, Inc. 2012. *BenMAP User’s Manual Appendices*. Prepared for U.S. Environmental Protection Agency Office of Air Quality Planning and Standards. Research Triangle Park, NC. September. Available on the Internet at <http://www.epa.gov/air/benmap/models/BenMAPAppendicesOct2012.pdf>.

Abt Associates, Inc. 2010. *Environmental Benefits and Mapping Program (Version 4.0)*. Bethesda, MD. Prepared for U.S. Environmental Protection Agency Office of Air Quality Planning and Standards. Research Triangle Park, NC. Available on the Internet at <http://www.epa.gov/air/benmap>.

Adams PF, Hendershot GE, Marano MA. 1999. “Current Estimates from the National Health Interview Survey, 1996.” *Vital Health Stat* 10(200):1-212.

Agency for Healthcare Research and Quality (AHRQ). 2000. *HCUPnet, Healthcare Cost and Utilization Project.* Rockville, MD. Available on the Internet at <http://hcupnet.ahrq.gov>.

Agency for Healthcare Research and Quality (AHRQ). 2007. *HCUPnet, Healthcare Cost and Utilization Project.* Rockville, MD. Available on the Internet at <http://hcupnet.ahrq.gov>.

Agency for Healthcare Research and Quality (AHRQ). 2009. *HCUPnet, Healthcare Cost and Utilization Project.* Rockville, MD. Available on the Internet at <http://hcupnet.ahrq.gov>.American Lung Association (ALA). 1999. *Chronic Bronchitis*. Available on the Internet at <http://www.lungusa.org/diseases/lungchronic.html>.

Alberini A, Cropper M, Krupnick A, Simon NB. 2004. “Does the value of a statistical life vary with age and health status? Evidence from the US and Canada.” *Journal of Environmental Economics and Management* 48(1): 769-792.

Alberini, Anna & Milan Ščasný. 2011. "Context and the VSL: Evidence from a Stated Preference Study in Italy and the Czech Republic," *Environmental & Resource Economics, European Association of Environmental and Resource Economists* 49(4), pages 511-538.

Aldy JE, Viscusi WK. 2008. “Adjusting the value of a statistical life for age and cohort effects.” *The Review of Economics and Statistics* 90(3):573-581.

American Lung Association (ALA). 2002. *Trends in Asthma Morbidity and Mortality*. American Lung Association, Best Practices and Program Services, Epidemiology and Statistics Unit. Available on the Internet at <http://www.lungusa.org/data/asthma/ASTHMAdt.pdf>.

American Lung Association (ALA). 2010a. *Trends in COPD (Chronic Bronchitis and Emphysema): Morbidity and Mortality*. American Lung Association Epidemiology and Statistics Unit Research and Program Services Division. February. Table 4. Based on NHIS data (CDC, 2008). Available at http://www.lungusa.org/finding-cures/our-research/trend-reports/copdtrend-report.pdf.

American Lung Association (ALA). 2010b. *Trends in Asthma Morbidity and Mortality*. American Lung Association Epidemiology and Statistics Unit Research and Program Services Division. February. Table 7. Based on NHIS data (CDC, 2008). Available at http://www.lungusa.org/finding-cures/our-research/trend-reports/asthma-trend-report.pdf.

Anderson HR, Bremner SA, Atkinson RW, Harrison RM, Walters S. 2001. “Particulate matter and daily mortality and hospital admissions in the west midlands conurbation of the UK: associations with fine and coarse particles, black smoke and sulphate.” *Occup Environ Med* 58:504–510.

Ayyub, B.M. 2002. Elicitation of Expert Opinions for Uncertainty and Risk. CRC Press, Florida.

Babin, S. M., H. S. Burkom, et al. 2007. “Pediatric patient asthma-related emergency department visits and admissions in Washington, DC, from 2001–2004, and associations with air quality, socio-economic status and age group.” *Environ Health* 6: 9.

Barnett, Adrian G. et al. 2006. “The Effects of Air Pollution on Hospitalizations for Cardiovascular Disease in Elderly People in Australian and New Zealand Cities.” *Environmental Health Perspectives* 114(7): 1018-1023.

Belanger, K., W. Beckett, E. Triche, M.B. Bracken, T. Holford, P. Ren, J.E. McSharry, D.R. Gold, T.A. Platts-Mills, and B.P. Leaderer. 2003. “Symptoms of Wheeze and Persistent Cough in the First Year of Life: Associations with Indoor Allergens, Air Contaminants, and Maternal History of Asthma.” *American Journal of Epidemiology* 158:195-202.

Bell, M. L., K. Ebisu, et al. 2008. “Seasonal and Regional Short-term Effects of Fine Particles on Hospital Admissions in 202 US Counties, 1999–2005.” *American Journal of Epidemiology* 168(11): 1301-1310.

Berger, M.C., G.C. Blomquist, D. Kenkel, and G.S. Tolley. 1987. "Valuing Changes in Health Risks: A Comparison of Alternative Measures." *The Southern Economic Journal* 53:977-984.

Centers for Disease Control and Prevention (CDC). 2003. *Health, United States. Table 30. Years of Potential Life Lost Before Age 75 for Selected Causes of Death, According to Sex, Race and Hispanic Origin: United States, Selected Years 1980– 2000*. Available on the Internet at <http://www.cdc.gov/nchs/data/hus/hus03.pdf>.

Centers for Disease Control and Prevention (CDC). 2008. National Center for Health Statistics. National Health Interview Survey, 1999-2008.

Centers for Disease Control and Prevention (CDC). 2011. United States Life Tables, 2007 National Vital Statistics Reports. Volume 59, Number 9. September. Available on the Internet at <http://www.cdc.gov/nchs/data/nvsr/nvsr59/nvsr59_09.pdf>.

Chay, K.Y., and M. Greenstone. 2003. "The Impact of Air Pollution on Infant Mortality: Evidence from Geographic Variation in Pollution Shocks Induced by a Recession." *Quarterly Journal of Economics* 118(3).

Chen L, Jennison BL, Yang W, and Omaye ST. 2000. "Elementary school absenteeism and air pollution." *Inhal Toxicol* 12(11):997-1016.

Cropper, M. L. and A. J. Krupnick. 1990. *The Social Costs of Chronic Heart and Lung Disease*. Resources for the Future. Washington, DC. Discussion Paper QE 89-16-REV.

Crouse DL, Peters PA, van Donkelaar A, Goldberg MS, Villeneuve PJ, Brion O, et al. 2012. "Risk of Non-accidental and Cardiovascular Mortality in Relation to Long-term Exposure to Low Concentrations of Fine Particulate Matter: A Canadian National-level Cohort Study." *Environ Health Perspect*. In press. http://dx.doi.org/10.1289/ehp.1104049.

Dockery, D.W., C.A. Pope, X.P. Xu, J.D. Spengler, J.H. Ware, M.E. Fay, B.G. Ferris, and F.E. Speizer. 1993. "An Association between Air Pollution and Mortality in Six U.S. Cities." *New England Journal of Medicine* 329(24):1753-1759.

Dockery, D.W., J. Cunningham, A.I. Damokosh, L.M. Neas, J.D. Spengler, P. Koutrakis, J.H. Ware, M. Raizenne, and F.E. Speizer. 1996. "Health Effects of Acid Aerosols On North American Children-Respiratory Symptoms." *Environmental Health Perspectives* 104(5):500-505.

Dominici, F., R.D. Peng, M.L. Bell, L. Pham, A. McDermott, S.L. Zeger, J.M. Samet. 2006. "Fine Particulate Air Pollution and Hospital Admission for Cardiovascular and Respiratory Diseases." *Journal of the American Medical Association* 295: 1127-1134.

Eftim SE, Samet JM, Janes H, McDermott A, Dominici F. 2008. "Fine particulate matter and mortality: a comparison of the Six Cities and American Cancer Society cohorts with a Medicare cohort." *Epidemiology* 19:209–216.

Eisenstein, E.L., L.K. Shaw, K.J. Anstrom, C.L. Nelson, Z. Hakim, V. Hasselblad and D.B. Mark. 2001. "Assessing the Clinical and Economic Burden of Coronary Artery Disease: 1986-1998. "*Medical Care* 39(8):824-35.

Eisner M, N Anthonisen, D Coultas et al. 2010. "An official American Thoracic Society public policy statement: novel risk factors and the global burden of chronic obstructive pulmonary disease," *Am J Respir Crit Care Med* 182: 693–718.

Fann N, Lamson A, Wesson K, Risley D, Anenberg SC, Hubbell BJ. 2012a. "Estimating the National Public Health Burden Associated with Exposure to Ambient $PM_{2.5}$ and Ozone. Risk Analysis," *Risk Analysis*. 32(1): 81-95.

Fann N, Roman HR, Fulcher CF, Gentile M, Wesson K, Hubbell BJ, Levy JI. 2012b. "Maximizing Health Benefits and Minimizing Inequality: Incorporating Local Scale Data in the Design and Evaluation of Air Quality Policies." *Risk Analysis*. 32(6): 908—922.

Freeman (III), AM. 1993. The *Measurement of Environmental and Resource Values: Theory and Methods*. Washington, DC: Resources for the Future.

Gan, W. Q., H. W. Davies, et al. 2012. "Association of Long-term Exposure to Community Noise and Traffic-related Air Pollution With Coronary Heart Disease Mortality." *American Journal of Epidemiology.* In press. doi: 10.1093/aje/kwr424.

GeoLytics Inc. 2002. GeoLytics CensusCD® 2000 Short Form Blocks. CD-ROM Release 1.0. GeoLytics, Inc. East Brunswick, NJ. Available: http://www.geolytics.com/ [accessed 29 September 2004].

Gilliland FD, Berhane K, Rappaport EB, Thomas DC, Avol E, Gauderman WJ, et al. 2001. "The effects of ambient air pollution on school absenteeism due to respiratory illnesses." *Epidemiology* 12(1):43-54.

Glad, J.A., L.L. Brink, E.O. Talbott, P.C. Lee, X. Xu, M. Saul, and J. Rager. 2012. The Relationship of Ambient Ozone and PM2.5 Levels and Asthma Emergency Department Visits: Possible Influence of Gender and Ethnicity. Archives of Environmental & Occupational Health. Vol 62 (2): 103-108.

Hall JV, Brajer V, Lurmann FW. 2003. Economic Valuation of Ozone-related School Absences in the South Coast Air Basin of California. *Contemporary Economic Policy* 21(4):407-417.

Harrington, W., and P.R. Portney. 1987. "Valuing the Benefits of Health and Safety Regulation." *Journal of Urban Economics* 22:101-112.

Hollman, F.W., T.J. Mulder, and J.E. Kallan. 2000. *Methodology and Assumptions for the Population Projections of the United States: 1999 to 2100*. Population Division Working Paper No. 38, Population Projections Branch, Population Division, U.S. Census Bureau, Department of Commerce. January. Available on the Internet at <http://www.census.gov/population/www/documentation/twps0038/twps0038.html>.

Hubbell, B., A. Hallberg, D.R. McCubbin, and E. Post. 2005. "Health-Related Benefits of Attaining the 8-Hr Ozone Standard." Environmental Health Perspectives 113:73–82. ICD, 1979

Hubbell BL. 2006. "Implementing QALYs in the analysis of air pollution regulations." *Environmental and Resource Economics* 34:365–384.

Industrial Economics, Incorporated (IEc). 2006. *Expanded Expert Judgment Assessment of the Concentration-Response Relationship Between $PM_{2.5}$ Exposure and Mortality*. Prepared for: Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. September. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Uncertainty/pm_ee_tsd_expert_interview_summaries.pdf>.

Industrial Economics, Incorporated (IEc). 2012. *Updating BenMAP Income Elasticity Estimates—Literature Review* . Prepared for: Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. September. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/IncomeElasticityUpdate_Recommendationswithappendices.pdf>.

Kleckner, N., and J. Neumann. 1999. *Recommended Approach to Adjusting WTP Estimates to Reflect Changes in Real Income*. Memorandum to Jim Democker, U.S. EPA/OPAR. June 3. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/klecknerandneumann1999.pdf>.

Kochi, I., B. Hubbell, and R. Kramer. 2006. "An Empirical Bayes Approach to Combining Estimates of the Value of Statistical Life for Environmental Policy Analysis." *Environmental and Resource Economics*. 34: 385-406.

Kloog, I., B.A. Coull, A. Zanobetti, P. Koutrakis, J.D. Schwartz. 2012. Acute and Chronic Effects of Particles on Hospital Admissions in New-England. PLoS ONE. Vol 7 (4): 1-8.

Krewski D, Jerrett M, Burnett RT, Ma R, Hughes E, Shi, Y, et al. 2009. "Extended follow-up and spatial analysis of the American Cancer Society study linking particulate air pollution and mortality." *HEI Research Report, 140*, Health Effects Institute, Boston, MA.

Krewski, D., R.T. Burnett, M.S. Goldbert, K. Hoover, J. Siemiatycki, M. Jerrett, M. Abrahamowicz, and W.H. White. 2000. "Reanalysis of the Harvard Six Cities Study and the American Cancer Society Study of Particulate Air Pollution and Mortality." Special Report to the Health Effects Institute. Cambridge MA. July.

Krupnick, A.J., and M.L. Cropper. 1992. "The Effect of Information on Health Risk Valuations." *Journal of Risk and Uncerta*inty 5(2):29-48.

Kunzli, N., R. Kaiser, S. Medina, M. Studnicka, O. Chanel, P. Filliger, M. Herry, F. Horak Jr., V. Puybonnieux-Texier, P. Quenel, J. Schneider, R. Seethaler, J-C Vergnaud, and H. Sommer. 2000. "Public-Health Impact of Outdoor and Traffic-Related Air Pollution: A European Assessment." *The Lancet* 356:795-801.

Kunzli, N., S. Medina, R. Kaiser, P. Quenel, F. Horak Jr, and M. Studnicka. 2001. "Assessment of Deaths Attributable to Air Pollution: Should We Use Risk Estimates Based on Time Series or on Cohort Studies?" *American Journal of Epidemiology* 153(11):1050-55.

Laden, F., J. Schwartz, F.E. Speizer, and D.W. Dockery. 2006. "Reduction in Fine Particulate Air Pollution and Mortality." *American Journal of Respiratory and Critical Care Medicine* 173:667-672.

Lave, L.B., and E.P. Seskin. 1977. *Air Pollution and Human Health*. Baltimore: Johns Hopkins University Press for Resources for the Future.

Lepeule J, Laden F, Dockery D, Schwartz J 2012. "Chronic Exposure to Fine Particles and Mortality: An Extended Follow-Up of the Harvard Six Cities Study from 1974 to 2009." *Environ Health Perspect.* Jul;120(7):965-70.

Levy JI, Baxter LK, Schwartz J. 2009. "Uncertainty and variability in health-related damages from coal-fired power plants in the United States." *Risk Anal*. 29(7) 1000-1014.

Lichtenstein, S. and P. Slovic, eds. 2006. *The construction of preference.* New York: Cambridge University Press.

Lipfert, F.W., H. Mitchell Perry Jr., J. Philip Miller, Jack D. Baty, Ronald E. Wyzg, and Sharon E. Carmody. 2000. "The Washington University-EPRI Veterans' Cohort Mortality Study: Preliminary Results." *Inhalation Toxicology* 12:41-74.

Lipfert, F.W., S.C. Morris, and R.E. Wyzga. 1989. "Acid Aerosols—the Next Criteria Air Pollutant." *Environmental Science & Technology* 23(11):1316-1322.

Lipfert, F.W.; Perry, H.M., Jr.; Miller, J.P.; Baty, J.D.; Wyzga, R.E.; Carmody, S.E. 2003. "Air Pollution, Blood Pressure, and Their Long-Term Associations with Mortality." *Inhalation Toxicology.* 15, 493-512.

Lipfert, F.W.; Wyzga, R.E.; Baty, J.D.; Miller, J.P. 2006. “Traffic Density as a Surrogate Measure of Environmental Exposures in Studies of Air Pollution Health Effects: Long-Term Mortality in a Cohort of US Veterans.” *Atmospheric Environment* 40: 154-169.

Lippmann M, Ito K, Nádas A, Burnett RT. 2000. “Association of particulate matter components with daily mortality and morbidity in urban populations.” *Res Rep Health Eff Inst.* 95:5-72, discussion 73-82.

Lisabeth, L.D. et al. 2008. “Ambient air pollution and risk for ischemic stroke and transient ischemic attack.” *Annals of Neurology*. July (64): 53—59.

Mansfield, Carol; Paramita Sinha; Max Henrion. 2009. Influence Analysis in Support of Characterizing Uncertainty in Human Health Benefits Analysis: Final Report. Prepared for U.S. EPA, Office of Air Quality Planning and Standards. November. Available on the internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/influence_analysis_final_report_psg.pdf>.

Mar, T. F., J. Q. Koenig and J. Primomo. 2010. “Associations between asthma emergency visits and particulate matter sources, including diesel emissions from stationary generators in Tacoma, Washington.” *Inhal Toxicol*. Vol. 22 (6): 445-8.Mar, T. F., T. V. Larson, et al. 2004. “An analysis of the association between respiratory symptoms in subjects with asthma and daily air pollution in Spokane, Washington.” *Inhal Toxicol* 16(13): 809-15.

Mathes, R.W., K. Ito, and T. Matte. 2011. Assessing Syndromic Surveillance of Cardiovascular Outcomes from Emergency Department Chief Complaint Data in New York City. PloS ONE. Vol 6(2): 1-10.

Metzger, K. B., P. E. Tolbert, et al. 2004. “Ambient air pollution and cardiovascular emergency department visits.” *Epidemiology* 15(1): 46-56.

Miller, Kristin A., David S. Siscovick, Lianne Sheppard, Kristen Shepherd, Jeffrey H. Sullivan, Garnet L. Anderson, and Joel D. Kaufman. 2007. “Long-Term Exposure to Air Pollution and Incidence of Cardiovascular Events in Women.” *New England Journal of Medicine*. 356 (5) :447-458.

Moolgavkar, S.H. 2000. “Air Pollution and Hospital Admissions for Diseases of the Circulatory System in Three U.S. Metropolitan Areas.” *Journal of the Air and Waste Management Association* 50:1199-1206.

Moolgavkar, S.H. 2003. “Air Pollution and Daily Deaths and Hospital Admissions in Los Angeles and Cook Counties.” In *Revised Analyses of Time-Series Studies of Air Pollution and Health*. Special Report. Boston, MA: Health Effects Institute.

Mrozek, J.R., and L.O. Taylor. 2002. “What Determines the Value of Life? A Meta-Analysis.” *Journal of Policy Analysis and Management* 21(2):253-270.

Mustafić H, Jabre P, Caussin C et al. 2012. "Main Air Pollutants and Myocardial Infarction: A Systematic Review and Meta-analysis" *JAMA*. 2012;307(7):713-721.

National Center for Education Statistics (NCHS). 1996. *The Condition of Education 1996, Indicator 42: Student Absenteeism and Tardiness*. U.S. Department of Education. Washington, DC.

National Research Council (NRC). 2002. *Estimating the Public Health Benefits of Proposed Air Pollution Regulations*. Washington, DC: The National Academies Press. Washington, DC.

National Research Council (NRC). 2008. *Estimating Mortality Risk Reduction and Economic Benefits from Controlling Ozone Air Pollution*. National Academies Press. Washington, DC.

Neumann, J.E., M.T. Dickie, and R.E. Unsworth. 1994. "Linkage Between Health Effects Estimation and Morbidity Valuation in the Section 812 Analysis—Draft Valuation Document." Industrial Economics Incorporated (IEc) Memorandum to Jim DeMocker, U.S. Environmental Protection Agency, Office of Air and Radiation, Office of Policy Analysis and Review. March 31. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/neumannn1994.pdf>.

Office of Management and Budget (OMB). 2003. *Circular A-4: Regulatory Analysis.* Washington, DC. Available on the Internet at <http://www.whitehouse.gov/omb/circulars/a004/a-4.html.>

Ostro, B., M. Lipsett, J. Mann, H. Braxton-Owens, and M. White. 2001. "Air Pollution and Exacerbation of Asthma in African-American Children in Los Angeles." *Epidemiology* 12(2):200–208.

Ostro, B.D. 1987. "Air Pollution and Morbidity Revisited: A Specification Test." *Journal of Environmental Economics Management* 14:87-98.

Ostro, B.D. and S. Rothschild. 1989. "Air Pollution and Acute Respiratory Morbidity: An Observational Study of Multiple Pollutants." *Environmental Research* 50:238-247.

Ozkaynak, H., and G.D. Thurston. 1987. "Associations between 1980 U.S. Mortality Rates and Alternative Measures of Airborne Particle Concentration." *Risk Analysis* 7(4):449-461.

Patel, M. M., S. N. Chillrud, J. C. Correa, Y. Hazi, M. Feinberg, D. Kc, S. Prakash, J. M. Ross, D. Levy and P. L. Kinney. 2010. "Traffic-Related Particulate Matter and Acute Respiratory Symptoms among New York City Area Adolescents." *Environ Health Perspect*. Sep;118(9):1338-43.

Peel JL; Metzger KB; Klein M; Flanders WD; Mulholland JA; Tolbert PE. 2007. "Ambient air pollution and cardiovascular emergency department visits in potentially sensitive groups." *Am J Epidemiol* 165: 625-633.

Peng, R. D., M. L. Bell, et al. 2009. “Emergency admissions for cardiovascular and respiratory diseases and the chemical composition of fine particle air pollution.” *Environ Health Perspect* 117(6): 957-63.

Peng, R. D., H. H. Chang, et al. 2008. “Coarse particulate matter air pollution and hospital admissions for cardiovascular and respiratory diseases among Medicare patients.” *JAMA* 299(18): 2172-9.

Peters A. 2005. “Particulate matter and heart disease: evidence from epidemiological studies.” *Toxicol Appl Pharmacol*. Sep 1;207(2 Suppl):477-82.

Peters A. 2006. “When a myocardial infarction comes out of the not-so-blue air.” *Circulation*. Dec 5;114(23):2430-1

Peters, A., D. W. Dockery, J. E. Muller and M. A. Mittleman. 2001. “Increased particulate air pollution and the triggering of myocardial infarction.” *Circulation*. Vol. 103 (23): 2810-5.

Poloniecki, J.D., R.W. Atkinson., A.P. de Leon., and H.R. Anderson. 1997. “Daily Time Series for Cardiovascular Hospital Admissions and Previous Day’s Air Pollution in London, UK.” *Occupational and Environmental Medicine* 54(8):535-540

Pope, C.A., III, R.T. Burnett, G.D. Thurston, M.J. Thun, E.E. Calle, D. Krewski, and J.J. Godleski. 2004. “Cardiovascular Mortality and Long-term Exposure to Particulate Air Pollution.” *Circulation* 109: 71-77.

Pope, C.A., III, R.T. Burnett, M.J. Thun, E.E. Calle, D. Krewski, K. Ito, and G.D. Thurston. 2002. “Lung Cancer, Cardiopulmonary Mortality, and Long-term Exposure to Fine Particulate Air Pollution.” *Journal of the American Medical Association* 287:1132-1141.

Pope, C.A., III, D.W. Dockery, J.D. Spengler, and M.E. Raizenne. 1991. “Respiratory Health and PM_{10} Pollution: A Daily Time Series Analysis.” *American Review of Respiratory Diseases* 144:668-674.

Pope, CA III, E Majid, D Dockery. 2009. “Fine Particle Air Pollution and Life Expectancy in the United States.” *New England Journal of Medicine* 360: 376–386.

Pope, C. A., III, J. B. Muhlestein, et al. 2006. “Ischemic heart disease events triggered by short-term exposure to fine particulate air pollution.” *Circulation* 114(23): 2443-8.

Pope, C.A., III, M.J. Thun, M.M. Namboodiri, D.W. Dockery, J.S. Evans, F.E. Speizer, and C.W. Heath, Jr. 1995. “Particulate Air Pollution as a Predictor of Mortality in a Prospective Study of U.S. Adults.” *American Journal of Respiratory Critical Care Medicine* 151:669-674.

Puett RC, Hart JE, Suh H, Mittleman M, Laden F. 2011. "Particulate Matter Exposures, Mortality and Cardiovascular Disease in the Health Professionals Follow-up Study." *Environ Health Perspect*. August; 119(8): 1130–1135.

Puett RC, Hart JE, Yanosky JD, Paciorek C, Schwartz J, Suh H, et al. 2009. "Chronic fine and coarse particulate exposure, mortality and coronary heart disease in the Nurses' Health Study." *Environ Health Perspect* 117:1697–1701.

Roman, Henry A., Katherine D. Walker, Tyra L. Walsh, Lisa Conner, Harvey M. Richmond, Bryan J. Hubbell, and Patrick L. Kinney. 2008. "Expert Judgment Assessment of the Mortality Impact of Changes in Ambient Fine Particulate Matter in the U.S." *Environ. Sci. Technol*., 42(7):2268-2274.

Rosamond, W., G. Broda, E. Kawalec, S. Rywik, A. Pajak, L. Cooper, and L. Chambless. 1999. "Comparison of Medical Care and Survival of Hospitalized Patients with Acute Myocardial Infarction in Poland and the United States." *American Journal of Cardiology* 83:1180-1185

Rowe, R.D., and L.G. Chestnut. 1986. "Oxidants and Asthmatics in Los Angeles: A Benefits Analysis—Executive Summary." Prepared by Energy and Resource Consultants, Inc. Report to the U.S. Environmental Protection Agency, Office of Policy Analysis. EPA- 230-09-86-018. Washington, DC.

Rowlatt, P., Spackman, M., Jones, S., Jones-Lee, M., Loomes, G. 1998. *Valuation of deaths from air pollution. A Report for the Department of Environment, Transport and the Regions and the Department of Trade and Industry*. National Economic Research Associates (NERA), London.

Russell, M.W., D.M. Huse, S. Drowns, E.C. Hamel, and S.C. Hartz. 1998. "Direct Medical Costs of Coronary Artery Disease in the United States." *American Journal of Cardiology* 81(9):1110-1115.

Schwartz, J. 1993. "Particulate Air Pollution and Chronic Respiratory Disease." *Environment Research* 62:7-13.

Schwartz, J., and L.M. Neas. 2000. "Fine Particles are More Strongly Associated than Coarse Particles with Acute Respiratory Health Effects in Schoolchildren." *Epidemiology* 11:6-10.

Schwartz J, Dockery DW, Neas LM, Wypij D, Ware JH, Spengler JD, Koutrakis P, Speizer FE, Ferris BG Jr. 1994. "Acute effects of summer air pollution on respiratory symptom reporting in children." *American Journal of Respiratory and Critical Care Medicine* 150(5 Pt 1):1234-42.

Schwartz J, Coull B, Laden F. 2008. "The Effect of Dose and Timing of Dose on the Association between Airborne Particles and Survival." *Environmental Health Perspectives* 116: 64-69.

Sheppard, L. 2003. "Ambient Air Pollution and Nonelderly Asthma Hospital Admissions in Seattle, Washington, 1987-1994." In *Revised Analyses of Time-Series Studies of Air Pollution and Health*. Special Report. Boston, MA: Health Effects Institute.

Shogren, J., and T. Stamland. 2002. "Skill and the Value of Life." *Journal of Political Economy* 110:1168-1197.

Slaughter, James C et al. 2005. "Association between particulate matter and emergency room visits, hospital admissions and mortality in Spokane, Washington." *Journal of Exposure Analysis and Environmental Epidemiology* (2005) 15, 153–159. doi:10.1038/sj.jea.7500382

Smith et al. 2004. "Do the Near Elderly Value Mortality Risks Differently?" *The Review of Economics and Statistics* 86(1): 423-429.

Smith, D.H., D.C. Malone, K.A. Lawson, L.J. Okamoto, C. Battista, and W.B. Saunders. 1997. "A National Estimate of the Economic Costs of Asthma." A*merican Journal of Respiratory and Critical Care Medicine* 156(3 Pt 1):787-793.

Smith, K., Pattanayak SK, Van Houtven G. 2006. "Structural benefit transfer: An example using VSL estimates." *Ecological Economics* 60: 361-371.

Standard and Poor's. 2000. "The U.S. Economy: The 25 Year Focus." Winter. Available on the Internet at http://:www.standardandpoors.com

Stanford, R., T. McLaughlin and L. J. Okamoto. 1999. "The cost of asthma in the emergency department and hospital." *Am J Respir Crit Care Med.*160 (1): 211-5.

Sullivan, J., L. Sheppard, et al. 2005. "Relation between short-term fine-particulate matter exposure and onset of myocardial infarction." *Epidemiology* 16(1): 41-8.

Tagaris E, Liao KJ, Delucia AJ, et al. 2009. "Potential impact of climate change on air-pollution related human health effects." *Environ. Sci. Technol*. 43: 4979–4988.

Tolbert, P. E., M. Klein, et al. 2007. "Multipollutant modeling issues in a study of ambient air quality and emergency department visits in Atlanta." *J Expo Sci Environ Epidemiol* 17 Suppl 2: S29-35.

Tolley, G.S. et al. 1986. *Valuation of Reductions in Human Health Symptoms and Risks.* University of Chicago. Final Report for the U.S. Environmental Protection Agency, Office of Policy Analysis. January. Available at <http://yosemite.epa.gov/ee/epa/eerm.nsf/a7a2ee5c6158cedd852563970080ee30/eaa2842b3d2bc2798525644d0053bf48!OpenDocument#_Section6>.

Tsuge T, Kishimot A, Takeuchi K. 2005. "A choice experiment approach to the valuation of mortality." *Journal of Risk and Uncertainty* 31(1):73-95.

U.S. Bureau of Economic Analysis (U.S. BEA). 2004. *BEA Economic Areas (EAs).* U.S. Department of Commerce. Available on the Internet at <http://www.bea.gov/regional/docs/econlist.cfm>.

U.S. Bureau of Economic Analysis (U.S. BEA). 1997. *Real Gross Domestic Product—Table 2A.* U.S. Department of Commerce.

U.S. Bureau of Economic Analysis (U.S. BEA). 1997. *The Economic and Budget Outlook: An Update*, Table 4—*Economic Projections for Calendar Years 1997 Through 2007.* September. U.S. Department of Commerce.

U.S. Bureau of the Census. 2001. *Statistical Abstract of the United States*, 2001, Section 12: Labor Force, Employment, and Earnings, Table No. 521. Washington, DC.

U.S. Bureau of the Census. 2002a. Modified Race Data Summary File 2000 Technical Documentation. Census of Population and Housing: Washington DC. Available on the Internet at <http://www.census.gov/popest/archives/files/MRSF-01-US1.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 1999a. *The Benefits and Costs of the Clean Air Act, 1990–2010.* Prepared for U.S. Congress by U.S. EPA, Office of Air and Radiation/Office of Policy Analysis and Review, Washington, DC, November; EPA report no. EPA-410-R-99-001. Available on the Internet at <http://www.epa.gov/air/sect812/1990-2010/fullrept.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2000. *Guidelines for Preparing Economic Analyses*. EPA 240-R-00-003. National Center for Environmental Economics, Office of Policy Economics and Innovation. Washington, DC. September. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eed.nsf/webpages/Guidelines.html/$file/cover.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2004. *Air Quality Criteria for Particulate Matter*. National Center for Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC EPA/600/P-99/002bF. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=87903>.

U.S. Environmental Protection Agency (U.S. EPA). 2006a. *Regulatory Impact Analysis, 2006 National Ambient Air Quality Standards for Particulate Matter, Chapter 5*. Office of Air Quality Planning and Standards, Research Triangle Park, NC. October. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/Chapter%205—Benefits.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2006b. *Air Quality Criteria for Ozone and Related Photochemical Oxidants (Final)*. EPA/600/R-05/004aF-cF. Washington, DC: U.S. EPA. February. Available on the Internet at http://cfpub.epa.gov/ncea/CFM/recordisplay.cfm?deid=149923.U.S. Environmental Protection Agency (U.S. EPA). 2008. *Regulatory Impact Analysis, 2008 National Ambient Air Quality Standards for Ground-level Ozone, Chapter 6*. Office of Air Quality Planning and Standards, Research Triangle Park, NC. March. Available at <http://www.epa.gov/ttn/ecas/regdata/RIAs/6-ozoneriachapter6.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2008a. Regulatory Impact Analysis, 2008 National Ambient Air Quality Standards for Ground-level Ozone, Chapter 6. Office of Air Quality Planning and Standards, Research Triangle Park, NC. March. Available at <http://www.epa.gov/ttn/ecas/regdata/RIAs/6-ozoneriachapter6.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2008b. *Integrated Science Assessment for Oxides of Nitrogen—Health Criteria (Final Report)*. National Center for Environmental Assessment, Research Triangle Park, NC. July. Available on the Internet at http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=194645

U.S. Environmental Protection Agency (U.S. EPA). 2008c. *Integrated Science Assessment for Sulfur Oxides—Health Criteria (Final Report)*. National Center for Environmental Assessment, Research Triangle Park, NC. September. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=198843>.

U.S. Environmental Protection Agency (U.S. EPA). 2009a. *Regulatory Impact Analysis: National Emission Standards for Hazardous Air Pollutants from the Portland Cement Manufacturing Industry*. Office of Air Quality Planning and Standards, Research Triangle Park, NC. April. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/portlandcementria_4-20-09.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2009b. *Integrated Science Assessment for Particulate Matter (Final Report)*. EPA-600-R-08-139F. National Center for Environmental Assessment—RTP Division. December. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=216546>.

U.S. Environmental Protection Agency (U.S. EPA). 2010a. *Final Regulatory Impact Analysis (RIA) for the NO_2 National Ambient Air Quality Standards (NAAQS).* Office of Air Quality Planning and Standards, Research Triangle Park, NC. January. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/FinalNO2RIAfulldocument.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2010b. *Quantitative Health Risk Assessment for Particulate Matter—Final Repor*t. EPA-452/R-10-005. Office of Air Quality Planning and Standards, Research Triangle Park, NC. September. Available on the Internet at <http://www.epa.gov/ttnnaaqs/standards/pm/data/PM_RA_FINAL_June_2010.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2010c. *Final Regulatory Impact Analysis (RIA) for the SO_2 National Ambient Air Quality Standards (NAAQS).* Office of Air Quality Planning and Standards, Research Triangle Park, NC. June. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/fso2ria100602full.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2010d. *Regulatory Impact Analysis (RIA) for the Proposed Reconsideration of the Ozone National Ambient Air Quality Standards (NAAQS).* Office of Air Quality Planning and Standards, Research Triangle Park, NC. January. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/s1-supplemental_analysis_full.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2010e. *Guidelines for Preparing Economic Analyses*. EPA 240-R-10-001. National Center for Environmental Economics, Office of Policy Economics and Innovation. Washington, DC. December. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0568-50.pdf/$file/EE-0568-50.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2010f. *Technical Support Document: Summary of Expert Opinions on the Existence of a Threshold in the Concentration-Response Function for $PM_{2.5}$-related Mortality*. Research Triangle Park, NC. June. Available on the Internet at <www.epa.gov/ttn/ecas/regdata/Benefits/thresholdstsd.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2010g. *Proposed Regulatory Impact Analysis (RIA) for the Transport Rule*. Office of Air Quality Planning and Standards, Research Triangle Park, NC. January. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/proposaltrria_final.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2011a. *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress*. Office of Air and Radiation, Office of Policy, Washington, DC. March. Available on the Internet at <http://www.epa.gov/oar/sect812/feb11/fullreport.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2011b. *Policy Assessment for the Review of the Particulate Matter National Ambient Air Quality Standards*. EPA-452/D-11-003. April. Available on the Internet at <http://www.epa.gov/ttnnaaqs/standards/pm/s_pm_2007_pa.html>.

U.S. Environmental Protection Agency (U.S. EPA). 2011c. *Regulatory Impact Analysis for the Federal Implementation Plans to Reduce Interstate Transport of Fine Particulate Matter and Ozone in 27 States; Correction of SIP Approvals for 22 States*. June. Available on the Internet at <http://www.epa.gov/airtransport/pdfs/FinalRIA.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2011d. *Regulatory Impact Analysis for the Final Mercury and Air Toxics Standards.* EPA-452/R-11-011. December. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/matsriafinal.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2011e. *Integrated Science Assessment for Ozone and Related Photochemical Oxidants (Second External Review Draft).* EPA/600/R-10/076B. U.S. Environmental Protection Agency, National Center for Environmental Assessment—RTP Division. September. Available at <http://oaspub.epa.gov/eims/eimscomm.getfile?p_download_id=503567>.

U.S. Environmental Protection Agency (U.S. EPA). 2011e. *Regulatory Impact Results for the Reconsideration Proposal for National Emission Standards for Hazardous Air Pollutants for Industrial, Commercial, and Institutional Boilers and Process Heaters at Major Sources.* December. Available on the Internet at <http://www.epa.gov/ttnecas1/regdata/RIAs/boilermactreconria.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2012a. *Regulatory Impact Analysis for the Proposed Revisions to the National Ambient Air Quality Standards for Particulate Matter.* EPA-452/R-12-003. Office of Air Quality Planning and Standards, Health and Environmental Impacts Division. June. Available at <http://www.epa.gov/ttnecas1/regdata/RIAs/PMRIACombinedFile_Bookmarked.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2012b. *Provisional Assessment of Recent Studies on Health Effects of Particulate Matter Exposure.* EPA/600/R-12/056A. National Center for Environmental Assessment—RTP Division. December.

U.S. Environmental Protection Agency (U.S. EPA). 2012c. *Integrated Science Assessment for Particulate* Ozone and Related Photochemical Oxidants *(Third External Review Draft)*. EPA/600/R-10/076C. National Center for Environmental Assessment—RTP Division. June. Available on the Internet at <http://cfpub.epa.gov/ncea/isa/recordisplay.cfm?deid=242490>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 1999. *An SAB Advisory on the Clean Air Act Section 812 Prospective Study Health and Ecological Initial Studies.* Prepared by the Health and Ecological Effects Subcommittee (HEES) of the Advisory Council on the Clean Air Compliance Analysis, Science Advisory Board, U.S. Environmental Protection Agency. Washington DC. EPA-SAB-COUNCIL-ADV-99-005. Available on the Internet at <http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/1AFDEBB45C57F6E585257193005 8C1A5/$File/coua9905.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2000. *An SAB Report on EPA's White Paper Valuing the Benefits of Fatal Cancer Risk Reduction.* EPA-SAB-EEAC-00-013. July. Available on the Internet at <http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/41334524148BCCD6852571A700516498/$File/eeacf013.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2004a. *Advisory on Plans for Health Effects Analysis in the Analytical Plan for EPA's Second Prospective Analysis—Benefits and Costs of the Clean Air Act, 1990–2020*. EPA-SAB-COUNCIL-ADV-04-002. March. Available on the Internet at <http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/08E1155AD24F871C85256E5400433D5D/$File/council_adv_04002.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2004b. *Review of the Draft Analytical Plan for EPA's Second Prospective Analysis—Benefits and Costs of the Clean Air Act, 1990-2020: An Advisory by the Advisory Council for Clean Air Compliance Analysis*. EPA-SAB-COUNCIL-ADV-04-004. May. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/7CCBBFE15CD4C8B185256F17005E3079/$File/council_adv_04004.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2004c. *Advisory Council on Clean Air Compliance Analysis Response to Agency Request on Cessation Lag.* EPA-COUNCIL-LTR-05-001. December. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/0/39F44B098DB49F3C85257170005293E0/$File/council_ltr_05_001.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2007. *SAB Advisory on EPA's Issues in Valuing Mortality Risk Reduction*. EPA-SAB-08-001. October. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/4128007E7876B8F0852573760058A978/$File/sab-08-001.pdf>.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 2008. *Characterizing Uncertainty in Particulate Matter Benefits Using Expert Elicitation.* EPA-COUNCIL-08-002. July. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/0/43B91173651AED9E85257487004EA6CB/$File/EPA-COUNCIL-08-002-unsigned.pdf>.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 2009a. *Consultation on EPA's Particulate Matter National Ambient Air Quality Standards: Scope and Methods Plan for Health Risk and Exposure Assessment*. EPA-COUNCIL-09-009. May. Available on the Internet at <http://yosemite.epa.gov/sab/SABPRODUCT.NSF/81e39f4c09954fcb85256ead006be86e/723FE644C5D758DF852575BD00763A32/$File/EPA-CASAC-09-009-unsigned.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2009b. *Review of EPA's Integrated Science Assessment for Particulate Matter (First External Review Draft, December 2008)*. EPA-COUNCIL-09-008. May. Available on the Internet at <http://yosemite.epa.gov/sab/SABPRODUCT.NSF/81e39f4c09954fcb85256ead006be86e/73ACCA834AB44A10852575BD0064346B/$File/EPA-CASAC-09-008-unsigned.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2009c. *Review of Integrated Science Assessment for Particulate Matter (Second External Review Draft, July 2009)*. EPA-CASAC-10-001. November. Available on the Internet at <http://yosemite.epa.gov/sab/SABPRODUCT.NSF/81e39f4c09954fcb85256ead006be86e/151B1F83B023145585257678006836B9/$File/EPA-CASAC-10-001-unsigned.pdf>.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 2010a. *Review of EPA's DRAFT Health Benefits of the Second Section 812 Prospective Study of the Clean Air Act*. EPA-COUNCIL-10-001. June. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/9288428b8eeea4c885257242006935a3/59e06b6c5ca66597852575e7006c5d09!OpenDocument&TableRow=2.3#2.>.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 2010b. *Review of Risk Assessment to Support the Review of the Particulate Matter (PM) Primary National Ambient Air Quality Standards—External Review Draft (September 2009).* EPA-CASAC-10-003. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/264cb1227d55e02c85257402007446a4/BC1ECC5D539EF72385257678006D5754/$File/EPA-CASAC-10-003-unsigned.pdf>.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 2010c. *CASAC Review of Quantitative Health Risk Assessment for Particulate Matter—Second External Review Draft (February 2010).* EPA-CASAC-10-008. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/264cb1227d55e02c85257402007446a4/BC4F6E77B6385155852577070002F09F/$File/EPA-CASAC-10-008-unsigned.pdf>.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 2010d. *CASAC Review of Policy Assessment for the Review of the PM NAAQS—Second External Review Draft (June 2010).* EPA-CASAC-10-015. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/264cb1227d55e02c85257402007446a4/CCF9F4C0500C500F8525779D0073C593/$File/EPA-CASAC-10-015-unsigned.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2011. *Review of Valuing Mortality Risk Reductions for Environmental Policy: A White Paper (December 10, 2010).* EPA-SAB-11-011 July. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/298E1F50F844BC23852578DC0059A616/$File/EPA-SAB-11-011-unsigned.pdf>.

Viscusi, V.K., and J.E. Aldy. 2003. "The Value of a Statistical Life: A Critical Review of Market Estimates throughout the World." *Journal of Risk and Uncertainty* 27(1):5-76.

Viscusi, W.K., W.A. Magat, and J. Huber. 1991. “Pricing Environmental Health Risks: Survey Assessments of Risk-Risk and Risk-Dollar Trade-Offs for Chronic Bronchitis.” *Journal of Environmental Economics and Management* 21:32-51.

Wellenius, G.A., M.R. Burger, B.A. Coull, J. Schwartz, H.H. Suh, P. Koutrakis, G. Schlaug, D.R. Gold, M.A. Mittlemen. 2012.”Ambient air pollution and the risk of acute ischemic stroke.” *Arch. Intern. Med*., 172 (3), pp. 229–234.

Wittels, E.H., J.W. Hay, and A.M. Gotto, Jr. 1990. “Medical Costs of Coronary Artery Disease in the United States.” *American Journal of Cardiology* 65(7):432-440.

Woodruff, T.J., J. Grillo, and K.C. Schoendorf. 1997. “The Relationship Between Selected of postneonatal infant mortality and particulate air pollution in the United States.” *Environmental Health Perspectives*. Vol. 105 (6): 608-612.

Woodruff TJ, Parker JD, Schoendorf KC. 2006. “Fine particulate matter (PM2.5) air pollution and selected causes of postneonatal infant mortality in California.” *Environmental Health Perspectives* 114(5):786-90.

Woods & Poole Economics Inc. 2007. *Population by Single Year of Age*. CD-ROM. Woods & Poole Economics, Inc. Washington, D.C.

World Health Organization (WHO). 1977. International Classification of Diseases, 9th Revision (ICD-9). Geneva: WHO.

World Health Organization (WHO). 2008. *Part 1: Guidance Document on Characterizing and Communicating Uncertainty in Exposure Assessment, Harmonization Project Document No. 6.* Published under joint sponsorship of the World Health Organization, the International Labour Organization and the United Nations Environment Programme. WHO Press: Geneva, Switzerland. Available on the Internet at <www.who.int/ipcs/methods/harmonization/areas/uncertainty%20.pdf>.

Zanobetti, A., M. Franklin, et al. 2009. “Fine particulate air pollution and its components in association with cause-specific emergency admissions.” *Environmental Health* 8: 58-60.Zeger S; Dominici F; McDermott A; Samet J. 2008. “Mortality in the Medicare population and chronic exposure to fine particulate air pollution in urban centers (2000-2005).” *Environ Health Perspect* 116: 1614-1619.

Zanobetti A. and Schwartz, J. 2006. “Air pollution and emergency admissions in Boston, MA.” *J Epidemiol Community Health* 60(10): 890-5.

APPENDIX 5.A

ADDITIONAL SENSITIVITY ANALYSES RELATED TO THE HEALTH BENEFITS ANALYSIS

The analysis presented in Chapter 5 of this RIA is based on our current interpretation of the scientific and economic literature. That interpretation requires judgments regarding the best available data, models, and analytical methodologies and the assumptions that are most appropriate to adopt in the face of important uncertainties. The majority of the analytical assumptions used to develop the main estimates of benefits have been reviewed and approved by EPA's independent Science Advisory Board (SAB). Both EPA and the SAB recognize that data and modeling limitations as well as simplifying assumptions can introduce significant uncertainty into the estimates of benefits and that alternative choices exist for some inputs to the analysis, such as the concentration-response functions for mortality.

This appendix supplements our main analysis of benefits with five additional sensitivity calculations. The supplemental estimates examine sensitivity to assumptions about both physical effects (i.e., the structure of the cessation lag; estimates of the number of avoided cerebrovascular events, cardiovascular emergency department visits and cases of chronic bronchitis; and alternate effect estimates for cohorts in California) and valuation issues (i.e., the appropriate income elasticity, updated cost-of-illness estimates). We conducted these sensitivity analyses for the selected annual standard of 12 $\mu g/m^3$ as an illustrative example. These supplemental estimates are not meant to be comprehensive. Rather, they reflect some of the key issues identified by EPA or commenters as likely to have a significant impact on total benefits, or they are health endpoints for which the health data are still evolving, or for which we lack an appropriate method to estimate the economic value. The individual income growth and lag adjustments in the tables should not simply be added together because (1) there may be overlap among the alternative assumptions, and (2) the joint probability among certain sets of alternative assumptions may be low.

5.A.1 Cessation Lag Structure for $PM_{2.5}$-Related Premature Mortality

Based in part on prior advice from the EPA's independent Science Advisory Board (SAB), EPA typically assumes that there is a time lag between reductions in particulate matter (PM) exposures in a population and the full realization of reductions in premature mortality. Within the context of benefits analyses, this term is often referred to as "cessation lag." The existence of such a lag is important for the valuation of reductions in premature mortality because economic theory suggests that dollar-based representations of health effect incidence changes occurring in the future should be discounted to reflect time preferences for consumption in the population (e.g., people generally prefer to consume now rather than later and will generally give up greater consumption in the future for earlier consumption).

Over the last 15 years, there has been a continuing discussion and evolving advice regarding the timing of changes in health effects following changes in ambient air pollution. It has been hypothesized that some reductions in premature mortality from exposure to ambient $PM_{2.5}$ will occur over short periods of time in individuals with compromised health status, but other effects are likely to occur among individuals who, at baseline, have reasonably good health that will deteriorate because of continued exposure. The SAB-HES has recognized this lack of direct evidence. However, in early advice, they also note that "although there is substantial evidence that a portion of the mortality effect of PM is manifest within a short period of time, i.e., less than one year, it can be argued that, if no lag assumption is made, the entire mortality excess observed in the cohort studies will be analyzed as immediate effects, and this will result in an overestimate of the health benefits of improved air quality. Thus some time lag is appropriate for distributing the cumulative mortality effect of PM in the population" (EPA-SAB-COUNCIL-ADV-00-001, 1999, p. 9). In more recent advice, the SAB-HES suggests that appropriate lag structures may be developed based on the distribution of cause-specific deaths within the overall all-cause estimate (EPA-SAB-COUNCIL-ADV-04-002, 2004). They suggest that diseases with longer progressions should be characterized by longer-term lag structures, while air pollution impacts occurring in populations with existing disease may be characterized by shorter-term lags.

A key question is the distribution of causes of death within the relatively broad categories analyzed in the long-term cohort studies. Although it may be reasonable to assume the cessation lag for lung cancer deaths mirrors the long latency of the disease, it is not at all clear what the appropriate lag structure should be for cardiopulmonary deaths, which include both respiratory and cardiovascular causes. Some respiratory diseases, such as chronic obstructive pulmonary disease, may have a long period of progression, while others, such as pneumonia, have a very short duration. In the case of cardiovascular disease, there is an important question of whether air pollution is causing the disease, which would imply a relatively long cessation lag, or whether air pollution is causing premature death in individuals with preexisting heart disease, which would imply very short cessation lags (in theory, air pollution may both cause cardiovascular disease and cause premature death in those with preexisting cardiovascular disease). The SAB-HES provides several recommendations for future research that could support the development of defensible lag structures, including using disease-specific lag models and constructing a segmented lag distribution to combine differential lags across causes of death (EPA-SAB-COUNCIL-ADV-04-002, 2004). The SAB-HES indicated support for using “a Weibull distribution or a simpler distributional form made up of several segments to cover the response mechanisms outlined above, given our lack of knowledge on the specific form of the distributions” (EPA-SAB-COUNCIL-ADV-04-002, 2004, p. 24). However, they noted that “an important question to be resolved is what the relative magnitudes of these segments should be, and how many of the acute effects are assumed to be included in the cohort effect estimate” (EPA-SAB-COUNCIL-ADV-04-002, 2004, p. 24-25). Since the publication of that report in March 2004, EPA has sought additional clarification from this committee. In its follow-up advice provided in December 2004, this SAB suggested that until additional research has been completed, EPA should assume a segmented lag structure characterized by 30% of mortality reductions occurring in the first year, 50% occurring evenly over years 2 to 5 after the reduction in $PM_{2.5}$, and 20% occurring evenly over the years 6 to 20 after the reduction in $PM_{2.5}$ (EPA-COUNCIL-LTR-05-001, 2004). The distribution of deaths over the latency period is intended to reflect the contribution of short-term exposures in the first year, cardiopulmonary deaths in the 2- to 5-year period, and long-term lung disease and lung cancer in the 6- to 20-year period. Furthermore, in their advisory letter, the SAB-HES recommended that EPA include sensitivity analyses on other possible lag structures. In this appendix, we investigate the sensitivity of premature mortality-reduction related benefits to alternative cessation lag structures, noting that ongoing and future research may result in changes to the lag structure used for the main analysis.

In previous advice from the SAB-HES, they recommended an analysis of 0-, 8-, and 15-year lags, as well as variations on the proportions of mortality allocated to each segment in the segmented lag structure (EPA-SAB-COUNCIL-ADV-00-001, 1999, (EPA-COUNCIL-LTR-05-001, 2004). The 0-year lag is representative of EPA’s assumption in previous RIAs. The 8- and 15-year lags are based on the study periods from the Pope et al. (1995) and Dockery et al. (1993) studies, respectively.[1] However, neither the Pope et al. nor Dockery et al. studies assumed any lag structure when estimating the relative risks from PM exposure. In fact, the Pope et al. and Dockery et al. analyses do not supporting or refute the existence of a lag. Therefore, any lag structure applied to the avoided incidences estimated from either of these studies will be an assumed structure. The 8- and 15-year lags implicitly assume that all premature mortalities occur at the end of the study periods (i.e., at 8 and 15 years).

[1] Although these studies were conducted for 8 and 15 years, respectively, the choice of the duration of the study by the authors was not likely due to observations of a lag in effects but is more likely due to the expense of conducting long-term exposure studies or the amount of satisfactory data that could be collected during this time period.

In addition to the simple 8- and 15-year lags, we have added several additional sensitivity analyses examining the impact of assuming different allocations of mortality to the segmented lag of the type suggested by the SAB-HES. The first alternate lag structure assumes that more of the mortality impact is associated with chronic lung diseases or lung cancer and less with acute cardiopulmonary causes. This illustrative lag structure ("alternate segmented") is characterized by 20% of mortality reductions occurring in the first year, 50% occurring evenly over years 2 to 5 after the reduction in $PM_{2.5}$, and 30% occurring evenly over the years 6 to 20 after the reduction in $PM_{2.5}$. The second alternate lag structure ("5-year distributed") assumes the 5-year distributed lag structure used in previous analyses, which is equivalent to a three-segment lag structure with 50% in the first 2-year segment, 50% in the second 3-year segment, and 0% in the 6- to 20-year segment. The third alternate lag structure assumes a smooth negative exponential relationship between the reduction in exposure and the reduction in mortality risk, which is described in more detail below.

In 2004, SAB-HES (U.S. EPA-SAB, 2004) urged EPA to consider using smoothed lag distributions, incorporating information from the smoking cessation literature. In June 2010, the SAB-HES provided additional advice regarding alternate cessation lags (U.S. EPA-SAB, 2010). For $PM_{2.5}$-related benefits, the SAB-HES continued to support the previous 20-year distributed lag as the main estimate. while recommending that EPA further examine additional exponential decay functions. Specifically, the SAB-HES suggested varying the rate constant with the risk coefficient from in the cohort studies. EPA intends to incorporate these new alternate cessation lag for $PM_{2.5}$-related benefits in the final PM NAAQS RIA.

In response to these suggestions, EPA identified Röösli et al. (2005) as model that combines empirical data on the relationship between changes in exposure and changes in mortality and the timing of the cessation of those effects for the smooth decay function.[2] Because an exponential model is often observed in biological systems, Röösli et al. (2005) developed a dynamic model that assumes that mortality risks decrease exponentially after exposure termination. This model assumes the form risk=$\exp^{-kt}$, where k is the time constant and t is the time after t_0. The relative risk from air pollution (RR) at a given time (t) can be calculated from the excess relative risk (ERR) attributable to air pollution from PM cohort studies ($ERR=RR-R_0$), as follows:

[2] In the 2006 PM NAAQS RIA (U.S. EPA, 2006), EPA applied equations and the time constant from a conference presentation by Röösli et al. (2004). We have updated this sensitivity analysis in this assessment to reflect the published version in Röösli et al. (2005) and generated additional time constants.

$$RR(t) = ERR \times exp^{-kt} + R_0 \quad (5.A.1)$$

where R_0 is the baseline relative risk in the absence of air pollution (R_0=1). After cessation of exposure, mortality will start to decline and approach the baseline level. The change in mortality (ΔM), in units of percent-years, can be derived from Equation (5.A.1) as follows:

$$\Delta M = ERR \times t - \int_0^t ERR \times exp^{-kt} dt \quad (5.A.2)$$

Integrating Equation (5.A.2) gives:

$$\Delta M = ERR \times t - \frac{ERR}{k} + \frac{ERR}{k} \times exp^{-kt} \quad (5.A.3)$$

In order to calculate values for the time constant, k, we applied the ΔM values from the two intervention studies that provide data on the time course of the change in mortality along with the ERR values from cohort studies on $PM_{2.5}$-related mortality. We applied the intervention studies by Clancy et al. (2002), which analyzed the change in mortality following the ban of coal sales in Dublin, and by Pope et al. (1992), which examined the change in mortality resulting from the closure of a steel mill in the Utah Valley. We applied effect estimates from the American Cancer Society (ACS) cohort by Krewski et al. (2002)[3] and the Six Cities cohort by Laden et al. (2006).[4] Applying combinations of these studies to equation 5.A.3 generates four estimates of k that range from 0.05 to 1.24. For additional context, the time constant calculated using on a smoking cessation study (i.e., Leksell and Rabl (2001)) is in the middle of this range (k=0.10). For this sensitivity analysis, we applied a time constant of k=0.45 as a reasonable parameter for the exponential decay function, but we acknowledge the range of estimates that we could have chosen. This k constant is calculated as the average of the average k constants corresponding to each cohort study.[5] Table 5.A.1 provides the time constants for each of these combinations and averages, and Figure 5.A.2 illustrates the exponential decay lag structures.

[3] The relative risk coefficient from Krewski et al. (2009) (1.06 per 10 µg/m^3 change in average $PM_{2.5}$ exposure for all-cause mortality) is the same as the risk estimate from Pope et al. (2002).

[4] We have not updated this analysis to reflect the newest Six Cities cohort from Lepeule et al. (2012). While the relative risk coefficient from Lepeule et al. (2012) is slightly less than the relative risk coefficient from Laden et al. (2006), this difference is unlikely to have a substantial difference in the value of k.

[5] The general approach for calculating the time constants based on the combination of the intervention study and cohort study is consistent with the 812 analysis (U.S. EPA, 2011), which was reviewed by SAB. However, in this analysis we have applied a single time constant (k=0.45) rather than presenting the monetized benefits results for every exponential lag function applying the various time constants.

Table 5.A-1. Values of the Time Constant (k) for the Exponential Decay Lag Function

Value of k	$PM_{2.5}$ Cohort Study	Intervention Study
0.05	H6C—Laden et al. (2006)	Dublin—Clancy et al. (2002)
0.15	ACS—Krewski et al. (2009)	Dublin—Clancy et al. (2002)
0.37	H6C—Laden et al. (2006)	Utah Valley—Pope et al. (1992)
1.24	ACS—Krewski et al. (2009)	Utah Valley—Pope et al. (1992)
0.70	Average k for ACS—Krewski et al. (2009)	
0.21	Average k for H6C—Laden et al. (2006)	
0.45	Average of average k for each cohort study	

The estimated impacts of alternative lag structures on the monetary benefits associated with reductions in PM-related premature mortality (estimated using the effect estimate from Krewski et al. (2009)) are presented in Table 5.A-2. These monetized estimates are calculated using the value of a statistical life (i.e., $6.3 million per incidence adjusted for inflation and income growth) and are presented for both a 3 and 7% discount rate over the lag period). The choice of mortality risk study and mortality valuation approach are described in detail in Chapter 5 of this RIA. Figure 5.A.1 illustrates the cumulative distributions of the cessation lags applied in this appendix. Because we applied an income adjustment factor specific to the analysis year (see section 5.6.8 of this RIA), we do not adjust for income growth over the 20-year cessation lag. This approach could underestimate the benefits for the later years of the lag.

The results of this sensitivity analyses demonstrate that because of discounting of delayed benefits, the lag structure may also have a large impact on monetized benefits, reducing benefits by 27% if an extreme assumption that no effects occur until after 15 years is applied at a 3% discount rate and 53% at a 7% discount rate. However, for most reasonable distributed lag structures, differences in the specific shape of the lag function have relatively small impacts on overall benefits. For example, the overall impact of moving from the previous 5-year distributed lag to the segmented lag recommended by the SAB-HES in 2004 in the main estimate is relatively modest, reducing benefits by approximately 5% when a 3% discount rate is used and 9% when a 7% discount rate is used. If no lag is assumed, benefits are increased by approximately 10% relative to the segmented lag at a 3% discount rate and 22% at a 7% discount rate.

Table 5.A-2. Sensitivity of Monetized $PM_{2.5}$-Related Premature Mortality Benefits to Alternative Cessation Lag Structures, Using Effect Estimate from Krewski et al. (2009)

Alternative Lag Structures for PM-Related Premature Mortality		12 μg/m³	
		Value (billion 2006$)[a,b]	Percent Difference from Base Estimate
SAB Segmented (Main estimate)	30% of incidences occur in 1st year, 50% in years 2 to 5, and 20% in years 6 to 20		
	3% discount rate	$4.0	N/A
	7% discount rate	$3.6	N/A
No lag	Incidences all occur in the first year		
	3% discount rate	$4.4	10.4%
	7% discount rate	$4.4	22.5%
8-year	Incidences all occur in the 8th year		
	3% discount rate	$3.6	−10.3%
	7% discount rate	$2.7	−23.7%
15-year	Incidences all occur in the 15th year		
	3% discount rate	$2.9	−27.0%
	7% discount rate	$1.7	−52.5%
Alternative Segmented	20% of incidences occur in 1st year, 50% in years 2 to 5, and 30% in years 6 to 20		
	3% discount rate	$3.8	−3.2%
	7% discount rate	$3.3	−6.6%
5-Year Distributed	50% of incidences occur in years 1 and 2 and 50% in years 2 to 5		
	3% discount rate	$4.2	4.9%
	7% discount rate	$3.9	9.4%
Exponential Decay (k=0.45)	Incidences occur at an exponentially declining rate		
	3% discount rate	$4.2	5.0%
	7% discount rate	$3.9	9.9%

[a] Dollar values rounded to two significant digits. The percent difference using the monetized benefits estimated using the Lepeule et al. (2012) study would be identical, but the value would be approximately 2.3 times higher.

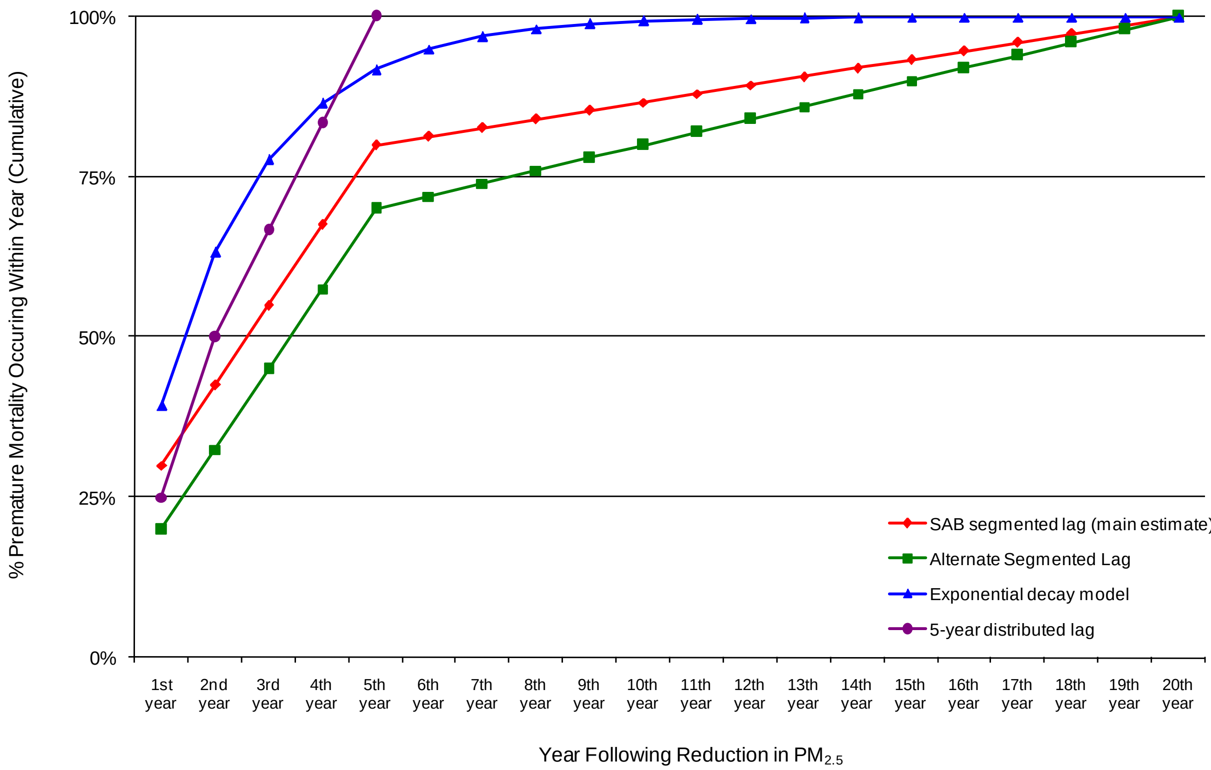


Figure 5.A-1. Alternate Lag Structures for $PM_{2.5}$ Premature Mortality (Cumulative)

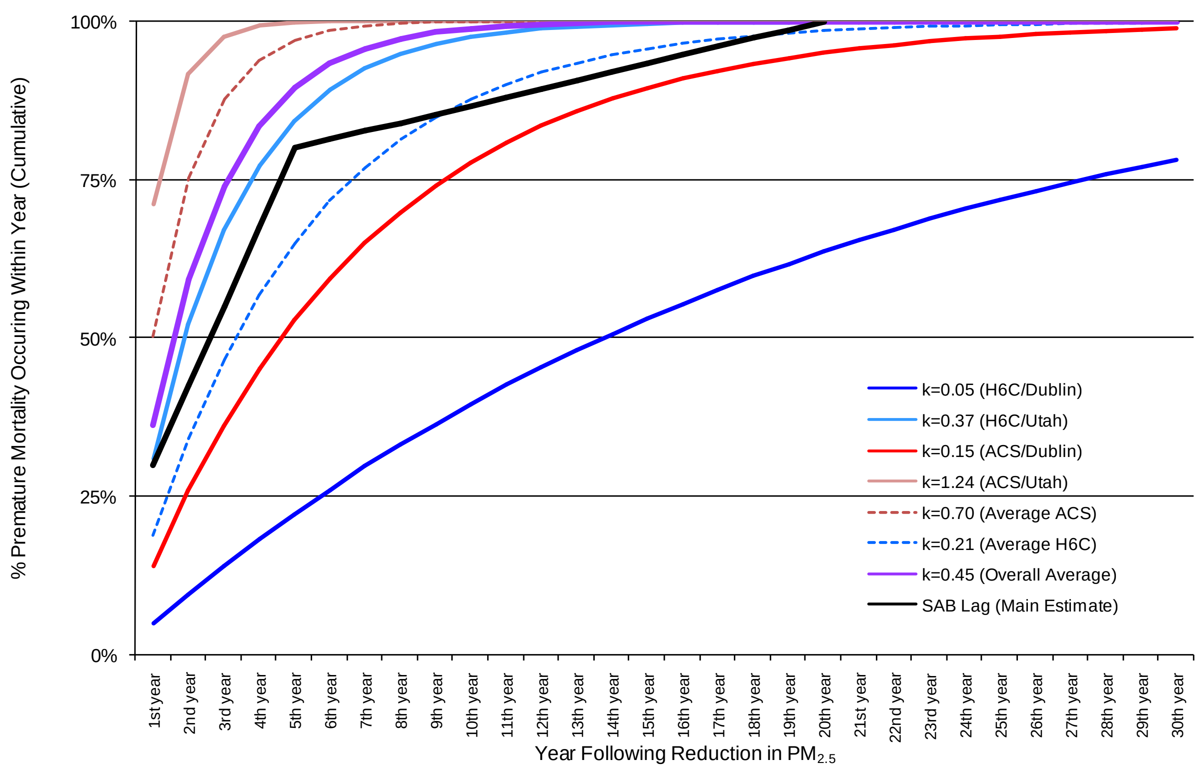


Figure 5.A-2. Exponential Lag Structures for $PM_{2.5}$ Premature Mortality (Cumulative)

5.A.2 Income Elasticity of Willingness to Pay

As discussed in Chapter 5, our estimates of monetized benefits account for growth in real GDP per capita by adjusting the WTP for individual endpoints based on the central estimate of the adjustment factor for each of the categories (minor health effects, severe and chronic health effects, premature mortality, and visibility). We examined how sensitive the estimate of total benefits is to alternative estimates of the income elasticities. Table 5.A-3 lists the ranges of elasticity values used to calculate the income adjustment factors, while Table 5.A-4 lists the ranges of corresponding adjustment factors. The results of this sensitivity analysis, giving the monetized benefit subtotals for the four benefit categories, are presented in Table 5.A-5.

Table 5.A-3. Ranges of Elasticity Values Used to Account for Projected Real Income Growth[a]

Benefit Category	Lower Sensitivity Bound	Upper Sensitivity Bound
Minor Health Effect	0.04	0.30
Premature Mortality	0.08	1.00

[a] Derivation of these ranges can be found in Kleckner and Neumann (1999). COI estimates are assigned an adjustment factor of 1.0.

Table 5.A-4. Ranges of Adjustment Factors Used to Account for Projected Real Income Growth[a]

Benefit Category	Lower Sensitivity Bound	Upper Sensitivity Bound
Minor Health Effect	1.018	1.147
Premature Mortality	1.037	1.591

[a] Based on elasticity values reported in Table C-4, U.S. Census population projections, and projections of real GDP per capita.

Table 5.A-5. Sensitivity of Monetized Benefits to Alternative Income Elasticities[a]

Benefit Category	Benefits Incremental to Analytical Baseline (Millions of 2006$) 12 μg/m³ No adjustment	Lower Sensitivity Bound	Upper Sensitivity Bound
Minor Health Effect	$30	$31	$35
Premature Mortality [b]	$3,600	$3,800	$5,800

[a] All estimates rounded to two significant digits.

[b] Using mortality effect estimate from Krewski et al. (2009) and 3% discount rate. Results using Lepeule et al. (2012) or a 7% discount rate would show the same proportional range.

Consistent with the impact of mortality on total benefits, the adjustment factor for mortality has the largest impact on total benefits. The value of mortality in 2020 ranges from 86% to 133% of the main estimate for mortality based on the lower and upper sensitivity bounds on the mortality income adjustment factor. The effect on the value of minor health effects is much less pronounced, ranging from 96% to 108% of the main estimate for minor effects.

5.A.3 Analysis of Cardiovascular Emergency Department Visits, Cerebrovascular Events and Chronic Bronchitis

Below we summarize the results of a sensitivity analysis of three health endpoints: cardiovascular emergency department visits, cerebrovascular events (stroke) and chronic bronchitis (Table 5.A-6). While in the benefits chapter we provide a full description of the rationale for treating these endpoints only in a sensitivity analysis, it is worth summarizing these reasons here. In the case of cardiovascular emergency department visits, we lack the necessary economic valuation functions to quantify the monetary value of these avoided cases. We treat cerebrovascular events as a sensitivity analysis for three reasons: (1) the epidemiological literature examining PM-related cerebrovascular events is still evolving; (2) there are special uncertainties associated with quantifying this endpoint; (3) we have not yet identified an appropriate means for estimating the economic value of this endpoint. Finally, we now quantify chronic bronchitis in a sensitivity analysis because of the absence of newer studies finding a relationship between long-term $PM_{2.5}$ exposure and this endpoint, and the relative weakness of the study available to quantify this endpoint.

To quantify cardiovascular hospital admissions, we apply risk coefficient drawn from three epidemiology studies: Metzger et al. (2004) (RR= 1.033, 95th percentile confidence intervals 1.01–1.056 per 10 μg/m^3 $PM_{2.5}$, age 0-99), Tolbert et al. (2007) (RR= 1.005, 95th percentile confidence intervals 0.993–1.017 per 10 μg/m^3 $PM_{2.5}$, age 0-99), and Mathes et al. (2011) (excess risk =0.8%, 95th percentile confidence intervals 0.0%-1.6% per 10 μg/m^3 $PM_{2.5}$, age 40-99) . To estimate cerebrovascular events, we apply a risk coefficient drawn from Miller et al. (2007) (RR= 1.28, 95% confidence intervals 1.02–1.61 per 10 μg/m^3 $PM_{2.5}$). To estimate chronic bronchitis, we use a risk coefficient drawn from Abbey et al. (1995) (RR= 1.81, 95% confidence intervals 0.98–3.25 per 45 μg/m^3 $PM_{2.5}$). Additional information, including the rationale for incorporating these new endpoints into the analysis, the baseline incidence rates for these endpoints, and the prevalence rate for chronic bronchitis are described in Chapter 5 of this RIA.

Table 5.A-6. Avoided Cases of Cardiovascular Emergency Department Visits, Stroke and Chronic Bronchitis in 2020 (95th percentile confidence intervals)[a]

Endpoint	12 μg/m^3
Cardiovascular emergency department visits	
Metzger et al. (2004) (ages 0–99)	440 (160–720)
Tolbert et al. (2007) (ages 0–99)	62 (−72–200)
Mathes et al. (2011) (ages 40-99)	101 (8-193)
Stroke	
Miller et al. (2007) (ages 50–79)	130 (20–230)
Chronic Bronchitis	
Abbey et al. (1995) (ages 27–99)	360 (39–670)

[a] All estimates rounded to two significant digits.

5.A.4 Updating Basis for Population Projections to 2010 Census

In this RIA, we updated the population demographic data in BenMAP to reflect the 2010 Census and future projections based on economic forecasting models developed by Woods and Poole, Inc. (Woods and Poole, 2012). These data replace the earlier demographic projection data from Woods and Poole (2007) that were projected from the 2000 Census. We use projections based on economic forecasting models developed by Woods and Poole, Inc. (Woods and Poole, 2012). Table 5.A-7 provides the results of a sensitivity analysis using the older population projections compared to the newer projections.

Table 5.A-7. Change in Total Monetized Benefits for 2000 and 2010 Census projections for 12 µg/m^3 in 2020 (2010$)[a]

	Projections from 2000 Census	Projections from 2010 Census	Percent Change
Total Monetized Benefits (3% discount rate)	$3.8 to $8.7 billion	$4.0 to $9.1 billion	+4.4%

[a] Percent change is based on the unrounded estimates.

5.A.5 Long-term $PM_{2.5}$ Mortality Estimates Using Cohort Studies in California

In Chapter 5, we described the multi-state cohort studies we used to estimate the $PM_{2.5}$-related mortality (i.e., Krewski et al., 2009; Lepeule et al., 2012), as well as summarized the effect estimates for additional cohort studies. In this appendix, we provide additional information about cohort studies in California.[6] As shown in Table 5.8 in the health benefits chapter, all of the monetized human health benefits associated with the illustrative control strategy to attain the revised annual standard of 12 µg/m^3 are projected to occur in California. For this reason, we determined that it was appropriate to consider the sensitivity of the benefits results using effect estimates for cohorts in California specifically. Although we have not calculated the benefits results using these cohort studies, it is possible to use the effect estimates themselves to determine how much the monetized benefits in California would have changed if we used effect estimates from the California cohorts. Each of the California cohort studies are summarized in the PM ISA (U.S. EAP, 2009) or the *Provisional Assessment* (U.S. EPA, 2012). Table 5.A.8 provides the effect estimates from each of these cohort studies for all-cause, cardiovascular, cardiopulmonary, and ischemic heart disease (IHD) mortality for each of the California cohort studies.

[6] In addition to cohorts studies conducted in California, we have also identified a cross-sectional studies (Hankey et al., 2012). However, we have not summarized that study here.

Table 5.A-8 Summary of Effect Estimates From Associated With Change in Long-Term Exposure to $PM_{2.5}$ in Recent Cohort Studies in California

		Hazard Ratios per 10 μg/m^3 Change in $PM_{2.5}$ (95[th] percentile confidence interval)		
Authors	**Cohort**	**All Causes**	**Cardiopulmonary**	**Ischemic Heart Disease**
McDonnell et al. (2000)[a]	Adventist Health Study (AHS) cohort (age > 27)	1.09 (.98–1.24)	N/A	N/A
Jerrett et al. (2005)[b]	Subset of the ACS cohort living in the Los Angeles metropolitan area (age > 30)	1.15 (1.03–1.29)	1.10 (0.94–1.28)	1.32 (1.03–1.29)
Chen et al. (2005)[c]	Adventist Health Study (AHS) cohort living in San Francisco, South Coast (i.e., Los Angeles and eastward), and San Diego air basins (age > 25)	N/A	N/A	1.42 (1.06–1.90)
Enstrom et al. (2005)[d]	California Prevention Study (age >65)	1.04 (1.01–1.07)	N/A	N/A
Krewski et al. (2009)[e]	Subset of the ACS cohort living in the 5-county Los Angeles Metropolitan Statistical Area (age > 30)	1.42 (1.26–1.27)	1.11 (0.95–1.23)	1.32 (1.06–1.64)
Ostro et al. (2010)[c]	California Teacher's study. Current and former female public school professionals (age > 22)	1.84 (1.66–2.05)	2.05 (1.80–2.36)	2.89 (2.27–3.67)
Ostro et al. (2011)[c,f]		1.06 (0.96–1.16)	1.19 (1.05–1.36)	1.55 (1.24–1.93)
Lipsett et al. (2011)[c]	California Teacher's study. Current and former female public school professionals (age > 22)	1.01 (0.95–1.09)	N/A	1.20 (1.02–1.41)

[a] Table 3, adjusted for 10 μg/m^3 change in $PM_{2.5}$.

[b] Table 1. 44 individual-level co-variates + all social (i.e., ecologic) factors specified (principal component analysis).

[c] Women only.

[d] Represents deaths occurring from 1973–1982, but no significant associations were reported with deaths in later time periods. The PM ISA (U.S. EPA, 2009) concludes that the use of average values for California counties as exposure surrogates likely leads to significant exposure error, as many California counties are large and quite topographically variable.

[e] Table 23. 44 individual-level co-variates + all social (i.e., ecologic) factors specified.

[f] Erratum Table 2.

As shown in Table 5.A.8, most of the cohort studies conducted in California report central effect estimates similar to the (nationwide) all-cause mortality risk estimate we applied from Krewski et al. (2009) and Lepeule et al. (2012) albeit with wider confidence intervals. Three cohort studies conducted in California indicate statistically significant higher risks than the risk estimates we applied from Lepeule et al. (2012), and four studies showed insignificant results.

5.A.6 Analysis of Health Benefits Estimated for 2025

In this RIA, we assumed an analysis year of 2020 for estimating costs and benefits with an adjustment to the San Joaquin and South Coast areas in California for NOx emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025. Because of population growth, population aging, and income growth over time, the health benefits estimated for an analysis year of 2025 would be higher. We have only conducted this sensitivity analysis for the revised standard of 12 $\mu g/m^3$, for which the health benefits occur entirely in California and are almost entirely in the South Coast and San Joaquin air basins. Table 5.A-9 provides the comparison of the health benefits (including avoided premature mortality and the total monetized health benefits) estimated for 2020 and 2025.

Table 5.A-9. Comparison of Health Benefits Estimated for 2000 and 2025 for 12 $\mu g/m^3$ [a]

	2020	2025	Percent Difference
Avoided Premature Mortality	460 to 1,000	510 to 1,200	+12%
Total Monetized Benefits (3% discount rate, 2010$)	$4.0 to $9.1 billion	$4.5 to $10 billion	+12%

[a] Percent change is based on the unrounded estimates.

5.A.7 References

Abbey, David E et al. 1995. “Chronic Respiratory Symptoms Associated with Estimated Long-Term Ambient Concentrations of Fine Particulates less than 2.5 Microns in aerodynamic diameter (PM2.5) and Other Air Pollutants.” *Journal of Exposure Analysis and Environmental Epidemiology* 5(2): 137-158.

Agency for Healthcare Research and Quality (AHRQ). 2000. *HCUPnet, Healthcare Cost and Utilization Project.* Rockville, MD. Available on the Internet at <http://hcupnet.ahrq.gov>.

Agency for Healthcare Research and Quality (AHRQ). 2007. *HCUPnet, Healthcare Cost and Utilization Project.* Rockville, MD. Available on the Internet at <http://hcupnet.ahrq.gov>.

Chen LH, Knutsen SF, Shavlik D, Beeson WL, Petersen F, Ghamsary M, et al. 2005. "The association between fatal coronary heart disease and ambient particulate air pollution: are females at greater risk?" *Environ Health Perspect* 113:1723–1729.

Clancy, L., P. Goodman, H. Sinclair, and D.W. Dockery. 2002. "Effect of Air-pollution Control on Death Rates in Dublin, Ireland: An Intervention Study." *Lancet* Oct 19;360(9341):1210-4.

Desvousges, W.H., F.R. Johnson, and H.S. Banzhaf. 1998. *Environmental Policy Analysis With Limited Information: Principles and Applications of the Transfer Method* (New Horizons in Environmental Economics.) Edward Elgar Pub: London.

Dockery, D.W., C.A. Pope, X.P. Xu, J.D. Spengler, J.H. Ware, M.E. Fay, B.G. Ferris, and F.E. Speizer. 1993. "An Association between Air Pollution and Mortality in Six U.S. Cities." *New England Journal of Medicine* 329(24):1753-1759.

Enstrom JE 2005. "Fine particulate air pollution and total mortality among elderly Californians, 1973-2002." *Inhal Toxicol* 17: 803-816.

Hankey, S., J. D. Marshall, et al. (2011). "Health Impacts of the Built Environment: Within-Urban Variability in Physical Inactivity, Air Pollution, and Ischemic Heart Disease Mortality." Environ Health Perspect 120(2).

Jerrett M, Burnett RT, Ma R, Pope CA III, Krewski D, Newbold KB, et al. 2005. "Spatial analysis of air pollution and mortality in Los Angeles." *Epidemiology* 16(6):727–736.

Kleckner, N., and J. Neumann. 1999. "Recommended Approach to Adjusting WTP Estimates to Reflect Changes in Real Income." Memorandum to Jim Democker, US EPA/OPAR. June 3, Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/klecknerandneumann1999.pdf>.

Krewski D, Jerrett M, Burnett RT, Ma R, Hughes E, Shi, Y, et al. 2009. "Extended follow-up and spatial analysis of the American Cancer Society study linking particulate air pollution and mortality." *HEI Research Report, 140*, Health Effects Institute, Boston, MA.

Laden F, Schwartz J, Speizer FE, Dockery DW. 2006. "Reduction in fine particulate air pollution and mortality: extended follow-up of the Harvard Six Cities study." *Am J Respir Crit Care Med* 173:667–672.

Lepeule J, Laden F, Dockery D, Schwartz J. 2012. "Chronic Exposure to Fine Particles and Mortality: An Extended Follow-Up of the Harvard Six Cities Study from 1974 to 2009." *Environ Health Perspect.* Jul;120(7):965-70.

Lipsett MJ*,* Ostro BD*,* Reynolds P et al. 2011. *"*Long-term exposure to air pollution and cardiorespiratory disease in the California teachers study cohort*." Am J Respir Crit Care Med* 184*:* 828–35*.*

Mathes, R.W., K. Ito, and T. Matte. 2011. Assessing Syndromic Surveillance of Cardiovascular Outcomes from Emergency Department Chief Complaint Data in New York City. PloS ONE. Vol 6(2): 1-10.

McDonnell WF; Nishino-Ishikawa N; Petersen FF; Chen LH; Abbey DE. 2000. "Relationships of mortality with the fine and coarse fractions of long-term ambient PM10 concentrations in nonsmokers." *J Expo Sci Environ Epidemiol*, 10: 427-436.

Metzger, K. B., P. E. Tolbert, et al. (2004). "Ambient air pollution and cardiovascular emergency department visits." *Epidemiology* 15(1): 46-56.

Miller, Kristin A., David S. Siscovick, Lianne Sheppard, Kristen Shepherd, Jeffrey H. Sullivan, Garnet L. Anderson, and Joel D. Kaufman. 2007. "Long-Term Exposure to Air Pollution and Incidence of Cardiovascular Events in Women." *New England Journal of Medicine*. 356 (5) :447-458.

Ostro B, Lipsett M, Reynolds P, Goldberg D, Hertz A, Garcia C, et al. 2010. "Long-Term Exposure to Constituents of Fine Particulate Air Pollution and Mortality: Results from the California Teachers Study." *Environ Health Perspect* 118:363-369.

Ostro B, Reynolds P, Goldberg D, Hertz A, et al. 2011. "Assessing Long-Term Exposure in the California Teachers Study." *Environ Health Perspect* June; 119(6): A242–A243.

Peters, A., D. W. Dockery, J. E. Muller and M. A. Mittleman. 2001. "Increased particulate air pollution and the triggering of myocardial infarction." *Circulation*. Vol. 103 (23): 2810-5.

Pope, C.A., III, M.J. Thun, M.M. Namboodiri, D.W. Dockery, J.S. Evans, F.E. Speizer, and C.W. Heath, Jr. 1995. "Particulate Air Pollution as a Predictor of Mortality in a Prospective Study of U.S. Adults." *American Journal of Respiratory Critical Care Medicine* 151:669-674.

Röösli M, Kunzli N, Braun-Fahrlander C. 2004. "Use of Air Pollution 'Intervention-Type' Studies in Health Risk Assessment." *16th Conference of the International Society for Environmental Epidemiology,* New York, August 1-4, 2004.

Röösli M, Kunzli N, Braun-Fahrlander C, Egger M. 2005. "Years of life lost attributable to air pollution in Switzerland: dynamic exposure-response model." *International Journal of Epidemiology* 34(5):1029-35.

Tolbert, P. E., M. Klein, et al. 2007. "Multipollutant modeling issues in a study of ambient air quality and emergency department visits in Atlanta." *J Expo Sci Environ Epidemiol* 17 Suppl 2: S29-35.

U.S. Environmental Protection Agency (U.S. EPA). 2009. *Integrated Science Assessment for Particulate Matter (Final Report)*. EPA-600-R-08-139F. National Center for Environmental Assessment—RTP Division. December. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=216546>.

U.S. Environmental Protection Agency (U.S. EPA). 2012. *Provisional Assessment of Recent Studies on Health Effects of Particulate Matter Exposure.* EPA/600/R-12/056A. National Center for Environmental Assessment—RTP Division. December.

U.S. Environmental Protection Agency—Science Advisory Board (EPA-SAB). 1999a. *The Clean Air Act Amendments (CAAA) Section 812 Prospective Study of Costs and Benefits (1999): Advisory by the Health and Ecological Effects Subcommittee on Initial Assessments of Health and Ecological Effects. Part 1.* EPA-SAB-COUNCIL-ADV-99-012. July. Available on the Internet at <http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/F64A21824D19766885257193005E51CA/$File/conadv12.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (EPA-SAB). 1999b. *The Clean Air Act Amendments (CAAA) Section 812 Prospective Study of Costs and Benefits (1999): Advisory by the Health and Ecological Effects Subcommittee on Initial Assessments of Health and Ecological Effects. Part 2.* EPA-SAB-COUNCIL-ADV-00-001. October. Available on the Internet at <http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/7D54675178C12703852571B90044B4F8/$File/coua0001.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (EPA-SAB). 2001. *Review of the Draft Analytical Plan for EPA's Second Prospective Analysis—Benefits and Costs of the Clean Air Act 1990-2020: An Advisory by a Special Panel of the Advisory Council on Clean Air Compliance Analysis.* EPA-SAB-COUNCIL-ADV-01-004. September. Available on the Internet at <http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/572DC1989A3986CC8525718D006BAB8B/$File/councila01004.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (EPA-SAB). 2004. *Advisory on Plans for Health Effects Analysis in the Analytical Plan for EPA's Second Prospective Analysis—Benefits and Costs of the Clean Air Act, 1990-2020: Advisory by the Health Effects Subcommittee of the Advisory Council on Clean Air Compliance Analysis.* EPA-SAB-COUNCIL-ADV-04-002. March. Available on the Internet at <http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/08E1155AD24F871C85256E5400433D5D/$File/council_adv_04002.pdf>.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 2004. *Advisory Council on Clean Air Compliance Analysis Response to Agency Request on Cessation Lag.* EPA-COUNCIL-LTR-05-001 December. Available on the Internet at <http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/39F44B098DB49F3C85257170005293E0/$File/council_ltr_05_001.pdf>.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 2010. *Review of EPA's DRAFT Health Benefits of the Second Section 812 Prospective Study of the Clean Air Act*. EPA-COUNCIL-10-001. June. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/9288428b8eeea4c885257242006935a3/59e06b6c5ca66597852575e7006c5d09!OpenDocument&TableRow=2.3#2.>.

APPENDIX 5.B

COMPREHENSIVE CHARACTERIZATION OF UNCERTAINTY IN BENEFITS ANALYSIS

As noted in Chapter 5, the benefits analysis relies on an array of data inputs—including air quality modeling, health impact functions and valuation estimates among others—which are themselves subject to uncertainty and may also in turn contribute to the overall uncertainty in this analysis. The analysis employs a variety of analytic approaches designed to reduce the extent of the uncertainty and/or characterize the impact that uncertainty has on the final estimate. We strive to incorporate as many quantitative assessments of uncertainty as possible (e.g., by using monte carlo assessments, concentration benchmark analyses, alternative concentration-response functions, sensitivity analyses, distributional assessments, and influence analyses); however, there are some aspects we are only able to characterize qualitatively.

To more comprehensively and systematically address these uncertainties, including those we cannot quantify, we adapt the WHO uncertainty framework (WHO, 2008), which provides a means for systematically linking the characterization of uncertainty to the sophistication of the underlying health impact assessment. EPA has applied similar approaches in peer-reviewed analyses of $PM_{2.5}$-related impacts (U.S. EPA, 2010b, 2011). EPA's Science Advisory Board (SAB) has supported using a tabular format to qualitatively assess the uncertainties inherent in the quantification and monetization of health impacts, including identifying potential bias, potential magnitude, confidence in our approach, and the level of quantitative assessment of each uncertainty (U.S. EPA-SAB, 1999, 2001, 2004, 2011a, 2011b). The assessments presented here are largely consistent with those previous peer-reviewed assessments.

5.B.1 Description of Classifications Applied in the Uncertainty Characterization

Table 5.B-1 catalogs the most significant sources of uncertainty in the PM benefits analysis and then characterizes four dimensions of that uncertainty. The first two dimensions focus on the nature of the uncertainty. The third and fourth dimensions focus on the extent to which the analytic approach chosen in the benefits analysis either minimizes the impact of the uncertainty or quantitatively characterizes its impact.

1) The direction of the bias that a given uncertainty may introduce into the benefits assessment if not taken into account in the analysis approach;

2) The magnitude of the impact that uncertainty is likely to have on the benefits estimate if not taken into account in the analysis approach;

3) The extent to which the analytic approach chosen is likely to minimize the impact of that uncertainty on the benefits estimate; and

4) The extent to which EPA has been able to quantify the residual uncertainty after the preferred analytic approach has been incorporated into the benefits model.

5.B.1.1 Direction of Bias

The "direction of bias" column in Table 5.B-1 is an assessment of whether, if left unaddressed, an uncertainty would likely lead to an underestimate or overestimate the total monetized benefits. In some cases we indicate that there are reasons why the bias might go either direction, depending upon the true nature of the underlying relationship. Where available, we base the classification of the "direction of bias" on the analysis in the Integrated Science Assessment for Particulate Matter (hereafter, "PM ISA") (U.S. EPA, 2009) . Additional sources of information include advice from SAB and the National Academies of Science (NAS), as well as studies from the peer-reviewed literature. In some cases we indicate that there is not sufficient information to estimate whether the uncertainty would likely lead to under or overestimation of benefits; these cases are identified as "unable to determine."

5.B.1.2 Magnitude of Impact

The "magnitude of impact" column in Table 5.B-1 is an assessment of how much plausible alternative assumptions about the underlying relationship about which we are uncertain could influence the overall monetary benefits. EPA has applied similar classifications in previous risk and benefit analyses (U.S. EPA, 2010b, 2011a), but we have slightly revised the category names and the cut-offs here.[1] The definitions used here are provided below.

- **High**—If the uncertainty associated with an assumption could influence the total monetized benefits by more than 25%.

- **Medium**—If the uncertainty associated with an assumption could influence the total monetized benefits by 5% to 25%.

- **Low**—If the uncertainty associated with an assumption could influence the total monetized benefits by less than 5%.

---

[1] In *The Benefits and Costs of the Clean Air Act from 1990 to 2020* (U.S. EPA, 2011a), EPA applied a classification of "potentially major" if a plausible alternative assumption or approach could influence the overall monetary benefit estimate by 5% percent or more and "probably minor" if an alternative assumption or approach is likely to change the total benefit estimate by less than five percent. In the *Quantitative Health Risk Assessment for Particulate Matter* (U.S. EPA, 2010b), EPA applied classifications of "low" if the impact would not be expected to impact the interpretation of risk estimates in the context of the PM NAAQS review, "medium" if the impact had the potential to change the interpretation; and "high" if it was are likely to influence the interpretation of risk in the context of the PM NAAQS review.

For each uncertainty, we provide as much quantitative information as is available in the table to support the classification.

Although many of the sources of uncertainty could affect both morbidity and mortality endpoints, because $PM_{2.5}$-related mortality benefits comprise over 98% of the monetized benefits that we are able to quantify in this analysis, uncertainties that affect the mortality estimate have the potential to have larger impacts on the total monetized benefits than uncertainties affecting only morbidity endpoints. One morbidity-related uncertainty that could have a significant impact on benefits estimate is the extent to which omitted morbidity endpoints are included in the benefits analysis. Including additional morbidity endpoints that are currently not monetized would reduce the fraction of total benefits from mortality. Ultimately, the magnitude classification is determined by professional judgment of EPA staff based on the results of available information, including other U.S. EPA assessments of uncertainty (U.S. EPA, 2010b, 2011)

Based on this assessment, the uncertainties which we classified as high or medium-high impact are: the causal relationship between $PM_{2.5}$ and mortality, regional differences in $PM_{2.5}$ mixtures, shape of the concentration-response function, mortality valuation, and cessation lag. The classification of these uncertainties as "high magnitude" is generally consistent with the results of EPA's *Influence Analysis* (Mansfield et al., 2008), the *Quantitative Health Risk Assessment for Particulate Matter* (U.S. EPA, 2010b), and the *Benefits and Costs of the Clean Air Act 1990 to 2020* (U.S. EPA, 2011).

5.B.1.3 Confidence in Analytic Approach

The "confidence in analytic approach" column of Table 5.B-1 is an assessment of the scientific support for the analytic approach chosen (or the inherent assumption made) to account for the relationship about which we are uncertain. In other words, based on the available evidence, how certain are we that EPA's selected approach is the most plausible of the potential alternatives. Similar classifications have been included in previous risk and benefits analyses (U.S. EPA, 2010b, 2011).[2] The three categories used to characterize the degree of confidence are:

- **High**—the current evidence is plentiful and strongly supports the selected approach;

[2] We have applied the same classification as *The Benefits and Costs of the Clean Air Act from 1990 to 2020* (U.S. EPA, 2011a) in this analysis. In the *Quantitative Health Risk Assessment for Particulate Matter* (U.S. EPA, 2010b), EPA assessed the degree of uncertainty (low, medium, or high) associated with the knowledge-base (i.e., assessed how well we understand each source of uncertainty), but did not provide specific criteria for the classification.

- **Medium**—some evidence exists to support the selected approach, but data gaps are present; and
- **Low**—limited data exists to support the selected approach.

Ultimately, the degree of confidence in the analytic approach is EPA staff's professional judgment based on the volume and consistency of supporting evidence, much of which has been evaluated in the PM ISA (U.S. EPA, 2009) and EPA's independent Science Advisory Board (SAB). The PM ISA evaluated the entire body of scientific literature on PM science and was twice peer-reviewed by EPA's Clean Air Scientific Advisory Committee (CASAC). In general, we regard a conclusion in the PM ISA or specific advice from SAB as supporting a high degree of confidence in the selected approach.

Based on this assessment, we have low or low-medium confidence in the evidence available to assess exposure error in epidemiology studies, morbidity valuation, baseline incidence projections for morbidity, and omitted morbidity endpoints. However, because these uncertainties have been classified as having a low or low-medium impact on the magnitude of the benefits, further investment in improving the available evidence would not have a substantial impact on the total monetized benefits.

5.B.1.4 Uncertainty Quantification

The column of Table 5.B-1 labeled "uncertainty quantification" is an assessment of the extent to which we were able to use quantitative methods to characterize the residual uncertainty in the benefits analysis, after addressing it to the extent feasible in the analytic approach for this RIA. We categorize the level of quantification using the four tiers used in the WHO uncertainty framework (WHO, 2008). The WHO uncertainty framework is a well-established approach to assess uncertainty in risk estimates that systematically links the characterization of uncertainty to the sophistication of the health impact assessment. The advantage of using this framework is that it clearly highlights the level of uncertainty quantification applied in this assessment and the potential sources of uncertainty that require methods development in order to assess quantitatively. Specifically, EPA applied this framework in the *Quantitative Risk Assessment for Particulate Matter* (U.S. EPA, 2010b), and it has been recommended in EPA guidance documents assessing air toxics-related risk and Superfund site risks (U.S. EPA, 2004b and 2001, respectively). Ultimately, the tier decision is professional judgment of EPA staff based on the availability of information for this assessment. The tiers used in this assessment are defined below.

- **Tier 0**—Screening level, generic qualitative characterization.

- **Tier 1**—Scenario-specific qualitative characterization.
- **Tier 2**—Scenario-specific sensitivity analysis.
- **Tier 3**—Scenario-specific probabilistic assessment of individual and combined uncertainty.

Within the limits of the data, we strive to use a more sophisticated approaches (e.g., Tier 2 or 3) for characterizing uncertainties that have the largest magnitudes and could not be completely addressed through the analytic approach. The uncertainties for which we have conducted probabilistic (Tier 3) assessments in this analysis are PM-mortality causality, the shape of the concentration-response function, and mortality and morbidity valuation. For lower magnitude uncertainties, we include qualitative discussions of the potential impact of uncertainty on risk results (WHO Tier 0/1) and/or completed sensitivity analyses assessing the potential impact of sources of uncertainty on risk results (WHO Tier 2).

5.B.2 Organization of the Qualitative Uncertainty Table

Table 5.B-1 is organized as follows. The uncertainties are grouped by category (i.e., concentration-response function, valuation, population and baseline incidence, omitted benefits categories, and exposure changes). Within each category, the uncertainties are sorted by magnitude of impact (i.e., high to low) then by confidence in our approach (i.e., low to high). In the table, red (bold) text is used to indicate the uncertainties that likely have a high magnitude of impact on the total benefits estimate. This organization highlights the uncertainty with the largest potential impact and the lowest confidence at the top of each category.

Table 5.B-1. Summary of Qualitative Uncertainty for Key Modeling Elements in $PM_{2.5}$ Benefits

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Uncertainties Associated with Concentration-Response Functions				
Variation in effect estimates reflecting differential toxicity of particle components and regional differences in $PM_{2.5}$ composition (mixtures)	Either direction, depending on the species.	Potentially **High**	Medium	Tier 2 (sensitivity analysis)
	PM composition and the size distribution of those particles vary within and between areas due to source characteristics. Any specific location could have higher or lower contributions of certain PM species and other pollutants than the national average, meaning potential regional differences in health impact of given control strategies. Depending on the toxicity of each PM species reduced in the control strategies, assuming equal toxicity could over or underestimate benefits.	Epidemiology studies examining regional differences in $PM_{2.5}$-related health effects have found differences in the magnitude of those effects, and composition remains one potential explanatory factor (PM ISA, section 2.3.2). In addition to differences in the contribution of any given species to the baseline concentrations, use of different control strategies would have a differing magnitude of the effect in different regions. Depending on the extent of the differences in toxicity and the exact mix if species controlled, different control strategies could have a differing magnitude of the effect in different regions.	Consistent with SAB advice, we assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality (U.S. EPA-SAB, 2010, pg. 18). The PM ISA concluded many compounds can be linked with multiple health effects and the evidence is not yet sufficient to allow differentiation of effects estimates by particle type (pg. 2-17). We also use national risk coefficients with no local variations due to differential exposure. The PM ISA states that available evidence and the limited amount of city-specific speciated $PM_{2.5}$ data does not allow differentiation of PM effects in different locations (pg. 2–17). Using national risk coefficients is supported by SAB (U.S. EPA-SAB, 2010) and NAS (NRC, 2002).	Regional differences in hazard ratios from studies conducted in California shown in Table 5.A-8. The hazard ratios from the California studies range from -83% to +1300% compared to the national estimate applied from Krewski et al. (2009).
Causal relationship between $PM_{2.5}$ exposure and premature mortality	Overestimate, if $PM_{2.5}$ does not have a causal relationship with premature mortality.	**High**	High	Tier 3 (probabilistic)
		Mortality generally dominates monetized benefits, so small uncertainties could have large impacts on the total monetized benefits.	Our approach is consistent with the PM ISA, which concluded that premature mortality is causally related to $PM_{2.5}$ exposure. This conclusion is based on the consistency of the effects observed across epidemiology studies and biological plausibility) (pg. 2–9, 2–11).In addition, in the $PM_{2.5}$ expert elicitation, 10 of 12 experts provided likelihood of causality of 90% or higher (Roman et al., 2008).	Each expert in the $PM_{2.5}$ expert elicitation had the opportunity to specify the likelihood of a causal relationship into their function (Roman et al., 2008). Using these expert-derived functions is a probabilistic assessment of causality (see Figure 5-4).

(continued)

Table 5.B-1. Summary of Qualitative Uncertainty for Key Modeling Elements in $PM_{2.5}$ Benefits (continued)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Uncertainties Associated with Concentration-Response Functions (cont'd)				
Causal relationship between $PM_{2.5}$ exposure and premature mortality	Overestimate, if $PM_{2.5}$ does not have a causal relationship with premature mortality.	**High** Mortality generally dominates monetized benefits, so small uncertainties could have large impacts on the total monetized benefits.	High Our approach is consistent with the PM ISA, which concluded that premature mortality is causally related to $PM_{2.5}$ exposure. This conclusion is based on the consistency of the effects observed across epidemiology studies and biological plausibility) (pg. 2–9, 2–11).In addition, in the $PM_{2.5}$ expert elicitation, 10 of 12 experts provided likelihood of causality of 90% or higher (Roman et al., 2008).	Tier 3 (probabilistic) Each expert in the $PM_{2.5}$ expert elicitation had the opportunity to specify the likelihood of a causal relationship into their function(Roman et al., 2008). Using these expert-derived functions is a probabilistic assessment of causality (see Figure 5-4).
Shape of the C-R functions, particularly at low concentrations	Either The direction of bias that assuming linear-no threshold model or alternative model introduces depends upon the "true" functional from of the relationship and the specific assumptions and data in a particular analysis. For example, if the true function identifies a threshold below which health effects do not occur, benefits may be overestimated if a substantial portion of those benefits were estimated to occur below that threshold. Alternately, if a substantial portion of the benefits occurred above that threshold, the benefits may be underestimated because an assumed linear no-threshold function may not reflect the steeper slope above that threshold to account for all health effects occurring above that threshold.	Medium-**High** Krewski et al. (2009) considered alternative model forms and found that the choice of functional relationship can make a considerable difference in the predicted risk at lower concentrations. Specifically, they found a 58% increase in risk at lower concentrations for all-cause mortality. The magnitude of this impact depends on the fraction of benefits occurring in areas with lower concentrations. Mortality generally dominates monetized benefits, so small uncertainties could have large impacts on total monetized benefits.	High Consistent with the PM ISA, we assume a log-linear no-threshold model for the concentration-response function. In previous RIAs (between 2006 and 2009), we assumed a threshold in the mortality relationship, which shifted the slope of the function to account for all health effects occurring above the threshold concentration. The PM ISA concluded that the studies overall support the use of a no-threshold log-linear model for $PM_{2.5}$-related mortality (pg. 2-25). Our approach also follows recommendations from the SAB (U.S. EPA-SAB, 2010, pg. 13).	Tier 3 (probabilistic) The experts in the $PM_{2.5}$ expert elicitation specified the shape of the C-R function (Roman et al., 2008). Only expert K assumed a threshold, but experts B, F, and L included different slopes at lower concentrations. Using the expert-derived functions is a probabilistic assessment of the shape of the C-R function (see Figure 5-4). Also, the concentration benchmark assessment shows the premature deaths estimated at various concentrations. Specifically, 92% of the monetized benefits occur at or above the lowest annual concentration of PM in the Krewski et al (2009) study.

(continued)

Table 5.B-1. Summary of Qualitative Uncertainty for Key Modeling Elements in $PM_{2.5}$ Benefits (continued)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Uncertainties Associated with Concentration-Response Functions (cont'd)				
Exposure error in epidemiology studies	Underestimate (generally) The PM ISA states that the results from the Krewski et al. (2009) and Jerrett et al. (2005) studies suggest that exposure error can underestimate effect estimates (pg. 7–90). The PM ISA states that exposure error can potentially bias an estimate of a health effect endpoint towards the null or increase the size of confidence intervals (pg. 3–152). The PM ISA states that reducing exposure error can result in stronger associations between pollutants and effect estimates than generally observed in studies having less exposure detail (pg. 7–90).	Medium Recent analyses reported in Krewski et al. (2009) demonstrate the potentially significant effect (approximately 18% increase in hazard ratio for all-cause mortality in Los Angeles by improving the exposure assessment) that this source of uncertainty can have on effect estimates. These analyses also illustrate the complexity and site-specific nature of this source of uncertainty.	Low-Medium Although this underestimation is well documented, including in the PM ISA, SAB has not suggested an approach to adjust for this bias.	Tier 1 (qualitative) *(No quantitative method available)*
Modification of Mortality C-R function by socio-economic status (SES)	Underestimate for ACS cohort (Krewski et al., 2009) because of the demographics of the cohort (NRC, 2002, pg. 101).	Medium for ACS cohort	Medium We have not modified the function for SES in the core analysis.	Tier 2 (sensitivity analysis)
	Unknown for Six Cities cohort (Lepeule et al., 2012), but educational attainment in cohort is likely more representative of the general population (NRC, 2002, pg. 101). Experts suggested that the educational attainment may also modify the risk in the Six Cities cohort, but estimates are not available, (IEc, 2006, pg. 3–16). Using both cohorts may balance any potential bias.	Krewski et al. (2009) found that educational attainment, which is a surrogate for SES, modifies the risk coefficient (i.e., ranging from -8% for individuals with more than Grade 12 to +37% for individual with less than Grade 12 after controlling for ecologic covariates compared to the national estimate). The overall impact would depend on the mixture of educational attainment in the target population. Unknown magnitude for Six Cities cohort.	The PM ISA concluded that there is evidence that SES, measured using surrogates such as educational attainment, modifies the association between PM and health effects (pg. 8–15), but gender (pg. 8–6) and race (pg. 8-7) do not seem to modify the association between PM and health effects. The PM ISA also concluded that some evidence suggests that Hispanic ethnicity may modify the relationship (pg. 8-7).	Effect modification for educational attainment evaluated in the distributional analysis in Appendix 5A of the proposal RIA. For 12/35, the percent of premature deaths ranged from 4.4% for individuals with more than Grade 12 education up to 6.5% for individuals with less than Grade 12 education.

(continued)

Table 5.B-1. Summary of Qualitative Uncertainty for Key Modeling Elements in $PM_{2.5}$ Benefits (continued)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Uncertainties Associated with Concentration-Response Functions (cont'd)				
Confounding by individual risk factors, other than SES—e.g., smoking, or ecologic factors, which represent the neighborhood, such as unemployment	Either, depending on the factor and study Individual, social, economic, and demographic covariates can bias the relationship between particulate air pollution and mortality, particularly in cohort studies that rely on regional air pollution levels.	Medium Because mortality dominates monetized benefits, even a small amount of confounding could have medium impacts on total monetized benefits.	Medium To minimize confounding effects, to the extent possible, we use risk coefficients that control for 44 individual and 7 ecologic factors from Krewski et al. (2009). Although Krewski et al. (2000, 2009) found that ecologic covariates did not exert a significant confounding influence on PM-related mortality, they highlighted the "vital need for further study of the role that ecologic covariates have in the association between air pollution and mortality." We use risk coefficients that control for 3 individual factors (e.g., BMI, smoking, and education) from Lepeule et al. (2012).	Tier 2 (sensitivity analysis) (Quantitative methods available but not assessed in this analysis.)
Confounding and effect modification by co-pollutants	Either, depending upon the pollutant. Disentangling the health responses of combustion-related pollutants (i.e., PM, SO_x, NO_x, ozone, and CO is a challenge. Ambient PM may be an indicator of complex mixtures that share emission sources (e.g., traffic and power generation). The PM ISA states that co-pollutants may mediate the effects of PM or PM may influence the toxicity of co-pollutants (pg. 1–16). Alternately, effects attributed to one pollutants may be due to another.	Medium Because this uncertainty could affect mortality and because mortality generally dominates monetized benefits, even a small uncertainties could have medium impacts on total monetized benefits.	Medium When modeling effects of pollutants jointly, we apply multi-pollutant effect estimates when those estimates are available to avoid double-counting when those estimates are available and satisfy other selection criteria. The PM ISA states that evidence from the limited number of studies suggests that gaseous co-pollutants do not confound the $PM_{2.5}$-mortality relationship (pg. 2–11). EPA's current approach to modeling has been supported during peer-reviews by SAB (U.S. EPA-SAB, 2010) and NAS (NRC, 2002).	Tier 1 (qualitative) *(No quantitative method available)*

(continued)

Table 5.B-1. Summary of Qualitative Uncertainty for Key Modeling Elements in $PM_{2.5}$ Benefits (continued)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Uncertainties Associated with Concentration-Response Functions (Cont'd)				
Not including short-term exposure studies in PM mortality calculations	Underestimate Long-term PM exposure studies likely capture a large part of the impact of short-term peak exposure on mortality; however, the extent of overlap between the two study types is unclear (NRC, 2002, pg. 116).	Medium If short-term mortality is not fully captured within the cohort mortality estimates, then the benefits could be underestimated.	Medium Consistent with the NAS, we assume that long-term cohort studies capture most of the mortality benefits. However, NAS acknowledges that the effects of short-term exposures are unlikely to be fully captured in the cohort studies (NRC, 2002, pgs. 108, 116).	Tier 1 (qualitative) *(No quantitative method available)*
Impact of historical exposure on mortality effect estimates	Either Long-term studies of mortality suggest that different periods of PM exposure can produce different effects estimates, raising the issue of uncertainty in relation to determining which exposure window to use when estimating mortality benefits.	Low-Medium Krewski et al. (2009) evaluated exposure windows for PM-related mortality and suggested that differences in effect estimates are associated with the use of different exposure periods (with the more recent period having larger estimates) but those differences were small. Lepeule et al. (2012) reported similar findings for different exposure periods. Mortality generally dominates monetized benefits, so small uncertainties could have medium impacts on total monetized benefits.	Medium We do not make adjustments for temporal variation in exposure. The PM ISA concludes that the overall evidence for determining the appropriate exposure window suggests that the health benefits from reducing air pollution do not require a long latency period and would be expected within a few years after intervention (pg. 7–95).	Tier 2 (sensitivity analysis) (Quantitative methods available but not assessed in this analysis.)
Application of C-R relationships only to the original study population	Underestimate Estimating health effects for only to the original study population may underestimate the whole population benefits of reductions in pollutant exposures.	Low Mortality generally dominates monetized benefits, so further age range expansions for morbidity would have a small impact on total monetized benefits. With respect to adult mortality—the baseline rate is significantly lower under age 25, so lowering the age range to 18 would have minimal impact on total monetized benefits.	High For mortality, we estimate health effects only for the ages matching the original study population (i.e., 25+ or 30+). Following advice from the SAB (U.S. EPA-SAB, 2004a, pg. 7) and NAS (NRC, 2002, pg. 114), we expanded the age range for childhood asthma exacerbations beyond the original study population to ages 6-18.	Tier 2 (sensitivity analysis) (Quantitative methods available but not assessed in this analysis.)

(continued)

Table 5.B-1. Summary of Qualitative Uncertainty for Key Modeling Elements in $PM_{2.5}$ Benefits (continued)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Uncertainties Associated with Economic Valuation				
Mortality Risk Valuation / Value-of-a-Statistical-Life (VSL)	Unknown Some studies suggest that EPA's mortality valuation is too high, while other studies suggest that it is too low. Differences in age, income, risk aversion, altruism, nature of risk (e.g., cancer), and study design could lead to higher or lower estimates of mortality valuation.	**High** Mortality generally dominates monetized benefits, so moderate uncertainties could have a large effect on total monetized benefits.	Medium The VSL used by EPA is based on 26 labor market and stated preference studies published between 1974 and 1991. EPA is in the process of reviewing this estimate and will issue revised guidance based on the most up-to-date literature and recommendations from the SAB-EEAC in the near future (U.S. EPA, 2010a, U.S. EPA-SAB, 2011c).	Tier 3 (probabilistic) Assessed uncertainty in mortality valuation using a Weibull distribution.
Cessation lag structure for long term PM mortality	Underestimate Recent studies (Schwartz, 2008; Puett et al., 2009; Lepeule et al., 2012) estimate that the majority of the risk occurs within 2 years of reduced exposure. Because we do not adjust for income growth over the 20-year cessation lag, this approach could also underestimate the benefits for the later years of the lag.	Medium-**High** Although the cessation lag does not affect the number of premature deaths attributable to PM2.5 exposure, it affects the timing of those deaths and thus the discounted monetized benefits. Mortality generally dominates monetized benefits, so moderate uncertainties could have a large effect on total monetized benefits.	Medium Consistent with SAB advice, we estimate that 30% of mortality reductions in the first year, 50% over years 2 to 5, and 20% over the years 6 to 20 after the reduction in PM2.5 (U.S. EPA-SAB, 2004c). The PM ISA concludes that health benefits from reducing air pollution would be expected within a few years of intervention (pg. 7–95). Despite recent studies providing new evidence of the timing of mortality risk reduction after changes in exposure, the SAB did not suggest changes to the default cessation lag applied in EPA's main benefits estimates in the most recent review (U.S. EPA-SAB, 2010).	Tier 2 (sensitivity analysis) As shown in Appendix 5-A, the use of an alternate lag structure would change the monetized benefits by +10% to -52% depending on the discount rate and lag structure assumed.

(continued)

Table 5.B-1. Summary of Qualitative Uncertainty for Key Modeling Elements in $PM_{2.5}$ Benefits (continued)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Uncertainties Associated with Economic Valuation (cont'd)				
Income growth adjustments	Either Income growth increases willingness-to-pay (WTP) valuation estimates, including mortality, over time. From 1997 to 2010, personal income and GDP growth have begun to diverge. If this trend continues, the assumption that per capita GDP growth is a reasonable proxy for income growth may lead to an overstatement of benefits. (IEc, 2012).	Medium Income growth from 1990 to 2020 increases mortality valuation by 20%. Alternate estimates for this adjustment vary by 20% (IEc, 2012). Because we do not adjust for income growth over the 20-year cessation lag, this approach could also underestimate the benefits for the later years of the lag.	Medium Consistent with SAB recommendations (U.S. EPA,-SAB, 2000, pg. 16), we adjust WTP for income growth. Difficult to forecast future income growth. However, in the absence of readily available income data projections, per capita GDP is the best available option.	Tier 2 (sensitivity analysis) As shown in Appendix 5-A, the use of alternate income growth adjustments would change the monetized benefits by +33% to −14%.
Morbidity valuation	Underestimate Morbidity benefits such as hospital admissions and heart attacks are calculated using cost-of-illness (COI) estimates, which are generally half the willingness-to-pay to avoid the illness (Alberini and Krupnick, 2000). In addition, the morbidity costs do not reflect physiological responses or sequelae events, such as increased susceptibility for future morbidity.	Low Even if we doubled the monetized valuation of morbidity endpoints using COI valuation that are currently included in the RIA, the change would still be less than 5% of the monetized benefits. It is unknown how much including sequelae events could increase morbidity valuation.	Low Although the COI estimates for hospitalizations reflect recent data, we have not yet updated other COI estimates such as for AMI. The SAB concluded that COI estimates could be used as placeholders where WTP estimates are unavailable, but it is reasonable to presume that this strategy typically understates WTP values (U.S. EPA-SAB, 2004b, pg. 3).	Tier 3 (probabilistic), where available Assessed uncertainty in morbidity valuation using distributions specified in the underlying literature, where available (see Table 5.9).

(continued)

Table 5.B-1. Summary of Qualitative Uncertainty for Key Modeling Elements in $PM_{2.5}$ Benefits (continued)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Uncertainties Associated with Baseline Incidence and Population Projections				
Population estimates and projections	Either The monetized benefits would change in the same direction as the over- or underestimate in population projections in areas where exposure changes.	Low–Medium Monetized benefits are substantially affected by population density. Comparisons using historical census data show that population projections are ±5% nationally, but projection accuracy can vary by locality. Historical error for Woods & Poole's population projections has been ±8.1% for county-level projections and ±4.1% for states (Woods and Poole, 2012). The magnitude of impact on total monetized benefits depends on the specific location where PM is reduced.	Medium We use population projections for 5-year increments for 304 race/ethnicity/gender/age groups (Woods and Poole, 2012) at Census blocks. Population forecasting is well-established but projections of future migration due to possible catastrophic events are not considered. In addition, projections at the small spatial scales used in this analysis are inherently more uncertain than projections at the county- or state-level.	Tier 1 (qualitative) *(No quantitative method available)*
Uncertainty in projecting baseline incidence rates for mortality	Unknown Because the mortality rate projections for future years reflect changes in mortality patterns as well as population growth, the projections are unlikely to be biased.	Low-Medium Because mortality generally dominates monetized benefits, small uncertainties could have medium impacts on total monetized benefits.	Medium The county-level baseline mortality rates reflect recent databases (i.e., 2004–2006 data) and projected for 5-year increments for multiple age groups. This database is generally considered to have relatively low uncertainty (CDC Wonder, 2008). The projections account for both spatial and temporal changes in the population.	Tier 1 (qualitative) *(No quantitative method available)*
Uncertainty in projecting baseline incidence rates and prevalence rates for morbidity	Either, depending on the health endpoint Morbidity baseline incidence is available for current year only (i.e., no projections available). Assuming current year levels can bias the benefits for a specific endpoint if the data has clear trends over time. Specifically, asthma prevalence rates have increased substantially over the past few years while hospital admissions have decreased substantially.	Low The magnitude varies with the health endpoint, but the overall impact on the total benefits estimate from these morbidity endpoints is likely to be low.	Low-Medium We do not have a method to project future baseline morbidity rates, thus we assume current year levels will continue. While we try to update the baseline incidence and prevalence rates as frequently as practicable, this does not continue trends into the future. Some endpoints such as hospitalizations and ER visits have more recent data (i.e., 2007) stratified by age and geographic location. Other endpoints, such as respiratory symptoms reflect a national average. Asthma prevalence rates reflect recent increases in baseline asthma rates (i.e., 2008).	Tier 1 (qualitative) *(No quantitative method available)*

(continued)

Table 5.B-1. Summary of Qualitative Uncertainty for Key Modeling Elements in $PM_{2.5}$ Benefits (continued)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Uncertainties Associated with Omitted Benefits Categories				
Unquantified PM health benefit categories, such as pulmonary function, cerebrovascular events, low birth weight, or cancer	Underestimate EPA has not included monetized estimates of these benefits categories in the core benefits estimate.	Medium Although the potential magnitude is unknown, including all of the additional morbidity endpoints associated with $PM_{2.5}$ exposure that are currently not monetized could increase the total benefits by a moderate amount.	Low Current data and methods are insufficient to develop (and value) national quantitative estimates of these health effects. The PM ISA determined that respiratory morbidity (e.g., decreases in lung function) is causally associated with $PM_{2.5}$ exposure (pg. 2–12).	Tier 2 (sensitivity analysis) In Table 5.A-6, EPA estimates avoided incidence of strokes, cardiovascular emergency department visits, and chronic bronchitis.
Uncertainties Associated with Estimated Exposure Changes				
Spatial matching of air quality estimates from epidemiology studies to air quality estimates from air quality modeling	Unknown Epidemiology studies often assume one air quality concentration is representative of an entire urban area when calculating hazard ratios, while benefits are calculated using air quality modeling conducted at 12 km spatial resolution. This spatial mismatch could introduce uncertainty.	Unknown	Low We have not controlled for this potential bias, and the SAB has not suggested an approach to adjust for this bias.	Tier 1 (qualitative) *(No quantitative method available)*

5.B.3 References

Alberini, Anna and Alan Krupnick. 2000. "Cost-of-Illness and Willingness-to-Pay Estimates of the Benefits of Improved Air Quality: Evidence from Taiwan." *Land Economics* Vol. 76, No. 1 (Feb), pp. 37-53.

Centers for Disease Control and Prevention (CDC). 2008. *National Center for Health Statistics. National Health Interview Survey, 1999-2008*.

Industrial Economics, Incorporated (IEc). 2006. *Expanded Expert Judgment Assessment of the Concentration-Response Relationship Between $PM_{2.5}$ Exposure and Mortality*. Prepared for: Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. September. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Uncertainty/pm_ee_tsd_expert_interview_summaries.pdf>.

Industrial Economics, Incorporated (IEc). 2012. *Updating BenMAP Income Elasticity Estimates—Literature Review.* Memo to Neal Fann. March. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/Income%20Elasticity%20Update_Recommendations_noAppD.pdf>

Jerrett M, Burnett RT, Ma R, et al. 2005. "Spatial analysis of air pollution and mortality in Los Angeles." *Epidemiology* 16:727-736.

Krewski D, Jerrett M, Burnett RT, Ma R, Hughes E, Shi, Y, et al. 2009. "Extended follow-up and spatial analysis of the American Cancer Society study linking particulate air pollution and mortality." HEI Research Report, 140, Health Effects Institute, Boston, MA.

Lepeule J, Laden F, Dockery D, Schwartz .J 2012. "Chronic Exposure to Fine Particles and Mortality: An Extended Follow-Up of the Harvard Six Cities Study from 1974 to 2009." *Environ Health Perspect.* 120: 965-970.

Mansfield, Carol; Paramita Sinha; Max Henrion. 2009. Influence Analysis in Support of Characterizing Uncertainty in Human Health Benefits Analysis: Final Report. Prepared for U.S. EPA, Office of Air Quality Planning and Standards. November. Available on the internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/influence_analysis_final_report_psg.pdf>.

National Research Council (NRC). 2002. *Estimating the Public Health Benefits of Proposed Air Pollution Regulations*. Washington, DC: The National Academies Press. Washington, DC.

Puett RC, Hart JE, Yanosky JD, Paciorek C, Schwartz J, Suh H, et al. 2009. "Chronic fine and coarse particulate exposure, mortality and coronary heart disease in the Nurses' Health Study." *Environ Health Perspect* 117:1697–1701.

Roman, Henry A., Katherine D. Walker, Tyra L. Walsh, Lisa Conner, Harvey M. Richmond, Bryan J. Hubbell, and Patrick L. Kinney. 2008. "Expert Judgment Assessment of the Mortality Impact of Changes in Ambient Fine Particulate Matter in the U.S." *Environ. Sci. Technol*., 42(7):2268-2274.

Schwartz J, Coull B, Laden F. 2008. "The Effect of Dose and Timing of Dose on the Association between Airborne Particles and Survival." *Environmental Health Perspectives* 116: 64-69.

U.S. Environmental Protection Agency (U.S. EPA). 2001. *Risk assessment guidance for Superfund. Vol. III, Part A. Process for conducting probabilistic risk assessment* (RAGS 3A). Washington, DC, United States Environmental Protection Agency (EPA 540-R-02-002; OSWER 9285.7-45; PB2002 963302. Available on the Internet at <http://www.epa.gov/oswer/riskassessment/rags3adt/index.htm>.

U.S. Environmental Protection Agency (U.S. EPA). 2004. *EPA's risk assessment process for air toxics: History and overview.* In: Air toxics risk assessment reference library. Vol. 1. Technical resource manual. Washington, DC, United States Environmental Protection Agency, pp. 3-1–3-30 (EPA-453-K-04-001A. Available on the Internet at <http://www.epa.gov/ttn/fera/data/risk/vol_1/chapter_03.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2009. *Integrated Science Assessment for Particulate Matter (Final Report)*. EPA-600-R-08-139F. National Center for Environmental Assessment—RTP Division. December. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=216546>.

U.S. Environmental Protection Agency (U.S. EPA). 2010a. *Valuing Mortality Risk Reductions for Environmental Policy: A White Paper (SAB Review Draft).* December. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0563-1.pdf/$file/EE-0563-1.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2010b. *Quantitative Health Risk Assessment for Particulate Matter—Final Repor*t. EPA-452/R-10-005. Office of Air Quality Planning and Standards, Research Triangle Park, NC. September. Available on the Internet at <http://www.epa.gov/ttnnaaqs/standards/pm/data/PM_RA_FINAL_June_2010.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2011a. *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress*. Office of Air and Radiation, Office of Policy, Washington, DC. March. Available on the Internet at <http://www.epa.gov/oar/sect812/feb11/fullreport.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 1999. *An SAB Advisory: The Clean Air Act Section 812 Prospective Study Health and Ecological Initial Studies*. EPA-SAB-COUNCIL-ADV-99-005, Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/1AFDEBB45C57F6E5852571930058C1A5/$File/coua9905.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2000. *The Clean Air Act Amendments (CAAA) Section 812 Prospective Study of Costs and Benefits (1999): Advisory by the Advisory Council on Clean Air Compliance Analysis: Costs and Benefits of the CAAA.* EPA-SAB-COUNCIL-ADV-00-002. October. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/E992F918DF32131A852571B9004571F7/$File/coua0002.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2001. *REVIEW OF THE DRAFT ANALYTICAL PLAN FOR EPA'S SECOND PROSPECTIVE ANALYSIS - BENEFITS AND COSTS OF THE CLEAN AIR ACT 1990- 2020*. EPA-SAB-COUNCIL-ADV-01-004, Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/572DC1989A3986CC8525718D006BAB8B/$File/councila01004.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2004a. *Advisory on Plans for Health Effects Analysis in the Analytical Plan for EPA's Second Prospective Analysis—Benefits and Costs of the Clean Air Act, 1990–2020*. EPA-SAB-COUNCIL-ADV-04-002. March. Available on the Internet at <http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/08E1155AD24F871C85256E5400433D5D/$File/council_adv_04002.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2004b. *Review of the Draft Analytical Plan for EPA's Second Prospective Analysis—Benefits and Costs of the Clean Air Act, 1990-2020: An Advisory by the Advisory Council for Clean Air Compliance Analysis*. EPA-SAB-COUNCIL-ADV-04-004. May. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/7CCBBFE15CD4C8B185256F17005E3079/$File/council_adv_04004.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2004c. *Advisory Council on Clean Air Compliance Analysis Response to Agency Request on Cessation Lag.* EPA-COUNCIL-LTR-05-001. December. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/0/39F44B098DB49F3C85257170005293E0/$File/council_ltr_05_001.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2010. *Review of EPA's DRAFT Health Benefits of the Second Section 812 Prospective Study of the Clean Air Act*. EPA-COUNCIL-10-001. June. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/9288428b8eeea4c885257242006935a3/72D4EFA39E48CDB28525774500738776/$File/EPA-COUNCIL-10-001-unsigned.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2011a. *CASAC Review of Quantitative Health Risk Assessment for Particulate Matter—Second External Review Draft (February 2010).* EPA-CASAC-10-008. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/BC4F6E77B6385155852577070002F09F/$File/EPA-CASAC-10-008-unsigned.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2011b. *Review of the Final Integrated Report for the Second Section 812 Prospective Study of the Benefits and Costs of the Clean Air Act (August 2010).* EPA-COUNCIL-11-001. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/1E6218DE3BFF682E852577FB005D46F1/$File/EPA-COUNCIL-11-001-unsigned.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2011c. *Review of Valuing Mortality Risk Reductions for Environmental Policy: A White Paper (December 10, 2010).* EPA-SAB-11-011. July. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/298E1F50F844BC23852578DC0059A616/$File/EPA-SAB-11-011-unsigned.pdf>.

Woods & Poole Economics, Inc. 2012. *Complete Economic and Demographic Data Source (CEDDS)*. CD-ROM. Woods & Poole Economics, Inc. Washington, D.C.

World Health Organization (WHO). 2008. *Part 1: Guidance Document on Characterizing and Communicating Uncertainty in Exposure Assessment, Harmonization Project Document No. 6.* Published under joint sponsorship of the World Health Organization, the International Labour Organization and the United Nations Environment Programme. WHO Press: Geneva, Switzerland. Available on the Internet at <www.who.int/ipcs/methods/harmonization/areas/uncertainty%20.pdf>.

CHAPTER 6
WELFARE CO-BENEFITS OF THE PRIMARY STANDARD

6.1 Synopsis

The emission reductions to meet the revised primary standard will have "welfare" co-benefits in addition to human health benefits, including changes in visibility, materials damage, ecological effects from PM deposition, ecological effects from nitrogen and sulfur emissions, vegetation effects from ozone exposure, ecological effects from mercury deposition, and climate effects.[1] Despite our goal to quantify and monetize as many of the benefits as possible for the revised primary standard, the welfare co-benefits of the revised primary standard remain unquantified and nonmonetized in this RIA due to data, methodology, and resource limitations. Specifically, we do not have air quality model runs for the regulatory baseline and the alternative standard levels that would allow us to calculate the visibility co-benefits of attaining the revised primary standard even though we have a complete methodology for estimating these co-benefits. However, using the approach described in this chapter, we provide the results of an illustrative analysis in Appendix 6.B using the 2020 base case and 2020 control case simulations that were used to develop the air quality ratios.

6.2 Introduction to Welfare Benefits

Illustrative emission reduction strategies to attain the revised and alternative annual primary standard have numerous documented effects on environmental quality that affect human welfare. The Clean Air Act defines welfare effects to include any non-health effects, including direct damages to property, either through impacts on material structures or by soiling of surfaces, direct economic damages in the form of lost productivity of crops and trees, indirect damages through alteration of ecosystem functions, and indirect economic damages through the loss in value of recreational experiences or the existence value of important resources. The EPA's *Integrated Science Assessments for Particulate Matter* (hereafter, "PM ISA") (U.S. EPA, 2009b), NO_X/SO_X—Ecological Criteria (U.S. EPA, 2008), and Ozone (U.S. EPA, 2012a) identify numerous physical and ecological effects known to be causally linked to these pollutants. This chapter describes these individual effects and how we would quantify and monetize them if there is enough data to do so. These welfare effects include changes in visibility, materials damage, ecological effects from PM deposition, ecological effects from

[1] While we understand that "welfare" can include health and non-health effects in the economic sense, we use the term "welfare" in this RIA in the same context as the definitions in the Clean Air Act for NAAQS. In practice, welfare benefits represent non-health effects.

nitrogen and sulfur emissions, vegetation effects from ozone exposure, ecological effects from mercury deposition, and climate effects.

These welfare co-benefits are associated with reductions in emissions of specific pollutants resulting from illustrative emission reduction strategies to attain the revised and alternative annual primary standard, not the form or intent of any specific standard. Even though the primary standards are designed to protect against adverse effects to human health, the emission reductions have welfare co-benefits in addition to the direct human health benefits.

The impacts of the illustrative emission reduction strategies can be grouped into four categories: directly emitted PM (e.g., metals, organic compounds, dust), reductions of $PM_{2.5}$ precursors (e.g., NO_x, SO_x, VOCs), other ancillary reductions from the illustrative emission reduction strategies (e.g., mercury and CO_2), and secondary co-pollutant formation from PM precursors (e.g., ozone from NO_x and VOCs). Regardless of the category, these emission changes are anticipated to affect ambient concentrations and deposition, and consequently affect public welfare. It is therefore appropriate and reasonable to include all the benefits and co-benefits associated with these emission reductions to provide a comprehensive understanding of the likely public impacts of attaining the revised or alternative annual standards. Table 6-1 shows the welfare effects associated with the various pollutants (either directly or as a precursor to secondary formation of PM or ozone) that would be reduced by the illustrative emission reduction strategies to attain the revised and alternative annual standard.

Based on the EPA's previous analyses, we believe the welfare co-benefits associated with these non-health benefit categories could be significant (U.S. EPA, 2011b). Despite our goal to quantify and monetize as many of the benefits and co-benefits as possible, welfare co-benefits of the revised primary standard remain unquantified and nonmonetized in this RIA due to data, methodology, and resource limitations. Therefore, the total benefits would be larger than we have estimated. The monetized value of these unquantified effects is represented by adding an unknown "B," which includes both unmonetized health benefits and welfare co-benefits, to the aggregate total for the cost-benefit comparison. These unquantified benefits and co-benefits may be substantial, although the magnitude is highly uncertain. We include a qualitative description of the anticipated welfare effects in this chapter to characterize the type and potential extent of those co-benefits, as identified in Table 6-2.

Table 6-1. Welfare Effects by Pollutants Potentially Affected by Attainment of the PM NAAQS

Pollutant	Atmospheric Effects			Atmospheric and Deposition Effects		Deposition Effects			
	Visibility Impairment	Vegetation Injury (SO_2)	Vegetation Injury (Ozone)	Materials Damage	Climate	Ecosystem Effects—(Organics & Metals)	Acidification (freshwater)	Nitrogen Enrichment	Mercury Methylation
Direct $PM_{2.5}$	✓			✓	✓	✓			
NO_x	✓		✓	✓	✓		✓	✓	
SO_2	✓	✓		✓	✓		✓		✓
VOCs	✓		✓	✓		✓			
PM_{10}	✓			✓	✓				
Hg						✓			✓
CO_2					✓				

✓ = Welfare category affected by this pollutant.

The remainder of this chapter is organized as follows: Section 6.3 provides a qualitative discussion of the visibility co-benefits and describes our approach to estimate those visibility co-benefits if we had the data to do so. Sections 6.4 through 6.6 provide qualitative co-benefits for the unquantified benefits categories of materials damage, climate, and ecosystem effects. References are provided in Section 6.7. Additional information regarding technical details of the visibility co-benefits approach is provided in Appendix 6A. The illustrative visibility co-benefits results for the specific modeled scenario (not the revised standard scenario) are provided in Appendix 6.B.

Table 6-2. Quantified and Unquantified Welfare Co-Benefits

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Improved Environment				
Reduced visibility impairment	Visibility in Class 1 areas in SE, SW, and CA regions	[a]	[a]	Section 6.3, Appendix 6.B
	Visibility in Class 1 areas in other regions	—	[a]	Section 6.3, Appendix 6.B
	Visibility in 8 cities	—	[a]	Section 6.3, Appendix 6.B
	Visibility in other residential areas	—	[a]	Section 6.3, Appendix 6.B
Reduced climate effects	Global climate impacts from CO_2	—	—	Section 6.5, SCC TSD[b]
	Climate impacts from ozone and PM	—	—	Section 6.5, Ozone ISA, PM ISA[c]
	Other climate impacts (e.g., other GHGs, other impacts)	—	—	Section 6.5, IPCC[c]
Reduced effects on materials	Household soiling	—	—	Section 6.4, PM ISA[c]
	Materials damage (e.g., corrosion, increased wear)	—	—	Section 6.4, PM ISA[c]
Reduced effects from PM deposition (metals and organics)	Effects on Individual organisms and ecosystems	—	—	Section 6.6.1, PM ISA[c]
Reduced vegetation and ecosystem effects from exposure to ozone	Visible foliar injury on vegetation	—	—	Section 6.6.4, Ozone ISA[c]
	Reduced vegetation growth and reproduction	—	—	Section 6.6.4, Ozone ISA[b]
	Yield and quality of commercial forest products and crops	—	—	Section 6.6.4, Ozone ISA[b,d]
	Damage to urban ornamental plants	—	—	Section 6.6.4, Ozone ISA[c]
	Carbon sequestration in terrestrial ecosystems	—	—	Ozone ISA[c]
	Recreational demand associated with forest aesthetics	—	—	Ozone ISA[c]
	Other non-use effects			Ozone ISA[c]
	Ecosystem functions (e.g., water cycling, biogeochemical cycles, net primary productivity, leaf-gas exchange, community composition)	—	—	Ozone ISA[2]

(continued)

Table 6-2. Quantified and Unquantified Welfare Co-Benefits (Cont.)

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Improved Environment (Cont.)				
Reduced effects from acid deposition	Recreational fishing	—	—	Section 6.6.2, NO_x SO_x ISA[b]
	Tree mortality and decline	—	—	Section 6.6.2, NO_x SO_x ISA[c]
	Commercial fishing and forestry effects	—	—	Section 6.6.2, NO_x SO_x ISA[c]
	Recreational demand in terrestrial and aquatic ecosystems	—	—	Section 6.6.2, NO_x SO_x ISA[c]
	Other non-use effects			Section 6.6.2, NO_x SO_x ISA[c]
	Ecosystem functions (e.g., biogeochemical cycles)	—	—	Section 6.6.2, NO_x SO_x ISA[c]
Reduced effects from nutrient enrichment	Species composition and biodiversity in terrestrial and estuarine ecosystems	—	—	Section 6.6.2, NO_x SO_x ISA[c]
	Coastal eutrophication	—	—	Section 6.6.2, NO_x SO_x ISA[c]
	Recreational demand in terrestrial and estuarine ecosystems	—	—	Section 6.6.2, NO_x SO_x ISA[c]
	Other non-use effects			Section 6.6.2, NO_x SO_x ISA[c]
	Ecosystem functions (e.g., biogeochemical cycles, fire regulation)	—	—	Section 6.6.2, NO_x SO_x ISA[c]
Reduced vegetation effects from ambient exposure to SO_2 and NO_x	Injury to vegetation from SO_2 exposure	—	—	Section 6.6.2, NO_x SO_x ISA[c]
	Injury to vegetation from NO_x exposure	—	—	Section 6.6.2, NO_x SO_x ISA[c]
Reduced ecosystem effects from exposure to methylmercury (through the role of sulfate in methylation)	Effects on fish, birds, and mammals (e.g., reproductive effects)	—	—	Section 6.2 and 6.6.3, Mercury Study RTC[c,d]
	Commercial, subsistence and recreational fishing	—	—	Section 6.2 and 6.6.3, Mercury Study RTC[c]

[a] We quantify these co-benefits in an illustrative analysis using the methods discussed in this chapter for the specific modeled scenario. These results are provided in Appendix 6.B, but these results of that illustrative scenario are not an estimate of the co-benefits for the revised primary standard.

[b] We assess these co-benefits qualitatively due to time and resource limitations for this RIA.

[c] We assess these co-benefits qualitatively because we do not have sufficient confidence in available data or methods.

[d] We assess these co-benefits qualitatively because current evidence is only suggestive of causality or there are other significant concerns over the strength of the association.

6.3 Visibility Co-Benefits Approach

6.3.1 Visibility and Light Extinction Background

The illustrative emission reduction strategies designed to attain the revised and alternative annual standards would reduce emissions of directly emitted $PM_{2.5}$ as well as precursor emissions such as NO_x and SO_2 for an alternative annual standard at 11 µg/m^3. These emission reductions would improve the level of visibility because these suspended particles and gases impair visibility by scattering and absorbing light (U.S. EPA, 2009b). Visibility is also referred to as visual air quality (VAQ),[2] and it directly affects people's enjoyment of a variety of daily activities (U.S. EPA, 2009b). Good visibility increases quality of life where individuals live and work, and where they travel for recreational activities, including sites of unique public value, such as the Great Smoky Mountains National Park (U. S. EPA, 2009b). This section discusses the economic co-benefits associated with improved visibility as a result of emission reductions associated with the revised and alternative annual standards.

Air pollution affects light extinction, a measure of how much the components of the atmosphere scatter and absorb light. More light extinction means that the clarity of visual images and visual range is reduced, all else held constant. Light extinction is the optical characteristic of the atmosphere that occurs when light is either scattered or absorbed, which converts the light to heat. Particulate matter and gases can both scatter and absorb light. Fine particles with significant light-extinction efficiencies include sulfates, nitrates, organic carbon, elemental carbon, and soil (Sisler, 1996). The extent to which any amount of light extinction affects a person's ability to view a scene depends on both scene and light characteristics. For example, the appearance of a nearby object (e.g., a building) is generally less sensitive to a change in light extinction than the appearance of a similar object at a greater distance. See Figure 6-1 for an illustration of the important factors affecting visibility.

According to the PM ISA, there is strong and consistent evidence that PM is the overwhelming source of visibility impairment in both urban and remote areas (U.S. EPA, 2009b). After reviewing all of the evidence, the PM ISA concluded that the evidence was sufficient to conclude that a causal relationship exists between PM and visibility impairment.

[2] We use the term VAQ to refer to the visibility effects caused solely by air quality conditions, excluding fog.

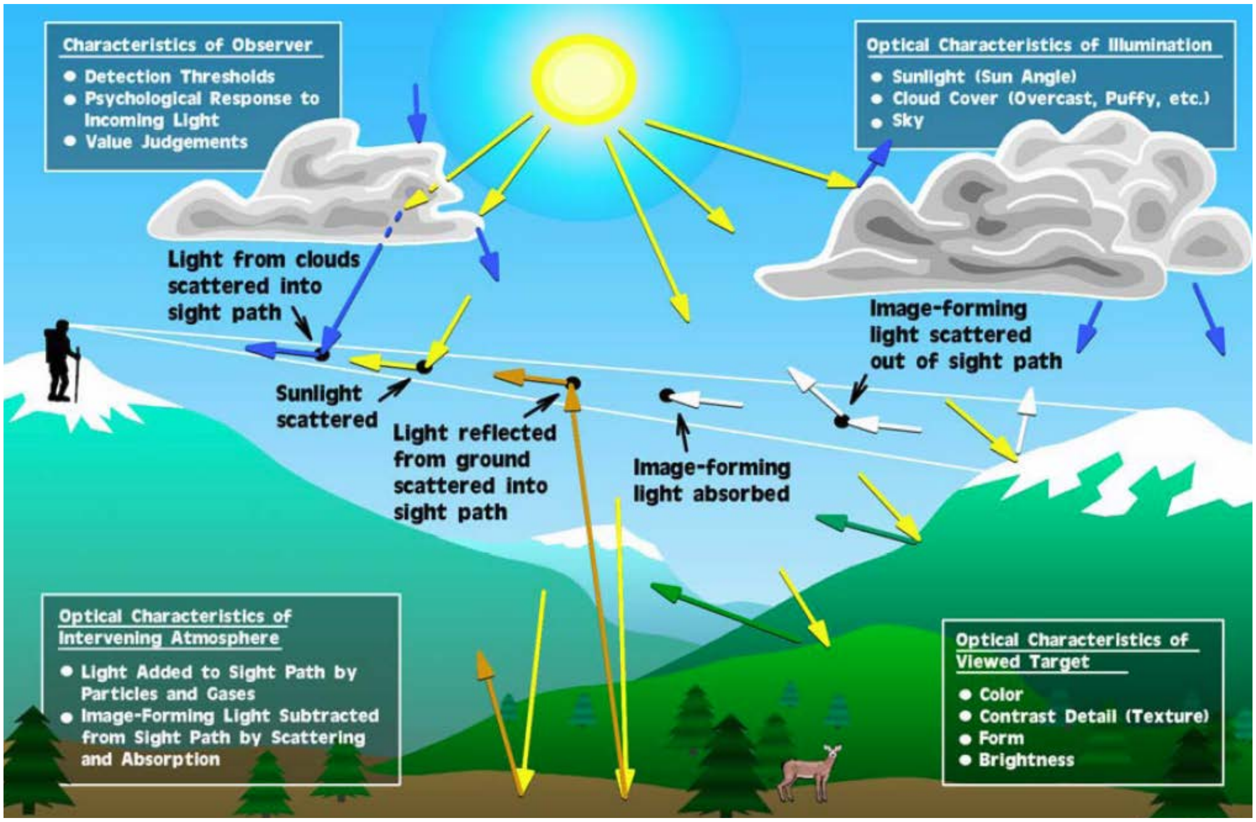


Figure 6-1. Important Factors Involved in Seeing a Scenic Vista (Malm, 1999)

Visibility is commonly measured as either light extinction (β_{ext}), which is defined as the loss of light per unit of distance in terms of inverse megameters (Mm^{-1}), or using the deciview (dv) metric, which is a logarithmic function of extinction (Pitchford and Malm, 1994). Deciviews, a unitless measure of visibility, are standardized for a reference distance in such a way that one deciview corresponds to a change of about 10% in available light.[3] Pitchford and Malm (1994) characterize a change of one deciview as "a small but perceptible scenic change under many circumstances."[4] Extinction and deciviews are both physical measures of the amount of visibility impairment (e.g., the amount of "haze"), with both extinction and deciview increasing as the amount of haze increases. Using the relationships derived by Pitchford and Malm (1994),

---

[3] Note that deciviews are inversely related to visual range, such that a decrease in deciviews implies an increase in visual range (i.e., improved visibility). Conversely, an increase in deciviews implies a decrease in visual range (i.e., decreased visibility). Deciview, in effect, is a measure of the *lack* of visibility.

[4] An instantaneous change of less than 1 deciview (i.e., less than 10% in the light extinction budget) represents a measurable improvement in visibility but may not be perceptible to the eye. The visibility co-benefits approach described in this chapter reflects annual average changes in visibility, which are likely made up of periods with changes less than one deciview and periods with changes exceeding one deciview. Annual averages appear to more closely correspond to the economic literature relied upon for valuation of visibility changes in this analysis.

$$Deciviews = 10 * \ln\left(\frac{391}{VR}\right) = 10 \; * \ln\left(\frac{\beta_{ext}}{10}\right) \quad (6.1)$$

where VR denotes visual range (in kilometers) and β_{ext} denotes light extinction (in Mm^{-1}).[5]

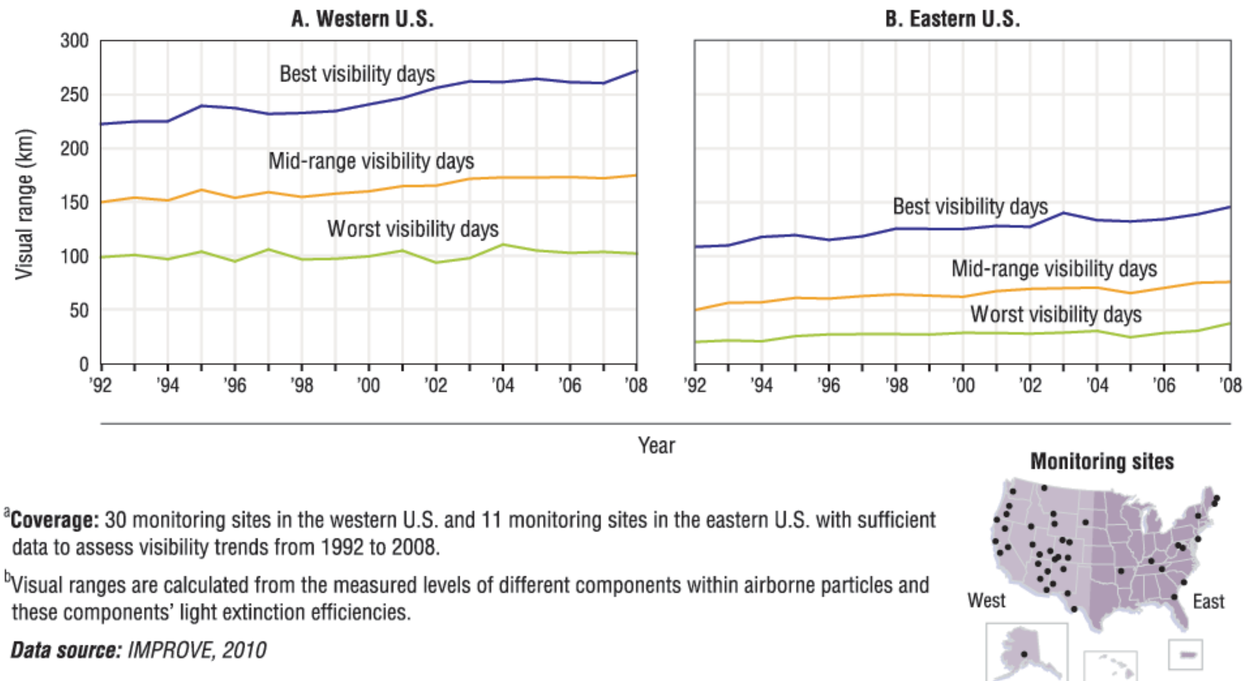
Annual average visibility conditions (reflecting light extinction due to both anthropogenic and non-anthropogenic sources) vary regionally across the U.S. and by season (U.S. EPA, 2009b). Particulate sulfate is the dominant source of regional haze in the eastern U.S. (>50% of the particulate light extinction) and an important contributor to haze elsewhere in the country (>20% of particulate light extinction) (U.S. EPA, 2009b). Particulate nitrate is an important contributor to light extinction in California and the upper Midwestern U.S., particularly during winter (U.S. EPA, 2009b). Smoke plumes from large wildfires dominate many of the worst haze periods in the western U.S., while Asian dust only caused a few of the worst haze episodes, primarily in the more northerly regions of the west (U.S. EPA, 2009b). Higher visibility impairment levels in the East are due to generally higher concentrations of fine particles, particularly sulfates, and higher average relative humidity levels (U.S. EPA, 2009b). Humidity increases visibility impairment because some particles such as ammonium sulfate and ammonium nitrate absorb water and form droplets that become larger when relative humidity increases, thus resulting in increased light scattering (U.S. EPA, 2009b).

Reductions in air pollution from implementation of various programs associated with the Clean Air Act Amendments of 1990 (CAAA) provisions have resulted in substantial improvements in visibility, and will continue to do so in the future. Because trends in haze are closely associated with trends in particulate sulfate and nitrate due to the simple relationship between their concentration and light extinction, visibility trends have improved as emissions of SO_2 and NO_x have decreased over time due to air pollution regulations such as the Acid Rain Program (U.S. EPA, 2009b). For example, Figure 6-2 shows that visual range increased nearly 50% in the eastern U.S. since 1992.[6] The EPA's recent regulations such as the Cross-State Air Pollution Rule (U.S. EPA, 2011c) and the Mercury and Air Toxics Standard (U.S. EPA, 2011d) are anticipated to reduce SO_2 emissions down to 2 million tons nationally, which would lead to substantial further improvement in visibility levels in the Eastern U.S. Calculated from light extinction efficiencies from Trijonis et al. (1987, 1988), annual average visual range under natural conditions in the East is estimated to be 150 km ± 45 km (i.e., 65 to 120 miles) and 230 km ± 35 km (i.e., 120 to 165 miles) in the West (Irving, 1991). Figure 6-2 reflects the average

[5] It has been noted that, for a given deciview value, there can be many different visual ranges, depending on the other factors that affect visual range—such as light angle and altitude. See Appendix 6a for more detail.

[6] In Figure 6-2, the "best days" are defined as the best 20% of days, the "mid-range days" are defined as the middle 20%, and the "worst days" are defined as the worst 20% of days (IMPROVE, 2010).

trends in visual ranges at select monitors in the eastern and western areas of the U.S. since 1992 using data from the IMPROVE monitoring network (U.S. EPA (2008) updated; IMPROVE (2010)). As an illustration of the improvements in visibility attributable to the CAAA, Figure 6-3 depicts the modeled improvements in visibility associated with all the CAAA provisions in 2020 compared to a counterfactual scenario without the CAAA (U.S. EPA, 2011b). While visibility trends have improved in most National Parks, the recent data show that these areas continue to suffer from visibility impairment beyond natural background levels (U.S. EPA, 2009b).



[a]**Coverage:** 30 monitoring sites in the western U.S. and 11 monitoring sites in the eastern U.S. with sufficient data to assess visibility trends from 1992 to 2008.

[b]Visual ranges are calculated from the measured levels of different components within airborne particles and these components' light extinction efficiencies.

Data source: *IMPROVE, 2010*

Figure 6-2. Visibility in Selected National Parks and Wilderness Areas in the U.S., 1992–2008[a,b]

Source: U.S. EPA (2008) updated, IMPROVE (2010).

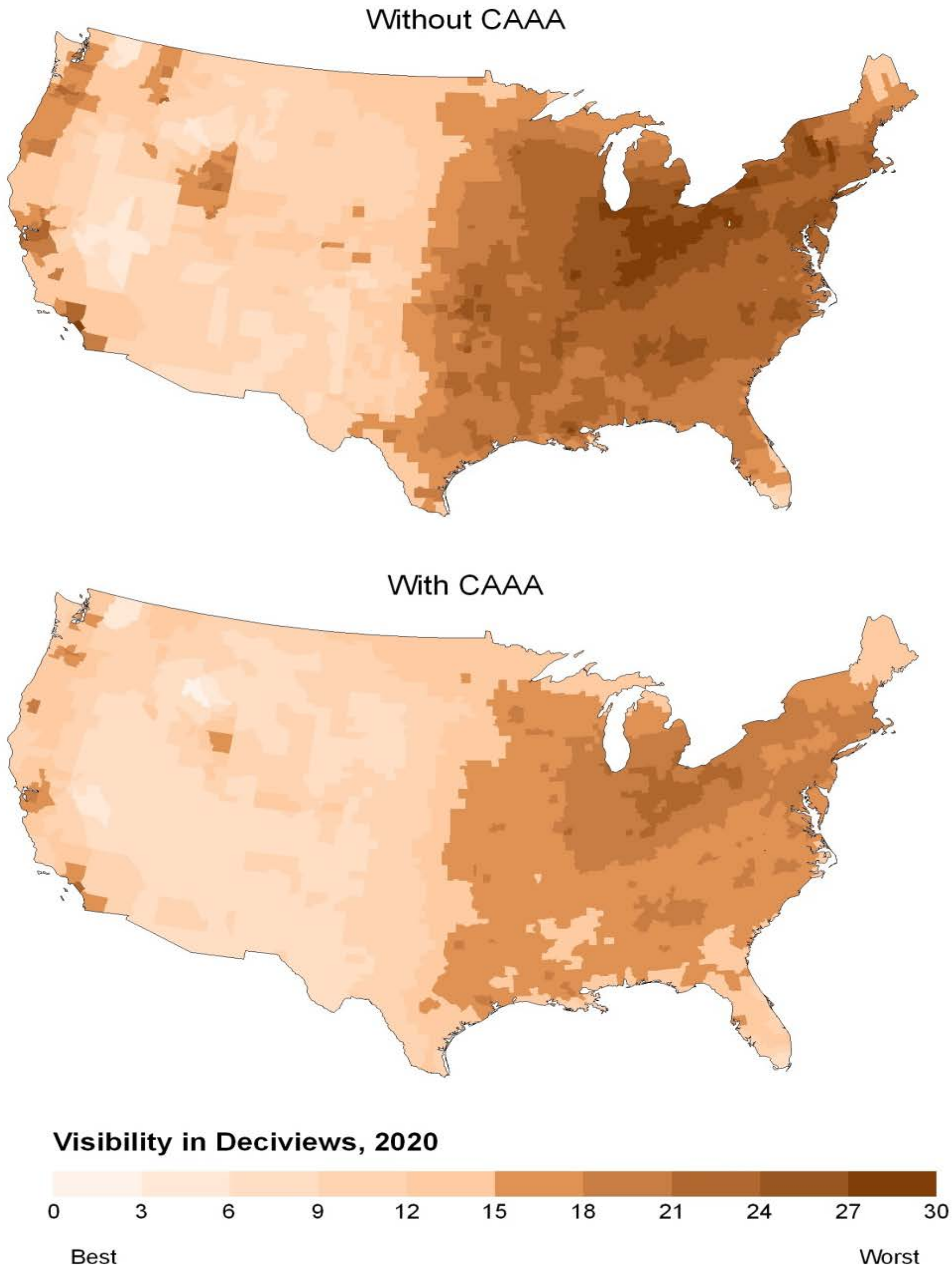


Figure 6-3. Estimated Improvement in Annual Average Visibility Levels Associated with the CAAA Provisions in 2020

Source: U.S. EPA, 2011b.[7]

[7] It is important to note that visibility levels shown in these maps were modeled differently than the modeling conducted for this analysis using an earlier method that we would currently use, including coarser grid resolution (i.e., 36 km instead of 12 km). In addition, please note that these maps present annual average visibility levels, which are different than the short-term averages being considered for the secondary standard.

6.3.2 *Quantifying Light Extinction for Assessing Visibility Co-benefits*

For this RIA, we do not have air quality model runs for the regulatory baseline and the alternative standard levels that would allow us to calculate the visibility co-benefits of attaining the revised primary standard. However, we provide an illustrative analysis in Appendix 6.B using the 2020 base case and 2020 control case simulations that were used to develop the air quality ratios.[8] In our approach, we generate light extinction estimates using the CMAQ model in conjunction with the IMPROVE (Interagency Monitoring of Protected Visual Environments) algorithm that estimates light extinction as a function of PM concentrations and relative humidity levels (U.S. EPA, 2009b).[9] The procedure for calculating light extinction associated with the revised and alternative annual standards is described in detail in Chapter 3 of this RIA. In addition, Appendix 6.A describes how the spatial resolution of the light extinction estimates would be adjusted in our approach.

It is important to note that the light extinction estimates used in our approach represent annual averages, which is different from the averaging times currently being considered for the secondary PM NAAQS. While the annual averages are influenced by days with extremely impaired visibility, the light extinction data is not sufficient to provide higher temporal resolution than quarterly averages. While we suspect that the most impaired days would have disproportionately improved visibility from the emission reductions associated with attaining the revised or alternative primary standards, we are not able to quantify those impacts. These data gaps result in an underestimate of visibility co-benefits associated with extreme days. We recognize that recent advice from the Science Advisory Board's Advisory Council on Clean Air Compliance Analysis (SAB-Council) recommends estimating visibility co-benefits considering daytime visibility on days with severe impairment (U.S. EPA-SAB, 2010a), but the available data and valuation studies do not allow such fine temporal resolution.

While our approach is a substantial improvement in the methods to estimate light extinction nationally, we are still developing a method to estimate coarse particle concentrations for the entire continental U.S. for estimating light extinction. As an interim solution, our approach includes sensitivity analyses to show the potential impact of omitting coarse particles from the light extinction estimates for recreational and residential visibility. For these sensitivity analyses, we selected the levels of coarse particles to represent the full range of possible annual concentrations from a recent report on the IMPROVE monitoring network

[8] These simulations are described in Chapter 3.

[9] According to the PM ISA, the IMPROVE algorithm performs reasonably well despite its simplicity (U.S. EPA, 2009b).

(Debell et al., 2006). Specifically, for these sensitivity analyses, we assume four levels of coarse particles: no coarse particles, 5 μg/m^3 nationwide, 15 μg/m^3 in the Southwest with 5 μg/m^3 in the rest of the country, and 15 μg/m^3 in the Southwest with 8 μg/m^3 in the rest of the country.[10] In Table 6-3, we provide a qualitative assessment of how key assumptions in the estimation of light extinction would affect the visibility co-benefits.

Table 6-3. Key Assumptions in the Light Extinction Estimates Affecting the Visibility Co-Benefits Approach[a]

Key Assumption	Direction of Bias	Magnitude of Effect
The light extinction estimates are annual averages to correspond with the valuation studies. People may value large changes to the haziest days differently than small changes to many days. We assume that annual average light extinction is the most appropriate temporal scale for estimating visibility benefits.	Potential Underestimate	Medium
Coarse particles are a component of light extinction, but we were unable to include coarse particles in the light extinction estimates. We provide sensitivity analyses with up to 15 μg/m^3 in the Southwest and 8 μg/m^3 in the rest of the country.	Potential Overestimate	Very Low

[a] A description of the classifications for magnitude of effects can be found in Appendix 5.B of this RIA.

6.3.3 Visibility Valuation Overview

In the Clean Air Act Amendments of 1977, the U.S. Government recognized visibility's value to society by establishing a national goal to protect national parks and wilderness areas from visibility impairment caused by manmade pollution.[11] Air pollution impairs visibility in both residential and recreational settings, and an individual's willingness to pay (WTP) to improve visibility differs in these two settings. Benefits of residential visibility relate to the impact of visibility changes on an individual's daily life (e.g., at home, at work, and while engaged in routine recreational activities). Benefits of recreational visibility relate to the impact of visibility changes manifested at parks and wilderness areas that are expected to be experienced by recreational visitors.

Both recreational and residential visibility benefits consist of use values and nonuse values. Use values include the aesthetic benefits of better visibility, improved road and air safety, and enhanced recreation in activities like hunting and birdwatching. Nonuse values are

[10] We define "Southwest" for this sensitivity analysis to be the states of California, Nevada, Utah, Arizona, New Mexico, Colorado, and Texas.
[11] See Section 169(a) of the Clean Air Act.

based on a belief that the environment ought to exist free of human-induced haze. This includes the value of better visibility for use by others now and in the future (bequest value). Nonuse values may be more important for recreational areas, particularly national parks and monuments.

The relationship between a household's WTP and changes in visibility can be derived from a number of contingent valuation (CV) studies published in the peer-reviewed economics literature. The studies used in the approach to estimate the residential and recreational visibility co-benefits associated with the revised and alternative annual standards are described in the following sections. In addition to CV studies, hedonic valuation studies (Beron et al., 2001, 2004) also demonstrate that visibility has value, but we are unable to apply these valuation estimates in the context of estimating the visibility co-benefits associated with national regulations that reduce air pollution (Leggett and Neumann, 2004).

6.3.3.1 Visibility Valuation Approach

In our approach, we assume that individuals value visibility for aesthetic reasons rather than viewing visibility as a proxy for other impacts associated with air pollution, such as health or ecological improvements. Some studies in the literature indicate that individuals may have difficulty distinguishing visibility from other aspects of air pollution (e.g., McClelland et al., 1993; Chestnut and Rowe, 1990c; Carson, Mitchell, and Rudd, 1990). Because visual air quality is inherently multi-attribute, it is a challenge for all visibility valuation studies to isolate the value of visibility from the collection of intertwined benefits. Each study used in our approach attempts to isolate visibility from other effect categories, but the different studies take different approaches (U.S. EPA, 2009b).[12] However, the degree to which the studies were successful in convincing respondents to focus solely on visibility is unclear

Similarly, it is important to try to distinguish residential visibility from recreational visibility co-benefits, specifically whether these can these can be treated as distinct and additive benefit categories based on the available literature. In our approach, we assume that residential and recreational visibility co-benefits are distinct and separable. It is conceivable that respondents to the recreational visibility survey may have partially included values for their own residential visibility when evaluating changes at national parks and wilderness areas located in their region of the country. In our approach, we take care to minimize the number of overlapping areas and their contributions. Specifically, we believe that the potential for double-counting recreational and residential visibility is minimal for several reasons. First, in our

[12] See Leggett and Neumann (2004) for a more detailed discussion of this issue.

approach, we only include a subset of areas in the primary estimates of recreational and residential visibility co-benefits, which overlap in only a few places.[13] Second, a number of the overlapping counties are wilderness areas, which would contribute little to the overall monetized visibility co-benefits due to low visitation rates, rather than highly visited national parks. For example, Los Angeles County is home to the San Gabriel Wilderness Area, which has 10 thousand annual visitors (NPS, 2008). If we were to exclude the residential visibility co-benefits that accrue to 10 million residents in Los Angeles County and only include the very small recreational visibility co-benefits for the wilderness area, we would be substantially biasing the overall estimates downward. For these reasons, we believe that the potential for double-counting is minimal.

In the next sections, we describe the methodology and limitations of the recreational and residential visibility approach. Consistent with the health benefits analysis, the monetized visibility co-benefits would be adjusted for inflation and income growth. These co-benefits would be specific to the analysis year, and as population and income increase over time, these co-benefits would be expected to increase each year for the same incremental change in light extinction.

6.3.4 Recreational Visibility

6.3.4.1 Methodology

Our approach for estimating recreational visibility co-benefits is well-established and has been used in numerous analyses by the EPA (U.S. EPA, 1999; 2005; 2006; 2010; 2011b). In our approach, recreational visibility co-benefits apply to Class 1 areas, such as National Parks and Wilderness Areas.[14] Although other recreational settings such as National Forests, state parks, or even hiking trails or roadside areas have important scenic vistas, a lack of suitable economic valuation literature to identify these other areas and/or a lack of visitation data prevents us from generating estimates for those recreational vista areas.

Under the 1999 Regional Haze Rule (64 FR 35714), states are required to set goals develop long-term strategies to improve visibility in Class 1 areas, with the goal of achieving

[13] As described in detail in Sections 6.3.3 and 6.3.4, our approach includes only a subset of visibility co-benefits in the main benefits estimates, while providing the rest of the visibility co-benefits in sensitivity analyses.

[14] Hereafter referred to as Class 1 areas, which are defined as areas of the country such as national parks, national wilderness areas, and national monuments that have been set aside under Section 169(a) of the Clean Air Act to receive the most stringent degree of air quality protection. Mandatory Class 1 federal lands fall under the jurisdiction of three federal agencies, the National Park Service, the Fish and Wildlife Service, and the Forest Service. EPA has designated 156 areas as mandatory Class 1 federal areas for visibility protection, including national parks that exceed 6,000 acres and wilderness areas that exceed 5,000 acres (40 CFR §81.400).

natural background visibility levels by 2064. In conjunction with the U.S. National Park Service (NPS), the U.S. Forest Service (USFS), other Federal land managers, and State organizations in the U.S., the EPA has supported visibility monitoring in national parks and wilderness areas since 1988. The monitoring network known as IMPROVE includes 156 sites that represent the Class 1 areas across the country (U.S. EPA, 2009b).[15] The IMPROVE monitoring network measures fine particles, coarse particles, and key $PM_{2.5}$ constituents that affect visibility, such as sulfate, nitrate, organic and elemental carbon, soil dust, and several other elements. Figure 6-4 identifies where each of these parks are located in the U.S.

Figure 6-4. Mandatory Class 1 Areas in the U.S.

For recreational visibility, the EPA relies upon a contingent valuation (CV) survey conducted in 1988 (Chestnut and Rowe, 1990a; 1990b) to estimate the recreational visibility co-benefits. Although there are several other studies in the literature on recreational visibility valuation, they are even older and use less robust methods. In the EPA's judgment, despite the inherent limitations in the survey, the Chestnut and Rowe study served as the basis for

[15] The formula used to estimate light extinction from concentrations of PM constituents and relative humidity is referred to as the IMPROVE algorithm.

monetary estimates of the co-benefits of visibility changes in recreational areas for a number of previous EPA rulemakings. This study serves as an essential input to our approach for estimating the co-benefits from improving recreational visibility.

In our approach, we assume that the household WTP is higher if the Class 1 recreational area is located close to the person's home (i.e., in the same region of the country). People appear to be willing to pay more for visibility improvements at parks and wilderness areas that are in the same region as their household than at those that are not in the same region as their household (Chestnut and Rowe, 1990a, 1990b). This is plausible, because people are more likely to visit, be familiar with, and care about parks and wilderness areas in their own part of the country. However, studies have also found many people who had never visited and never planned to visit the parks still had positive values for visibility improvements in those locations (Chestnut and Rowe, 1990b).

The Chestnut and Rowe survey measured the demand for visibility in Class 1 areas managed by the NPS in three broad regions of the country: California, the Colorado Plateau (Southwest), and the Southeast.[16] Respondents in five states were asked about their WTP to protect national parks or NPS-managed wilderness areas within a particular region. The survey used photographs reflecting different visibility levels in the specified recreational areas. The authors used the survey data to estimate household WTP values for improved visibility in each region.

The separate regions were developed to capture differences in household WTP values based on proximity to recreational areas. Chestnut (1997) also concluded that, for a given region, a substantial proportion of the WTP is attributable to one specific park within the region. This so called "indicator park" is the most well-known and frequently visited park within a particular region. The indicator parks for the three studied park regions are Yosemite National Park for the California region, the Grand Canyon National Park for the Southwest region, and Shenandoah National Park for the Southeast region. In accordance with the methodology in Chestnut (1997), our approach calculates the benefits from households for a particular region for a given change in visibility at a particular Class 1 area. In theory, summing benefits from households in all regions would yield the total monetary benefits associated with a given visibility improvement at a particular park, which could then be summed with other parks and

[16] The Colorado Plateau (Southwest) region is defined as the states of Colorado, New Mexico, Arizona, and Utah. The Southeast region is defined as the states of West Virginia, Virginia, North Carolina, South Carolina, Georgia, Florida, Alabama, Mississippi, Louisiana, Tennessee, and Kentucky. The California region includes the state of California and one wilderness area in Nevada.

regions to estimate national benefits. Because recreational visibility benefits may reflect the value an individual places on visibility improvements regardless of whether the person plans to visit the park, all households in the U.S. are assumed to derive some benefit from improvements to Class 1 areas.

To value recreational visibility improvements associated with its rulemakings, the EPA developed a valuation WTP equation function based on the baseline of visibility, the magnitude of the visibility improvement, and household income. This function requires light extinction estimates measured as visual range. The behavioral parameters of this equation were taken from an analysis of the survey described in Chestnut and Rowe (1990a, 1990b). These parameters were used to calibrate WTP for the visibility changes resulting from this rule.[17] As an example, household WTP for a visibility improvement at a park in its region takes the following form:

$$WTP(\Delta Q_{ik}) = m - [m^{\rho} + \gamma_{ik} * (Q_{0ik}^{\rho} - Q_{1ik}^{\rho})]^{\frac{1}{\rho}} \tag{6.2}$$

where:

i indexes region,

k indexes park,

m = household income,

ρ = shape parameter (0.1),

γ = parameter corresponding to the visibility at in-region parks,

Q_0 = starting visibility, and

Q_1 = visibility after change.

As discussed in more detail in Appendix 6.A of this RIA, our approach to valuing recreational visibility changes is an application of the Constant Elasticity of Substitution (CES) utility function approach and is based on the preference calibration method developed by Smith, Van Houtven, and Pattanayak (2002).[18] Available evidence indicates that households are willing to pay more for a given visibility improvement as their income increases (Chestnut,

[17] The parameters for each region are available in Appendix 6a of this RIA.
[18] The Constant Elasticity of Substitution utility function has been chosen for use in this analysis due to its flexibility when illustrating the degree of substitutability present in various economic relationships (in this case, the tradeoff between income and improvements in visibility).

1997). Using the income elasticity calculated by Chestnut (1997), the recreational visibility benefits assume a 1% increase in income is associated with a 0.9% increase in WTP for a given change in visibility. WTP responses reported in Chestnut and Rowe (1990a, 1990b) were also region-specific, rather than park-specific. As visibility improvements are not constant across all parks in a region, we must infer park-specific visibility parameters in order to calculate WTP for projected visibility changes. As the quantity and quality of parks differs between regions, we apportion the regional WTP parameters based on relative visitation rates at the different parks, because this statistic likely captures both park quality (more people visit parks with more desirable attributes, so collective WTP is likely higher) and quantity (more people visit parks in a region if the parks are more numerous, so collective WTP is likely higher).[19] We also adjust the co-benefits for inflation and growth in real income.

Recreational visibility co-benefits can be calculated as the sum of the household WTPs for changes in light extinction. We assume that each household is valuing the first or only visibility change that occurs in a particular area. The co-benefits at particular areas can be calculated by assuming that the subset of visibility changes of interest is the first or the only set of changes being valued by households. Estimating benefit components in this way will yield slightly upwardly biased estimates of co-benefits, because disposable income is not reduced by the WTPs for any prior visibility improvements. The upward bias should be extremely small, however, because all of the WTPs for visibility changes are very small relative to income.

In our approach, the primary estimate for recreational visibility only includes co-benefits for 86 Class 1 areas in the original study regions (i.e., California, the Southwest, and the Southeast).[20] These co-benefits reflect the value to households living in the same region as the Class 1 area as well as values for all households in the United States living outside the state containing the Class 1 area.

[19] We use 2008 park visitation data from the National Park Service Statistical Abstracts (NPS, 2008), as this is the most current data available. Where the data for a particular park was not representative of normal visitation rates at that park (for example due to fire damage that occurred during that year), we substitute data from the prior year. We use 1997 visitation data for those wilderness areas not included in the National Park Service Statistical Abstracts, as more current data is not readily available. As visitation rates for Wilderness Areas are small compared to visitation rates in National Parks, the inaccuracies generated by using 1997 data are likely to also be small.

[20] The 86 Class 1 areas in the three studied park regions represented 68% of the total visitor days to Class 1 areas in 2008 (NPS, 2008).

The Chestnut and Rowe study did not measure values for visibility improvement in Class 1 areas in the Northwest, Northern Rockies, and Rest of U.S. regions.[21] In order to obtain estimates of WTP for visibility changes for the 70 additional Class 1 areas in these non-studied regions, we have to transfer the WTP values from the studied regions. [22] This co-benefits transfer approach introduces additional uncertainty into the estimates. However, we have taken steps to adjust the WTP values to account for the possibility that a visibility improvement in parks within one region may not necessarily represent the same visibility improvement at parks within a different region in terms of environmental improvement. This may be due to differences in the scenic vistas at different parks, uniqueness of the parks, or other factors, such as public familiarity with the park resource. To account for this potential difference, we adjusted the transferred WTP being transferred by the ratio of visitor days in the two regions.[23] A complete description of the co-benefits transfer method used to infer values for visibility changes in Class 1 areas outside the study regions is provided in Appendix 6a of this RIA.

Table 6-4 indicates which studied park regions we used to estimate the value in the non-studied park regions in our approach. Figure 6-5 shows how the visitation rates vary across Class 1 areas and regions and indicates whether each Class 1 area is located within one of the studied regions.

Table 6-4. WTP for Visibility Improvements in Class 1 Areas in Non-Studied Park Regions

Park Region	Source of WTP Estimate
1. Northwest	Benefits transfer from California
2. Northern Rockies	Benefits transfer from Colorado Plateau
3. Rest of U.S.	Benefits transfer from Southeast

[21] The Northwest region is defined as the states of Washington and Oregon. The Northern Rockies region includes the states of Idaho, Montana, Wyoming, North Dakota, and South Dakota. The Rest of the U.S. region includes all other states not included in the other 5 regions.
[22] The 70 additional Class 1 areas represented 32% of the total visitor days to Class 1 areas in 2008 (NPS, 2008).
[23] For example, if total park visitation in a transfer region was less than visitation in a study region, transferred WTP would be adjusted downward by the ratio of the two.

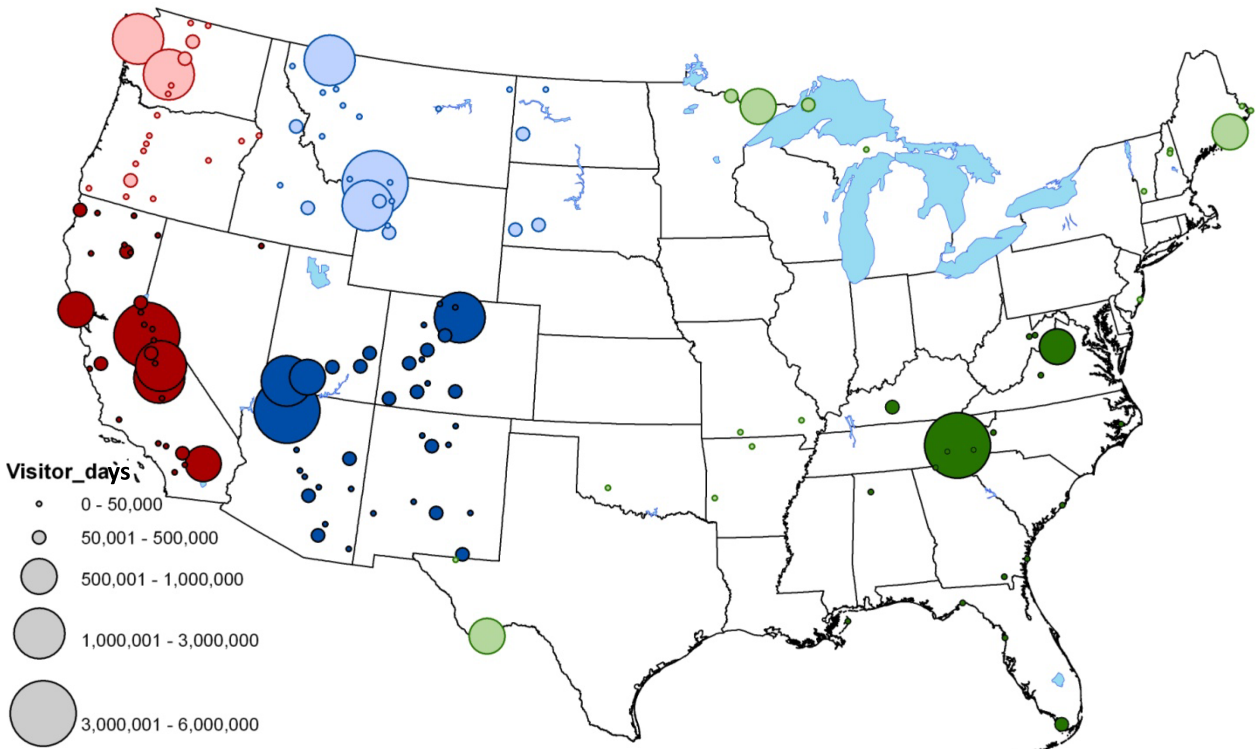


Figure 6-5. Visitation Rates and Park Regions for Class 1 Areas[a]

[a] The colors in this map correspond to the park regions used in the valuation study and the extrapolation to parks in other regions. Red = California, light red = Northwest (extrapolated from California), blue = Colorado Plateau, light blue = Northern Rockies (extrapolated from Colorado Plateau), green = Southeast, light green = Rest of U.S. (extrapolated from Southeast).

In a more recent study, Smith et al. (2005) conducted a contingent valuation survey that updated and expanded a portion of the 1988 survey by Chestnut and Rowe (1990). Specifically, the Smith et al. (2005) survey relied on a panel maintained by *Knowledge Networks* with 2,020 participants completing the survey. Similar to the Chestnut and Rowe survey, the Smith et al. survey assessed WTP for changes in summertime visibility using the base photograph of Shenandoah National Park. Unlike the Chestnut and Rowe survey, the Smith et al. survey only assessed the Shenandoah National Park, did not estimate in-region estimates of WTP, and evaluated several options for incorporating budgetary constraints into the survey. The authors concluded that WTP for recreational visibility is skewed and sensitive to information about budgetary constraints. We are still evaluating the potential error identified by Smith et al. (2005) regarding the visibility levels in the photographs for Shenandoah National Park in the Chestnut and Rowe survey (1990a,b).

Even though this survey represents several advantages over the older survey (e.g., more recent, national, demographically representative, larger sample, etc.), we are unable incorporate the results generated by this survey into our existing method for calculating recreational visibility co-benefits because the survey did not account for the differences between WTP for in-region parks and out-of-region parks. This omission precludes us from combining this new survey for only 1 region of the country with the WTP for the other regions of the country from Chestnut and Rowe (1990). Furthermore, Smith et al. (2005) provide a variety of WTP estimates reflecting different versions of the survey and different methods of summarizing the typical response, which makes it difficult to select estimates to incorporate into the recreational visibility benefits calculation.

6.3.4.2 Recreational Visibility Limitations, Caveats, and Uncertainties

Our approach relies upon several data sources as inputs, including emission inventories, air quality data from models (with their associated parameters and inputs), relative humidity measurements, park information, economic data and assumptions for monetizing co-benefits. Each of these inputs may contain uncertainty that would affect the recreational visibility co-benefits estimates. Though we are unable to quantify the cumulative effect of all of these uncertainties in our approach, we do provide information on uncertainty based on the available data, including model evaluation[24] and sensitivity analyses to characterize major omissions (i.e., benefits from parks in non-studied park regions and inclusion of coarse particles). Although we strive to incorporate as many quantitative assessments of uncertainty as possible, we are severely limited by the available data, and there are several aspects that we are only able to address qualitatively. A summary of the key assumptions including direction and magnitude of bias is provided in Table 6-5.

One major source of uncertainty for the estimation of recreational visibility co-benefits is the benefits transfer process. Choices regarding the functional form and key parameters of the estimating equation for WTP for the affected population could have significant effects on the magnitude of the estimates. Assumptions about how individuals respond to changes in visibility that are either very small or outside the range covered in the Chestnut and Rowe study could also affect the estimates.

[24] See Chapter 4 for more information on model evaluation.

Table 6-5. Summary of Key Assumptions in Estimating Recreational Visibility Co-benefits[a]

Key Assumption	Direction of Bias	Potential Magnitude of Effect
Chestnut and Rowe study covers parks in three regions: California, Southwest, and Southeast. Benefits to other regions in the U.S. are not included in the primary benefits estimate.	Underestimate	Medium
Benefits to other recreational settings, such as National Forests and state parks, are not included in our approach.	Underestimate	Medium-Low
Chestnut and Rowe study conducted on populations in five states. These results are applied to the entire U.S. population.	Unclear	Unclear
Individuals have a greater WTP for visibility changes in parks within their region.	Unclear	Unclear
WTP values reflect only visibility improvements and not overall air quality improvements.	Potential Overestimate	Unclear
We assume that there are 2.68 people per household. Because this estimate has been decreasing over time, this may underestimate the number of households.	Potential Underestimate	Medium-Low

[a] A description of the classifications for magnitude of effects can be found in Appendix 5.B of this RIA.

Since the valuation of recreational visibility co-benefits relies upon one study (Chestnut and Rowe, 1990a; 1990b), all of the uncertainties within that study also pertain to any analysis that uses it. In general, the survey design and implementation reflect the period in which the Chestnut and Rowe study was conducted. Since that time, many improvements to the design of stated preference surveys have been developed (e.g., Arrow, 1993), but we are currently unaware of newer studies that we could incorporate into our visibility co-benefits methodology. Although Chestnut and Rowe still offers the best available WTP estimates, the study has a number of limitations, including:

- The vintage of the survey (late 1980s) invites questions whether the values would still be valid for current populations, or more importantly for our approach, future populations in 2020.
- The survey focused on visibility improvements in and around specific national parks and wilderness areas. Given that national parks and wilderness areas exhibit unique characteristics, it is not clear whether the WTP estimate obtained from this survey can be transferred to other national parks and wilderness areas, even other parks within the studied park regions, without introducing additional uncertainty.

- The survey focused only on populations in five states, so the application of the estimated values to populations outside those states requires that preferences of populations in the five surveyed states be similar to those of non-surveyed states.

- There is an inherent difficulty in separating values expressed for visibility improvements from an overall value for improved air quality. The survey attempted to control for this by informing respondents that "other households are being asked about visibility, human health, and vegetation protections in urban areas and at national parks in other regions." However, most of the respondents did not feel that they were able to segregate recreational visibility at national parks entirely from residential visibility and health effects.

- It is not clear exactly what visibility improvements the respondents to the survey were valuing. The WTP question asked about changes in average visibility, but the survey respondents were shown photographs of only daytime, summer conditions, when visibility is generally at its worst. It is possible that the respondents believed those visibility conditions held year-round, in which case they would have been valuing much larger overall improvements in visibility than what otherwise would be the case. In our approach, the EPA assumed that respondents provided values for changes in annual average visibility. Because most policies would result in a shift in the distribution of visibility (usually affecting the worst days more than the best days), the annual average may not be the most relevant metric for policy analysis.

- The survey did not include reminders of possible substitutes (e.g., visibility at other parks) or budget constraints. These reminders are considered to be best practice for stated preference surveys.

6.3.5 Residential Visibility

6.3.5.1 Methodology

Residential visibility co-benefits are those that occur from visibility changes in urban, suburban, and rural areas where people live. These co-benefits are important because some people living in certain urban areas may place a high value on unique scenic resources in or near these areas that are outside of Class 1 areas. For example, the State of Colorado established a local visibility standard for the Denver metropolitan area in 1990 (Ely et al., 1991). In our approach, residential visibility improvements are defined as those that occur specifically in Metropolitan Statistical Areas (MSAs).

In the *Urban-focused Visibility Assessment* (U.S. EPA, 2010b) and the *Policy Assessment for the Review of the PM NAAQS* (U.S. EPA, 2011a), several preference studies provide the foundation for the secondary PM NAAQS.[25] The three completed survey studies (all in the west) included Denver, Colorado (Ely et al., 1991), one in the lower Fraser River valley near Vancouver, British Columbia (BC), Canada (Pryor, 1996), and one in Phoenix, Arizona (BBC Research & Consulting, 2003). A pilot focus group study was conducted in Washington, DC on behalf of the EPA to inform the 2006 PM NAAQS review (Abt Associates Inc., 2001). Although these studies indicate that some individuals considered the visual air quality associated with ambient levels of air pollution in urban areas to be unacceptable, these studies do not provide sufficient information on which to develop monetized co-benefits estimates. Specifically, the public perception studies do not provide preferences expressed in dollar values, even though they do suggest that the co-benefits associated with improving residential visibility are positive.

Studies in the peer-reviewed literature support a non-zero value for residential visibility (e.g., Brookshire et al., 1982; Loehman et al., 1994). Furthermore, Chestnut and Rowe (1990c) conclude that residential visibility co-benefits are likely to be at least as high as recreational visibility co-benefits because of the quantity of time most people spend in and near their homes and the substantial number of people affected. In previous assessments, the EPA used a study on residential visibility valuation conducted in 1990 (McClelland et al., 1993). Consistent with advice from SAB-Council, the EPA designated the McClelland et al. study as significantly less reliable for regulatory benefit-cost analysis, although it does provide useful estimates on the order of magnitude of residential visibility co-benefits (U.S. EPA-SAB, 1999).[26] In our approach for estimating residential visibility co-benefits, we replaced the previous methodology with a new benefits transfer approach and incorporated additional valuation studies. This new approach was developed for *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress* (U.S. EPA, 2011)[27] and reviewed by the SAB-Council (U. S. EPA-SAB, 2004, 2010a, 2010b).

[25] For more detail about these preference studies, including information about study designs and sampling protocols, please see Section 2 of the *Particulate Matter Urban-Focused Visibility Assessment* (U.S. EPA, 2010b).

[26] EPA's Advisory Council on Clean Air Compliance Analysis noted that the McClelland et al. (1993) study may not incorporate two potentially important adjustments. First, their study does not account for the "warm glow" effect, in which respondents may provide higher willingness to pay estimates simply because they favor "good causes" such as environmental improvement. Second, while the study accounts for non-response bias, it may not employ the best available methods. As a result of these concerns, the Council recommended that residential visibility be omitted from the overall primary benefits estimate. (U.S. EPA-SAB, 1999)

[27] This report is also known as the Second Prospective 812 analysis.

To value residential visibility improvements, the new approach draws upon information reported in the Brookshire et al. (1979), Loehman et al. (1985) and Tolley et al. (1986) studies.[28] Each of the studies provides estimates of household WTP to improve visibility conditions. While uncertainty exists regarding the precision of these older, stated-preference residential valuation studies, we believe their results support the argument that individuals have a non-zero value for residential visibility improvements. These studies provide primary visibility values for Atlanta, Boston, Chicago, Denver, Los Angeles, Mobile, San Francisco, and Washington D.C.[29]

In accordance with Chestnut and Rowe (1990c), we utilize the WTP estimates and the associated change in visual range from each study to estimate the β parameter for the eight study areas. The β parameter represents the WTP for a specific improvement in visibility in a specific location. Where studies provide multiple estimates for visual range improvements, we estimate β by regressing the natural log of the ratio of visual range following and prior to improvement against WTP. To express these value estimates in comparable terms across study locations, we express household WTP for a change in visual range in a specific MSA using the following function:

$$WTP\ (\Delta VR) = \beta^* \ln\left(\frac{VR_1}{VR_0}\right) \qquad (6.3)$$

where:

VR_0 = mean annual visual range in miles before the improvement,

VR_1 = mean annual visual range in miles after the improvement, and

β = parameter.

[28] Loehman et al. (1994) and Brookshire et al. (1982) published results in peer-reviewed journals based on the same underlying data we obtained from Loehman et al. (1985) and Brookshire et al. (1979). While the specific details need to compute visibility benefits using Tolley et al. (1986) were not published in a peer-reviewed journal, the overall work including study and survey design was subject to peer review during study development (see Leggett and Neumann, 2004 and Patterson et al., 2005). In addition, Tolley subsequently published a book (Tolley and Fabian, 1988) based on this research, which notes in the preface that the methods were critiqued throughout by various external economists. The EPA does not claim that this external critique necessarily constituted a formal peer review process, but we provide this information for transparency regarding the review of this work. The use of these studies as the only available information to estimate residential visibility co-benefits in the main estimate was supported by the SAB-Council (U.S. EPA-SAB, 1999a, 2010a).

[29] Recognizing potential fundamental issues associated with data collected in Cincinnati and Miami (e.g., see Chestnut et al. (1986) and Chestnut and Rowe (1990c), we do not include values for these cities in our analysis. The 8 MSAs where the valuation studies were conducted represent 15% of the total US population in 2020 (U.S. Census).

Total residential visibility co-benefits within a particular MSA are driven by visibility improvements, population density, and the WTP value applied. Only those people living within in an MSA are assumed to receive co-benefits from improved residential visibility. In other words, unlike recreational visibility, we do not assume a non-use value by people who live outside the MSA for residential visibility. Table 6-6 provides a summary of these valuation estimates for each study location, as well as an illustrative implied WTP value for a 10% improvement in visual range. As shown, the implied annual per-household WTP estimates for a hypothetical 10% improvement ranges from $21 to $220, depending on the study area. It is not surprising that such a range of values exists, as these study areas all feature different landscapes and vistas, populations and prevailing visibility conditions.

Table 6-6. Summary of Residential Visibility Valuation Estimates

City	Study	β Estimate	Implied WTP for 10% Improvement in Visual Range (1990$, 1990 income)	Implied WTP for 10% Improvement in Visual Range (2006$, 2020 income)
Atlanta	Tolley et al. (1986)	321	$31	$72
Boston	Tolley et al. (1986)	398	$38	$89
Chicago	Tolley et al. (1986)	310	$30	$69
Denver	Tolley et al. (1986)	696	$66	$155
Los Angeles	Brookshire et al. (1979)	94	$9	$21
Mobile	Tolley et al. (1986)	313	$30	$70
San Francisco	Loehman et al. (1985)	989	$94	$220
Washington, DC	Tolley et al. (1986)	614	$59	$137

Similar to recreational visibility co-benefits, we then incorporate preference calibration using the method developed by Smith, Van Houtven, and Pattanayak (2002), which is discussed in more detail in Appendix 6a of this RIA. This preference calibration is a change since *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress* (U.S. EPA, 2011) intended to address the SAB-Council's concern (U.S. EPA-SAB, 2010a) regarding the inconsistency regarding household income in the estimation of recreational and residential visibility. To express these "preference-calibrated" value estimates across study locations, we express household WTP for a change in visual range in a specific MSA using the following function:

$$WTP(\Delta VR) = m - [m^{\rho} + \theta * (VR_0^{\rho} - VR_1^{\rho})]^{\frac{1}{\rho}} \tag{6-4}$$

where:

m = household income,

ρ = shape parameter (0.1),

θ = WTP parameter corresponding to the visibility at MSA,

VR_0 = starting visibility, and

VR_1 = visibility after change.

While the primary estimate for residential visibility includes co-benefits in only the eight MSAs included in the valuation studies, people living in other urban areas also have non-zero values for residential visibility. For this reason, our approach includes a sensitivity analysis for the extrapolated residential visibility in the 351 additional MSAs.[30] Because there is considerable uncertainty about the validity of this benefit transfer approach, these extrapolated co-benefits are included in a sensitivity analysis only. This is an important distinction between the approach used in *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress* (U.S. EPA, 2011), where all cities were included in the total benefits approach. We believe that it is appropriate to deviate from the previous approach in order to be consistent with the approach used to estimate recreational visibility co-benefits and to recognize the uncertainty associated with extrapolating beyond the studied cities. Figure 6-6 indicates the study cities as well as the assignment of the other MSAs to the study cities.

The degree to which the three studies were successful in convincing respondents to focus solely on visibility is unclear, as none of the three studies includes follow-up questions necessary to investigate the issue. Furthermore, no other residential visibility CV studies provide evidence regarding the degree to which health effects are embedded in visibility values. Although the McClelland et al. (1991) study has a follow-up question designed to allocate WTP across several categories, the CV question in the McClelland et al. study was focused on air pollution generally rather than visibility. As a result, we do not adjust the results from these studies to account for potentially embedded health effects.

There are many factors that could influence WTP for residential visibility, and these factors vary across urban areas. In our approach, we utilize the benefit transfer approach

[30] The 351 additional MSAs plus the 8 study area MSAs represent 84% of the total US population in 2020 (U.S. Census).

developed for *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress* (U.S. EPA, 2011) report, but we recognize that there are alternative methods that we could have used. We assigned a valuation study area to each MSA based on two factors: geographic proximity to one of the eight study cities and elevation. Any MSA with a county elevation above 1,500 meters was assigned the Denver valuation instead of the nearest study area.[31] Because residents of Denver have a dramatic view of the Rocky Mountains that is rarely obstructed by trees, it is plausible that they might have a greater interest in protecting visibility than a city without nearby mountains. The geographic proximity factor is constrained in two areas. The San Francisco valuation study is only assigned to the six counties in the San Francisco Bay area MSAs because the study is unique among the three regarding the temporal description of visibility conditions, landscape/vistas, and prevailing weather conditions. In addition, the Los Angeles valuation was assigned to the Riverside MSA despite exceeding the elevation threshold.[32]

[31] Elevation data represent the county-level maximum, which were calculated using the ArcGIS Spatial Analyst tool "Zonal Statistics" using the geographic database HYDRO1K for North America (U.S. Geological Survey, 1997). This dataset and associated documentation are available on the Internet at DEMhttp://eros.usgs.gov/#/Find_Data/Products_and_Data_Available/gtopo30/hydro/namerica.

[32] Riverside MSA is assigned to the Los Angeles study area because a significant portion of Riverside County itself is located in the South Coast Air Quality Management District, which can be considered by to be part of the same regulated airshed as Los Angeles. The geographic assignment is preserved despite exceeding the elevation threshold because Riverside is adjacent to one of the study cities and this region has a particular set of location-specific characteristics that set it apart from Denver.

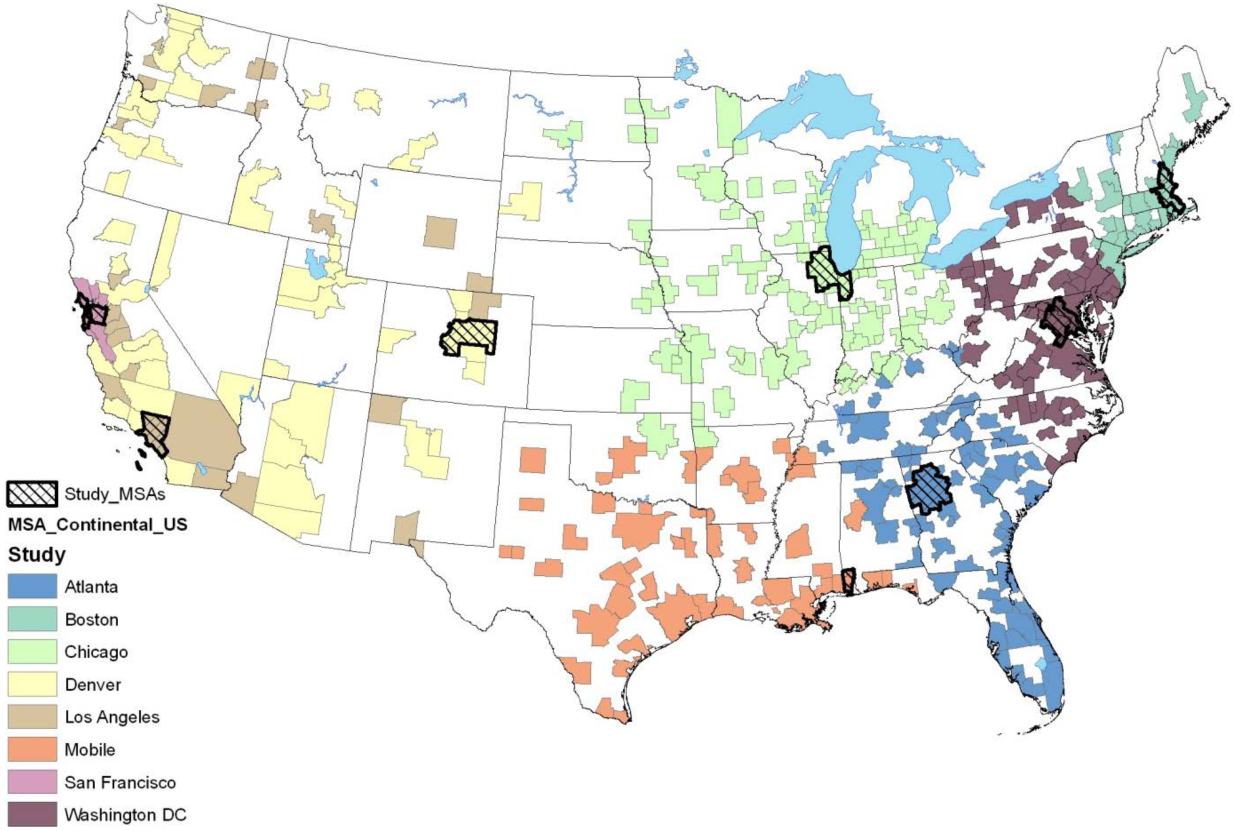


Figure 6-6. Residential Visibility Study City Assignment

6.3.5.2 Residential Visibility Limitations, Caveats, and Uncertainties

Similar to recreational visibility co-benefits, there are many data inputs into the residential visibility co-benefits that contribute to overall uncertainty. Our approach includes sensitivity analyses to characterize major omissions (i.e., co-benefits in other MSAs and coarse particles). A summary of the key assumptions including direction and magnitude of bias in our approach is provided in Table 6-7.

The valuation studies relied upon for the residential visibility co-benefits, although representing the best available estimates, have a number of limitations. These include the following:

- The survey design and implementation reflects the period in which the surveys were conducted. Since that time, many improvements to the stated preference methods have been developed.
- The vintage of the surveys (1970s and 1980s) invites questions whether the values are still valid for current populations, or more importantly for our approach, future populations in 2020.

- The survey focused only on populations in eight cities, so the transfer of the WTP estimates values to populations outside those cities requires that their preferences be similar to those in non-surveyed cities, as well as the visibility attributes be similar across study and transfer MSAs.

- There is an inherent difficulty in separating values expressed for visibility improvements from an overall value for improved air quality. The studies attempted to control for this, but most of the respondents did not feel that they were able to segregate residential visibility entirely from recreational visibility and health effects.

Table 6-7. Summary of Key Assumptions in the Residential Visibility Co-benefits[a]

Key Assumption	Direction of Bias	Magnitude of Effect
Residential and recreational visibility benefits are distinct and separable.	Potential Overestimate	Medium-Low
Estimates residential visibility benefits are limited to populations within the boundaries of MSAs. Areas outside of an MSA are not included in our approach.	Underestimate	Low
WTP values reflect only visibility improvements and not overall air quality improvements.	Potential Overestimate	Medium-Low
WTP values from studies in Atlanta, Boston, Chicago, Denver, Los Angeles, Mobile, San Francisco, and Washington D.C. can be accurately transferred to MSAs across the U.S. based on proximity and elevation	Unclear	Unclear
We assume that there are 2.68 people per household. Because this estimate has been decreasing over time, this may underestimate the number of households.	Potential Underestimate	Medium-Low

[a] A description of the classifications for magnitude of effects can be found in Appendix 5.B of this RIA.

6.3.5.3 Using Hedonic Economic Literature to Estimate Visibility Co-benefits

The hedonic model assumes that consumers do not value the consumption of a good directly, but rather value the characteristics contained within a good. In the context of property values, the consumer values both the physical attributes of the property (i.e., number of rooms, square footage, etc.) as well as geographic and environmental attributes (e.g., proximity to parks, visibility, etc.). Following the technique developed by Rosen (1974), property characteristics are regressed on the observed price of the properties within a given housing market to estimate the contribution of each characteristic to the overall price.

Numerous studies have applied hedonic methods to estimate the willingness to pay (WTP) for air quality changes (see Smith and Huang (1995) and Boyle and Kiel (2001) for literature reviews), but fewer researchers have focused specifically on visibility. Studies that have estimated the WTP to improve visibility have focused on specific housing markets like Los Angeles (Murdoch and Thayer, 1998; Beron et al., 2001) or San Francisco (Graves et al., 1998), and it is unclear if these results would be more broadly applicable to the rest of the county. While the literature demonstrates a link between pollutant concentrations and home-buying behavior, it is difficult to partition the WTP for changes in pollution between health and aesthetic concerns. Murdoch and Thayer (1998) use visibility as a surrogate for environmental quality and Beron et al. (2001) acknowledge that their parameter estimates likely reflect a combination of visual aesthetics and an absence of health effects. Delucchi et al. (2002) deals with this issue by partitioning WTP estimates from a hedonic model into health and visibility components using results from previous contingent valuation studies, and find that the estimate of visibility co-benefits is similar to estimates based on contingent valuation alone.

In 2004, the Advisory Council on Clean Air Compliance Analysis (SAB-Council) recommended that the EPA evaluate the available studies addressing residential visibility and consider the possibility of using hedonic property models to estimate residential visibility co-benefits (U.S. EPA-SAB, 2004). In response to this recommendation, the EPA evaluated the existing economic literature, and determined that there were substantial limitations that precluded the Agency from using these studies to make inferences regarding individuals' WTP for improved visibility (Leggett and Neumann, 2004). Specifically, the literature did not provide support for the assumption that market participants are aware of the spatial variation in visibility, and consider this variation when purchasing a home, and can successfully separate visibility effects from health effects (Leggett and Neumann, 2004). This conclusion is also supported by Delucchi et al. (2004), which found that hedonic price analysis does not capture all of the health effects of air pollution because homebuyers may not be fully informed about these effects .

Research since 2004 has attempted to address limitations of the hedonic method through the use of U.S. Census microdata (Bayer et al., 2009), spatial statistical methods (Anselin and Le Gallo, 2006; Anselin and Lozana-Gracia, 2009; Beron et al., 2004; Kim et al., 2010) and more complete air quality data and information about nonattainment status (Chay and Greenstone, 2005). However, none of these studies specifically address visibility, and they are therefore of limited use at this time. While the current state of the literature does not provide a basis for using hedonics-based approaches, continued innovations in methodology and the further development of national, micro-level housing and demographic datasets may open possibilities for national-scale hedonics-based benefit analysis in the future.

Regardless of whether we use hedonic models or stated preference surveys to estimate co-benefits arising from improved visibility, it is important to emphasize that estimates of WTP for residential or recreational visibility improvements are not substitutes for health benefits. As previously mentioned, people often have difficulty separating their health concerns from their aesthetic concerns when evaluating preferences for visibility, which could overestimate visibility co-benefits if not properly controlled. However, because we use a damage-function approach to estimate health benefits (see Chapter 5 of this RIA), the health benefits estimates are unaffected by any potential confounding with visibility preferences.

6.3.6 Discussion of Visibility Co-benefits

As described in the previous sections of this chapter, the estimation of visibility co-benefits is complex and suffers from unavoidable limitations. While we are confident that the underlying scientific literature supports a non-zero estimate for visibility co-benefits attributable to emission reductions, we are less confident in the magnitude of those co-benefits outside of previously studied locations. While acknowledging these limitations, it is important to note that this general approach was included in *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress* (U.S. EPA, 2011), which was reviewed by the SAB-Council (U.S. EPA-SAB, 2010a, 2010b). Although the SAB-Council highlighted concerns with the visibility approach used in the study, it did not recommend that visibility benefits be excluded. We have addressed the SAB-Council's concern regarding inconsistency between estimation of residential and recreational visibility in our approach. However, we do not have the data to address the SAB-Council's concern regarding inclusion of night-time benefits of visibility improvements in our annual average, which may lead to an underestimation of visibility benefits. To minimize uncertainties related to extrapolation and geographical double counting, our approach only includes a subset of monetized visibility co-benefits in the core monetized visibility co-benefits estimate to correspond with our higher level of confidence in recreational co-benefits within the study regions and residential co-benefits within the study cities. Although we are confident that visibility co-benefits extend beyond these studied areas, we are less confident about the magnitude of those co-benefits. However, it is unclear the degree to which the visibility valuation surveys were successful in controlling for potential double counting embedded health benefits.

Consistent with the approach described in the proposal RIA, we have described a revised approach for estimating visibility co-benefits, including light extinction estimation methods, visitation data for Class 1 areas (used in extrapolating co-benefits), valuation studies for residential visibility co-benefits, and the benefit transfer technique for residential co-benefits. Including residential visibility co-benefits in the core visibility co-benefits estimates reflects an evolution in our understanding of the nature and importance of the effect on public welfare from visibility impairment to a more multifaceted approach that includes non-Class 1 areas, such as urban areas. This evolution has occurred in conjunction with the expansion of available PM data and information from associated studies of public perception, valuation and personal comfort and well-being. While visibility preference studies (Abt Associates Inc., 2001, Ely et al., 1991, Pryor, 1996, BBC Research & Consulting, 2003) also provide support for a non-zero benefits estimate, these surveys did not include questions that would enable monetization of those preferences.

Despite these improvements, we are limited by the available peer-reviewed studies on visibility co-benefits, which have not undergone a similar expansion as the health literature. Each of these valuation studies has limitations, which are identified in the sections 6.3.4.2 and 6.3.5.2. When the SAB-Council reviewed the visibility benefits analysis for *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress* (U.S. EPA, 2011), they also lamented on the need for additional research to improve methods and estimates (U.S. EPA-SAB, 2010a, 2010b). Because of time and resource constraints, performing original research for regulatory analyses of specific policy actions is infeasible. Most importantly, we are interested in recently published national-scale visibility valuation studies that incorporate current CV best practices, as the existing studies are limited to specific subset of geographic areas. Other important research questions that remain unresolved include identifying factors that affect valuation preferences in order to facilitate benefits transfer from the original studies to transfer sites across localities, disentangling health and ecosystem valuation from visibility valuation, usefulness of preference calibration, and potential role of hedonic valuation approaches. Many of these same research needs were identified by Cropper (2000), but they have yet to be addressed by the research community.

In Appendix 6.B, we provide the results of an illustrative analysis of the visibility co-benefits associated with the 2020 base case 2020 control case simulation described in Chapter 3 that were used to develop the air quality ratios; however, we do not have air quality model runs for the regulatory baseline and the alternative standard levels that would allow us to calculate the visibility co-benefits of attaining the revised primary standard.

6.4 Materials Damage Co-benefits

Building materials including metals, stones, cements, and paints undergo natural weathering processes from exposure to environmental elements (e.g., wind, moisture, temperature fluctuations, sunlight, etc.). Pollution can worsen and accelerate these effects. Deposition of PM is associated with both physical damage (materials damage effects) and impaired aesthetic qualities (soiling effects). Wet and dry deposition of PM can physically affect materials, adding to the effects of natural weathering processes, by potentially promoting or accelerating the corrosion of metals, by degrading paints and by deteriorating building materials such as stone, concrete and marble (U.S. EPA, 2009b). The effects of PM are exacerbated by the presence of acidic gases and can be additive or synergistic due to the complex mixture of pollutants in the air and surface characteristics of the material. Acidic deposition has been shown to have an effect on materials including zinc/galvanized steel and other metal, carbonate stone (as monuments and building facings), and surface coatings (paints) (Irving, 1991). The effects on historic buildings and outdoor works of art are of particular concern because of the uniqueness and irreplaceability of many of these objects.

The PM ISA concludes that evidence is sufficient to support a causal relationship between PM and effects on materials (U.S. EPA, 2009b). Considerable research has been conducted on the effects of air pollutants on metal surfaces due to the economic importance of these materials, especially steel, zinc, aluminum, and copper. Moisture is the single greatest factor promoting metal corrosion; however, deposited PM can have additive, antagonistic or synergistic effects. In general, SO_2 is more corrosive than NO_x although mixtures of NO_x, SO_2 and other particulate matter corrode some metals at a faster rate than either pollutant alone (U.S. EPA, 2008). Metal structures are usually coated by alkaline corrosion product layers and thus are subject to increased corrosion by acidic deposition. In addition, research has demonstrated that iron, copper, and aluminum-based products are subject to increased corrosion due to pollution (Irving, 1991). Information from both the PM ISA (U.S. EPA, 2009b) and NO_x/SO_x ISA (U.S. EPA, 2008) suggest that the extent of damage to metals due to ambient PM is variable and dependent upon the type of metal, prevailing environmental conditions, rate of natural weathering and presence or absence of other pollutants

In addition, the deposition of PM can cause soiling, which is the accumulation of dirt, dust, and ash on exposed surfaces such as metal, glass, stone and paint. Particles consisting primarily of carbonaceous compounds can cause soiling of commonly used building materials and culturally important items such as statues and works of art. Soiling occurs when PM accumulates on an object and alters the optical characteristics (appearance). The reflectivity of a surface may be changed or presence of particulates may alter light transmission. These effects can reduce the aesthetic value of a structure or result in reversible or irreversible damage to statues, artwork and architecturally or culturally significant buildings. Due to soiling of building surfaces by PM, the frequency and duration of cleaning or repainting may be increased. In addition to natural factors, exposure to PM may give painted surfaces a dirty appearance. Pigments in works of art can be degraded or discolored by atmospheric pollutants, especially sulfates (U.S. EPA, 2008). Previous assessments estimated household soiling co-benefits based on the Manuel et al. (1982) study of consumer expenditures on cleaning and household maintenance. However, the data used to estimate household soiling damages in the Manuel et al. study is from a 1972 consumer expenditure survey and as such may not accurately represent consumer preferences in the future. In light of this significant limitation, we believe that this study cannot provide reliable estimates of the likely magnitude of the co-benefits of reduced PM household soiling.

In order to estimate the monetized co-benefits associated with reducing materials damage and household soiling, quantitative relationships are needed between particle size, concentration, chemical concentrations and frequency of maintenance and repair. Such an analysis would require three steps:

1. Develop a national inventory of sensitive materials;
2. Derive concentration-response functions that relate material damage to change in pollution concentration or deposition; and,
3. Estimate the value of lost materials and/or repair of damage.

Due to data limitations and uncertainties inherent in each of these steps, we have chosen not to include a monetized estimate of materials damage and household soiling in this analysis. The PM ISA concluded that there is considerable uncertainty with regard to interaction of co-pollutants in regards to materials damage and soiling processes (U.S. EPA, 2009b). Previous benefits analyses by the EPA have provided quantitative estimates of materials damage (U.S. EPA, 2011b) and household soiling damage (U.S. EPA, 1999). Consistent with SAB advice (U.S. EPA, 1998), we determined that the existing data are not sufficient to calculate a reliable estimate of future year household soiling damages (U.S. EPA, 1998). These previous analyses have shown that materials damage co-benefits are significantly smaller than the health benefits associated with reduced exposure to $PM_{2.5}$ and ozone, or even visibility co-benefits. However, studies of materials damage to historic buildings and outdoor artwork in Sweden (Grosclaude and Soguel, 1994) indicate that these co-benefits could be an order of magnitude larger than household soiling co-benefits.

In the absence of quantified co-benefits, we provide a qualitative description of the avoided damage associated with reducing PM and PM precursor pollutants. Table 6-8 shows the effect of various $PM_{2.5}$ precursor pollutants and other co-pollutants on various materials.

Table 6-8. Materials Damaged by Pollutants Affected by this Rule (U.S. EPA, 2011b)

Pollutant	Unquantified Effects / Damage to:
Sulfur oxides	Infrastructural materials—galvanized and painted carbon steel Commercial buildings—carbonate stone, metal, and painted wood surfaces Residential buildings—carbonate stone, metal, and painted wood surfaces Monuments—carbonate stone and metal Structural aesthetics Automotive finishes—painted metal
Hydrogen ion and nitrogen oxides	Infrastructural materials—galvanized and painted carbon steel Zinc-based metal products, such as galvanized steel Commercial and residential buildings—carbonate stone, metal, and wood surfaces Monuments—carbonate stone and metal Structural aesthetics Automotive finishes—painted metal
Carbon dioxide	Zinc-based metal products, such as galvanized steel
Formaldehyde	Zinc-based metal products, such as galvanized steel
Particulate matter	Household cleanliness (i.e., household soiling)
Ozone	Rubber products (e.g., tires)

6.5 Climate Co-benefits

Actions taken by state and local governments to implement the revised annual primary standard are likely to have implications for climate change because emission reductions ultimately implemented to meet the standard may have impacts on emissions of long-lived greenhouse gas (GHG) such as carbon dioxide (CO_2), short-lived climate forcers such as black carbon (BC), and cooling aerosols like organic carbon (OC). Our ability to quantify the climate effects of these revised standard is limited due to lack of available information on the co-controlled GHG emission reductions, the energy and associated climate gas implications of control technologies assumed in the illustrative regulatory alternatives, and remaining uncertainties regarding the impact of long-lived and short-lived climate forcer impacts on climate change. For this RIA, we discuss qualitatively the implications of potential emission reductions in warming and cooling aerosols and changes in long-lived GHG emissions such as

CO_2 for the regulatory alternatives. Implementation strategies undertaken by state and local governments to comply with the standards may differ from the illustrative emission reduction strategies in this RIA. It is important to note that the net climate forcing depends on the specific combinations of emission reductions chosen to meet the revised standard because of the differences in warming and cooling potential of the difference pollutants.

6.5.1 Climate Effects of Short Lived Climate Forcers

Pollutants that affect the energy balance of the earth are referred to as climate forcers. A pollutant that increases the amount of energy in the Earth's climate system is said to exert "positive radiative forcing," which leads to warming and climate change. In contrast, a pollutant that exerts negative radiative forcing reduces the amount of energy in the Earth's system and leads to cooling.

Long-lived gases such as CO_2 differ from short-lived pollutants such as BC in the length of time they remain in the atmosphere affecting the earth's energy balance. Long-lived gases remain in the atmosphere for hundreds to thousands of years. Short-lived climate forcers (SLCFs), in contrast, remain in the atmosphere for short periods of time ranging from days to weeks. The potential to affect near-term climate change and the rate of climate change with policies to address these emissions is gaining attention nationally and internationally (e.g., *Black Carbon Report to Congress* (U.S. EPA, 2012b), Arctic Council Task Force, Global Methane Initiative, and Convention on Long-Range Trans-boundary Air Pollution of the United Nations Economic Commission for Europe). A recent United Nations Environmental Programme (UNEP) study provides the most comprehensive analysis to date of the co-benefits of measures to reduce SLCFs including methane, ozone, and black carbon assessing the health, climate, and agricultural co-benefits of a suite of mitigation technologies. The report concludes that the climate is changing now, and these changes have the potential to "trigger abrupt transitions such as the release of carbon from thawing permafrost and biodiversity loss." While reducing long-lived GHGs such as CO_2 is necessary to protect against long-term climate change, reducing SLCF gases including BC and ozone is beneficial and will slow the rate of climate change within the first half of this century (UNEP, 2011).

6.5.1.1 Climate Effects of Black Carbon

Black carbon is the most strongly light-absorbing component of $PM_{2.5}$, and is formed by incomplete combustion of fossil fuels, biofuels, and biomass. The short atmospheric lifetime of BC lasting from days to weeks and the mechanisms by which BC affects climate distinguish it from long-lived GHGs like CO_2. This means that actions taken to reduce the BC constituents in direct $PM_{2.5}$ will have almost immediate effects on climate change. Emissions sources and ambient concentrations of BC vary geographically and temporally resulting in climate effects that are more regionally and seasonally dependent than the effects of long-lived, well-mixed GHGs. Likewise, mitigation actions for BC will produce different climate impacts depending on the region, season, and emission source category affected.

BC influences climate in multiple ways: directly absorbing light, reducing the reflectivity ("albedo") of snow and ice through deposition, and interacting with clouds. BC affects climate directly by absorbing both incoming and outgoing radiation of all wavelengths. In contrast, GHGs mainly trap outgoing infrared radiation from the earth's surface. Per unit of mass in the atmosphere, BC can absorb a million times more energy than CO_2 (Bond and Sun 2005). This strong absorptive capacity is the property most relevant to its potential to affect the Earth's climate. When BC is deposited on snow and ice, it darkens the surface and decreases albedo, thereby increasing absorption and accelerating melting. Finally, BC also affects climate indirectly by altering the properties of clouds, affecting cloud reflectivity, precipitation, and surface dimming. These indirect impacts of BC are associated with all ambient particles and may lead to cooling, but are not associated with long-lived well mixed GHGs.

Regional climate impacts of BC are highly variable, and sensitive regions such as the Arctic and the Himalayas are particularly vulnerable to the warming and melting effects of BC. Snow and ice cover in the Western U.S. has also been affected by BC. Specifically, deposition of BC on mountain glaciers and snow packs produces a positive snow and ice albedo effect, contributing to the melting of snowpack earlier in the spring and reducing the amount of snowmelt that normally would occur later in the spring and summer (Hadley et al. 2010). This has implications for freshwater resources in regions of the U.S. dependent on snow-fed or glacier-fed water systems. In the Sierra Nevada mountain range, Hadley et al. (2010) found BC at different depths in the snowpack, deposited over the winter months by snowfall. In the spring, the continuous uncovering of the BC contributed to the early melt. A model capturing the effects of soot on snow in the western U.S. shows significant decreases in snowpack between December and May (Qian et al., 2009). Snow water equivalent (the amount of water that would be produced by melting all the snow) is reduced 2-50 millimeters (mm) in mountainous areas, particularly over the Central Rockies, Sierra Nevadas, and western Canada. A study found that biomass burning emissions in Alaska and the Rocky Mountain region during the summer can enhance snowmelt. Dust deposition on snow, at high concentrations, can have similar effects to BC (Koch et al., 2007). Similarly, a study done by Painter et al. (2007) in the

San Juan Mountains in Colorado indicated a decrease in snow cover duration of 18-35 days as a result of dust transported from non-local desert sources.

The illustrative emission reduction strategies evaluated for this rule include reductions in BC emissions that will tend to have a beneficial cooling effect on the atmosphere. BC and elemental carbon (EC) (or particulate elemental carbon (PEC)) are used interchangeably in this report because the EPA traditionally estimates EC emissions rather than BC and for the purpose of this analysis these measures are essentially equivalent.

6.5.1.2 Climate Effects of Nitrates, Sulfate, and Organic Carbon (excluding BC)

The composition of the total emissions mixture is also relevant as to whether emissions are warming or cooling to the atmosphere. Pollutants such as SO_2, NO_X, and most OC particles tend to produce a cooling influence on climate. Exceptions include OC deposition on snow and ice, which leads to increased melting.

In addition, it is important to account for the indirect effects of all PM constituents on climate: all aerosols (including BC) affect climate indirectly by changing the reflectivity and lifetime of clouds. The net indirect effect of all aerosols is very uncertain but is thought to be a net cooling influence.

6.5.1.3 Climate Effects of Ozone

Ozone changes due to this revised annual standard are not estimated for this analysis but may occur due to the NO_x reductions estimated. Ozone is a well-known SLCF (U.S. EPA, 2006). Stratospheric ozone (the upper ozone layer) is beneficial because it protects life on Earth from the sun's harmful ultraviolet (UV) radiation. In contrast, tropospheric ozone (ozone in the lower atmosphere) is a harmful air pollutant that adversely affects human health and the environment and contributes significantly to regional and global climate change. Due to its short atmospheric lifetime, tropospheric ozone concentrations exhibit large spatial and temporal variability (U.S. EPA, 2009). The discernible influence of ground level ozone on climate leads to increases in global surface temperature and changes in hydrological cycles. While reducing long-lived GHGs such as CO_2 is necessary to protect against long-term climate change, reducing SLCF gases including ozone is beneficial and will slow the rate of climate change within the first half of this century (UNEP, 2011).

6.5.1.4 SLCFs Summary and Conclusions

Assessing the net climate impact of SLCFs for the illustrative emission reduction strategies is outside the scope of this regulatory analysis and requires climate atmospheric modeling not undertaken due to time and resource constraints. Information about the amount of BC relative to non-BC constituents emitted from a source is important. In general, these non-BC constituents are emitted in greater volume than BC, counteracting the warming influence of BC. Qualitatively, it seems likely that BC emission reductions associated with directly emitted $PM_{2.5}$ reductions will be beneficial for the climate in terms of reduced radiative forcing and deposition on snow and ice. Reductions in OC, sulfates and nitrates are likely to produce warming in the atmosphere. The indirect impacts of aerosols on clouds and precipitation remain the subject of great uncertainty making it more difficult to estimate the quantitative impact of aerosol reductions on climate.

6.5.2 Climate Effects of Long-Lived Greenhouse Gases

The EPA Administrator found in 2009 that elevated concentrations of the six major GHGs, including CO_2, endanger the public health and public welfare of current and future generations (FR 77 66496). While addressing short-lived climate forcers can result in near-term (and sometimes regionally specific) co-benefits as well as reductions in the rate of warming, reductions of long-term warming would require mitigation of long-lived GHGs. We are unable to quantify the impact of the illustrative emission reduction strategies for this rulemaking on long-lived climate gases due lack of available data. However, State and Local governments may want to consider human health, welfare, and climate implications of regulatory strategies undertaken to implement the promulgated PM standards.

6.6 Ecosystem Co-benefits and Services

The effects of air pollution on the health and stability of ecosystems are potentially very important. At present, it is difficult to measure the impact of reducing air pollution in a national scale analysis across different types of ecosystems and different pollutant effects. Previous science assessments by the EPA (U.S. EPA, 2006a; 2008c; 2009b) have determined that air pollution can be directly linked to aquatic and terrestrial acidification, nutrient enrichment, vegetation injury, and metal bioaccumulation in animals. Ecosystem services are a useful conceptual framework for analyzing the impact of ecosystem changes on public welfare.

Ecosystem services can be generally defined as the benefits that individuals and organizations obtain from ecosystems. The EPA has defined ecological goods and services as the "outputs of ecological functions or processes that directly or indirectly contribute to social welfare or have the potential to do so in the future. Some outputs may be bought and sold, but most are not marketed" (U.S. EPA, 2006c). Figure 6-7 provides the Millennium Ecosystem Assessment's schematic demonstrating the connections between the categories of ecosystem services and human well-being. The interrelatedness of these categories means that any one ecosystem may provide multiple services. Changes in these services can affect human well-being by affecting security, health, social relationships, and access to basic material goods (MEA, 2005).

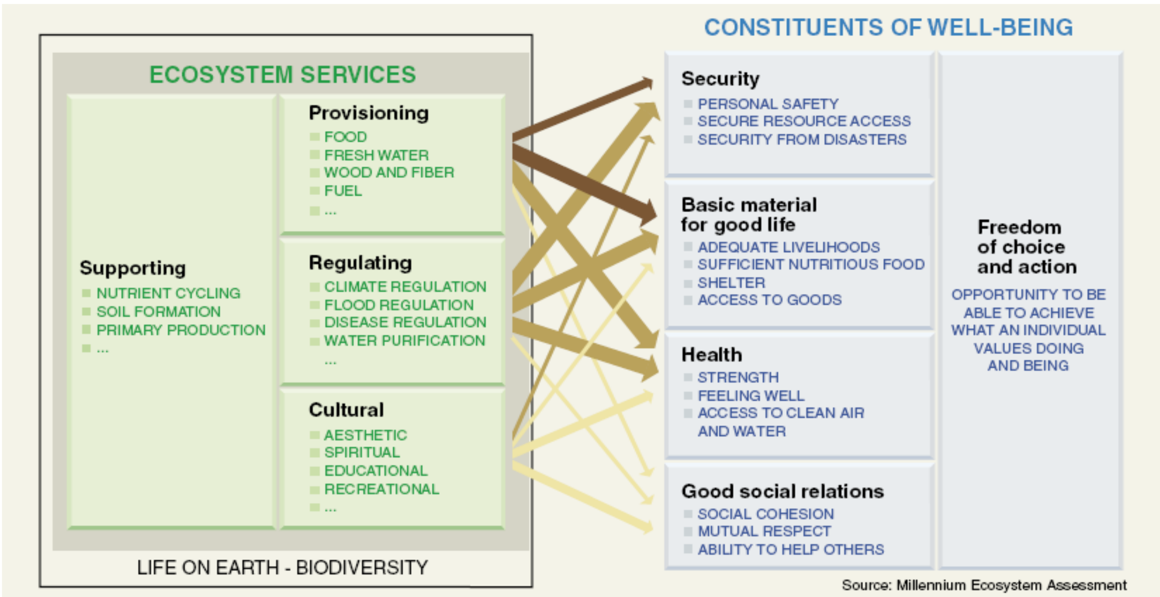


Figure 6-7. Linkages between Categories of Ecosystem Services and Components of Human Well-Being from Millennium Ecosystem Assessment (MEA, 2005)

In the Millennium Ecosystem Assessment (MEA, 2005), ecosystem services are classified into four main categories:

1. Provisioning: Products obtained from ecosystems, such as the production of food and water
2. Regulating: Benefits obtained from the regulation of ecosystem processes, such as the control of climate and disease
3. Cultural: Nonmaterial benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences
4. Supporting: Services necessary for the production of all other ecosystem services, such as nutrient cycles and crop pollination

The monetization of ecosystem services generally involves estimating the value of ecological goods and services based on what people are willing to pay (WTP) to increase ecological services or by what people are willing to accept (WTA) in compensation for reductions in them (U.S. EPA, 2006c). There are three primary approaches for estimating the monetary value of ecosystem services: market-based approaches, revealed preference methods, and stated preference methods (U.S. EPA, 2006c). Because economic valuation of ecosystem services can be difficult, nonmonetary valuation using biophysical measurements and concepts also can be used. An example of a nonmonetary valuation method is the use of relative-value indicators (e.g., a flow chart indicating uses of a water body, such as boatable, fishable, swimmable, etc.). It is necessary to recognize that in the analysis of the environmental responses associated with any particular policy or environmental management action, only a subset of the ecosystem services likely to be affected are readily identified. Of those ecosystem services that are identified, only a subset of the changes can be quantified. Within those services whose changes can be quantified, only a few will likely be monetized, and many will remain nonmonetized. The stepwise concept leading up to the valuation of ecosystems services is graphically depicted in Figure 6-8.

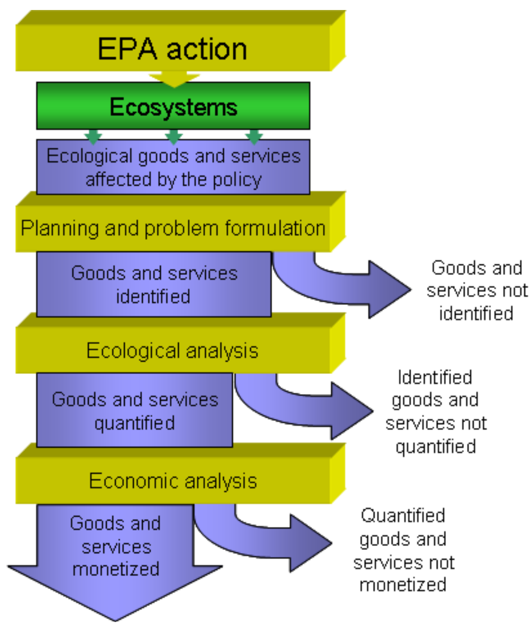


Figure 6-8. Schematic of the Benefits Assessment Process (U.S. EPA, 2006c)

6.6.1 Ecosystem Co-benefits for Metallic and Organic Constituents of PM

Several significant ecological effects are associated with deposition of chemical constituents of ambient PM such as metals and organics (U.S. EPA, 2009b). The trace metal constituents of PM include cadmium, copper, chromium, mercury, nickel, zinc, and lead. The organics include persistent organic pollutants (POPs), polyaromatic hydrocarbons (PAHs) and polybromiated diphenyl ethers (PBDEs). Exposure to PM for direct effects occur via deposition (e.g., wet, dry or occult) to vegetation surfaces, while indirect effects occur via deposition to ecosystem soils or surface waters where the deposited constituents of PM then interacts with biological organisms. While both fine and coarse-mode particles may affect plants and other organisms, more often the chemical constituents drive the ecosystem response to PM (Grantz et al., 2003). Ecological effects of PM include direct effects to metabolic processes of plant foliage; contribution to total metal loading resulting in alteration of soil biogeochemistry and microbiology, plant and animal growth and reproduction; and contribution to total organics loading resulting in bioaccumulation and biomagnification across trophic levels.

The PM ISA concludes that a causal relationship is likely to exist between deposition of PM and a variety of effects on individual organisms and ecosystems (U.S. EPA 2009b). Most direct ecosystem effects associated with particulate pollution occur in severely polluted areas near industrial point sources (quarries, cement kilns, metal smelting) (U.S. EPA, 2009b). However the PM ISA also finds, in many cases, it is difficult to characterize the nature and magnitude of effects and to quantify relationships between ambient concentrations of PM and ecosystem response due to significant data gaps and uncertainties as well as considerable variability that exists in the components of PM and their various ecological effects (U.S. EPA, 2009b).

Particulate matter can adversely impact plants and ecosystem services provided by plants by deposition to vegetative surfaces (U.S. EPA, 2009b). Particulates deposited on the surfaces of leaves and needles can block light, altering the radiation received by the plant. PM deposition near sources of heavy deposition can obstruct stomata limiting gas exchange, damage leaf cuticles and increase plant temperatures (U.S. EPA, 2009b). Plants growing on roadsides exhibit impact damage from near-road PM deposition, having higher levels of organics and heavy metals, and accumulate salt from road de-icing during winter months (U.S. EPA, 2009b). In addition, atmospheric PM can convert direct solar radiation to diffuse radiation, which is more uniformly distributed in a tree canopy, allowing radiation to reach lower leaves (U.S. EPA, 2009b). Decreases in crop yields (a provisioning service) due to reductions in solar radiation have been attributed to regional scale air pollution in other counties with especially severe regional haze (Chameides et al., 1999).

In addition to damage to plant surfaces, deposited PM can be taken up by plants from soil or foliage. Copper, zinc, and nickel have been shown to be directly toxic to vegetation under field conditions (U.S. EPA, 2009b). The ability of vegetation to take up heavy metals is dependent upon the amount, solubility and chemical composition of the deposited PM. Uptake of PM by plants from soils and vegetative surfaces can disrupt photosynthesis, alter pigments and mineral content, reduce plant vigor, decrease frost hardiness and impair root development.

Particulate matter can also contain organic air toxic pollutants, including PAHs, which are a class of polycyclic organic matter (POM). PAHs can accumulate in sediments and bioaccumulate in freshwater, flora and fauna. The uptake of organics depends on the plant species, site of deposition, physical and chemical properties of the organic compound and prevailing environmental conditions (U.S. EPA, 2009b). Different species can have different uptake rates of PAHs. For example, zucchini (*Cucurbita pepo*) accumulated significantly more PAHs than related plant species (Parrish et al., 2006). PAHs can accumulate to high enough concentrations in some coastal environments to pose an environmental health threat that includes cancer in fish populations, toxicity to organisms living in the sediment and risks to those (e.g., migratory birds) that consume these organisms (Simcik et al., 1996; Simcik et al., 1999). Atmospheric deposition of particles is thought to be the major source of PAHs to the sediments of Lake Michigan, Chesapeake Bay, Tampa Bay and other coastal areas of the U.S. (Arzavus, Dickhut, and Canuel, 2001).

Contamination of plant leaves by heavy metals can lead to elevated concentrations in the soil. Trace metals absorbed into the plant, frequently bind to the leaf tissue, and then are lost when the leaf drops. As the fallen leaves decompose, the heavy metals are transferred into the soil (Cotrufo et al., 1995; Niklinska et al., 1998). Many of the major indirect plant responses to PM deposition are chiefly soil-mediated and depend on the chemical composition of individual components of deposited PM. Upon entering the soil environment, PM pollutants can alter ecological processes of energy flow and nutrient cycling, inhibit nutrient uptake to plants, change microbial community structure and, affect biodiversity. Accumulation of heavy metals in soils depends on factors such as local soil characteristics, geologic origin of parent soils, and metal bioavailability. Heavy metals, such as zinc, copper, and cadmium, and some pesticides can interfere with microorganisms that are responsible for decomposition of soil litter, an important regulating ecosystem service that serves as a source of soil nutrients (U.S. EPA, 2009b). Surface litter decomposition is reduced in soils having high metal concentrations. Soil communities have associated bacteria, fungi, and invertebrates that are essential to soil nutrient cycling processes. Changes to the relative species abundance and community composition are associated with deposited PM to soil biota (U.S. EPA, 2009b).

Atmospheric deposition can be the primary source of some organics and metals to watersheds. Deposition of PM to surfaces in urban settings increases the metal and organic component of storm water runoff (U.S. EPA, 2009b). This atmospherically-associated pollutant burden can then be toxic to aquatic biota. The contribution of atmospherically deposited PAHs to aquatic food webs was demonstrated in high elevation mountain lakes with no other anthropogenic contaminant sources (U.S. EPA, 2009b). Metals associated with PM deposition limit phytoplankton growth, affecting aquatic trophic structure. Long-range atmospheric transport of 47 pesticides and degradation products to the snowpack in seven national parks in the Western U.S. was recently quantified indicating PM-associated contaminant inputs to receiving waters during spring snowmelt (Hageman et al., 2006).

The recently completed Western Airborne Contaminants Assessment Project (WACAP) is the most comprehensive database on contaminant transport and PM depositional effects on sensitive ecosystems in the Western U.S. (Landers et al., 2008). In this project, the transport, fate, and ecological impacts of anthropogenic contaminants from atmospheric sources were assessed from 2002 to 2007 in seven ecosystem components (air, snow, water, sediment, lichen, conifer needles and fish) in eight core national parks. The study concluded that bioaccumulation of semi-volatile organic compounds occurred throughout park ecosystems, an elevational gradient in PM deposition exists with greater accumulation in higher altitude areas, and contaminants accumulate in proximity to individual agriculture and industry sources, which is counter to the original working hypothesis that most of the contaminants would originate from Eastern Europe and Asia.

Although there is considerable data on impacts of PM on ecological receptors, few studies link ambient PM levels to observed effect. This is due, in part, to the nature, deposition, transport and fate of PM in ecosystems. Some of the difficulties in quantifying the ecosystem co-benefits associated with reduced PM deposition include the following:

- PM is not a single pollutant, but a heterogeneous mixture of particles differing in size, origin and chemical composition. Since vegetation and other ecosystem components are affected more by particulate chemistry than size fraction, exposure to a given mass concentration of airborne PM may lead to widely differing plant or ecosystem responses, depending on the particular mix of deposited particles.

- Composition of ambient PM varies in time and space and the particulate mixture may have synergistic, antagonistic or additive effects on ecological receptors depending upon the chemical species present.

- Presence of co-pollutants makes it difficult to attribute observed effects to ecological receptors to PM alone or one component of deposited PM.

- Ecosystem effects linked to PM are difficult to determine because the changes may not be observed until pollutant deposition has occurred for many decades. Furthermore, many PM components bioaccumulate over time in organisms or plants, making correlations to ambient levels of PM difficult.

- Multiple ecological stressors can confound attempts to link specific ecosystem responses to PM deposition. These stressors can be anthropogenic (e.g., habitat destruction, eutrophication, other pollutants) or natural (e.g., drought, fire, disease). Deposited PM interacts with other stressors to affect ecosystem patterns and processes.

- Each ecosystem has a unique topography, underlying bedrock, soils, climate, meteorology, hydrologic regime, natural and land use history, and species composition. Sensitivity of ecosystem response can be highly variable in space and time. Because of this variety and lack of data for most ecosystems, extrapolating these effects from one ecosystem to another is highly uncertain.

6.6.2 Ecosystem Co-benefits from Reductions in Nitrogen and Sulfur Emissions

Emissions of the PM precursors, such as nitrogen and sulfur oxides occur over large regions of North America. Once these pollutants are lofted to the middle and upper troposphere, they typically have a much longer lifetime and, with the generally stronger winds at these altitudes, can be transported long distances from their source regions. The length scale of this transport is highly variable owing to differing chemical and meteorological conditions encountered along the transport path (U.S. EPA, 2008c). Secondary particles are formed from NO_X and SO_2 gaseous emissions and associated chemical reactions in the atmosphere. Deposition can occur in either a wet (i.e., rain, snow, sleet, hail, clouds, or fog) or dry form (i.e., gases or particles). Together these emissions are deposited onto terrestrial and aquatic ecosystems across the U.S., contributing to the problems of acidification, nutrient enrichment, and methylmercury production as represented in Figures 6-9 and 6-10. Although there is some evidence that nitrogen deposition may have positive effects on agricultural and forest output through passive fertilization, it is likely that the overall value is very small relative to other health and welfare effects. In addition to deposition effects, SO_2 can affect vegetation at ambient levels near pollution sources.

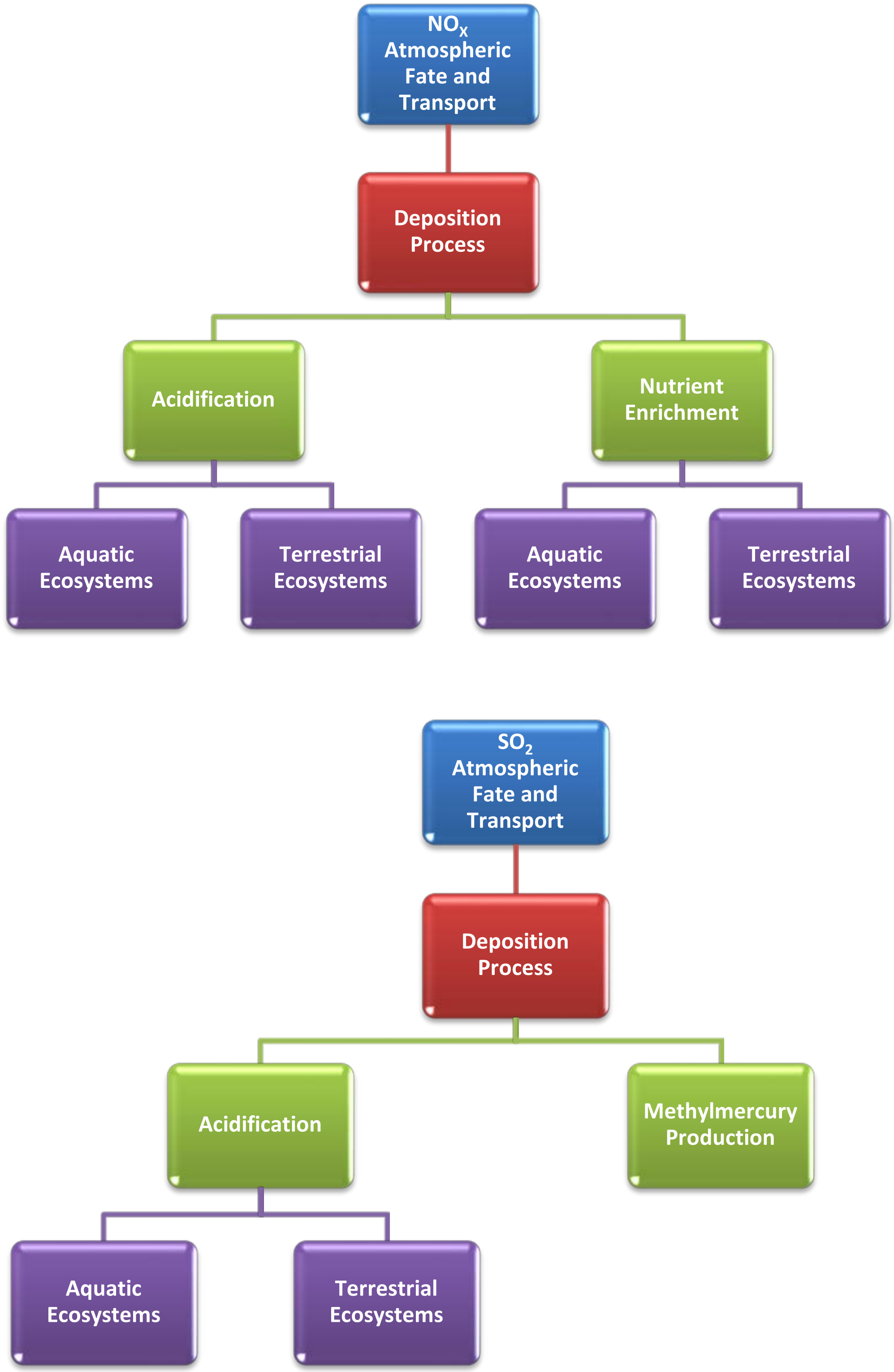


Figure 6-9. Schematics of Ecological Effects of Nitrogen and Sulfur Deposition

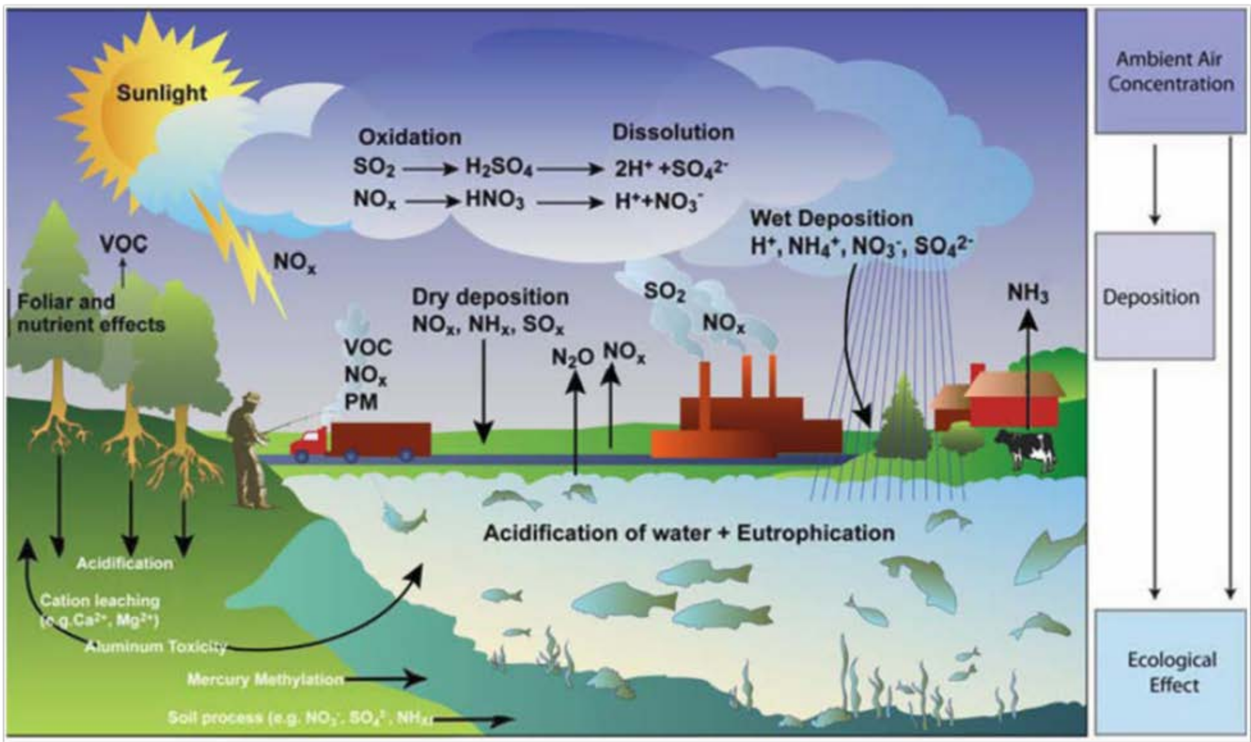


Figure 6-10. Nitrogen and Sulfur Cycling, and Interactions in the Environment

Source: U.S. EPA, 2008c.

The atmospheric lifetimes of particles vary with particle size. Accumulation-mode particles such as sulfates are kept in suspension by normal air motions and have a lower deposition velocity than coarse-mode particles; they can be transported thousands of kilometers and remain in the atmosphere for a number of days. They are removed from the atmosphere primarily by cloud processes. Particulates affect acid deposition by serving as cloud condensation nuclei and contribute directly to the acidification of rain. In addition, the gas-phase species that lead to the dry deposition of acidity are also precursors of particles. Therefore, reductions in NO_X and SO_2 emissions will decrease both acid deposition and PM concentrations, but not necessarily in a linear fashion (U.S. EPA, 2008c). Sulfuric acid is also deposited on surfaces by dry deposition and can contribute to environmental effects (U.S. EPA, 2008c).

6.6.2.1 Ecological Effects of Acidification

Deposition of nitrogen and sulfur can cause acidification, which alters biogeochemistry and affects animal and plant life in terrestrial and aquatic ecosystems across the U.S. Soil acidification is a natural process, but is often accelerated by acidifying deposition, which can decrease concentrations of exchangeable base cations in soils (U.S. EPA, 2008c). Major terrestrial effects include a decline in sensitive tree species, such as red spruce (*Picea rubens*) and sugar maple (*Acer saccharum*) (U.S. EPA, 2008c). Biological effects of acidification in terrestrial ecosystems are generally linked to aluminum toxicity and decreased ability of plant roots to take up base cations (U.S. EPA, 2008c). Decreases in the acid neutralizing capacity and increases in inorganic aluminum concentration contribute to declines in zooplankton, macro invertebrates, and fish species richness in aquatic ecosystems (U.S. EPA, 2008c).

Geology (particularly surficial geology) is the principal factor governing the sensitivity of terrestrial and aquatic ecosystems to acidification from nitrogen and sulfur deposition (U.S. EPA, 2008c). Geologic formations having low base cation supply generally underlie the watersheds of acid-sensitive lakes and streams. Other factors contribute to the sensitivity of soils and surface waters to acidifying deposition, including topography, soil chemistry, land use, and hydrologic flow path (U.S. EPA, 2008c).

Aquatic Acidification. Aquatic effects of acidification have been well studied in the U.S. and elsewhere at various trophic levels. These studies indicate that aquatic biota have been affected by acidification at virtually all levels of the food web in acid sensitive aquatic ecosystems. The ISA for NO_x/SO_x—Ecological Criteria concluded that the evidence is sufficient to infer a causal relationship between acidifying deposition and effects on biogeochemistry related to aquatic ecosystems and biota in aquatic ecosystems (U.S. EPA, 2008c). Effects have been most clearly documented for fish, aquatic insects, other invertebrates, and algae. Biological effects are primarily attributable to a combination of low pH and high inorganic aluminum concentrations. Such conditions occur more frequently during rainfall and snowmelt that cause high flows of water and less commonly during low-flow conditions, except where chronic acidity conditions are severe. Biological effects of episodes include reduced fish condition factor[33], changes in species composition and declines in aquatic species richness across multiple taxa, ecosystems and regions. These conditions may also result in direct fish mortality (Van Sickle et al., 1996). Biological effects in aquatic ecosystems can be divided into two major categories: effects on health, vigor, and reproductive success; and effects on biodiversity. Surface water with ANC values greater than 50 µeq/L generally provides moderate protection for most fish (i.e., brook trout, others) and other aquatic organisms (U.S. EPA, 2009c). Table 6-9 provides a summary of the biological effects experienced at various ANC levels.

[33] Condition factor is an index that describes the relationship between fish weight and length, and is one measure of sublethal acidification stress that has been used to quantify effects of acidification on an individual fish (U.S. EPA, 2008f).

Table 6-9. Aquatic Status Categories

Category Label	ANC Levels	Expected Ecological Effects
Acute Concern	<0 micro equivalent per Liter (μeq/L)	Near complete loss of fish populations is expected. Planktonic communities have extremely low diversity and are dominated by acidophilic forms. The number of individuals in plankton species that are present is greatly reduced.
Severe Concern	0–20 μeq/L	Highly sensitive to episodic acidification. During episodes of high acidifying deposition, brook trout populations may experience lethal effects. Diversity and distribution of zooplankton communities decline sharply.
Elevated Concern	20–50 μeq/L	Fish species richness is greatly reduced (i.e., more than half of expected species can be missing). On average, brook trout populations experience sublethal effects, including loss of health, reproduction capacity, and fitness. Diversity and distribution of zooplankton communities decline.
Moderate Concern	50–100 μeq/L	Fish species richness begins to decline (i.e., sensitive species are lost from lakes). Brook trout populations are sensitive and variable, with possible sublethal effects. Diversity and distribution of zooplankton communities also begin to decline as species that are sensitive to acidifying deposition are affected.
Low Concern	>100 μeq/L	Fish species richness may be unaffected. Reproducing brook trout populations are expected where habitat is suitable. Zooplankton communities are unaffected and exhibit expected diversity and distribution.

A number of national and regional assessments have been conducted to estimate the distribution and extent of surface water acidity in the U.S. (U.S. EPA, 2008c). As a result, several regions of the U.S. have been identified as containing a large number of lakes and streams that are seriously impacted by acidification. Figure 6-11 illustrates those areas of the U.S. where aquatic ecosystems are at risk from acidification.

Because acidification primarily affects the diversity and abundance of aquatic biota, it also affects the ecosystem services that are derived from the fish and other aquatic life found in these surface waters.

While acidification is unlikely to have serious negative effects on, for example, water supplies, it can limit the productivity of surface waters as a source of food (i.e., fish). In the northeastern United States, the surface waters affected by acidification are not a major source of commercially raised or caught fish; however, they are a source of food for some recreational and subsistence fishermen and for other consumers. For example, there is evidence that certain population subgroups in the northeastern United States, such as the Hmong and Chippewa ethnic groups, have particularly high rates of self-caught fish consumption (Hutchison and Kraft, 1994; Peterson et al., 1994). However, it is not known if and how their consumption patterns are affected by the reductions in available fish populations caused by surface water acidification.

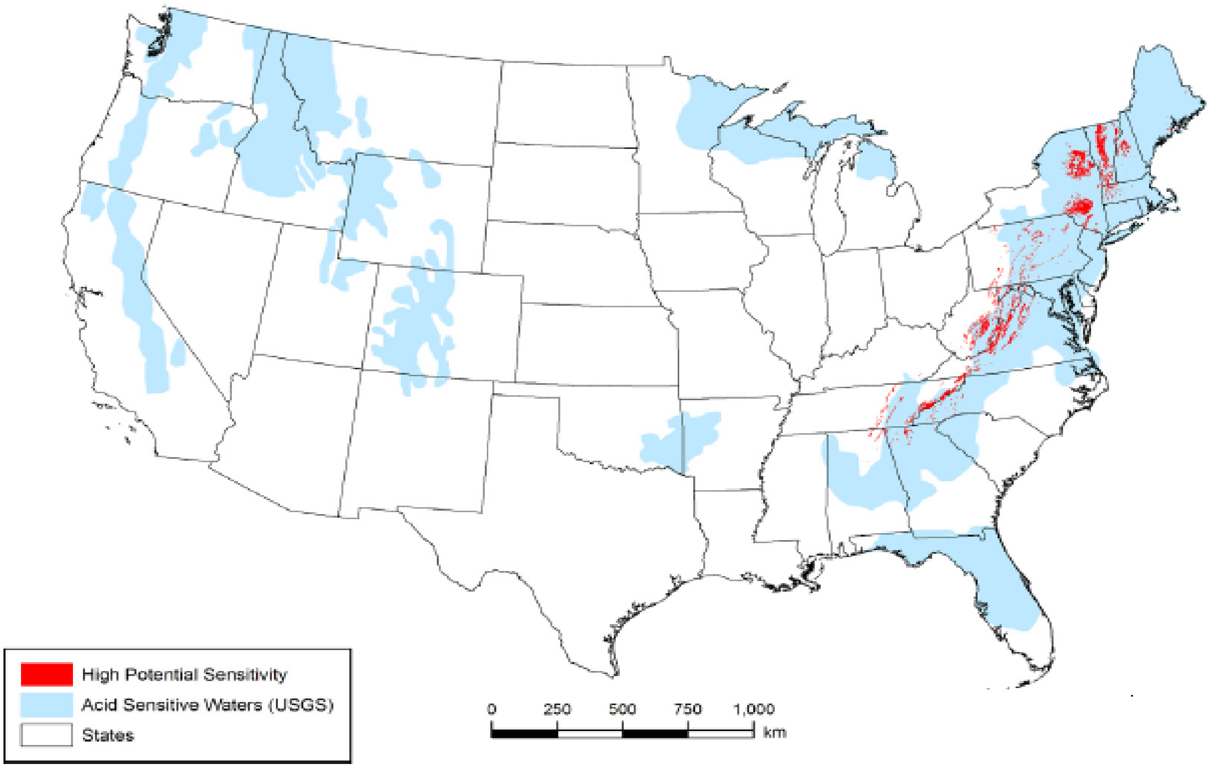


Figure 6-11. Areas Potentially Sensitive to Aquatic Acidification

Source: U.S. EPA, 2008c.

Inland surface waters support several cultural services, including aesthetic and educational services and recreational fishing. Recreational fishing in lakes and streams is among the most popular outdoor recreational activities in the northeastern United States. Based on studies conducted in the northeastern United States, Kaval and Loomis (2003) estimated average consumer surplus values per day of $36 for recreational fishing (in 2007 dollars); therefore, the implied total annual value of freshwater fishing in the northeastern United States was $5.1 billion in 2006.[34] For recreation days, consumer surplus value is most commonly measured using recreation demand, travel cost models.

Another estimate of the overarching ecological co-benefits associated with reducing lake acidification levels in Adirondacks National Park can be derived from the contingent valuation (CV) survey (Banzhaf et al., 2006), which elicited values for specific improvements in

[34] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

acidification-related water quality and ecological conditions in Adirondack lakes. The survey described a base version with minor improvements said to result from the program, and a scope version with large improvements due to the program and a gradually worsening status quo. After adapting and transferring the results of this study and converting the 10-year annual payments to permanent annual payments using discount rates of 3% and 5%, the WTP estimates ranged from $48 to $107 per year per household (in 2004 dollars) for the base version and $54 to $154 for the scope version. Using these estimates, the aggregate annual benefits of eliminating all anthropogenic sources of NO_x and SO_x emissions were estimated to range from $291 million to $829 million (U.S. EPA, 2009c).[35]

In addition, inland surface waters provide a number of regulating services associated with hydrological and climate regulation by providing environments that sustain aquatic food webs. These services are disrupted by the toxic effects of acidification on fish and other aquatic life. Although it is difficult to quantify these services and how they are affected by acidification, some of these services may be captured through measures of provisioning and cultural services.

Terrestrial Acidification. Acidifying deposition has altered major biogeochemical processes in the U.S. by increasing the nitrogen and sulfur content of soils, accelerating nitrate and sulfate leaching from soil to drainage waters, depleting base cations (especially calcium and magnesium) from soils, and increasing the mobility of aluminum. Inorganic aluminum is toxic to some tree roots. Plants affected by high levels of aluminum from the soil often have reduced root growth, which restricts the ability of the plant to take up water and nutrients, especially calcium (U. S. EPA, 2008c). These direct effects can, in turn, influence the response of these plants to climatic stresses such as droughts and cold temperatures. They can also influence the sensitivity of plants to other stresses, including insect pests and disease (Joslin et al., 1992) leading to increased mortality of canopy trees. In the U.S., terrestrial effects of acidification are best described for forested ecosystems (especially red spruce and sugar maple ecosystems) with additional information on other plant communities, including shrubs and lichen (U.S. EPA, 2008c). The ISA for NO_x/SO_x—Ecological Criteria concluded that the evidence is sufficient to infer a causal relationship between acidifying deposition and effects on biogeochemistry related to terrestrial ecosystems and biota in terrestrial ecosystems (U.S. EPA, 2008c).

Certain ecosystems in the continental U.S. are potentially sensitive to terrestrial acidification, which is the greatest concern regarding nitrogen and sulfur deposition U.S. EPA

[35] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

(2008c). Figure 6-12 depicts the areas across the U.S. that are potentially sensitive to terrestrial acidification.

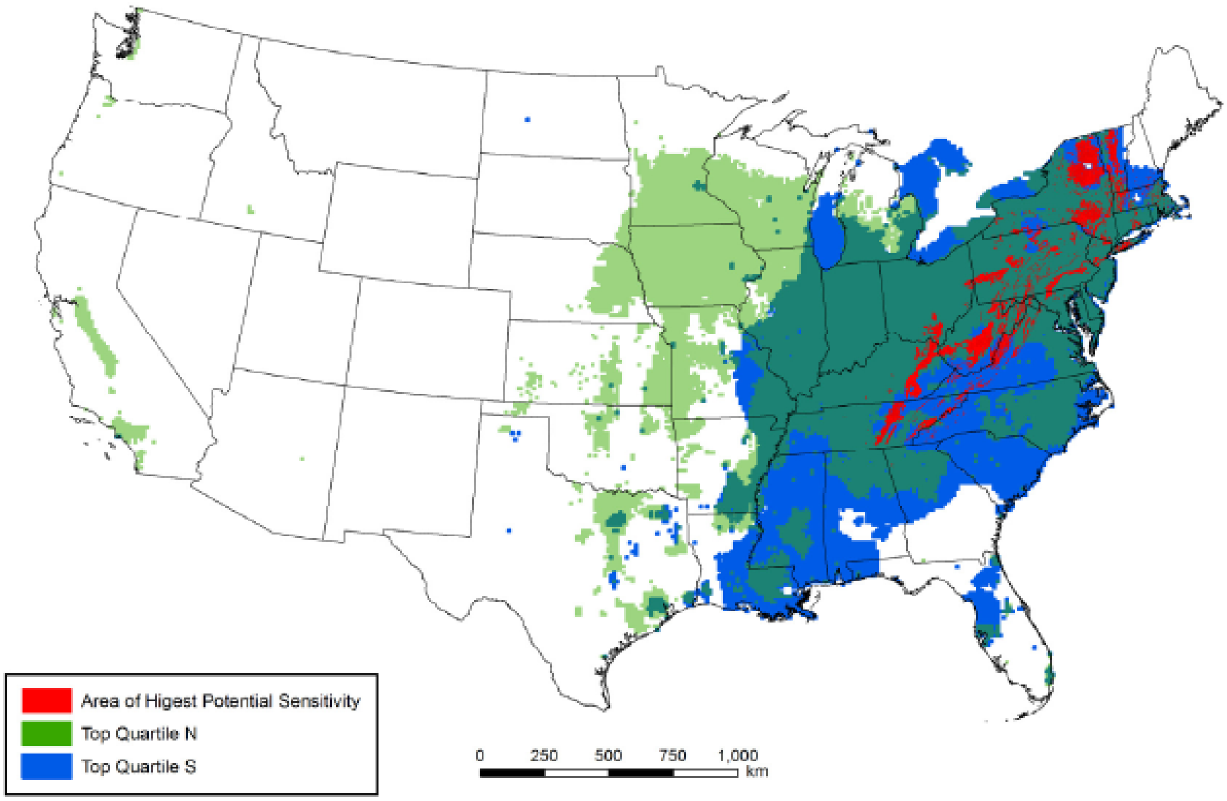


Figure 6-12. Areas Potentially Sensitive to Terrestrial Acidification

Source: U.S. EPA, 2008c.

Both coniferous and deciduous forests throughout the eastern U.S. are experiencing gradual losses of base cation nutrients from the soil due to accelerated leaching from acidifying deposition. This change in nutrient availability may reduce the quality of forest nutrition over the long term. Evidence suggests that red spruce and sugar maple in some areas in the eastern U.S. have experienced declining health because of this deposition. For red spruce, (*Picea rubens*) dieback or decline has been observed across high elevation landscapes of the northeastern U.S., and to a lesser extent, the southeastern U.S., and acidifying deposition has been implicated as a causal factor (DeHayes et al., 1999). Figure 6-13 shows the distribution of red spruce (brown) and sugar maple (green) in the eastern U.S.

Terrestrial acidification affects several important ecological endpoints, including declines in habitat for threatened and endangered species (cultural), declines in forest aesthetics (cultural), declines in forest productivity (provisioning), and increases in forest soil erosion and reductions in water retention (cultural and regulating).

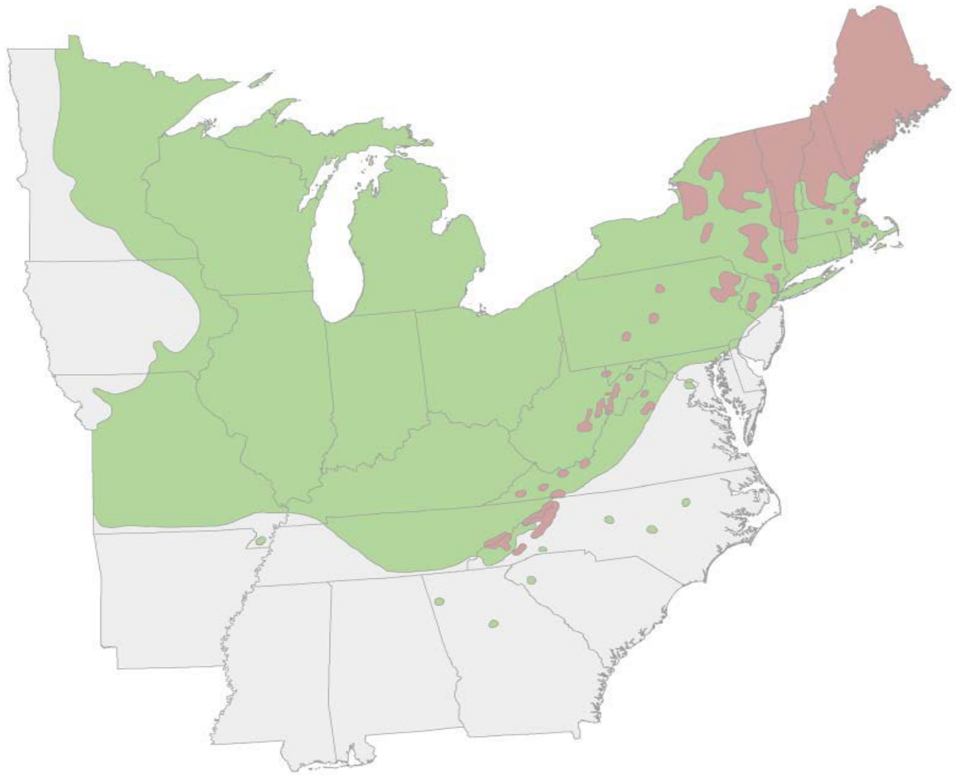


Figure 6-13. Distribution of Red Spruce (pink) and Sugar Maple (green) in the Eastern U.S.

Source: U.S. EPA, 2008c.

Forests in the northeastern United States provide several important and valuable provisioning services in the form of tree products. Sugar maples are a particularly important commercial hardwood tree species, providing timber and maple syrup. In the United States, sugar maple saw timber was nearly 900 million board feet in 2006 (USFS, 2006), and annual production of maple syrup was nearly 1.4 million gallons, accounting for approximately 19% of worldwide production. The total annual value of U.S. production in these years was approximately $160 million (NASS, 2008).[36] Red spruce is also used in a variety of products including lumber, pulpwood, poles, plywood, and musical instruments. The total removal of red spruce saw timber from timberland in the United States was over 300 million board feet in 2006 (USFS, 2006).

[36] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

Forests in the northeastern United States are also an important source of cultural ecosystem services—nonuse (i.e., existence value for threatened and endangered species), recreational, and aesthetic services. Red spruce forests are home to two federally listed species and one delisted species:

1. Spruce-fir moss spider (*Microhexura montivaga*)—endangered

2. Rock gnome lichen (*Gymnoderma lineare*)—endangered

3. Virginia northern flying squirrel (*Glaucomys sabrinus fuscus*)—delisted, but important

Forestlands support a wide variety of outdoor recreational activities, including fishing, hiking, camping, off-road driving, hunting, and wildlife viewing. Regional statistics on recreational activities that are specifically forest based are not available; however, more general data on outdoor recreation provide some insights into the overall level of recreational services provided by forests. More than 30% of the U.S. adult population visited a wilderness or primitive area during the previous year and engaged in day hiking (Cordell et al., 2008). From 1999 to 2004, 16% of adults in the northeastern United States participated in off-road vehicle recreation, for an average of 27 days per year (Cordell et al., 2005). The average consumer surplus value per day of off-road driving in the United States was $25 (in 2007 dollars), and the implied total annual value of off-road driving recreation in the northeastern United States was more than $9 billion (Kaval and Loomis, 2003). More than 5% of adults in the northeastern United States participated in nearly 84 million hunting days (U.S. FWS and U.S. Census Bureau, 2007). Ten percent of adults in northeastern states participated in wildlife viewing away from home on 122 million days in 2006. For these recreational activities in the northeastern United States, Kaval and Loomis (2003) estimated average consumer surplus values per day of $52 for hunting and $34 for wildlife viewing (in 2007 dollars). The implied total annual value of hunting and wildlife viewing in the northeastern United States was, therefore, $4.4 billion and $4.2 billion, respectively, in 2006 (U.S. EPA, 2009c).[37]

As previously mentioned, it is difficult to estimate the portion of these recreational services that are specifically attributable to forests and to the health of specific tree species. However, one recreational activity that is directly dependent on forest conditions is fall color viewing. Sugar maple trees, in particular, are known for their bright colors and are, therefore, an essential aesthetic component of most fall color landscapes. A survey of residents in the

[37] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

Great Lakes area found that roughly 30% of residents reported at least one trip in the previous year involving fall color viewing (Spencer and Holecek, 2007). In a separate study conducted in Vermont, Brown (2002) reported that more than 22% of households visiting Vermont in 2001 made the trip primarily for viewing fall colors.

Two studies estimated values for protecting high-elevation spruce forests in the southern Appalachian Mountains. Kramer et al. (2003) conducted a contingent valuation study estimating households' WTP for programs to protect remaining high-elevation spruce forests from damages associated with air pollution and insect infestation. Median household WTP was estimated to be roughly $29 (in 2007 dollars) for a smaller program, and $44 for the more extensive program. Jenkins et al. (2002) conducted a very similar study in seven Southern Appalachian states on a potential program to maintain forest conditions at status quo levels. The overall mean annual WTP for the forest protection programs was $208 (in 2007 dollars). Multiplying the average WTP estimate from these studies by the total number of households in the seven-state Appalachian region results in an aggregate annual range of $470 million to $3.4 billion for avoiding a significant decline in the health of high-elevation spruce forests in the Southern Appalachian region (U.S. EPA, 2009c).[38]

Forests in the northeastern United States also support and provide a wide variety of valuable regulating services, including soil stabilization and erosion control, water regulation, and climate regulation. The total value of these ecosystem services is very difficult to quantify in a meaningful way, as is the reduction in the value of these services associated with total nitrogen and sulfur deposition. As terrestrial acidification contributes to root damages, reduced biomass growth, and tree mortality, all of these services are likely to be affected; however, the magnitude of these impacts is currently very uncertain.

6.6.2.2 Ecological Effects from Nitrogen Enrichment

Aquatic Enrichment. The ISA for NO_x/SO_x—Ecological Criteria concluded that the evidence is sufficient to infer a causal relationship between nitrogen deposition and the alteration of species richness, species composition, and biodiversity in wetland, freshwater aquatic and coastal marine ecosystems (U.S. EPA, 2008c).

One of the main adverse ecological effects resulting from nitrogen deposition, particularly in the Mid-Atlantic region of the United States, is the effect associated with nutrient enrichment in estuarine waters. A recent assessment of 141 estuaries nationwide by the

[38] These estimates reflect the marginal value of the service for the hypothetical program described in the survey, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

National Oceanic and Atmospheric Administration (NOAA) concluded that 19 estuaries (13%) suffered from moderately high or high levels of eutrophication due to excessive inputs of both N and phosphorus, and a majority of these estuaries are located in the coastal area from North Carolina to Massachusetts (NOAA, 2007). For estuaries in the Mid-Atlantic region, the contribution of atmospheric distribution to total N loads is estimated to range between 10% and 58% (Valigura et al., 2001).

Eutrophication in estuaries is associated with a range of adverse ecological effects. The conceptual framework developed by NOAA emphasizes four main types of eutrophication effects—low dissolved oxygen (DO), harmful algal blooms (HABs), loss of submerged aquatic vegetation (SAV), and low water clarity. Low DO disrupts aquatic habitats, causing stress to fish and shellfish, which, in the short-term, can lead to episodic fish kills and, in the long-term, can damage overall growth in fish and shellfish populations. Low DO also degrades the aesthetic qualities of surface water. In addition to often being toxic to fish and shellfish, and leading to fish kills and aesthetic impairments of estuaries, HABs can, in some instances, also be harmful to human health. SAV provides critical habitat for many aquatic species in estuaries and, in some instances, can also protect shorelines by reducing wave strength; therefore, declines in SAV due to nutrient enrichment are an important source of concern. Low water clarity is in part the result of accumulations of both algae and sediments in estuarine waters. In addition to contributing to declines in SAV, high levels of turbidity also degrade the aesthetic qualities of the estuarine environment.

Estuaries in the eastern United States are an important source of food production, in particular fish and shellfish production. The estuaries are capable of supporting large stocks of resident commercial species, and they serve as the breeding grounds and interim habitat for several migratory species. To provide an indication of the magnitude of provisioning services associated with coastal fisheries, from 2005 to 2007, the average value of total catch was $1.5 billion per year. It is not known, however, what percentage of this value is directly attributable to or dependent upon the estuaries in these states.

In addition to affecting provisioning services through commercial fish harvests, eutrophication in estuaries may also affect the demand for seafood. For example, a well-publicized toxic pfiesteria bloom in the Maryland Eastern Shore in 1997, which involved thousands of dead and lesioned fish, led to an estimated $56 million (in 2007 dollars) in lost seafood sales for 360 seafood firms in Maryland in the months following the outbreak (Lipton, 1999).

Estuaries in the United States also provide an important and substantial variety of cultural ecosystem services, including water-based recreational and aesthetic services. The water quality in the estuary directly affects the quality of these experiences. For example, there were 26 million days of saltwater fishing coastal states from North Carolina to Massachusetts in 2006 (FWA and Census, 2007). Assuming an average consumer surplus value for a fishing day at $36 (in 2007 dollars) in the Northeast and $87 in the Southeast (Kaval and Loomis, 2003), the aggregate value was approximately $1.3 billion (in 2007 dollars) (U.S. EPA, 2009c). [39] In addition, almost 6 million adults participated in motorboating in coastal states from North Carolina to Massachusetts, for a total of nearly 63 million days annually during 1999–2000 (Leeworthy and Wiley, 2001). Using a national daily value estimate of $32 (in 2007 dollars) for motorboating (Kaval and Loomis (2003), the aggregate value of these coastal motorboating outings was $2 billion per year (U.S. EPA, 2009c). [40] Almost 7 million participated in birdwatching for 175 million days per year, and more than 3 million participated in visits to non-beach coastal waterside areas.

Estuaries and marshes have the potential to support a wide range of regulating services, including climate, biological, and water regulation; pollution detoxification; erosion prevention; and protection against natural hazards from declines in SAV (MEA, 2005). SAV can help reduce wave energy levels and thus protect shorelines against excessive erosion, which increases the risks of episodic flooding and associated damages to near-shore properties or public infrastructure or even contribute to shoreline retreat.

We are unable to provide an estimate of the aquatic enrichment co-benefits associated with the revised or alternative annual standards due to data, time, and resource limitations.

Terrestrial Enrichment. Terrestrial enrichment occurs when terrestrial ecosystems receive N loadings in excess of natural background levels, through either atmospheric deposition or direct application. Evidence presented in the ISA for NO_x/SO_x (U.S. EPA, 2008c) supports a causal relationship between atmospheric N deposition and biogeochemical cycling and fluxes of N and carbon in terrestrial systems. Furthermore, evidence summarized in the report supports a causal link between atmospheric N deposition and changes in the types and number of species and biodiversity in terrestrial systems. Nitrogen enrichment occurs over a long time period; as a result, it may take as much as 50 years or more to see changes in

[39] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

[40] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

ecosystem conditions and indicators. This long time scale also affects the timing of the ecosystem service changes. The ISA for NO_x/SO_x—Ecological Criteria concluded that the evidence is sufficient to infer a causal relationship between nitrogen deposition and the alteration of species richness, species composition, and biodiversity in terrestrial ecosystems (U.S. EPA, 2008c).

One of the main provisioning services potentially affected by N deposition is grazing opportunities offered by grasslands for livestock production in the Central U.S. Although N deposition on these grasslands can offer supplementary nutritive value and promote overall grass production, there are concerns that fertilization may favor invasive grasses and shift the species composition away from native grasses. This process may ultimately reduce the productivity of grasslands for livestock production. Losses due to invasive grasses can be significant; for example, based on a bioeconomic model of cattle grazing in the upper Great Plains, Leitch, Leistritz, and Bangsund (1996) and Leistritz, Bangsund, and Hodur (2004) estimated $130 million in losses due to a leafy spurge infestation in the Dakotas, Montana, and Wyoming.[41] However, the contribution of N deposition to these losses is still uncertain.

Terrestrial nutrient enrichment also affects cultural and regulating services. For example, in California, Coastal Sage Scrub (CSS) habitat concerns focus on a decline in CSS and an increase in nonnative grasses and other species, impacts on the viability of threatened and endangered species associated with CSS, and an increase in fire frequency. Changes in Mixed Conifer Forest (MCF) include changes in habitat suitability and increased tree mortality, increased fire intensity, and a change in the forest's nutrient cycling that may affect surface water quality through nitrate leaching (U.S. EPA, 2008c). CSS and MCF are an integral part of the California landscape, and together the ranges of these habitats include the densely populated and valuable coastline and the mountain areas. Numerous threatened and endangered species at both the state and federal levels reside in CSS and MCF. The value that California residents and the U.S. population as a whole place on CSS and MCF habitats is reflected in the various federal, state, and local government measures that have been put in place to protect these habitats, including the Endangered Species Act, conservation planning programs, and private and local land trusts. CSS and MCF habitat are showcased in many popular recreation areas in California, including several national parks and monuments. In addition, millions of individuals are involved in fishing, hunting, and wildlife viewing in California every year (DOI, 2007). The quality of these trips depends in part on the health of the

[41] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

ecosystems and their ability to support the diversity of plants and animals found in important habitats found in CSS or MCF ecosystems and the parks associated with those ecosystems. Based on analyses in the NO_x SO_x REA average values of the total benefits in 2006 from fishing, hunting, and wildlife viewing away from home in California were approximately $950 million, $170 million, and $3.6 billion, respectively (U.S. EPA, 2009c).[42] In addition, data from California State Parks (2003) indicate that in 2002, 69% of adult residents participated in trail hiking for an average of 24 days per year. The aggregate annual benefit for California residents from trail hiking in 2007 was $11 billion (U.S. EPA, 2009c).[43] It is not currently possible to quantify the loss in value of services due to nitrogen deposition as those losses are already reflected in the estimates of the contemporaneous total value of these recreational activities. Restoration of services through decreases in nitrogen deposition would likely increase the total value of recreational services.

Fire regulation is also an important regulating service that could be affected by nutrient enrichment of the CSS and MCF ecosystems by encouraging growth of more flammable grasses, increasing fuel loads, and altering the fire cycle. Over the 5-year period from 2004 to 2008, Southern California experienced, on average, over 4,000 fires per year burning, on average, over 400,000 acres per year (National Association of State Foresters [NASF], 2009). It is not possible at this time to quantify the contribution of nitrogen deposition, among many other factors, to increased fire risk.

We are unable to provide an estimate of the terrestrial nutrient enrichment co-benefits associated with the revised or alternative annual standards due to data, time, and resource limitations. Methods are not yet available to allow estimation of changes in ecosystem services due to nitrogen deposition.

6.7.2.3 Vegetation Effects Associated with Gaseous Sulfur Dioxide

Uptake of gaseous sulfur dioxide in a plant canopy is a complex process involving adsorption to surfaces (leaves, stems, and soil) and absorption into leaves. SO_2 penetrates into leaves through to the stomata, although there is evidence for limited pathways via the cuticle (U.S. EPA, 2008c). Pollutants must be transported from the bulk air to the leaf boundary layer in order to get to the stomata. When the stomata are closed, as occurs under dark or drought conditions, resistance to gas uptake is very high and the plant has a very low degree of

[42] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.
[43] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

susceptibility to injury. In contrast, mosses and lichens do not have a protective cuticle barrier to gaseous pollutants or stomates and are generally more sensitive to gaseous sulfur and nitrogen than vascular plants (U.S. EPA, 2008c). Acute foliar injury usually happens within hours of exposure, involves a rapid absorption of a toxic dose, and involves collapse or necrosis of plant tissues. Another type of visible injury is termed chronic injury and is usually a result of variable SO_2 exposures over the growing season. Besides foliar injury, chronic exposure to low SO_2 concentrations can result in reduced photosynthesis, growth, and yield of plants (U.S. EPA, 2008c). These effects are cumulative over the season and are often not associated with visible foliar injury. As with foliar injury, these effects vary among species and growing environment. SO_2 is also considered the primary factor causing the death of lichens in many urban and industrial areas (Hutchinson et al., 1996). The ISA for NO_x/SO_x—Ecological Criteria concluded that the evidence is sufficient to infer a causal relationship between SO_2 injury to vegetation (U.S. EPA, 2008c).

6.6.2.4 Mercury-Related Co-benefits Associated with the Role of Sulfate in Mercury Methylation

Mercury is a persistent, bioaccumulative toxic metal that is emitted from in three forms: gaseous elemental Hg (Hg^0), oxidized Hg compounds (Hg^{+2}), and particle-bound Hg (Hg_P). Methylmercury (MeHg) is formed by microbial action in the top layers of sediment and soils, after Hg has precipitated from the air and deposited into waterbodies or land. Once formed, MeHg is taken up by aquatic organisms and bioaccumulates up the aquatic food web. Larger predatory fish may have MeHg concentrations many times, typically on the order of one million times, that of the concentrations in the freshwater body in which they live.

The NO_x SO_x ISA—Ecological Criteria concluded that evidence is sufficient to infer a causal relationship between sulfur deposition and increased mercury methylation in wetlands and aquatic environments (U.S. EPA, 2008c). Specifically, there appears to be a relationship between SO_4^{2-} deposition and mercury methylation; however, the rate of mercury methylation varies according to several spatial and biogeochemical factors whose influence has not been fully quantified (see Figure 6-14). Therefore, the correlation between SO_4^{2-} deposition and MeHg could not be quantified for the purpose of interpolating the association across waterbodies or regions. Nevertheless, because changes in MeHg in ecosystems represent changes in significant human and ecological health risks, the association between sulfur and mercury cannot be neglected (U.S. EPA, 2008c).

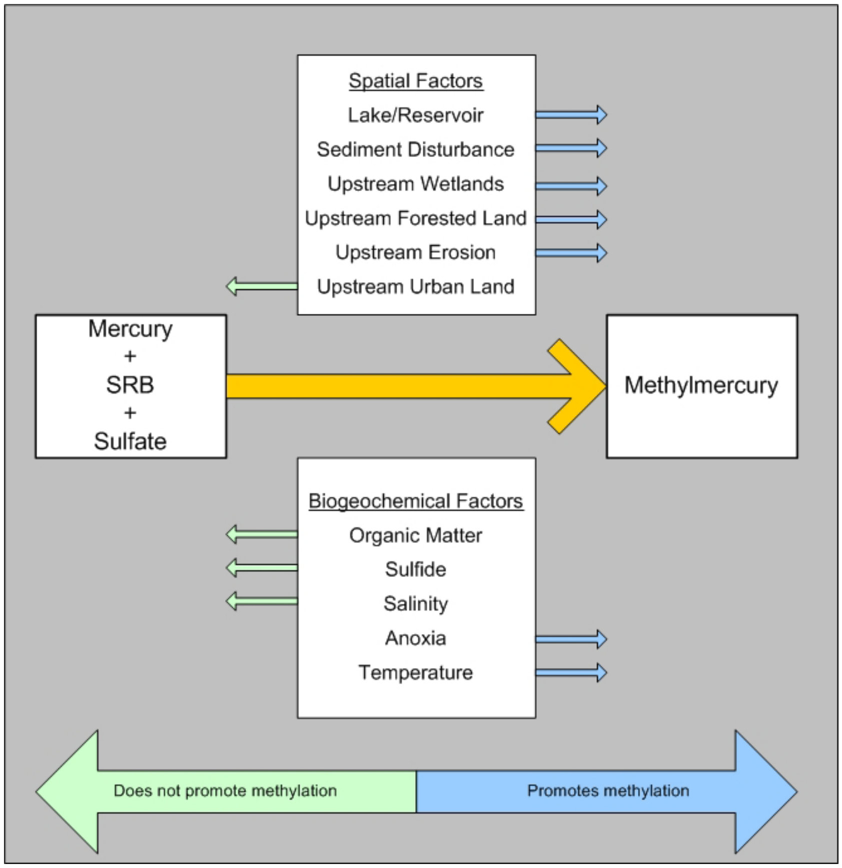


Figure 6-14. Spatial and Biogeochemical Factors Influencing MeHg Production

As research evolves and the computational capacity of models expands to meet the complexity of mercury methylation processes in ecosystems, the role of interacting factors may be better parsed out to identify ecosystems or regions that are more likely to generate higher concentrations of MeHg. Figure 6-15 illustrates the type of current and forward-looking research being developed by the U.S. Geological Survey (USGS) to synthesize the contributing factors of mercury and to develop a map of sensitive watersheds. The mercury score referenced in Figure 6-15 is based on SO_4^{2-} concentrations, acid neutralizing capacity (ANC), levels of dissolved organic carbon and pH, mercury species concentrations, and soil types to gauge the methylation sensitivity (Myers et al., 2007).

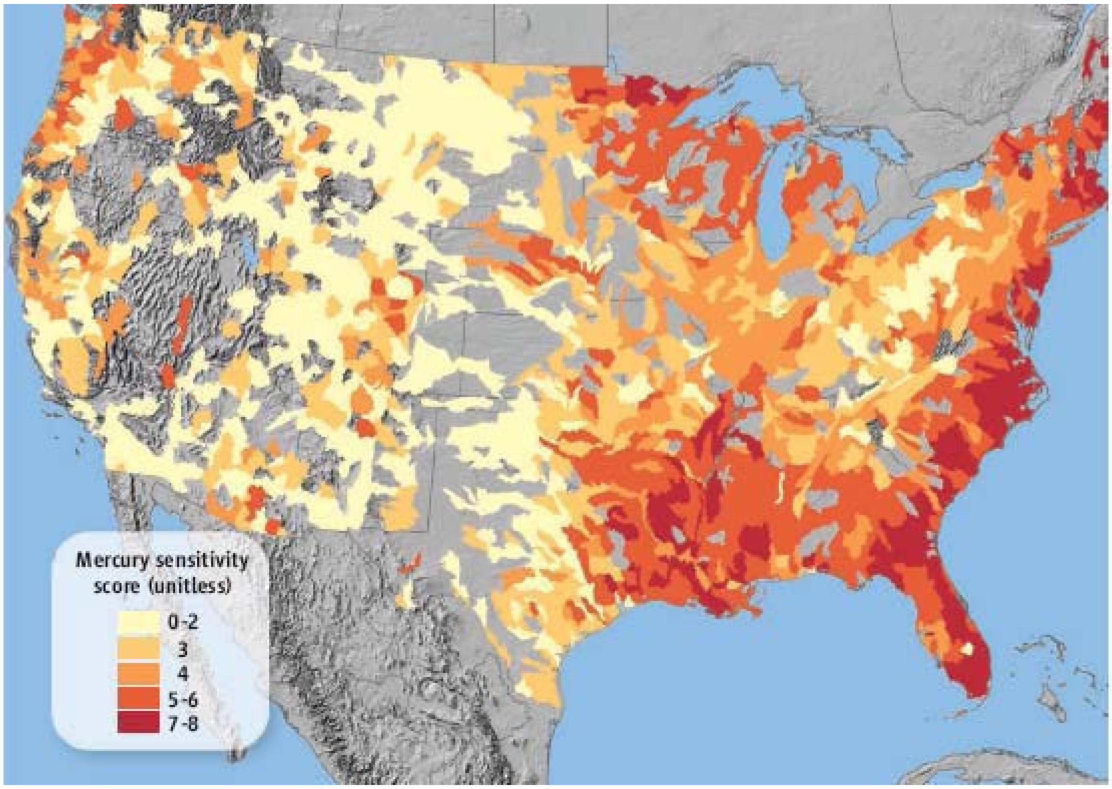


Figure 6-15. Preliminary USGS Map of Mercury Methylation–Sensitive Watersheds

Source: Myers et al., 2007.

Interdependent biogeochemical factors preclude the existence of simple sulfate-related mercury methylation models. It is clear that decreasing sulfate deposition is likely to result in decreased MeHg concentrations. Future research may allow for the characterization of a usable sulfate-MeHg response curve; however, no regional or classification calculation scale can be created at this time because of the number of confounding factors.

Decreases in SO_4^{2-} deposition have already shown promising reductions in MeHg. Observed decreases in MeHg fish tissue concentrations have been linked to decreased acidification and declining SO_4^{2-} and mercury deposition in Little Rock Lake, WI (Hrabik and Watras, 2002), and to decreased SO_4^{2-} deposition in Isle Royale in Lake Superior, MI (Drevnick et al., 2007). Although the possibility exists that reductions in SO_4^{2-} emissions could generate a pulse in MeHg production because of decreased sulfide inhibition in sulfate-saturated waters, this effect would likely involve a limited number of U.S. waters (Harmon et al., 2007). Also, because of the diffusion and outward flow of both mercury-sulfide complexes and SO_4^{2-}, increased mercury methylation downstream may still occur in sulfate-enriched ecosystems with increased organic matter and/or downstream transport capabilities.

Remediation of sediments heavily contaminated with mercury has yielded significant reductions of MeHg in biotic tissues. Establishing quantitative relations in biotic responses to MeHg levels as a result of changes in atmospheric mercury deposition, however, presents difficulties because direct associations can be confounded by all of the factors discussed in this section. Current research does suggest that the levels of MeHg and total mercury in ecosystems are positively correlated, so that reductions in mercury deposited into ecosystems would also eventually lead to reductions in MeHg in biotic tissues. Ultimately, an integrated approach that involves the reduction of both sulfur and mercury emissions may be most efficient because of the variability in ecosystem responses. Reducing SO_X emissions could have a beneficial effect on levels of MeHg in many waters of the United States.

MeHg is the only form of mercury that biomagnifies in the food web. Concentrations of MeHg in fish are generally on the order of a million times the MeHg concentration in water. In addition to mercury deposition, key factors affecting MeHg production and accumulation in fish include the amount and forms of sulfur and carbon species present in a given waterbody. Thus, two adjoining water bodies receiving the same deposition can have significantly different fish mercury concentrations.

Methylmercury builds up more in some types of fish and shellfish than in others. The levels of methylmercury in high and shellfish vary widely depending on what they eat, how long they live, and how high they are in the food chain. Most fish, including ocean species and local freshwater fish, contain some methylmercury. In general, higher mercury concentrations are expected in top predators, which are often large fish relative to other species in a waterbody.

The ecosystem service most directly affected by sulfate-mediated mercury methylation is the provision of fish for consumption as a food source. This service is of particular importance to groups engaged in subsistence fishing, pregnant women and young children.

6.6.3 Ecosystem Co-benefits from Reductions in Mercury Emissions

Deposition of mercury to waterbodies can also have an impact on ecosystems and wildlife. Mercury contamination is present in all environmental media with aquatic systems experiencing the greatest exposures due to bioaccumulation. Bioaccumulation refers to the net uptake of a contaminant from all possible pathways and includes the accumulation that may occur by direct exposure to contaminated media as well as uptake from food.

Atmospheric mercury enters freshwater ecosystems by direct deposition and through runoff from terrestrial watersheds. Once mercury deposits, it may be converted to organic methylmercury mediated primarily by sulfate-reducing bacteria. Methylation is enhanced in anaerobic and acidic environments, greatly increasing mercury toxicity and potential to bioaccumulate in aquatic foodwebs. A number of key biogeochemical controls influence the production of methylmercury in aquatic ecosystems. These include sulfur, pH, organic matter, iron, mercury "aging," and bacteria type and activity (Munthe et al., 2007).

Wet and dry deposition of oxidized mercury is a dominant pathway for bringing mercury to terrestrial surfaces. In forest ecosystems, elemental mercury may also be absorbed by plants stomatally, incorporated by foliar tissues and released in litterfall (Ericksen et al., 2003). Mercury in throughfall, direct deposition in precipitation, and uptake of dissolved mercury by roots (Rea et al., 2002) are also important in mercury accumulation in terrestrial ecosystems.

Soils have significant capacity to store large quantities of atmospherically deposited mercury where it can leach into groundwater and surface waters. The risk of mercury exposure extends to insectivorous terrestrial species such as songbirds, bats, spiders, and amphibians that receive mercury deposition or from aquatic systems near the forest areas they inhabit (Bergeron et al., 2010a, b; Cristol et al., 2008; Rimmer et al., 2005; Wada et al., 2009 & 2010).

Numerous studies have generated field data on the levels of mercury in a variety of wild species. Many of the data from these environmental studies are anecdotal in nature rather than representative or statistically designed studies. The body of work examining the effects of these exposures is growing but still incomplete given the complexities of the natural world. A large portion of the adverse effect research conducted to date has been carried out in the laboratory setting rather than in the wild; thus, conclusions about overarching ecosystem health and population effects are difficult to make at this time. In the sections that follow numerous effects have been identified at differing exposure levels.

6.6.3.1 Mercury Effects on Fish

A review of the literature on effects of mercury on fish (Crump and Trudeau, 2009) reports results for numerous species including trout, bass (large and smallmouth), northern pike, carp, walleye, salmon and others from laboratory and field studies. The effects studied are reproductive and include deficits in sperm and egg formation, histopathological changes in testes and ovaries, and disruption of reproductive hormone synthesis. These studies were conducted in areas from New York to Washington and while many were conducted by adding MeHg to water or diet many were conducted at current environmental levels. While we cannot determine at this time whether these reproductive deficits are affecting fish populations across the United States it should be noted that it is possible that over time reproductive deficits could have an effect on populations. Lower fish populations would conceivably impact the ecosystem services like recreational fishing derived from having healthy aquatic ecosystems quite apart from the effects of consumption advisories due to the human health effects of mercury.

6.6.3.2 Mercury Effects on Birds

In addition to effects on fish, mercury also affects avian species. In previous reports (U.S. EPA, 1997; 2005), much of the focus has been on large piscivorous species in particular the common loon. The loon is most visible to the public during the summer breeding season on northern lakes and they have become an important symbol of wilderness in these areas (McIntyre and Barr, 1997). A multitude of loon watch, preservation, and protection groups have formed over the past few decades and have been instrumental in promoting conservation, education, monitoring, and research of breeding loons (McIntyre and Evers, 2000; Evers, 2006). Significant adverse effects on breeding loons from mercury have been found to occur including behavioral (reduced nest-sitting), physiological (flight feather asymmetry) and reproductive (chicks fledged/territorial pair) effects (Evers, 2008). Additionally Evers, et al (2008) report that they believe that the weight of evidence indicates that population-level effects occur in parts of Maine and New Hampshire, and potentially in broad areas of the loon's range.

Recently attention has turned to other piscivorous species such as the white ibis, and great snowy egret. While considered to be fish-eating generally these wading birds have a very wide diet including crayfish, crabs, snails, insects and frogs. These species are experiencing a range of adverse effects due to exposure to mercury. The white ibis has been observed to have decreased foraging efficiency (Adams and Frederick, 2008). Additionally ibises have been shown to exhibit decreased reproductive success and altered pair behavior (Frederick and Jayasena, 2010). These effects include significantly more unproductive nests, male/male pairing, reduced courtship behavior (head bobbing and pair bowing) and lower nestling production by exposed males. In this study, a worst-case scenario suggested by the results could involve up to a 50% reduction in fledglings due to MeHg in diet. These estimates may be conservative if male/male pairing in the wild it could result in a shortage of partners for females and the effect of homosexual breeding would be magnified. In egrets, mercury has been implicated in the decline of the species in south Florida (Sepulveda, et al., 1999) and Hoffman (2010) has shown that egrets show liver and possibly kidney effects. While ibises and egrets are most abundant in coastal areas and these studies were conducted in south Florida and Nevada the ranges of ibises and egrets extend to a large portion of the United States. Ibis territory can range inland to Oklahoma, Arkansas and Tennessee. Egret range covers virtually the entire United States except the mountain west.

Insectivorous birds have also been shown to suffer adverse effects due to mercury exposure. These songbirds such as Bicknell's thrush, tree swallows and the great tit have shown reduced reproduction, survival, and changes in singing behavior. Exposed tree swallows produced fewer fledglings (Brasso, 2008), lower survival (Hallinger, 2010) and had compromised immune competence (Hawley, 2009). The great tit has exhibited reduced singing behavior and smaller song repertoire in an area of high contamination in the vicinity of a metallurgic smelter in Flanders (Gorissen, 2005).

6.6.3.3 Mercury Effects on Mammals

In mammals, adverse effects have been observed in mink and river otter, both fish eating species. For otter from Maine and Vermont maximum concentrations on Hg in fur nearly equal or exceed a concentration associated with mortality and concentration in liver for mink in Massachusetts/Connecticut and the levels in fur from mink in Maine exceed concentrations associated with acute mortality (Yates, 2005). Adverse sublethal effects may be associated with lower Hg concentrations and consequently be more widespread than potential acute effects. These effects may include increased activity, poorer maze performance, abnormal startle reflex, and impaired escape and avoidance behavior (Scheuhammer et al., 2007).

6.6.3.4 Mercury Ecological Conclusions

The studies cited here provide a glimpse of the scope of mercury effects on wildlife particularly reproductive and survival effects. These effects range across species from fish to mammals and spatially across a wide area of the United States. The literature is far from complete however. Much more research is required to establish a link between the ecological effects on wildlife and the effect on ecosystem services (services that the environment provides to people) for example recreational fishing, bird watching and wildlife viewing. The EPA is not, however, currently able to quantify or monetize the co-benefits of reducing mercury exposures affecting provision of ecosystem services.

6.6.4 Vegetation Co-benefits from Reductions in Ambient Ozone

Illustrative emission reduction strategies that include NO_x emission reductions would affect ambient ozone concentrations. Ozone causes discernible injury to a wide array of vegetation (U.S. EPA, 2006a; Fox and Mickler, 1996). Air pollution can affect the environment and affect ecological systems, leading to changes in the ecological community and influencing the diversity, health, and vigor of individual species (U.S. EPA, 2006a). In terms of forest productivity and ecosystem diversity, ozone may be the pollutant with the greatest potential for regional-scale forest impacts (U.S. EPA, 2006a). Studies have demonstrated repeatedly that ozone concentrations commonly observed in polluted areas can have substantial impacts on plant function (De Steiguer et al., 1990; Pye, 1988).

When ozone is present in the air, it can enter the leaves of plants, where it can cause significant cellular damage. Like carbon dioxide (CO_2) and other gaseous substances, ozone enters plant tissues primarily through the stomata in leaves in a process called "uptake" (Winner and Atkinson, 1986). Once sufficient levels of ozone (a highly reactive substance), or its reaction products, reaches the interior of plant cells, it can inhibit or damage essential cellular components and functions, including enzyme activities, lipids, and cellular membranes, disrupting the plant's osmotic (i.e., water) balance and energy utilization patterns (U.S. EPA, 2006a; Tingey and Taylor, 1982). With fewer resources available, the plant reallocates existing resources away from root growth and storage, above ground growth or yield, and reproductive processes, toward leaf repair and maintenance, leading to reduced growth and/or reproduction. Studies have shown that plants stressed in these ways may exhibit a general loss of vigor, which can lead to secondary impacts that modify plants' responses to other environmental factors. Specifically, plants may become more sensitive to other air pollutants, or more susceptible to disease, pest infestation, harsh weather (e.g., drought, frost) and other environmental stresses, which can all produce a loss in plant vigor in ozone-sensitive species that over time may lead to premature plant death. Furthermore, there is evidence that ozone can interfere with the formation of mycorrhiza, essential symbiotic fungi associated with the roots of most terrestrial plants, by reducing the amount of carbon available for transfer from the host to the symbiont (U.S. EPA, 2006a).

This ozone damage may or may not be accompanied by visible injury on leaves, and likewise, visible foliar injury may or may not be a symptom of the other types of plant damage described above. Foliar injury is usually the first visible sign of injury to plants from ozone exposure and indicates impaired physiological processes in the leaves (Grulke, 2003). When visible injury is present, it is commonly manifested as chlorotic or necrotic spots, and/or increased leaf senescence (accelerated leaf aging). Visible foliar injury reduces the aesthetic value of ornamental vegetation and trees in urban landscapes and negatively affects scenic vistas in protected natural areas.

Ozone can produce both acute and chronic injury in sensitive species depending on the concentration level and the duration of the exposure. Ozone effects also tend to accumulate over the growing season of the plant, so that even lower concentrations experienced for a longer duration have the potential to create chronic stress on sensitive vegetation. Not all plants, however, are equally sensitive to ozone. Much of the variation in sensitivity between individual plants or whole species is related to the plant's ability to regulate the extent of gas exchange via leaf stomata (e.g., avoidance of ozone uptake through closure of stomata) and the relative ability of species to detoxify ozone-generated reactive oxygen free radicals (U.S. EPA, 2006a; Winner, 1994). After injuries have occurred, plants may be capable of repairing the damage to a limited extent (U.S. EPA, 2006a). Because of the differing sensitivities among plants to ozone, ozone pollution can also exert a selective pressure that leads to changes in plant community composition. Given the range of plant sensitivities and the fact that numerous other environmental factors modify plant uptake and response to ozone, it is not possible to identify threshold values above which ozone is consistently toxic for all plants.

Because plants are at the base of the food web in many ecosystems, changes to the plant community can affect associated organisms and ecosystems (including the suitability of habitats that support threatened or endangered species and below ground organisms living in the root zone). Ozone impacts at the community and ecosystem level vary widely depending upon numerous factors, including concentration and temporal variation of tropospheric ozone, species composition, soil properties and climatic factors (U.S. EPA, 2006a). In most instances, responses to chronic or recurrent exposure in forested ecosystems are subtle and not observable for many years. These injuries can cause stand-level forest decline in sensitive ecosystems (U.S. EPA, 2006a, McBride et al., 1985; Miller et al., 1982). It is not yet possible to predict ecosystem responses to ozone with certainty; however, considerable knowledge of potential ecosystem responses is available through long-term observations in highly damaged forests in the U.S. (U.S. EPA, 2006a).

6.6.4.1 Ozone Effects on Forests

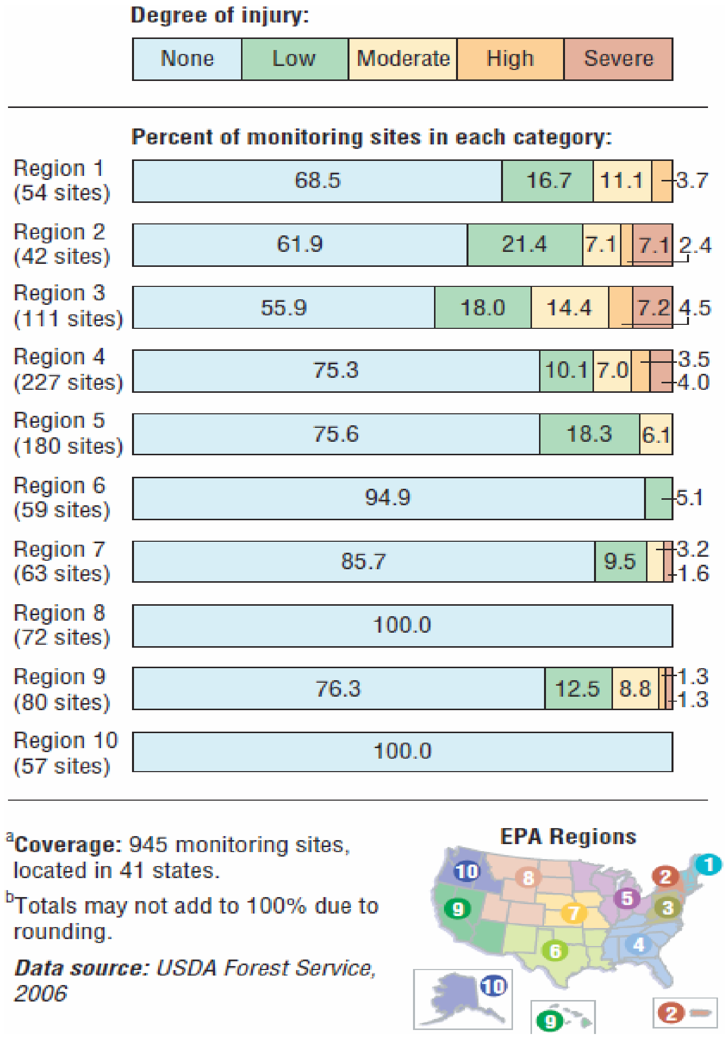
Air pollution can affect the environment and affect ecological systems, leading to changes in the ecological community and influencing the diversity, health, and vigor of individual species (U.S. EPA, 2006a). Ozone has been shown in numerous studies to have a strong effect on the health of many plants, including a variety of commercial and ecologically important forest tree species throughout the United States (U.S. EPA, 2007b).

In the U.S., this data comes from the U.S. Department of Agriculture (USDA) Forest Service Forest Inventory and Analysis (FIA) program. As part of its Phase 3 program (formerly known as Forest Health Monitoring), FIA looks for visible foliar injury of ozone-sensitive forest plant species at each ground monitoring site across the country (excluding woodlots and urban trees) that meets certain minimum criteria. Because ozone injury is cumulative over the course of the growing season, examinations are conducted in July and August, when ozone concentrations and associated injury are typically highest.

Monitoring of ozone injury to plants by the U.S. Forest Service has expanded over the last 15 years from monitoring sites in 10 states in 1994 to nearly 1,000 monitoring sites in 41 states in 2002. Since 2002, the monitoring program has further expanded to 1,130 monitoring sites in 45 states. Figure 6-16 shows the results of this monitoring program for the year 2002 broken down by U.S. EPA Regions.[44] Figure 6-17 identifies the counties that were included in Figure 6-16, and provides the county-level data regarding the presence or absence of ozone-related injury. As shown in Figure 6-16, large geographic areas of EPA Regions 6, 8, and 10 were not included in the assessment. Ozone damage to forest plants is classified using a subjective five-category biosite index based on expert opinion, but designed to be equivalent from site to site. Ranges of biosite values translate to no injury, low or moderate foliar injury (visible foliar injury to highly sensitive or moderately sensitive plants, respectively), and high or severe foliar injury, which would be expected to result in tree-level or ecosystem-level responses, respectively (U.S. EPA, 2006a; Coulston, 2004). The highest percentages of observed high and severe foliar injury, which are most likely to be associated with tree or ecosystem-level responses, are primarily found in the Mid-Atlantic and Southeast regions. While the assessment showed considerable regional variation in ozone injury, this assessment targeted different ozone-sensitive species in different parts of the country with varying ozone sensitivity, which contributes to the apparent regional differences. It is important to note that ozone can have other, more significant impacts on forest plants (e.g., reduced biomass growth in trees) prior to showing signs of visible foliar injury (U.S. EPA, 2006a).

Assessing the impact of ground-level ozone on forests in the U.S. involves understanding the risks to sensitive tree species from ambient ozone concentrations and accounting for the prevalence of those species within the forest. As a way to quantify the risks to particular plants from ground-level ozone, scientists have developed ozone-exposure/tree-response functions by exposing tree seedlings to different ozone levels and measuring reductions in growth as "biomass loss." Typically, seedlings are used because they are easy to manipulate and measure their growth loss from ozone pollution. The mechanisms of susceptibility to ozone within the leaves of seedlings and mature trees are identical, and the decreases predicted using the seedlings should be related to the decrease in overall plant fitness for mature trees, but the magnitude of the effect may be higher or lower depending on the tree species (Chappelka and Samuelson, 1998). In areas where certain ozone-sensitive species dominate the forest community, the biomass loss from ozone can be significant. Experts have identified 2% annual

[44] The data are based on averages of all observations collected in 2002, which is the last year for which data are publicly available. For more information, please consult EPA's 2008 Report on the Environment (U.S. EPA, 2008b).



[c] ***Degree of Injury:*** These categories reflect a subjective index based on expert opinion. Ozone can have other, more significant impacts on forest plants (e.g., reduced biomass growth in trees) prior to showing signs of visible foliar injury.

Figure 6-16. Visible Foliar Injury to Forest Plants from Ozone in U.S. by EPA Regions[a,b,c]

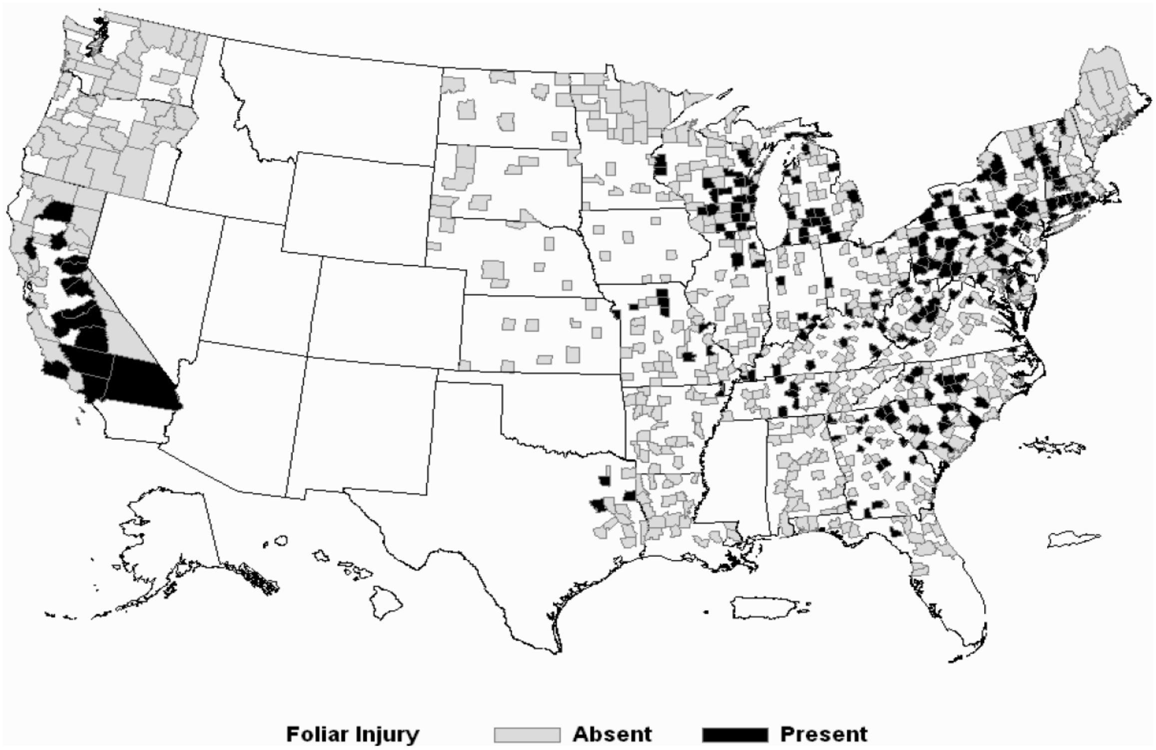


Figure 6-17. Presence and Absence of Visible Foliar Injury, as Measured by U.S. Forest Service, 2002

Source: U.S. EPA, 2007b.

biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health (Heck and Cowling, 1997).

Ozone damage to the plants including the trees and understory in a forest can affect the ability of the forest to sustain suitable habitat for associated species particularly threatened and endangered species that have existence value—a nonuse ecosystem service—for the public. Similarly, damage to trees and the loss of biomass can affect the forest's provisioning services in the form of timber for various commercial uses. In addition, ozone can cause discoloration of leaves and more rapid senescence (early shedding of leaves), which could negatively affect fall-color tourism because the fall foliage would be less available or less attractive. Beyond the aesthetic damage to fall color vistas, forests provide the public with many other recreational and educational services that may be affected by reduced forest health including hiking, wildlife viewing (including bird watching), camping, picnicking, and hunting. Another potential effect of biomass loss in forests is the subsequent loss of climate regulation service in the form of reduced ability to sequester carbon and alteration of hydrologic cycles.

Some of the common tree species in the United States that are sensitive to ozone are black cherry (*Prunus serotina*), tulip-poplar (*Liriodendron tulipifera)*, and eastern white pine (*Pinus strobus)*. Ozone-exposure/tree-response functions have been developed for each of these tree species, as well as for aspen (*Populus tremuliodes*), and ponderosa pine (*Pinus ponderosa*) (U.S. EPA, 2007b).

6.6.4.2 Ozone Effects on Crops

Laboratory and field experiments have shown reductions in yields for agronomic crops exposed to ozone, including vegetables (e.g., lettuce) and field crops (e.g., cotton and wheat). Damage to crops from ozone exposures includes yield losses (i.e., in terms of weight, number, or size of the plant part that is harvested), as well as changes in crop quality (i.e., physical appearance, chemical composition, or the ability to withstand storage) (U.S. EPA, 2007b). The most extensive field experiments, conducted under the National Crop Loss Assessment Network (NCLAN) examined 15 species and numerous cultivars. The NCLAN results show that "several economically important crop species are sensitive to ozone levels typical of those found in the United States" (U.S. EPA, 2006a). In addition, economic studies have shown reduced economic co-benefits as a result of predicted reductions in crop yields, directly affecting the amount and quality of the provisioning service provided by these crops, associated with observed ozone levels (Kopp et al., 1985; Adams et al., 1986; Adams et al., 1989). In addition, visible foliar injury by itself can reduce the market value of certain leafy crops (such as spinach, lettuce). According to the Ozone Staff Paper, there has been no evidence that crops are becoming more tolerant of ozone (U.S. EPA, 2007b). Using the Agriculture Simulation Model (AGSIM) (Taylor, 1994) to calculate the agricultural benefits of reductions in ozone exposure, the EPA estimated that attaining a W126 standard of 13 ppm-hr would produce monetized benefits of approximately $400 million to $620 million in 2006 (inflated to 2006 dollars) (U.S. EPA, 2007b).[45]

6.6.4.3 Ozone Effects on Ornamental Plants

Urban ornamental plants are an additional vegetation category likely to experience some degree of negative effects associated with exposure to ambient ozone levels. Several ornamental species have been listed as sensitive to ozone (Abt Associates, 1995). Because ozone causes visible foliar injury, the aesthetic value of ornamental plants (such as petunia, geranium, and poinsettia) in urban landscapes would be reduced (U.S. EPA, 2007b). Sensitive

[45]These estimates illustrate the value of vegetation effects from a substantial reduction of ozone concentrations, not the marginal change in ozone concentrations anticipated a result of the emission reductions achieved by this rule.

ornamental species would require more frequent replacement and/or increased maintenance (fertilizer or pesticide application) to maintain the desired appearance because of exposure to ambient ozone (U.S. EPA, 2007b). In addition, many businesses rely on healthy-looking vegetation for their livelihoods (e.g., horticulturalists, landscapers, Christmas tree growers, farmers of leafy crops, etc.). The ornamental landscaping industry is a multi-billion dollar industry that affects both private property owners/tenants and governmental units responsible for public areas (Abt Associates, 1995). Preliminary data from the 2007 Economic Census indicate that the landscaping services industry, which is primarily engaged in providing landscape care and maintenance services and installing trees, shrubs, plants, lawns, or gardens, was valued at $53 billion (U.S. Census Bureau, 2010). Therefore, urban ornamentals represent a potentially large unquantified benefit category. This aesthetic damage may affect the enjoyment of urban parks by the public and homeowners' enjoyment of their landscaping and gardening activities. In addition, homeowners may experience a reduction in home value or a home may linger on the market longer due to decreased aesthetic appeal. In the absence of adequate exposure-response functions and economic damage functions for the potential range of effects relevant to ornamental plants, we cannot conduct a quantitative analysis to estimate these effects.

We are unable to provide an estimate of the ozone crop co-benefits associated with the revised or alternative annual standards due to data, time, and resource limitations.

6.7 References

Abt Associates, Inc. 1995. *Urban ornamental plants: sensitivity to ozone and potential economic losses.* U.S. EPA, Office of Air Quality Planning and Standards, Research Triangle Park. Under contract to RADIAN Corporation, contract no. 68-D3-0033, WA no. 6. pp. 9-10. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/abt1995.pdf>.

Abt Associates, Inc. 2001. *Assessing Public Opinions on Visibility Impairment due to Air Pollution: Summary Report*. Prepared for EPA Office of Air Quality Planning and Standards; funded under EPA Contract No. 68-D-98-001. Abt Associates Inc., Bethesda, MD. January. Available on the Internet at <http://www.epa.gov/ttncaaa1/t1/reports/vis_rpt_final.pdf>.

Adams, Evan M., and Frederick, Peter C. 2008. "Effects of methylmercury and spatial complexity on foraging behavior and foraging efficiency in juvenile white ibises (*Eudocimus albus*)." *Environmental Toxicology and Chemistry*. Vol 27, No. 8.

Adams, R. M., Glyer, J. D., Johnson, S. L., McCarl, B. A. 1989. “A reassessment of the economic effects of ozone on U.S. agriculture.” *Journal of the Air Pollution Control Association, 39,* 960-968.

Adams, R. M., Hamilton, S. A., McCarl, B. A. 1986. “The benefits of pollution control: the case of ozone and U.S. agriculture.” *American Journal of Agricultural Economics, 34,* 3-19.

Anselin, L., Le Gallo, J. 2006. “Interpolation of air quality measures in hedonic house price models: Spatial aspects.” *Spatial Economic Analysis* 1: 31–52

Anselin, L., Lozano-Gracia, N. 2008. “Errors in variables and spatial effects in hedonic house price models of ambient air quality.” *Empirical Economics* 34: 5–34.

Arrow, K.J., et al. 1993. *Report of the NOAA Panel on Contingent Valuation*. Federal Register. Volume 58, Number 10. Pages 4601 to 4614. Available on the Internet at <http://www.darp.noaa.gov/library/pdf/cvblue.pdf>.

Arzavus K.M., Dickhut R.M., and Canuel E.A. 2001. “Fate of Atmospherically Deposited Polycyclic Aromatic Hydrocarbons (PAHs) in Chesapeake Bay.” *Environmental Science & Technology*, 35, 2178-2183.

Banzhaf, S., D. Burtraw, D. Evans, and A. Krupnick. 2006. “Valuation of Natural Resource Improvements in the Adirondacks.” *Land Economics* 82:445-464.

Bayer, P., Keohane, N.O. and Timmins, C. 2009. “Migration and hedonic valuation: the case of air quality,” *Journal of Environmental Economics and Management*, 58 (2009) 1–14.

BBC Research & Consulting. 2003. *Phoenix Area Visibility Survey. Draft Report*. Available on the Internet at <http://www.azdeq.gov/environ/air/download/vis_021903f.pdf>.

Bergeron, CM., Bodinof, CM., Unrine, JM., Hopkins, WA. 2010a. “Mercury accumulation along a contamination gradient and nondestructive indices of bioaccumulation in amphibians.” *Environmental Toxicology and Chemistry* 29(4), 980-988.

Bergeron, CM., Bodinof, CM., Unrine, JM., Hopkins, WA. 2010b. “Bioaccumulation and maternal transfer of mercury and selenium in amphibians.” *Environmental Toxicology and Chemistry* 29(4), 989-997.

Beron KJ, Murdoch J, Thayer M. 2001. “The benefits of visibility improvement: New evidence from Los Angeles metropolitan area.” *J Real Estate Finance Econ* 22(2–3):319–337.

Beron KJ, Hanson Y, Murdoch JC, Thayer MA. 2004. “Hedonic price functions and spatial dependence: implications for the demand for urban air quality.” In: Anselin L, Florax RJ, Rey SJ (eds) *Advances in spatial econometrics: methodology, tools and applications*. Springer-Verlag, Berlin, pp 267–281.

Bond, T.C., and Sun H.L. (2005) Can reducing BC emissions counteract global warming? *Environmental Science & Technology* 39 (16).

Brasso, Rebecka L., and Cristol, Daniel A. 2008. "Effects of mercury exposure in the reproductive success of tree swallows (*Tachycineta bicolor*)." *Ecotoxicology*. 17:133-141.

Brookshire, D.S., R.C. d'Arge, W.D. Schulze and M.A. Thayer. 1979. *Methods Development for Assessing Tradeoffs in Environmental Management, Vol. II: Experiments in Valuing Non-Market Goods: A Case Study of Alternative Benefit Measures of Air Pollution Control in the South Coast Air Basin of Southern California*. Prepared for the U.S. Environmental Protection Agency, Office of Research and Development. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0271B-1.pdf/$file/EE-0271B-1.pdf>.

Brookshire, D.S., Thayer, M.A., Schulze, W.D. & D'Arge, R.C. 1982. "Valuing Public Goods: A Comparison of Survey and Hedonic Approaches." *The American Economic Review.* 72(1): 165-177.

Burgess, Neil M., and Meyer, Michael W. 2008. "Methylmercury exposure associated with reduced productivity in common loons." *Ecotoxicology*. 17:83-91.

California State Parks. 2003. *Public Opinions and Attitudes on Outdoor Recreation in California 2002*. California State Parks, Sacramento, California. December 2003. Available on the Internet at <http://parks.ca.gov/pages/795/files/poa2002final.pdf>.

Carson, R.T., R.C. Mitchell, and P.A. Rudd. 1990. "Valuing Air Quality Improvements: Simulating a Hedonic Pricing Equation in the Context of a Contingent Valuation Scenario." In *Visibility and Fine Particles*, Transactions of an AWMA/EPA International Specialty Conference, C.V. Mathai, ed. Air and Waste Management Association, Pittsburgh. Available on the Internet at <http://www.econ.ucsd.edu/~rcarson/papers/AirQuality.pdf>.

Chameides. W.L., Yu, H., Liu, S.C., Bergin, M., Zhou, X., Mearns, L., Wang, G., Kiang, C.S., Saylor, R.D., Luo, C. Huang, Y., Steiner, A., and Giorgi, F. 1999. "Case study of the effects of atmospheric aerosols and regional haze on agriculture: an opportunity to enhance crop yields in China through emission controls?" *PNAS*, 96: 13626-13633.

Chappelka, A.H., Samuelson, L.J. 1998. "Ambient ozone effects on forest trees of the eastern United States: a review." *New Phytologist*, 139, 91-108.

Chay KY, Greenstone M. 2005. "Does air quality matter? evidence form the housing market." *J Polit Econ* 113(2):376–424.

Chestnut, L.G. 1997. *Methodology for Estimating Values for Changes in Visibility in National Parks*. Prepared for Office of Policy, Planning and Evaluation, May. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/chestnut1997.pdf>.

Chestnut, L.G. and R. Dennis. 1997. "Economic Benefits of Improvements in Visibility: Acid Rain Provisions of the 1990 Clean Air Act Amendments." *Journal of Air and Waste Management Association* 47: 395-402.

Chestnut, L.G., and R.D. Rowe. 1990a. "New National Park Visibility Value Estimates." In *Visibility and Fine Particles, Transactions of an AWMA/EPA International Specialty Conference*, C.V. Mathai, ed. Air and Waste Management Association, Pittsburgh.

Chestnut, L.G. and R.D. Rowe. 1990b. *Preservation Values for Visibility Protection at the National Parks*. Prepared for U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, and National Park Service, Air Quality Management Division. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwGA/EE97D10D71E3FAD08525662400766534 >.

Chestnut, L.G. and R.D. Rowe. 1990c. "Economic Valuation of Changes in Visibility: A State of the Science Assessment for NAPAP." Section B5 in Irving, Patricia M., e.d., 1991. *Acid Deposition: State of Science and Technology, Volume IV, Control Technologies, Future Emissions, and Effects Valuation*, The U.S. National Acid Precipitation Assessment Program, Report 27, page 27–153.

Chestnut, L.G., R.D. Rowe and J. Murdoch. 1986. *Review of 'Establishing and Valuing the Effects of Improved Visibility in Eastern United States.'* Prepared for the U.S. Environmental Protection Agency. October. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/chestnut1986.pdf>.

Cordell, H.K., C.J. Betz, G. Green, and M. Owens. 2005. *Off-Highway Vehicle Recreation in the United States, Regions and States: A National Report from the National Survey on Recreation and the Environment (NSRE)*. Prepared for the U.S. Department of Agriculture Forest Service, Southern Research Station, National OHV Policy and Implementation Teams, Athens, GA. Available on the Internet at <http://www.fs.fed.us/recreation/programs/ohv/OHV_final_report.pdf>

Cordell, K., B. Leeworthy, G.T. Green, C. Betz, B. Stephens. 2008. *The National Survey on Recreation & the Environment. Research Work Unit 4953. Pioneering Research on Changing Forest Values in the South and Nation*, U.S. Department of Agriculture Forest Service, Southern Research Station, Athens, GA. Available on the Internet at <http://www.srs.fs.fed.us/trends>.

Cotrufo, M.F., De Santo A.V., Alfani A., Bartoli G., De Cristofaro A. 1995. “Effects of urban heavy metal pollution on organic matter decomposition in Quercus ilex L. Woods.” *Environmental Pollution*, 89(1), 81-87.

Coulston, J.W., Riitters, K.H., Smith, G.C. 2004. “A preliminary assessment of the Montreal process indicators of air pollution for the United States.” *Environmental Monitoring and Assessment, 95*, 57-74.

Cristol D. A., Brasso R. L., Condon A. M., Fovargue R. E., Friedman S. L., Hallinger K. K., Monroe A. P., White A. E. 2008. “The movement of aquatic mercury through terrestrial food webs.” *Science* 320, 335–335.

Cropper, Maureen. 2000. “Has Economic Research Answered the Needs of Environmental Policy?” *Journal of Environmental Economics and Management* 39, 328-350.

Crump, Kate L., and Trudeau, Vance L. 2009. “Mercury-induced reproductive impairment in fish.” *Environmental Toxicology and Chemistry*. Vol. 28, No. 5.

De Steiguer, J., Pye, J., Love, C. 1990. “Air Pollution Damage to U.S. Forests.” *Journal of Forestry*, 88(8), 17-22.

DeBell, L. J.; Gebhart, K. A.; Hand, J. L.; Malm, W. C.; Pitchford, M. L.; Schichtel, B. A.; White, W. H. 2006. *Spatial and Seasonal Patterns and Temporal Variability of Haze and its Constituents in the United States, Report IV*; Cooperative Institute for Research in the Atmosphere, Nov; ISSN 0737-5352-74. Available on the Internet at <http://vista.cira.colostate.edu/improve/Publications/Reports/2006/2006.htm>.

DeHayes, D.H., P.G. Schaberg, G.J. Hawley, and G.R. Strimbeck. 1999. “Acid rain impacts on calcium nutrition and forest health.” *Bioscience* 49(10):789–800.

DeLucchi, M.A., J.J. Murphy, D.R. McCubbin. 2002. “The health and visibility cost of air pollution: A comparison of estimation methods,” *Journal of Environmental Management*, 64 (2), pp. 139–152.

Ely, D.W., Leary, J.T., Stewart, T.R., Ross, D.M. 1991. “The Establishment of the Denver Visibility Standard.” Presented at Air and Waste Management Association 84th Annual Meeting, June 16-21, Paper 91-48.4, 17pp. Available on the Internet at <http://www.albany.edu/cpr/stewart/Papers/T0210-ElyDenverVisStandard-1991cap.pdf>.

Ericksen, J. A., Gustin, M. S., Schorran, D. E., Johnson, D. W., Lindberg, S. E., & Coleman, J. S. (2003). “Accumulation of atmospheric mercury in forest foliage.” *Atmospheric Environment* 37(12), 1613-1622.

Evers, D.C., 2006. *Status assessment and conservation plan for the common loon (Gavia immer) in North America.* U.S. Fish and Wildlife Service, Hadley, MA, USA.

Evers, David C., Savoy, Lucas J., DeSorbo, Christopher R., Yates, David E., Hanson, William, Taylor, Kate M., Siegel, Lori S., Cooley, John H. Jr., Bank, Michael S., Major, Andrew, Munney, Kenneth, Mower, Barry F., Vogel, Harry S., Schoch, Nina, Pokras, Mark, Goodale, Morgan W., Fair, Jeff. 2008. "Adverse effects from environmental mercury loads on breeding common loons." *Ecotoxicology*. 17:69-81.

Fox, S., Mickler, R. A. (Eds.). 1996. *Impact of Air Pollutants on Southern Pine Forests, Ecological Studies.* (Vol. 118, 513 pp.) New York: Springer-Verlag.

Frederick, Peter, and Jayasena, Nilmini. 2010. "Altered pairing behavior and reproductive success in white ibises exposed to environmentally relevant concentrations of methylmercury." *Proceedings of the Royal Society B*. doi: 10-1098.

Gorissen, Leen, Snoeijs, Tinne, Van Duyse, Els, and Eens, Marcel. 2005. "Heavy metal pollution affects dawn singing behavior in a small passerine bird." *Oecologia*. 145: 540-509.

Grantz DA; Garner JHB; Johnson DW. 2003. "Ecological effects of particulate matter." *Environ Int*, 29: 213-239.

Graves, Phil, James C. Murdoch, Mark A. Thayer, and Don Waldman. 1988. "The Robustness of Hedonic Price Estimation: Urban Air Quality." *Land Economics*, 64: 220-33.

Grosclaude P. and N. Soguel. 1994. "Valuing damage to historic buildings using a contingent market: a case study of road traffic externalities," *Journal of Environmental Planning and Management* 37 (1994) (3), pp. 279–287.

Grulke, N.E. 2003. "The physiological basis of ozone injury assessment attributes in Sierran conifers." In A. Bytnerowicz, M.J. Arbaugh, & R. Alonso (Eds.), *Ozone air pollution in the Sierra Nevada: Distribution and effects on forests*. (pp. 55-81). New York, NY: Elsevier Science, Ltd.

Hadley O.L., Corrigan C.E., Kirchstetter T.W., Cliff S.S., and Ramanathan V. (2010) Measured black carbon deposition on the Sierra Nevada snow pack and implication for snow pack retreat. *Atmos. Chem. Phys. Discuss.* 10, 10463-10485 (doi:10.5194/acpd-10-10463-2010).

Hageman KJ; Simonich SL; Campbell DH; Wilson GR; Landers DH. 2006. "Atmospheric deposition of current-use and historic-use pesticides in snow at national parks in the western United States." *Environ Sci Technol*, 40: 3174-3180

Hallinger, Kelly K., Cornell, Kerri L., Brasso, Rebecka L., and Cristol, Daniel A. 2010. "Mercury exposure and survival in free-living tree swallows (*Tachycineta bicolor*)." *Ecotoxicology*. Doi: 10.1007/s10646-010-0554-4.

Hawley, Dana M., Hallinger, Kelly K., Cristol, Daniel A. 2009. “Compromised immune competence in free-living tree swallows exposed to mercury.” *Ecotoxicology*. 18:499-503.

Heck, W.W. &Cowling E.B. 1997. “The need for a long term cumulative secondary ozone standard—an ecological perspective.” *Environmental Management*, January, 23-33.

Hoffman, David J., Henny, Charles J., Hill, Elwood F., Grover, Robert A., Kaiser, James L., Stebbins, Katherine R. 2009. “Mercury and drought along the lower Carson River, Nevada: III. Effects on blood and organ biochemistry and histopathology of snowy egrets and black-crowned night-herons on Lahontan Reservoir, 2002–2006.” *Journal of Toxicology and Environmental Health, Part A*. 72: 20, 1223-1241.

Hutchison, R., and C.E. Kraft. 1994. “Hmong Fishing Activity and Fish Consumption.” Journal *of Great Lakes Research* 20(2):471–487.

Intergovernmental Panel on Climate Change 4^{th} Assessment Report: Climate Change 2007 Synthesis Report Summary for Policy Makers Pachauri, R.K. and Reisinger, A. (Eds.). Available on the Internet at <http://ipcc.ch/publications_and_data/ar4/syr/en/spm.html>.

IMPROVE (Interagency Monitoring of Protected Visual Environments). 2010. *1992–2008 data from the IMPROVE network based on the “New IMPROVE algorithm”* (updated November, 2008). Available on the Internet at <http://vista.cira.colostate.edu/improve/Data/IMPROVE/summary_data.htm.>

Irving, P.M., e.d. 1991. *Acid Deposition: State of Science and Technology, Volume III, Terrestrial, Materials, Health, and Visibility Effects*, The U.S. National Acid Precipitation Assessment Program, Chapter 24, page 24–76.

Joslin, J.D., Kelly, J.M., and van Miegroet, H. 1992. “Soil chemistry and nutrition of North American spruce-fir stands: evidence for recent change. “ *Journal of Environmental Quality*, 21, 12-30.

Kaval, P., and J. Loomis. 2003. *Updated Outdoor Recreation Use Values With Emphasis On National Park Recreation.* Final Report October 2003, under Cooperative Agreement CA 1200-99-009, Project number IMDE-02-0070. Available on the Internet at <http://www.mngt.waikato.ac.nz/departments/staff/pkaval/Kaval%26LoomisNPSReport10-03.pdf>.

Kim, CW, Phipps, T, and Anselin L. 2003. “Measuring the benefits of air quality improvement: a spatial hedonic approach.” *J Environ Econ Manage* 45:24–39.

Koch, D., Bond, T.C., Streets D., and Unger N. (2007) Linking future aerosol radiative forcing to shifts in source activities. *Geophys. Res. Lett.* 34 (L05821), AGU (doi:10.1029/2006GL028360, 2007). Available on the Internet at <http://dx.doi.org/10.1029/2006GL028360>.

Koch D., Bond T.C., Streets D., Unger N., and van der Werf G.R. (2007) Global impacts of aerosols from particular source regions and sectors. *Journal of Geophysical Research-Atmospheres* 112 (D02205).

Kopp, R. J., Vaughn, W. J., Hazilla, M., Carson, R. 1985. "Implications of environmental policy for U.S. agriculture: the case of ambient ozone standards." *Journal of Environmental Management, 20*, 321-331.

Kramer, A., T. Holmes, and M. Haefel. 2003. *Contingent valuation of forest ecosystem protection. Pp. 303–320 in Forests in a Market Economy*. Edited by E.O. Sills and K.L. Abt. Dordrecht, The Netherlands: Kluwer Academic Publishers.

Landers DH; Simonich SL; Jaffe DA; Geiser LH; Campbell DH; Schwindt AR; Schreck CB; Kent ML; Hafner WD; Taylor HE; Hageman KJ; Usenko S; Ackerman LK; Schrlau JE; Rose NL; Blett TF; Erway MM. (2008). *The Fate, Transport and Ecological Impacts of Airborne Contaminants in Western National Parks (USA)*. EPA/600/R-07/138. U.S. Environmental Protection Agency, Office of Research and Development, NHEERL, Western Ecology Division. Corvallis, Oregon.

Leeworthy, V.R., and P.C. Wiley. 2001. *Current Participation Patterns in Marine Recreation*. Silver Spring, MD: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Special Projects. November. Available on the Internet at < http://www.naturetourismplanning.com/pdfs/NSRE_2.pdf>.

Leggett, C.G. and J.E. Neumann. 2004. *Responding to SAB Council Comments on the May 2003 Draft Analytical Plan for the Section 812 Second Prospective—Visibility Benefits.* Prepared for U.S. Environmental Protection Agency OPAR. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/leggettandneumann2004.pdf>.

Leistritz, F. L., D. A. Bangsund, and N. M. Hodur. 2004. "Assessing the Economic Impact of Invasive Weeds: The Case of Leafy Spurge (*Euphorbia esula*)." *Weed Technology* 18:1392-1395.

Leitch, J. A, F. L Leistritz, and D. A Bangsund. 1996. "Economic effect of leafy spurge in the upper Great Plains: methods, models, and results." *Impact Assessment* 14: 419–434.

Lipton, D. W. 1999. “Pfiesteria’s economic impact on seafood industry sales and recreational fishing, p. 35–38.” In B. L. Gardner and L. Koch (ed.), *Proceedings of the Conference, Economics of Policy Options for Nutrient Management and Pfiesteria*. Center for Agricultural and Natural Resource Policy, University of Maryland, College Park.

Loehman, E.T., D. Boldt, K. Chaikin. 1985. *Measuring the Benefits of Air Quality* Improvements *in the San Francisco Bay Area*. From Methods Development for Environmental Control Benefits Assessment, Volume IV. Prepared for the U.S. Environmental Protection Agency, Office of Policy, Planning and Evaluation, September. Grant #R805059-01-0 Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwGA/e85705387833398f8525644d0053bdd0!OpenDocument#_Section6>.

Loehman, E.T., S. Park, and D. Boldt. 1994. “Willingness to Pay for Gains and Losses in Visibility and Health.” *Land Economics* 70(4): 478-498.

Malm, WC. 1999. *Introduction to visibility.* Colorado State University, Fort Collins, CO, USA, 1983, Revised edition, 1999. Available on the Internet at <vista.cira.colostate.edu/improve/Education/IntroToVisinstr.htm>.

McBride, J.R., Miller, P.R., Laven, R.D. 1985. “Effects of oxidant air pollutants on forest succession in the mixed conifer forest type of southern California.” In: *Air Pollutants Effects on Forest Ecosystems, Symposium Proceedings, St. P,* 1985, p. 157-167.

McClelland, G., W. Schulze, D. Waldman, J. Irwin, D. Schenk, T. Stewart, L. Deck and M. Thayer. 1993. *Valuing Eastern Visibility: A Field Test of the Contingent Valuation Method*. Prepared for U.S. Environmental Protection Agency, Office of Policy, Planning and Evaluation. September. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0008-1.pdf/$file/EE-0008-1.pdf >.

McIntyre, J.W., Barr, J.F. 1997. “Common Loon (Gavia immer)” *in*: Pool A, Gill F (eds.) *The Birds of North America*. Academy of Natural Sciences, Philadelphia, PA, 313

McIntrye, J.W., and Evers, D.C.,(eds). 2000. “Loons: old history and new finding.” *Proceedings of a Symposium from the 1997 meeting, American Ornithologists’ Union*. North American Loon Fund, 15 August 1997, Holderness, NH, USA.

Millennium Ecosystem Assessment Board (MEA). 2005. *Ecosystems and Human Well-being: Synthesis*. Washington, DC: World Resources Institute. Available on the Internet at <http://www.who.int/globalchange/ecosystems/ecosystems05/en/index.html>.

Miller, P.R., O.C. Taylor, R.G. Wilhour. 1982. *Oxidant air pollution effects on a western coniferous forest ecosystem*. Corvallis, OR: U.S. Environmental Protection Agency, Environmental Research Laboratory (EPA600-D-82-276).

Mitro, Matthew G., Evers, David C., Meyer, Michael W., and Piper, Walter H. 2008. "Common loon survival rates and mercury in New England and Wisconsin." *Journal of Wildlife Management* 72(3): 665-673.

Munthe, J., Bodaly, R. A., Branfireun, B. A., Driscoll, C. T., Gilmour, C. C., Harris, R., et al. (2007). "Recovery of Mercury-Contaminated Fisheries." *Environmental Science & Technology* 36(1), 33-44.

Murdoch, J. C. and M. A. Thayer. 1988. "Hedonic price estimation of variable urban air quality," *J. Environ. Econ. Management* 15, 143-146.

National Agricultural Statistics Service (NASS). 2008. *Maple Syrup—June 12, 2008: Maple Syrup Production Up 30 Percent Nationwide*. U.S. Department of Agriculture, National Agricultural Statistics Service, New England Agricultural Statistics, Concord, NH.

National Association of State Foresters (NASF). 2009. *Quadrennial Fire Review 2009.*Washington, DC: NASF. Quadrennial Fire and Fuel Review Final Report 2009. National Wildfire Coordinating Group Executive Board. January.

National Oceanic and Atmospheric Administration (NOAA). 2007. *Annual Commercial Landing Statistics*. August. Available on the Internet at <http://www.st.nmfs.noaa.gov/st1/commercial/landings/annual_landings.html>.

National Park Service (NPS). 1997. *Statistical Abstract: 1997.* Department of Interior, National Park Service Social Science Program, Public Use Statistics Office, Denver, Colorado. Available on the Internet at <http://www.nature.nps.gov/stats/abstracts/abst1997.pdf>.

National Park Service (NPS). 2008. *Statistical Abstract: 2008.* Department of Interior, National Park Service Social Science Program, Public Use Statistics Office, Denver, Colorado. Available on the Internet at <http://www.nature.nps.gov/stats/abstracts/abst2008.pdf>.

Neill, Helen R., David M. Hassenzahl and Djeto D. Assane. 2007. "Estimating the Effect of Air Quality: Spatial versus Traditional Hedonic Price Models," *Southern Economic Journal* 73, No. 4, pp. 1088-1111.

Neumann, J.E. and R.W. Patterson. 2009. *Alternative Approach to Estimating Monetized Benefits of Residential Visibility for the Section 812 Second Prospective.* Prepared for U.S. Environmental Protection Agency OAR/OPAR. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/neumannpaterson2009.pdf>.

Niklinska M., Laskowski R., Maryanski M. (1998). "Effect of heavy metals and storage time on two types of forest litter: basal respiration rate and exchangeable metals" *Ecotoxicological Environmental Safety*, 41, 8-18.

Office of Management and Budget (OMB). 2003. *Circular A-4: Regulatory Analysis.* Washington, DC. Available on the internet at <http://www.whitehouse.gov/omb/circulars/a004/a-4.html>.

Painter, et al. 2007. "Impact of disturbed desert soils on duration of mountain snow cover." *Geophys. Res. Lett*. 34, L12502.

Parrish ZD; White JC; Isleyen M; Gent MPN; Iannucci-Berger W; Eitzer BD; Kelsey JW; Mattina MI. 2006. "Accumulation of weathered polycyclic aromatic hydrocarbons (PAHs) by plant and earthworm species." *Chemosphere*, 64: 609-618.

Patterson, R.W., J.E. Neumann, and C.G. Leggett. 2005. *Recommended Residential Visibility Values for the Section 812 Second Prospective Analysis.* Prepared for U.S. Environmental Protection Agency OPAR. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/patersonneumanleggett2005.pdf>.

Peterson, D.E., M.S. Kanarek, M.A. Kuykendall, J.M. Diedrich, H.A. Anderson, P.L. Remington, and T.B. Sheffy. 1994. "Fish Consumption Patterns and Blood Mercury Levels in Wisconsin Chippewa Indians." *Archives of Environmental Health* 49(1):53–58.

Pitchford, M and W. Malm. 1994. "Development and Application of a Standard Visual Index." *Atmospheric Environment* 28(5): 1049-1054.

Prasad, A.M. and Iverson, L.R. 2003. "Little's range and FIA importance value database for 135 eastern U.S. tree species." Northeastern Research Station, USDA Forest Service. Available on the Internet at <http://www.fs.fed.us/ne/delaware/4153/global/littlefia/index.html>.

Pryor, S.C. 1996. "Assessing Public Perception of Visibility for Standard Setting Exercises." *Atmospheric Environment* 30, no. 15, pp. 2705-2716.

Pye, J.M. 1988. "Impact of ozone on the growth and yield of trees: A review." *Journal of Environmental Quality, 17*, 347-360.

Qian Y., Gustafson W.I., Leung L.R., and Ghan S.J. 2009. "Effects of soot-induced snow albedo change on snowpack and hydrological cycle in western United States based on Weather Research Forecasting Chemistry and Regional Climate Simulations." *Journal of Geophysical Research* 114 (DO2108).

Rae, D.A. 1983. *Benefits of Visual Air Quality in Cincinnati Results of a Contingent Ranking Survey*. Prepared for Electric Power Research Institute. May.

Rea, A. W., Lindberg, S. E., Scherbatskoy, T., & Keeler, G. J. 2002. "Mercury Accumulation in Foliage over Time in Two Northern Mixed-Hardwood Forests." *Water, Air, & Soil Pollution* 133(1), 49-67.

Rimmer, C. C., McFarland, K. P., Evers, D. C., Miller, E. K., Aubry, Y., Busby, D., et al. 2005. “Mercury Concentrations in Bicknell’s Thrush and Other Insectivorous Passerines in Montane Forests of Northeastern North America.” *Ecotoxicology* 14(1), 223-240.

Rosen, S. 1974. “Hedonic prices and implicit markets: product differentiation in pure competition.” *Journal of Political Economy* 82(1): 34-55.

Scheuhammer, Anton M., Meyer Michael W., Sandheinrich, Mark B., and Murray, Michael W. 2007. “Effects of environmental methylmercury on the health of wild birds, mammals, and fish,” *Ambio* Vol.36, No.1.

Sepulveda, Maria S., Frederick, Peter C., Spalding, Marilyn G., and Williams, Gary E. Jr. 1999. “Mercury contamination in free-ranging great egret nestlings (*Ardea albus*) from southern Florida, USA,” *Environmental Toxicology and Chemistry* Vol. 18, No.5.

Simcik M.F., Eisenreich S.J., Lioy P.J. 1999. “Source apportionment and source/sink relationship of PAHs in the coastal atmosphere of Chicago and Lake Michigan.” *Atmospheric Environment*, 33, 5071-5079.

Simcik M.F., Eisenreich, S.J., Golden K.A., et al. 1996. “Atmospheric Loading of Polycyclic Aromatic Hydrocarbons to Lake Michigan as Recorded in the Sediments.” *Environmental Science and Technology*, 30, 3039-3046.

Sisler, J.F. 1996. *Spatial and seasonal patterns and long-term variability of the composition of the haze in the United States: an analysis of data from the IMPROVE network*. CIRA Report, ISSN 0737-5352-32, Colorado State University.

Smith, G., Coulston, J., Jepsen, E., Prichard, T. 2003. “A national ozone biomonitoring program—results from field surveys of ozone sensitive plants in Northeastern forests (1994–2000).” *Environmental Monitoring and Assessment, 87*, 271-291.

Smith, A.E.; Kemp, M.A.; Savage, T.H.; Taylor, C.L. 2005. “Methods and Results from a New Survey of Values for Eastern Regional Haze Improvements,” *J. Air & Waste Manage Assoc* 55, 1767-1779.

Smith, V.K., G. Van Houtven, and S.K. Pattanayak. 2002. “Benefit Transfer via Preference Calibration.” *Land Economics* 78:132-152.

Smith, V.K., and J.C. Huang. 1995. “Can markets value air quality? A meta-analysis of hedonic property value models*,” J. Polit. Econ*, 103, pp. 209–227.

Spencer, D.M., and D.F. Holecek. 2007. “Basic characteristics of the fall tourism market.” *Tourism Management* 28:491–504.

Taylor R. 1994. “Deterministic versus stochastic evaluation of the aggregate economic effects of price support programs.” *Agricultural Systems* 44: 461-473.

Tingey, D.T., and Taylor, G.E. 1982. "Variation in plant response to ozone: a conceptual model of physiological events." In M.H. Unsworth & D.P. Omrod (Eds.), *Effects of Gaseous Air Pollution in Agriculture and Horticulture*. (pp.113-138). London, UK: Butterworth Scientific.

Tolley, G., A. Randall, G. Blomquist, M. Brien, R. Fabian, G. Fishelson, A. Frankel, M. Grenchik, J. Hoehn, A. Kelly, R. Krumm, E. Mensah, and T. Smith. 1986. *Establishing and Valuing the Effects of Improved Visibility in Eastern United States*. Prepared for U.S. Environmental Protection Agency, Office of Policy, Planning and Evaluation. March. U.S. Environmental Protection Agency Grant #807768-01-0. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwGA/e673ccafbda23639852565f00007412f!OpenDocument&ExpandSection=3>.

Tolley, George and Robert Fabian (eds). 1988. *The Economic Value of Visibility*. Series : Studies in urban and resource economics. Mount Pleasant, MO: Blackstone.

Trijonis, J.C. et al. 1987. "Preliminary extinction budget results from the RESOLVE program," pp. 872-883. In: P.J. Bhardwaja, et al. *Visibility Protection Research and Policy Aspects*. Air Pollution Control Assoc., Pittsburgh, PA.

Trijonis, J.C. et al. 1988. *RESOLVE Project Final Report: Visibility conditions and Causes of Visibility Degradation in the Mojave Desert of California.* NWC TP #6869. Naval Weapons Center, China Lake, CA. Available on the Internet at <www.dtic.mil/dtic/tr/fulltext/u2/a206322.pdf>.

United Nations Environmental Programme, World Meteorological Organization and Stockholm Environment Institute (UNEP). 2011. Integrated Assessment of Black Carbon and Tropospheric Ozone Summary for Decision Makers. Available on the Internet at <http://www.unep.org/dewa/Portals/67/pdf/Black_Carbon.pdf>.

U.S. Census Bureau. 2010. "Sector 56: EC075612: Administrative Support and Waste Management and Remediation Services: Industry Series: Preliminary Comparative Statistics for the United States (2002 NAICS Basis): 2007 and 2002." *American FactFinder* Available on the Internet at http://factfinder.census.gov (accessed August 25, 2010)

U.S. Department of the Interior (DOI). 2007. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*. U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. Washington, DC. Available on the Internet at <http://www.census.gov/prod/2008pubs/fhw06-nat.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 1999a. *The Clean Air Act Amendments (CAAA) Section 812 Prospective Study of Costs and Benefits (1999): Advisory by the Advisory Council on Clean Air Compliance Analysis: Costs and Benefits of the CAAA.* EPA-SAB-COUNCIL-ADV-00-002. October. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/E992F918DF32131A852571B9004571F7/$File/coua0002.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 1999b. *Final Advisory by the Advisory Council on Clean Air Compliance Analysis on the 1999 Prospective Study of Costs and Benefits (1999) of Implementation of the Clean Air Act Amendments (CAAA).* EPA-SAB-COUNCIL-ADV-00-003. November. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/93BE842F5CBA7BEF852571B900461197/$File/coua0003.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2004. *Review of the Draft Analytical Plan for EPA's Second Prospective Analysis—Benefits and Costs of the Clean Air Act, 1990-2020: An Advisory by the Advisory Council for Clean Air Compliance Analysis*. EPA-SAB-COUNCIL-ADV-04-004. May. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/7CCBBFE15.BD4C8B185256F17005E3079/$File/council_adv_04004.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2007. *Clean Air Scientific Advisory Committee's (CASAC) Review of the Agency's Final Ozone Staff Paper.* EPA-CASAC-07-002. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/FE915E916333D776852572AC007397B5/$File/casac-07-002.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2010a. *Advisory on a Preliminary Draft of the Second Section 812 Prospective Study of the Benefits and Costs of the Clean Air Act (April 2010)*. EPA-COUNCIL-10-004. July. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/553D59A280CDCF388525776D005EE83A/$File/EPA-COUNCIL-10-004-unsigned.pdf>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 2010b. *Review of the Final Integrated Report for the Second Section 812 Prospective Study of the Benefits and Costs of the Clean Air Act (August 2010).* EPA-COUNCIL-11-001. December. Available on the Internet at <http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/1E6218DE3BFF682E852577FB005D46F1/$File/EPA-COUNCIL-11-001-unsigned.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 1997. *Mercury Study Report to Congress. Volume V: Health Effects of Mercury and Mercury Compounds.* EPA-452/R-97-007. U.S. EPA Office of Air Quality Planning and Standards, and Office of Research and Development. Available on the Internet at <http://www.epa.gov/ttncaaa1/t3/reports/volume5.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 1998. *Advisory Council on Clean Air Compliance Analysis Advisory on the Clean Air Act Amendments (CAAA) of 1990 Section 812 Prospective Study: Overview of Air Quality and Emissions Estimates: Modeling, Health and Ecological Valuation Issues Initial Studies*. EPA-SAB-COUNCIL-ADV-98-003. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/D20F5AE06F98883F852571AF00474927/$File/coad9803.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 1999a. *Regional Haze Regulations*, July 1, 1999. (64 FR 35714, July 1, 1999).

U.S. Environmental Protection Agency (U.S. EPA). 1999b. *The Benefits and Costs of the Clean Air Act, 1990–2010.* Prepared for U.S. Congress by U.S. EPA, Office of Air and Radiation, Office of Policy Analysis and Review, Washington, DC, November; EPA report no. EPA410-R-99-001. Available on the Internet at <http://www.epa.gov/oar/sect812/1990-2010/chap1130.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2004. *Air Quality Criteria for Particulate Matter Volume II of II*. National Center for Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC EPA/600/P-99/002bF. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=87903>.

U.S. Environmental Protection Agency (U.S. EPA). 2005. *Regulatory Impact Analysis of the Final Clean Air Mercury Rule.* Office of Air Quality Planning and Standards, Research Triangle Park, NC., March; EPA report no. EPA-452/R-05-003. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/mercury_ria_final.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2006b. *Regulatory Impact Analysis, 2006 National Ambient Air Quality Standards for Particulate Matter*. Office of Air Quality Planning and Standards, Research Triangle Park, NC. October. Available on the Internet at < http://www.epa.gov/ttn/ecas/regdata/RIAs/Appendix%20I--Visibility%20Benefits%20Methodology.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2006c. *Ecological Benefits Assessment Strategic Plan*. EPA-240-R-06-001. Office of the Administrator. Washington, DC. October. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eed.nsf/webpages/EcologBenefitsPlan.html>.

U.S. Environmental Protection Agency (U.S. EPA). 2007a. Guidance *on the Use of Models and Other Analyses for Demonstrating Attainment of Air Quality Goals for Ozone, $PM_{2.5}$, and Regional Haze*. Office of Air Quality Planning and Standards. EPA-454/B-07-002. April. <Available on the Internet at http://www.epa.gov/scram001/guidance/guide/final-03-pm-rh-guidance.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2007b. *Review of the National Ambient Air Quality Standards for Ozone: Policy assessment of scientific and technical information*. Staff Paper. Office of Air Quality Planning and Standards. EPA-452/R-07-007a. July. Available on the Internet at <http://www.epa.gov/ttn/naaqs/standards/ozone/data/2007_07_ozone_staff_paper.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2007c. *Technical Report on Ozone Exposure, Risk, and Impact Assessments for Vegetation*. Office of Air Quality Planning and Standards. EPA-452/R-07-002. January. Available on the Internet at http://www.epa.gov/ttn/naaqs/standards/ozone/s_o3_pr.html

U.S. Environmental Protection Agency (U.S. EPA). 2008a. *Regulatory Impact Analysis, 2008 National Ambient Air Quality Standards for Ground-level Ozone, Chapter 6*. Office of Air Quality Planning and Standards, Research Triangle Park, NC. March. Available at <http://www.epa.gov/ttn/ecas/regdata/RIAs/6-ozoneriachapter6.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2008b. *U.S. EPA's 2008 Report on the Environment (Final Report).* Washington, D.C., EPA/600/R-07/045F (NTIS PB2008-112484). May. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=190806>. Updated in December 2010, available on the Internet at <http://cfpub.epa.gov/eroe/index.cfm?fuseaction=detail.viewInd&showQues=Air&ch=47&lShowInd=0,287&subtop=341&lv=list.listByQues&r=188207>.

U.S. Environmental Protection Agency (U.S. EPA). 2008c. *Integrated Science Assessment for Oxides of Nitrogen and Sulfur—Ecological Criteria National (Final Report)*. National Center for Environmental Assessment, Research Triangle Park, NC. EPA/600/R-08/139. December. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=201485>.

U.S. Environmental Protection Agency (U.S. EPA). 2009a. *The NO_x Budget Trading Program: 2008 Environmental Results*. Clean Air Markets Division. September. Available on the Internet at <http://www.epa.gov/airmarkt/progress/NBP_3/ NBP_2008_Environmental_Results.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2009b. *Integrated Science Assessment for Particulate Matter (Final Report)*. EPA-600-R-08-139F. National Center for Environmental Assessment—RTP Division. December. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=216546>.

U.S. Environmental Protection Agency (U.S. EPA). 2009c. *Risk and Exposure Assessment for Review of the Secondary National Ambient Air Quality Standards for Oxides of Nitrogen and Oxides of Sulfur (Final Report*). EPA-452/R-09-008a. Office of Air Quality Planning and Standards, Research Triangle Park, NC. September. Available on the Internet at <http://www.epa.gov/ttn/naaqs/standards/no2so2sec/data/NOxSOxREASep2009MainContent.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2009d. *Endangerment and Cause or Contribute Findings for Greenhouse Gases Under Section 202(a) of the Clean Air Act.* December 15, 2009. (74 Fed. Reg. 66,496, December 15, 2009).

U.S. Environmental Protection Agency (U.S. EPA). 2010a. *Updated Regulatory Impact Analysis (RIA) for the Reconsideration of the 2008 Ozone National Ambient Air Quality Standard (NAAQS)*. Office of Air Quality Planning and Standards, Research Triangle Park, NC. January. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/s1-supplemental_analysis_full.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2010b. *Particulate Matter Urban-Focused Visibility Assessment—Final Document (Corrected).* Office of Air Quality Planning and Standards, Research Triangle Park, NC. July. Available on the Internet at <Particulate Matter Urban-Focused Visibility Assessment—Final Document>.

U.S. Environmental Protection Agency (U.S. EPA). 2011a. *Policy Assessment for the Review of the PM NAAQS—Final Document.* Office of Air Quality Planning and Standards, Research Triangle Park, NC. April. Available on the Internet at <http://www.epa.gov/ttn/naaqs/standards/pm/s_pm_2007_pa.html>.

U.S. Environmental Protection Agency (U.S. EPA). 2011b*. The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress.* Office of Air and Radiation, Washington, DC. March. Available on the Internet at <http://www.epa.gov/air/sect812/feb11/fullreport.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2011c. *Regulatory Impact Analysis for the Federal Implementation Plans to Reduce Interstate Transport of Fine Particulate Matter and Ozone in 27 States; Correction of SIP Approvals for 22 States*. June. Available on the Internet at: <http://www.epa.gov/airtransport/pdfs/FinalRIA.pdf>

U.S. Environmental Protection Agency (U.S. EPA). 2011d. *Regulatory Impact Analysis for the Final Mercury and Air Toxics Standards.* EPA-452/R-11-011. December. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/matsriafinal.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2012a. *Integrated Science Assessment for Particulate* Ozone and Related Photochemical Oxidants *(Third External Review Draft)*. EPA/600/R-10/076C. National Center for Environmental Assessment—RTP Division. June. Available on the Internet at <http://cfpub.epa.gov/ncea/isa/recordisplay.cfm?deid=242490>.

U.S. Environmental Protection Agency (U.S. EPA). 2012b. *Black Carbon Report to Congress*. EPA-450/R-12-001. March. Available on the Internet at <http://www.epa.gov/blackcarbon/2012report/fullreport.pdf>.

U.S. Fish and Wildlife Service and U.S. Census Bureau (FWS and Census). 2007. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (FHWAR).* FHW/06-NAT. U.S. Department of the Interior, U.S. Fish and Wildlife Service, Washington, DC, and U.S. Department of Commerce, U.S. Census Bureau, Washington, DC.

U.S. Forest Service (USFS). 2006. *Forest Inventory and Analysis National Program: Forest Inventory Data Online*. Online database. U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis, Arlington, VA. Available on the Internet at <http://fia.fs.fed.us/tools-data>.

Valigura, R.A., R.B. Alexander, M.S. Castro, T.P. Meyers, H.W. Paerl, P.E. Stacy, and R.E. Turner. 2001. *Nitrogen Loading in Coastal Water Bodies: An Atmospheric Perspective*. Washington, DC: American Geophysical Union.

Van Sickle, J., Baker, J.P., Simonin, H.A., Baldigo, B.P., Kretser, W.A., Sharpe, W.E. 1996. "Episodic acidification of small streams in the northeastern United States: Fish mortality in field bioassays." *Ecological Applications*, 6, 408-421.

Wada, H. and Cristol, D.A. and McNabb, F.M.A. and Hopkins, W.A. 2009. "Suppressed adrenocortical responses and thyroid hormone levels in birds near a mercury-contaminated river." *Environmental Science & Technology* 43(15), 6031-6038.

Wada, H., Yates, DE., Evers, DC., Taylor, RJ., Hopkins, WA. 2010. "Tissue mercury concentrations and adrenocortical responses of female big brown bats (*Eptesicus fuscus*) near a contaminated river." *Ecotoxicology* 19(7), 1277-1284.

Winner, W.E. 1994. "Mechanistic analysis of plant responses to air pollution." *Ecological Applications, 4(4),* 651-661.

Winner, W.E., and C.J. Atkinson. 1986. "Absorption of air pollution by plants, and consequences for growth." *Trends in Ecology and Evolution* 1:15-18.

Wiscombe W. and Warren S. 1980. "A model for the spectral albedo of snow. Snow containing atmospheric aerosols." *J. Atmos. Sci.* 37, 2734-2274.

Yates, David E., Mayack, David T., Munney, Kenneth, Evers David C., Major, Andrew, Kaur, Taranjit, and Taylor, Robert J. 2005. “Mercury levels in mink (*Mustela vison*) and river otter (*Lonra canadensis*) from northeastern North America,” *Ecotoxicology* 14, 263-274.

APPENDIX 6.A

ADDITIONAL DETAILS REGARDING THE VISIBILITY BENEFITS METHODOLOGY

6.A.1 Introduction

Economic benefits may result from two broad categories of changes in light extinction: (1) changes in "residential" visibility—i.e., the visibility in and around the locations where people live; and (2) changes in "recreational" visibility at Class I areas—i.e., visibility at Class I national parks and wilderness areas.[1] In this analysis, only those recreational and residential benefits in areas that have been directly studied in the valuation literature are included in the primary presentation of benefits; recreational benefits in other U.S. Class I regions and residential benefits in other metropolitan areas are presented as sensitivity analyses of visibility benefits.

In Chapter 6 of this RIA, we provide an overview of the visibility benefits methodology and results. This appendix provides additional detail regarding specific aspects of the visibility benefits methodology and is organized as follows. Section 6.A.2 describes the process we used to convert the modeled light extinction data to match the spatial scale of the visibility benefits assessment. We present the basic utility model in Section 6.A.3. In Section 6.A.4 we discuss the measurement of visibility, and the mapping from environmental "bads" to environmental "goods." In Sections 6.A.5 and 6.A.6 we summarize the methodology for estimating the parameters of the model corresponding to visibility at in-region and out-of-region Class I areas, and visibility in residential areas, respectively, and we describe the methods used to estimate these parameters. Section 6.A.7 describes the process for aggregating the recreational and residential visibility benefits. Section 6.A.8 describes the adjustment to reflect income growth over time. Section 6.A.9 provides all the parameters used to calculate visibility benefits.

6.A.2 Converting Modeled Light Extinction Estimates

To calculate visibility benefits, we use light extinction estimates generated by the CMAQ model.[2] Modeled light extinction estimates are measured in units of inverse megameters (Mm^{-1}). Because the valuation studies measure visibility in terms of visual range, we convert the light extinction units from Mm^{-1} to visual range (in km) for both recreational and residential

---

[1] Hereafter referred to as Class I areas, which are defined as areas of the country such as national parks, national wilderness areas, and national monuments that have been set aside under Section 169(a) of the Clean Air Act to receive the most stringent degree of air quality protection. Class I federal lands fall under the jurisdiction of three federal agencies, the National Park Service, the Fish and Wildlife Service, and the Forest Service.

[2] For more information regarding the CMAQ modeling conducted for the PM NAAQS RIA, please see Chapter 3 of this RIA.

visibility benefits. Using the relationships derived by Pitchford and Malm (1994), the formulas for this conversion are

$$Deciviews = 10 * \ln\left(\frac{391}{VR}\right) = 10 \ * \ln\left(\frac{\beta_{ext}}{10}\right)$$

where VR denotes visual range (in kilometers) and β_{ext} denotes light extinction (in Mm^{-1}). Because we leverage the tools and data prepared for previous analyses (U.S. EPA, 2011), we use a two-step process to convert from Mm^{-1} to VR using deciviews as an intermediate conversion instead of converting directly. Therefore, the full formula incorporating the two-step conversion is

$$VR = 391 * e^{-0.1*(10*\ln\left(\frac{\beta_{ext}}{10}\right))}$$

The spatial scale of the modeled light extinction estimates must also be adjusted to correspond with the design of the valuation studies and the underlying population and economic data. For the residential visibility benefits analysis, we convert the spatial resolution of the light extinction estimates from 12-km grid to county-level. We use county-level light extinction to match the MSA boundaries, population data, and household income data. We used the geographic centroids of each 12-km grid cell with the Veronoi Neighborhood Averaging (VNA) interpolation method in the BenMAP model for this conversion (Abt Associates, 2010).

For the recreational visibility benefits analysis, we use the light extinction estimates from 12-km grid cell located at the geographic center of the Class I area. Although we considered using the IMPROVE monitor location instead, we selected the park centroid for three reasons:

1. Consistency with previous method for estimating recreational visibility benefits
2. Not all Class I areas have monitors, and shared monitors may be outside park
3. Siting criteria for IMPROVE monitors do not include iconic scenic vista location

6.A.3 Basic Utility Model

Within the category of recreational visibility, further distinctions have been made. There is evidence (Chestnut and Rowe, 1990) that an individual's WTP for improvements in visibility at a Class I area is influenced by whether it is in the region in which the individual lives, or whether it is somewhere else. In general, people appear to be willing to pay more for visibility improvements at parks and wilderness areas that are “in-region” than at those that are “out-of-region.” This is plausible, because people are more likely to visit, be familiar with, and care about parks and wilderness areas in their own part of the country.

To value estimated changes in visibility, we use an approach that is consistent with economic theory. Below we discuss an application of the Constant Elasticity of Substitution (CES) utility function approach[3] to value both residential visibility improvements and visibility improvements at Class I areas in the United States. This approach is based on the preference calibration method developed by Smith, Van Houtven, and Pattanayak (2002).

We begin with a CES utility function in which a household derives utility from

1. “all consumption goods,” X,
2. visibility in the residential area in which the household is located (“residential visibility”),[4]
3. visibility at Class I areas in the same region as the household (“in-region recreational visibility”), and
4. visibility at Class I areas outside the household’s region (“out-of-region recreational visibility”).

We have specified a total of six recreational visibility regions,[5] so there are five regions for which any household is out of region. The utility function of a household in the n[th] residential area and the *i*[th] region of the country is:

$$U_{ni} = \left(X^{\rho} + \theta Z_n^{\rho} + \sum_{k=1}^{N_i} \gamma_{ik} Q_{ik}^{\rho} + \sum_{j \neq i} \sum_{k=1}^{N_j} \delta_{jk} Q_{jk}^{\rho}\right)^{1/\rho} \quad ,$$

$$\theta > 0, \gamma_{ik} > 0, \forall i,k, \delta_{jk} > 0, \forall j,k, \ \rho \leq 1.$$

[3] The constant elasticity of substitution utility function has been chosen for use in this analysis because of its flexibility when illustrating the degree of substitutability present in various economic relationships (in this case, the trade-off between income and improvements in visibility).

[4] We remind the reader that, although residential and recreational visibility benefits estimation is discussed simultaneously in this section, benefits are calculated and presented separately for each visibility category.

[5] See Section 6.3.4 of this RIA for a description of the different recreational visibility considered in this analysis.

where

Z_n = the level of visibility in the nth residential area;

Q_{ik} = the level of visibility at the k^{th} in-region park (i.e., the k^{th} park in the i^{th} region);

Q_{jk} = the level of visibility at the k^{th} park in the j^{th} region (for which the household is out of region), $j \neq i$;

N_i = the number of Class I areas in the i^{th} region;

N_j = the number of Class I areas in the j^{th} region (for which the household is out of region), $j \neq i$; and

θ, the γ's and δ's are parameters of the utility function corresponding to the visibility levels at residential areas, and at in-region and out-of-region Class I areas, respectively.

In particular, the γ_{ik}'s are the parameters corresponding to visibility at in-region Class I areas; the δ_1's are the parameters corresponding to visibility at Class I areas in region 1 (California), if $i \neq 1$; the δ_2's are the parameters corresponding to visibility at Class I areas in region 2 (Colorado Plateau), if $i \neq 2$, and so forth. Because the model assumes that the relationship between residential visibility and utility is the same everywhere, there is only one θ. The parameter ρ in this CES utility function is an important determinant of the slope of the marginal WTP curve associated with any of the environmental quality variables. When ρ=1, the marginal WTP curve is horizontal. When ρ<1, it is downward sloping.

The household's budget constraint is:

$$m - p \cdot X \leq 0 \ ,$$

where m is income, and p is the price of X. Without loss of generality, set ρ = 1. The only choice variable is X. The household maximizes its utility by choosing X=m. The indirect utility function for a household in the n^{th} residential area and the i^{th} region is therefore

$$V_{ni}(m, Z_n, Q; \theta, \gamma, \delta, \rho) = (m^{\rho} + \theta Z_n^{\rho} + \sum_{k=1}^{N_i} \gamma_{ik} Q_{ik}^{\rho} + \sum_{j \neq i} \sum_{k=1}^{N_j} \delta_{jk} Q_{jk}^{\rho})^{1/\rho} \quad ,$$

where Q denotes the vector of vectors, Q_1, Q_2, Q_3, Q_4, Q_5, and Q_6, and the unsubscripted γ and δ denote vectors as well.

Given estimates of ρ, θ, the γ's and the δ's, the household's utility function and the corresponding WTP functions are fully specified. The household's WTP for any set of changes in the levels of visibility at in-region Class I areas, out-of-region Class I areas, and the household's residential area can be shown to be:

$$WTP_{ni}(\Delta Z, \Delta Q) = m - [m^{\rho} + \theta(Z_{0n}^{\rho} - Z_{1n}^{\rho}) + \sum_{k=1}^{N_i} \gamma_{ik}(Q_{0ik}^{\rho} - Q_{1ik}^{\rho}) + \sum_{j \neq i} \sum_{k=1}^{N_j} \delta_{jk}(Q_{0jk}^{\rho} - Q_{1jk}^{\rho})]^{1/\rho} \quad .$$

The household's WTP for a single visibility improvement will depend on its order in the series of visibility improvements the household is valuing. If it is the first visibility improvement to be valued, the household's WTP for it follows directly from the previous equation. For example, the household's WTP for an improvement in visibility at the first in-region park, from $Q_{i1} = Q_{0i1}$ to $Q_{i1} = Q_{1i1}$, is

$$WTP(\Delta Q_{i1}) = m - [m^{\rho} + \gamma_{i1}(Q_{0i1}^{\rho} - Q_{1i1}^{\rho})]^{1/\rho} \quad ,$$

if this is the first (or only) visibility change the household values.

6.A.4 Measure of Visibility: Environmental "Goods" Versus "Bads"

In the above model, Q and Z are environmental "goods." As the level of visibility increases, utility increases. The utility function and the corresponding WTP function both have reasonable properties. The first derivative of the indirect utility function with respect to Q (or Z) is positive; the second derivative is negative. WTP for a change from Q_0 to a higher (improved) level of visibility, Q_1, is therefore a concave function of Q_1, with decreasing marginal WTP.

The measure of visibility that is currently preferred by air quality scientists is the deciview, which increases as visibility *decreases*. Deciview, in effect, is a measure of the *lack* of visibility. As deciviews increase, visibility, and therefore utility, decreases. The deciview, then, is a measure of an environmental "bad." There are many examples of environmental "bads"—all types of pollution are environmental "bads." Utility decreases, for example, as the concentration of particulate matter in the atmosphere increases.

One way to value decreases in environmental bads is to consider the "goods" with which they are associated, and to incorporate those goods into the utility function. In particular, if B denotes an environmental "bad," such that:

$$\frac{\partial V}{\partial B} < 0 \quad ,$$

and the environmental "good," Q, is a function of B,

$$Q = F(B) \ ,$$

then the environmental "bad" can be related to utility via the corresponding environmental "good":[6]

$$V = V(m, Q) = V(m, F(B)) \ .$$

The relationship between Q and B, F(B), is an empirical relationship that must be estimated.

There is a potential problem with this approach, however. If the function relating B and Q is not the same everywhere (i.e., if for a given value of B, the value of Q depends on other factors as well), then there can be more than one value of the environmental good corresponding to any given value of the environmental bad, and it is not clear which value to use. This has been identified as a problem with translating deciviews (an environmental "bad") into visual range (an environmental "good"). It has been noted that, for a given deciview value, there can be many different visual ranges, depending on the other factors that affect visual range—such as light angle and altitude. We note here, however, that this problem is not unique to visibility, but is a general problem when trying to translate environmental "bads" into "goods."[7]

In order to translate deciviews (a "bad") into visual range (a "good"), we use a relationship derived by Pitchford and Malm (1994) in which

$$DV = 10 * ln(\frac{391}{VR}) \ ,$$

where DV denotes deciview and VR denotes visual range (in kilometers). Solving for VR as a function of DV yields

[6] There may be more than one "good" related to a given environmental "bad." To simplify the discussion, however, we assume only a single "good."

[7] Another example of an environmental "bad" is particulate matter air pollution (PM). The relationship between survival probability (Q) and the ambient PM level is generally taken to be of the form $Q = 1 - \alpha e^{\beta PM}$. where ∀ denotes the mortality rate (or level) when there is no ambient PM (i.e., when PM=0). However, α is implicitly a function of all the factors other than PM that affect mortality. As these factors change (e.g., from one location to another), αwill change (just as visual range changes as light angle changes). It is therefore possible to have many values of Q corresponding to a given value of PM, as the values of ∀ vary.

$$VR = 391 * e^{-0.1DV} \quad .$$

This conversion is based on specific assumptions characterizing the "average" conditions of those factors, such as light angle, that affect visual range. To the extent that specific locations depart from the average conditions, the relationship will be an imperfect approximation.[8]

6.A.5 Estimating the Parameters for Visibility at Class I Areas: the γ's and δ's

As noted in Section 6.A.3, if we consider a particular visibility change as the first or the only visibility change valued by the household, the household's WTP for that change in visibility can be calculated, given income (m), the "shape" parameter, ρ, and the corresponding recreational visibility parameter. For example, a Southeast household's WTP for a change in visibility at in-region parks (collectively) from $Q_1 = Q_{01}$ to $Q_1 = Q_{11}$ is:

$$WTP(DQ_1) = m - [m^r + g_1(Q_{01}^r - Q_{11}^r)]^{1/r}$$

if this is the first (or only) visibility change the household values.

Alternatively, if we have estimates of m as well as WTP_1^{in} and WTP_1^{out} of in-region and out-of-region households, respectively, for a given change in visibility from Q_{01} to Q_{11} in Southeast parks, we can solve for γ_1 and δ_1 as a function of our estimates of m, WTP_1^{in} and WTP_1^{out}, for any given value of ρ. Generalizing, we can derive the values of γ and δ for the j^{th} region as follows:

$$\gamma_j = \frac{(m - WTP_j^{in})^{\rho} - m^{\rho}}{(Q_{0j}^{\rho} - Q_{1j}^{\rho})}$$

and

$$\delta_j = \frac{(m - WTP_j^{out})^{\rho} - m^{\rho}}{(Q_{0j}^{\rho} - Q_{1j}^{\rho})} \quad .$$

Chestnut and Rowe (1990) and Chestnut (1997) estimated WTP (per household) for specific visibility changes at national parks in three regions of the United States—both for households that are in-region (in the same region as the park) and for households that are out-

[8] Ideally, we would want the location-, time-, and meteorological condition-specific relationships between deciviews and visual range, which could be applied as appropriate. This is probably not feasible, however.

of-region. The Chestnut and Rowe study asked study subjects what they would be willing to pay for each of three visibility improvements in the national parks in a given region. Study subjects were shown a map of the region, with dots indicating the locations of the parks in question. The WTP questions referred to the three visibility improvements in all the parks collectively; the survey did not ask subjects' WTP for these improvements in specific parks individually. Responses were categorized according to whether the respondents lived in the same region as the parks in question ("in-region" respondents) or in a different region ("out-of-region" respondents). The areas for which in-region and out-of-region WTP estimates are available from Chestnut and Rowe (1990), and the sources of benefits transfer-based estimates that we employ in the absence of estimates, are summarized in Table 6.A-1. In all cases, WTP refers to WTP per household.

Table 6.A-1. Available Information on WTP for Visibility Improvements in National Parks

	Region of Household	
Region of Park	**In Region[a]**	**Out of Region[b]**
1. California	WTP estimate from study	WTP estimate from study
2. Colorado Plateau	WTP estimate from study	WTP estimate from study
3. Southeast United States	WTP estimate from study	WTP estimate from study
4. Northwest United States	(based on benefits transfer from California)	
5. Northern Rockies	(based on benefits transfer from Colorado Plateau)	
6. Rest of United States	(based on benefits transfer from Southeast U.S.)	

[a] In-region" WTP is WTP for a visibility improvement in a park in the same region as that in which the household is located. For example, in-region WTP in the "Southeast" row is the estimate of the average Southeast household's WTP for a visibility improvement in a Southeast park.

[b] Out-of-region" WTP is WTP for a visibility improvement in a park that is not in the same region in which the household is located. For example, out-of-region WTP in the "Southeast" row is the estimate of WTP for a visibility improvement in a park in the Southeast by a household outside of the Southeast.

In the primary calculation of visibility benefits for this analysis, only visibility changes at parks within visibility regions for which a WTP estimate was available from Chestnut and Rowe (1990) are considered (for both in- and out-of-region benefits). Primary estimates will not include visibility benefits calculated by transferring WTP values to visibility changes at parks not included in the Chestnut and Rowe study. Transferred benefits at parks located outside of the Chestnut and Rowe visibility regions will, however, be included as an alternative calculation.

The values of the parameters in a household's utility function will depend on where the household is located. The region-specific parameters associated with visibility at Class I areas (that is, all parameters except the residential visibility parameter) are arrayed in Table 6.A-2. The parameters in columns 1 through 3 can be directly estimated using WTP estimates from Chestnut and Rowe (1990) (the columns labeled "Region 1," "Region 2," and "Region 3").

Table 6.A-2. Summary of Region-Specific Recreational Visibility Parameters to be Estimated in Household Utility Functions

	Region of Park					
Region of Household	**Region 1**	**Region 2**	**Region 3**	**Region 4**	**Region 5**	**Region 6**
Region 1	γ_1[a]	δ_2	δ_3	δ_4	δ_5	δ_6
Region 2	δ_1	γ_2	δ_3	δ_4	δ_5	δ_6
Region 3	δ_1	δ_2	γ_3	δ_4	δ_5	δ_6
Region 4	δ_1	δ_2	δ_3	γ_4	δ_5	δ_6
Region 5	δ_1	δ_2	δ_3	δ_4	γ_5	δ_6
Region 6	δ_1	δ_2	δ_3	δ_4	δ_5	γ_6

[a] The parameters arrayed in this table are region-specific rather than park-specific or wilderness area-specific. For example, δ_1 is the parameter associated with visibility at "Class I areas in region 1" for a household in any region other than region 1. The benefits analysis must derive Class I area-specific parameters (e.g., δ_{1k}, for the k[th] Class I area in the first region).

For the three regions covered in Chestnut and Rowe (1990a) (California, the Colorado Plateau, and the Southeast United States), we can directly use the in-region WTP estimates from the study to estimate the parameters in the utility functions corresponding to visibility at in-region parks (γ_1); similarly, we can directly use the out-of-region WTP estimates from the study to estimate the parameters for out-of-region parks (δ_1). For the other three regions not covered in the study, however, we must rely on benefits transfer to estimate the necessary parameters.

While Chestnut and Rowe (1990) provide useful information on households' WTP for visibility improvements in national parks, there are several significant gaps remaining between the information provided in that study and the information necessary for the benefits analysis. First, as noted above, the WTP responses were not park specific, but only region specific. Because visibility improvements vary from one park in a region to another, the benefits analysis must value park-specific visibility changes. Second, not all Class I areas in each of the three regions considered in the study were included on the maps shown to study subjects. Because the focus of the study was primarily national parks, most Class I wilderness areas were not included. Third, only three regions of the United States were included, leaving the three remaining regions without direct WTP estimates.

In addition, Chestnut and Rowe (1990) elicited WTP responses for *three different* visibility changes, rather than a single change. In theory, if the CES utility function accurately describes household preferences, and if all households in a region have the same preference structure, then households' three WTP responses corresponding to the three different visibility changes should all produce the same value of the associated recreational visibility parameter, given a value of ρ and an income, m. In practice, of course, this is not the case.

In addressing these issues, we take a three-phase approach:

1. We estimate region-specific parameters for the region in the modeled domain covered by Chestnut and Rowe (1990a) (California, the Colorado Plateau, and the Southeast)—γ_1, γ_2, and γ_3 and δ_1, $\delta_{2,}$ and δ_3.
2. We infer region-specific parameters for those regions not covered by the Chestnut and Rowe study (the Northwest United States, the Northern Rockies, and the rest of the U.S.)—γ_4, γ_5, and γ_6 and δ_4, δ_5, and $\delta_{6.}$
3. We derive park- and wilderness area-specific parameters within each region (γ_{1k} and δ_{1k}, for k=1, …, N_1; γ_{2k} and δ_{2k}, for k=1, …, N_2; and so forth).

The question that must be addressed in the first phase is how to estimate a single region-specific in-region parameter and a single region-specific out-of-region parameter for each of the three regions covered in Chestnut and Rowe (1990) from study respondents' WTPs for *three different* visibility changes in each region. All parks in a region are treated collectively as if they were a single "regional park" in this first phase. In the second phase, we infer region-specific recreational visibility parameters for regions not covered in the Chestnut and Rowe study (the Northwest United States, the Northern Rockies, and the rest of the United States). As in the first phase, we ignore the necessity to derive park-specific parameters at this phase. Finally, in the third phase, we derive park- and wilderness area-specific parameters for each region.

6.A.5.1 Estimating Region-Specific Recreational Visibility Parameters for the Region Covered in the Chestnut and Rowe Study (Regions 1, 2, and 3)

Given a value of ρ and estimates of m and in-region and out-of-region WTPs for a change from Q_0 to Q_1 in a given region, the in-region parameter, γ, and the out-of-region parameter, δ, for that region can be solved for. Chestnut and Rowe (1990), however, considered not just one, but three visibility changes in each region, each of which results in a different calibrated γ and a different calibrated δ, even though in theory all the γ's should be the same and similarly, all the δ's should be the same. For each region, however, we must have only a single γ and a single δ.

Denoting $\widehat{\gamma}_j$ as our estimate of γ for the j^{th} region, based on all three visibility changes, we chose $\widehat{\gamma}_j$ to best predict the three WTPs observed in the study for the three visibility improvements in the j^{th} region. First, we calculated $\hat{\gamma}_{ji}$, i=1, 2, 3, corresponding to each of the three visibility improvements considered in the study. Then, using a grid search method beginning at the average of the three's $\hat{\gamma}_{ji}$, we chose to minimize the sum of the squared differences between the WTPs we predict using $\widehat{\gamma}_j$ and the three region-specific WTPs observed in the study. That is, we selected to minimize:

$$\sum_{i=1}^{3} (WTP_{ij}(\hat{\gamma}_j) - WTP_{ij})^2$$

where WTP_{ij} and $WTP_{ij}()$ are the observed and the predicted WTPs for a change in visibility in the j^{th} region from $Q_0 = Q_{0i}$ to $Q_1 = Q_{1i}$, i=1, …, 3. An analogous procedure was used to select an optimal δ, for each of the three regions in the Chestnut and Rowe study.

6.A.5.2 Inferring Region-Specific Recreational Visibility Parameters for Regions Not Covered in the Chestnut and Rowe Study (Regions 4, 5, and 6)

One possible approach to estimating region-specific parameters for regions not covered by Chestnut and Rowe (1990a) (γ_4, γ_5, and γ_6 and δ_4, δ_5, and δ_6) is to simply assume that households' utility functions are the same everywhere, and that the environmental goods being valued are the same—e.g., that a change in visibility at national parks in California is the same environmental good to a Californian as a change in visibility at national parks in Minnesota is to a Minnesotan.

For example, to estimate δ_4 in the utility function of a California household, corresponding to visibility at national parks in the Northwest United States, we might assume that out-of-region WTP for a given visibility change at national parks in the Northwest United States is the same as out-of-region WTP for the same visibility change at national parks in California (income held constant). Suppose, for example, that we have an estimated mean WTP of out-of-region households for a visibility change from Q_{01} to Q_{11} at national parks in California (region 1), denoted WTP_1^{out}. Suppose the mean income of the out-of-region subjects in the study was m. We might assume that, for the same change in visibility at national parks in the Northwest United States, $WTP_4^{out} = WTP_1^{out}$ among out-of-region individuals with income m.

We could then derive the value of δ_4, given a value of ρ as follows:

$$\delta_4 = \frac{(m - WTP_4^{out})^\rho - m^\rho}{Q_{04}^\rho - Q_{14}^\rho}$$

where $Q_{04} = Q_{01}$ and $Q_{14} = Q_{11}$, (i.e., where it is *the same* visibility change in parks in region 4 that was valued at parks in the region 1).

This benefits transfer method assumes that (1) all households have the same preference structures and (2) what is being valued in the Northwest United States (by a California household) is the same as what is being valued in the California (by all out-of-region households). While we cannot know the extent to which the first assumption approximates reality, the second assumption is clearly problematic. National parks in one region are likely to differ from national parks in another region in both quality and quantity (i.e., number of parks).

One statistic that is likely to reflect both the quality and quantity of national parks in a region is the average annual visitation rate to the parks in that region. A reasonable way to gauge the extent to which out-of-region people would be willing to pay for visibility changes in parks in the Northwest United States versus in California might be to compare visitation rates in the two regions.[9] Suppose, for example, that twice as many visitor-days are spent in California parks per year as in parks in the Northwest United States per year. This could be an indication that the parks in California are in some way more desirable than those in the Northwest United States and/or that there are more of them—i.e., that the environmental goods being valued in the two regions ("visibility at national parks") are not the same.

A preferable way to estimate δ_4, then, might be to assume the following relationship:

$$\frac{WTP_4^{out}}{WTP_1^{out}} = \frac{n_4}{n_1}$$

(income held constant), where n_1 = the average annual number of visitor-days to California parks and n_4 = the average annual number of visitor-days to parks in the Northwest United States. This implies that

[9] We acknowledge that reliance on visitation rates does not get at nonuse value.

$$WTP_4^{out} = \frac{n_4}{n_1} * WTP_1^{out}$$

for the same change in visibility in region 4 parks among out-of-region individuals with income m. If, for example, $n_1 = 2n_4$, WTP_4^{out} would be half of WTP_1^{out}. The interpretation would be the following: California national parks have twice as many visitor-days per year as national parks in the Northwest United States; therefore they must be twice as desirable/plentiful; therefore, out-of-region people would be willing to pay twice as much for visibility changes in California parks as in parks in the Northwest United States; therefore a Californian would be willing to pay only half as much for a visibility change in national parks in the Northwest United States as an out-of-region individual would be willing to pay for the same visibility change in national parks in California. This adjustment, then, is based on the premise that the environmental goods being valued (by people out of region) are not the same in all regions.

The parameter δ_4 is estimated as shown above, using this adjusted WTP_4^{out}. The same procedure is used to estimate δ_5 and $\delta_{6.}$ We estimate $\gamma_{4,}$ γ, and γ_6 in an analogous way, using the in-region WTP estimates from the transfer regions, e.g.,

$$WTP_4^{in} = \frac{n_4}{n_1} * WTP_1^{in} \quad .$$

6.A.5.3 Estimating Park- and Wilderness Area-Specific Parameters

As noted above, Chestnut and Rowe (1990) estimated WTP for a region's national parks collectively, rather than providing park-specific WTP estimates. The □s and □s are therefore the parameters that would be in household utility functions if there were only a single park in each region, or if the many parks in a region were effectively indistinguishable from one another. Also noted above is the fact that the Chestnut and Rowe study did not include all Class I areas in the regions it covered, focusing primarily on national parks rather than wilderness areas. Most Class I wilderness areas were not represented on the maps shown to study subjects. In California, for example, there are 31 Class I areas, including 6 national parks and 25 wilderness areas. The Chestnut and Rowe study map of California included only 10 of these Class I areas, including all 6 of the national parks. It is unclear whether subjects had in mind "all parks and wilderness areas" when they offered their WTPs for visibility improvements, or whether they had in mind the specific number of (mostly) parks that were shown on the maps. The derivation of park- and wilderness area-specific parameters depends on this.

6.A.5.4 Derivation of Region-Specific WTP for National Parks and Wilderness Areas

If study subjects were lumping all Class I areas together in their minds when giving their WTP responses, then it would be reasonable to allocate that WTP among the specific parks and wilderness areas in the region to derive park- and wilderness area-specific γ's and δ's for the region. If, on the other hand, study subjects were thinking only of the (mostly) parks shown on the map when they gave their WTP response, then there are two possible approaches that could be taken. One approach assumes that households would be willing to pay some additional amount for the same visibility improvement in additional Class I areas that were not shown, and that this additional amount can be estimated using the same benefits transfer approach used to estimate region-specific WTPs in regions not covered by Chestnut and Rowe (1990a).

However, even if we believe that households would be willing to pay some additional amount for the same visibility improvement in additional Class I areas that were not shown, it is open to question whether this additional amount can be estimated using benefits transfer methods. A third possibility, then, is to simply omit wilderness areas from the benefits analysis. For this analysis we calculate visibility benefits assuming that study subjects lumped all Class I areas together when stating their WTP, even if these Class I areas were not present on the map.

6.A.5.5 Derivation of Park- and Wilderness Area-Specific WTPs, Given Region-Specific WTPs for National Parks and Wilderness Areas

The first step in deriving park- and wilderness area-specific parameters is the estimation of park- and wilderness area-specific WTPs. To derive park and wilderness area-specific WTPs, we apportion the region-specific WTP to the specific Class I areas in the region according to each area's share of the region's visitor-days. For example, if WTP_1^{in} and WTP_1^{out} denote the mean household WTPs in the Chestnut and Rowe (1990) study among respondents who were in-region-1 and out-of-region-1, respectively, n_{1k} denotes the annual average number of visitor-days to the kth Class I area in California, and n_1 denotes the annual average number of visitor-days to all Class I areas in California (that are included in the benefits analysis), then we assume that

$$WTP_{1k}^{in} = \frac{n_{1k}}{n_1} * WTP_1^{in} \ ,$$

and

$$WTP_{1k}^{out} = \frac{n_{1k}}{n_1} * WTP_1^{out} \ .$$

Using WTP_j^{in} and WTP_j^{out}, either from the Chestnut and Rowe study (for j = 1, 2, and 3) or derived by the benefits transfer method (for j = 4, 5, and 6), the same method is used to derive Class I area-specific WTPs in each of the six regions.

While this is not a perfect allocation scheme, it is a reasonable scheme, given the limitations of data. Visitors to national parks in the United States are not all from the United States, and certainly not all from the region in which the park is located. A very large proportion of the visitors to Yosemite National Park in California, for example, may come from outside the United States. The above allocation scheme implicitly assumes that the relative frequencies of visits to the parks in a region *from everyone in the world* is a reasonable index of the relative WTP of an average household in that region (WTP_j^{in}) or out of that region (but in the United States) (WTP_j^{out}) for visibility improvements at these parks.[10]

A possible problem with this allocation scheme is that the relative frequency of visits is an indicator of use value but not necessarily of nonuse value, which may be a substantial component of the household's total WTP for a visibility improvement at Class I areas. If park A is twice as popular (i.e., has twice as many visitors per year) as park B, this does not necessarily imply that a household's WTP for an improvement in visibility at park A is twice its WTP for the same improvement at park B. Although an allocation scheme based on relative visitation frequencies has some obvious problems, however, it is still probably the best way to allocate a collective WTP.

6.A.5.6 Derivation of Park- and Wilderness Area-Specific Parameters, Given Park- and Wilderness-Specific WTP

Once the Class I area-specific WTPs have been estimated, we could derive the park- and wilderness area-specific γ's and δ's using the method used to derive region-specific γ's and δ's. Recall that that method involved (1) calibrating γ and δ to each of the three visibility improvements in the Chestnut and Rowe study (producing three γ's and three δ's),
(2) averaging the three γ's and averaging the three δ's, and finally, (3) using these average γ and δ as starting points for a grid search to find the optimal γ and the optimal δ—i.e., the γ and δ

[10] This might be thought of as two assumptions: (1) that the relative frequencies of visits to the parks in a region *from everyone in the world* is a reasonable representation of the relative frequency of visits *from people in the United States*—i.e., that the parks that are most popular (receive the most visitors per year) in general are also the most popular among Americans; and (2) that the relative frequency with which Americans visit each of their parks is a good index of their relative WTPs for visibility improvements at these parks.

that would allow us to reproduce, as closely as possible, the three in-region and three out-of-region WTPs in the study for the three visibility changes being valued.

Going through this procedure for each national park and each wilderness area separately would be very time consuming, however. We therefore used a simpler approach, which produces very close approximations to the γ's and δ's produced using the above approach. If:

- WTP_j^{in} = the in-region WTP for the change in visibility from Q_0 to Q_1 in the j^{th} region;
- WTP_{jk}^{in} = the in-region WTP for the same visibility change (from Q_0 to Q_1) in the k^{th} Class I area in the jth region (= s_{jk}*WTP_j^{in}, where s_{jk} is the k^{th} area's share of visitor-days in the j^{th} region);
- m = income;
- γ_j* = the optimal value of γ for the jth region; and
- γ_{jk} = the value of γ_{jk} calibrated to WTP_{jk}^{in} and the change from Q_0 to Q_1;

then[11]:

$$\gamma_j^* \approx \frac{(m - WTP_j^{in})^{\rho} - m^{\rho}}{(Q_0^{\rho} - Q_1^{\rho})}$$

and

$$\gamma_{jk} = \frac{(m - WTP_{jk}^{in})^{\rho} - m^{\rho}}{(Q_0^{\rho} - Q_1^{\rho})}$$

which implies that:

$$\gamma_{jk} \approx a_{jk} * \gamma_j^* ,$$

where:

$$a_{jk} = \frac{(m - WTP_{jk}^{in})^{\rho} - m^{\rho}}{(m - WTP_j^{in})^{\rho} - m^{\rho}} .$$

[11] γ_j* is only approximately equal to the right-hand side because, although it is the optimal value designed to reproduce as closely as possible all three of the WTPs corresponding to the three visibility changes in the Chestnut and Rowe study, γ_j* will not exactly reproduce any of these WTPs.

We use the adjustment factor, a_{jk}, to derive γ_{jk} from γ_j*, for the k^{th} Class I area in the j^{th} region. We use an analogous procedure to derive δ_{jk} from δ_j* for the k^{th} Class I area in the j^{th} region (where, in this case, we use WTP_j^{out} and WTP_{jk}^{out} instead of WTP_j^{in} and WTP_{jk}^{in}).[12]

6.A.6 Estimating the Parameter for Visibility in Residential Areas: θ

In previous assessments, EPA used a study on residential visibility valuation conducted in 1990 (McClelland et al., 1993). Consistent with advice from EPA's Science Advisory Board (SAB), EPA designated the McClelland et al. study as significantly less reliable for regulatory benefit-cost analysis, although it does provide useful estimates on the order of magnitude of residential visibility benefits (U.S. EPA-SAB, 1999).[13] In order to estimate residential visibility benefits in this analysis, we have replaced the previous methodology with a new benefits transfer approach and incorporated additional valuation studies. This new approach was developed for *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress* (U.S. EPA, 2011) and reviewed by the SAB (U. S. EPA-SAB, 2010). To value residential visibility improvements, the new approach draws upon information from the Brookshire et al. (1979), Loehman et al. (1985) and Tolley et al. (1984) studies.[14] These studies provide primary visibility values for Atlanta, Boston, Chicago, Denver, Los Angeles, Mobile, San Francisco, and Washington D.C.[15]

The estimation of θ is a simpler procedure for residential visibility benefits, involving a straightforward calibration using the study income and WTP for each study city:

$$\theta = \frac{(m - WTP)^{\rho} - m^{\rho}}{(Z_0^{\rho} - Z_1^{\rho})}.$$

---

[12] This method uses a single in-region WTP and a single out-of-region WTP per region. Although the choice of WTP will affect the resulting adjustment factors (the a_{jk}'s) and therefore the resulting γ_{jk}'s and δ_{jk}'s, the effect is negligible. We confirmed this by using each of the three in-region WTPs in California and comparing the resulting three sets of γ_{jk}'s and δ_{jk}'s, which were different from each other by about one one-hundredth of a percent.

[13] EPA's Advisory Council on Clean Air Compliance Analysis noted that the McClelland et al. (1993) study may not incorporate two potentially important adjustments. First, their study does not account for the "warm glow" effect, in which respondents may provide higher willingness to pay estimates simply because they favor "good causes" such as environmental improvement. Second, while the study accounts for non-response bias, it may not employ the best available methods. As a result of these concerns, the Council recommended that residential visibility be omitted from the overall primary benefits estimate. (U.S. EPA-SAB, 1999)

[14] Loehman et al. (1985) and Brookshire et al. (1979) were subsequently published in peer-reviewed journals (see Loehman et al. (1994) and Brookshire et al. (1982). The Tolley et al. (1984) work was not published, but was subject to peer review during study development.

[15] Recognizing potential fundamental issues associated with data collected in Cincinnati and Miami (e.g., see Chestnut et al. (1986) and Chestnut and Rowe (1990c), we do not include values for these cities in our analysis.

where:

- m = household income,
- ρ = shape parameter (0.1),
- θ = WTP parameter corresponding to the visibility at MSA,
- Z_0 = starting visibility, and
- Z_1 = visibility after change.

Where studies provide multiple estimates for visual range improvements for a single study city, we estimate one θ as the simple average of the θ calculated for each set of visual range improvements.

6.A.7 Putting It All Together: The Household Utility and WTP Functions

Given an estimate of θ, derived as shown in Section 6.A.4, and estimates of the γ's and δ's, derived as shown in Section 6.A.3, based on an assumed or estimated value of ρ, the utility and WTP functions for a household in any region are fully specified. We could therefore estimate the value to that household of visibility changes from any baseline level to any alternative level in the household's residential area and/or at any or all of the Class I areas in the United States, in a way that is consistent with economic theory. In particular, the WTP of a household in the i^{th} region and the n^{th} residential area for any set of changes in the levels of visibility at in-region Class I areas, out-of-region Class I areas, and the household's residential area is:

$$WTP_{ni}(\Delta Z, \Delta Q) = m - [\, m^{\rho} + \theta(Z_{0n}^{\rho} - Z_{1n}^{\rho}) + \sum_{k=1}^{N_i} \gamma_{ik}(Q_{0ik}^{\rho} - Q_{1ik}^{\rho}) + \sum_{j \neq i} \sum_{k=1}^{N_j} \delta_{jk}(Q_{0jk}^{\rho} - Q_{1jk}^{\rho})\,]^{1/\rho} \quad .$$

The national benefits associated with any suite of visibility changes would be calculated as the sum of these household WTPs for those changes. The benefit of any subset of visibility changes (e.g., changes in visibility only at Class I areas in California) can be calculated by setting all the other components of the WTP function to zero (that is, by assuming that all other visibility changes that are not of interest are zero). This is effectively the same as assuming that the subset of visibility changes of interest is the first or the only set of changes being valued by households. Estimating benefit components in this way will yield slightly upward biased estimates of benefits, because disposable income, m, is not being reduced by the WTPs for any prior visibility improvements. That is, each visibility improvement (e.g., visibility at Class I areas in the California) is assumed to be the first, and they cannot all be the first. The upward bias should be extremely small, however, because all of the WTPs for visibility changes are very small relative to income.

Although we recognize that the approach described above is most consistent with economic theory, we have chosen to not use this function with income constraints on overall WTP. Instead, we simply add the total preference calibrated recreational visibility benefits to the preference-calibrated residential visibility benefits. Again, because all of the WTPs for visibility changes are very small relative to income, the upward bias should be extremely small.

6.A.8 Income Elasticity and Income Growth Adjustment for Visibility Benefits

Growth in real income over time is an important component of benefits analysis. Economic theory argues that WTP for most goods (such as environmental protection) will increase if real incomes increase. There is substantial empirical evidence that the income elasticity[16] of WTP for health risk reductions is positive, although there is uncertainty about its exact value. Thus, as real income increases, the WTP for environmental improvements also increases. Although many analyses assume that the income elasticity of WTP is unit elastic (i.e., a 10% higher real income level implies a 10% higher WTP to reduce risk changes), empirical evidence suggests that income elasticity is substantially less than one and thus relatively inelastic. As real income rises, the WTP value also rises but at a slower rate than real income.

The effects of real income changes on WTP estimates can influence benefits estimates in two different ways: through real income growth between the year a WTP study was conducted and the year for which benefits are estimated, and through differences in income between study populations and the affected populations at a particular time. Empirical evidence of the effect of real income on WTP gathered to date is based on studies examining the former. The Environmental Economics Advisory Committee (EEAC) of the Science Advisory Board (SAB) advised EPA to adjust WTP for increases in real income over time but not to adjust WTP to account for cross-sectional income differences "because of the sensitivity of making such distinctions, and because of insufficient evidence available at present" (U.S. EPA-SAB, 2000a). A recent advisory by another committee associated with the SAB, the Advisory Council on Clean Air Compliance Analysis, has provided conflicting advice. While agreeing with "the general principle that the willingness to pay to reduce mortality risks is likely to increase with growth in real income (U.S. EPA-SAB, 2004)" and that "The same increase should be assumed for the WTP for serious nonfatal health effects (U.S. EPA-SAB, 2004)," they note that "given the

[16] Income elasticity is a common economic measure equal to the percentage change in WTP for a 1% change in income.

limitations and uncertainties in the available empirical evidence, the Council does not support the use of the proposed adjustments for aggregate income growth as part of the primary analysis (U.S. EPA-SAB, 2004)." Until these conflicting advisories have been reconciled, EPA will continue to adjust valuation estimates to reflect income growth using the methods described below, while providing sensitivity analyses for alternative income growth adjustment factors.

We assume that the WTP for improved visibility would increase with growth in real income. The relative magnitude of the income elasticity of WTP for visibility compared with those for health effects suggests that visibility is not as much of a necessity as health, thus, WTP is more elastic with respect to income.

Details of the general procedure to account for projected growth in real U.S. income between 1990 and 2020 can be found in Kleckner and Neumann (1999). Specifically, we use the elasticity for visibility benefits provided in Chestnut (1997).

In addition to elasticity estimates, projections of real gross domestic product (GDP) and populations from 1990 to 2020 are needed to adjust benefits to reflect real per capita income growth. We used projections of real GDP provided in Kleckner and Neumann (1999) for the years 1990 to 2010.[17] We used projections of real GDP provided by Standard and Poor's (2000) for the years 2010 to 2020.[18] Visibility benefits are adjusted by multiplying the unadjusted benefits by the appropriate adjustment factor.

6.A.9 Summary of Parameters

In Tables 6.A-3 through 6.A-6, we provide the parameters used to calculate recreational and residential visibility benefits.

[17] U.S. Bureau of Economic Analysis, Table 2A (available at http://www.bea.doc.gov/bea/dn/0897nip2/ tab2a.htm.) and U.S. Bureau of Economic Analysis, Economics and Budget Outlook. Note that projections for 2007 to 2010 are based on average GDP growth rates between 1999 and 2007.

[18] In previous analyses, we used the Standard and Poor's projections of GDP directly. This led to an apparent discontinuity in the adjustment factors between 2010 and 2011. We refined the method by applying the relative growth rates for GDP derived from the Standard and Poor's projections to the 2010 projected GDP based on the Bureau of Economic Analysis projections.

Table 6.A-3. Mean Annual Household WTP for Changes in Visual Range for Recreational Visibility (1990$)[a]

Region	WTP In-region	WTP Out-of-region	Starting Visual Range (miles)	Ending Visual Range (miles)	Study Household Income
California	$66.41	$43.85	90	125	
	$80.19	$53.88	90	150	$48,759
	$71.42	$51.37	45	90	
Southwest	$50.12	$45.11	155	200	
	$72.67	$55.13	155	250	$48,759
	$61.40	$48.87	115	155	
Southeast	$66.41	$35.08	25	50	
	$82.70	$53.88	25	75	$48,759
	$75.18	$47.61	10	25	

[a] Based on Chestnut (1997) and adjusted for study sample income and currency year

Table 6.A-4. Region-Specific Parameters for Recreational Visibility Benefits[a]

Region	Optimal γ	Optimal δ
California	0.00517633	0.003629603
Southwest	0.006402706	0.005092572
Southeast	0.003552379	0.002163346
Northwest	0.001172669	0.000823398
Northern Rockies	0.005263445	0.004176339
Rest of U.S.	0.001211215	0.000738149

[a] Calculated using methodology described in sections 6.A.3 through 6.A.4

Table 6.A-5. Mean Annual Household WTP for Changes in Visual Range for Residential Visibility

City	WTP in Original Year's $	Starting Visual Range (miles)	Ending Visual Range (miles)	Study Household Income	Year of Original Data	θ if ρ = 0.1 (1990$, 1990 income)	θ if ρ = 0.1 (Simple Average)
Atlanta (Tolley et al., 1984)	$188	12	22	$19,900[a]	1982	0.033446	0.021316
	$281	12	32	$19,900[a]	1982	0.031661	
	$82	12	22	$27,600[d]	1984	0.010738	
	$119	12	32	$27,600[d]	1984	0.009417	
Boston (Tolley et al., 1984)	$139	18	28	$25,000[a]	1982	0.026636	0.022843
	$171	18	38	$25,000[a]	1982	0.019049	
Chicago (Tolley et al., 1984)	$202	9	18	$30,000[b]	1981	0.022313	0.015480
	$269	9	30	$30,000[b]	1981	0.016696	
	$121	10	20	$29,400[d]	1984	0.013180	
	$144	10	30	$29,400[d]	1984	0.009732	
Denver (Tolley et al., 1984)	$115	50	60	$32,000[c]	1984	0.038558	0.033181
	$154	50	70	$32,000[c]	1984	0.027803	
Los Angeles (Brookshire et al., 1979)	$43	2	12	$15,200[d]	1978	0.003866	0.007428
	$116	2	28	$15,200[d]	1978	0.006716	
	$71	12	28	$15,200[d]	1978	0.011702	
Mobile (Tolley et al., 1984)	$168	10	20	$20,200[a]	1982	0.026078	0.022480
	$197	10	30	$20,200[a]	1982	0.018882	
San Francisco (Loehman et al., 1985)	$71	16.3	18.6	$26,100[c]	1980	0.045307	0.045307
Washington, DC (Tolley et al., 1984)	$238	15	25	$27,500[a]	1982	0.036866	0.032335
	$303	15	35	$27,500[a]	1982	0.027804	

[a] See Chestnut et al. (1986), pages 5-5 through 5-10.

[b] See Tolley et al., (1984), page 127.

[c] See Loehman et al. (1985), page 38.

[d] Historical median income data by MSA from U.S. Census (1990).

Table 6.A-6. Parameters for Income Growth Adjustment for Visibility Benefits

Adjustment Step	Parameter Estimate
Central Estimate of Elasticity [a]	0.90
Adjustment Factor Used to Account for Projected Real Income Growth in 2020[b]	1.517

[a] Derivation of estimates can be found in Kleckner and Neumann (1999) and Chestnut (1997).

[b] Based on elasticity values reported in Table 5-3, U.S. Census population projections, and projections of real GDP per capita.

6.A.10 References

Abt Associates, Inc. 2010. *Environmental Benefits and Mapping Program (Version 4.0).* Bethesda, MD. Prepared for U.S. Environmental Protection Agency Office of Air Quality Planning and Standards. Research Triangle Park, NC. Available on the Internet at <http://www.epa.gov/air/benmap>.

Brookshire, D.S., R.C. d'Arge, W.D. Schulze and M.A. Thayer. 1979. *Methods Development for Assessing Tradeoffs in Environmental Management, Vol. II: Experiments in Valuing Non-Market Goods: A Case Study of Alternative Benefit Measures of Air Pollution Control in the South Coast Air Basin of Southern California*. Prepared for the U.S. Environmental Protection Agency, Office of Research and Development. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0271B-1.pdf/$file/EE-0271B-1.pdf>.

Brookshire, D.S., Thayer, M.A., Schulze, W.D. & D'Arge, R.C. 1982. "Valuing Public Goods: A Comparison of Survey and Hedonic Approaches." *The American Economic Review.* 72(1): 165-177.

Chestnut, L.G. 1997. *Draft Memorandum: Methodology for Estimating Values for Changes in Visibility in National Parks*. Prepared for Office of Policy, Planning and Evaluation, May. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/chestnut1997.pdf>.

Chestnut, L.G., and R.D. Rowe. 1990. "A New National Park Visibility Value Estimates." In *Visibility and Fine Particles, Transactions of an AWMA/EPA International Specialty Conference*, C.V. Mathai, ed. Air and Waste Management Association, Pittsburgh.

Chestnut, L.G., R.D. Rowe and J. Murdoch. 1986. *Review of 'Establishing and Valuing the Effects of Improved Visibility in Eastern United States.'* Prepared for the U.S. Environmental Protection Agency. October. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/chestnut1986.pdf>.

Kleckner, N., and J. Neumann. 1999. *Recommended Approach to Adjusting WTP Estimates to Reflect Changes in Real Income.* Memorandum to Jim Democker, U.S. EPA/OPAR. June. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Benefits/background/klecknerandneumann1999.pdf>.

Loehman, E.T., D. Boldt, K. Chaikin. 1985. *Measuring the Benefits of Air Quality* Improvements *in the San Francisco Bay Area*. From Methods Development for Environmental Control Benefits Assessment, Volume IV. Prepared for the U.S. Environmental Protection Agency, Office of Policy, Planning and Evaluation, September. Grant #R805059-01-0 Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwGA/e85705387833398f8525644d0053bdd0!OpenDocument#_Section6>.

Loehman, E.T., S. Park, and D. Boldt. 1994. “Willingness to Pay for Gains and Losses in Visibility and Health.” *Land Economics* 70(4): 478-498.

McClelland, G., W. Schulze, D. Waldman, J. Irwin, D. Schenk, T. Stewart, L. Deck and M. Thayer. 1993. *Valuing Eastern Visibility: A Field Test of the Contingent Valuation Method*. Prepared for U.S. Environmental Protection Agency, Office of Policy, Planning and Evaluation. September. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0008-1.pdf/$file/EE-0008-1.pdf >.

Pitchford, M.L., and W.C. Malm. 1994. “Development and Applications of a Standard Visual Index.” *Atmospheric Environment* 28(5):1049-1054.

Sisler, J.F. 1996. *Spatial and Seasonal Patterns and Long Term Variability of the Composition of the Haze in the United States: An Analysis of Data from the IMPROVE Network*. Colorado State University, Cooperative Institute for Research in the Atmosphere (CIRA), ISSN 0737-5352-32. Fort Collins, CO. July. Available on the Internet at <http://vista.cira.colostate.edu/improve/Publications/Reports/1996/PDF/TBLCON1.96.pdf>.

Smith, V.K., G. Van Houtven, and S.K. Pattanayak. 2002. “Benefit Transfer via Preference Calibration.” *Land Economics* 78:132-152.

Tolley, G., A. Randall, G. Blomquist, M. Brien, R. Fabian, G. Fishelson, A. Frankel, M. Grenchik, J. Hoehn, A. Kelly, R. Krumm, E. Mensah, and T. Smith. 1984. *Establishing and Valuing the Effects of Improved Visibility in Eastern United States*. Prepared for U.S. Environmental Protection Agency, Office of Policy, Planning and Evaluation. March. U.S. Environmental Protection Agency Grant #807768-01-0. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0003-01.pdf/$file/EE-0003-01.pdf>.

U.S. Census. 1990. “Table 1: Median Household Income by Metropolitan Statistical Areas (MSA): 1969, 1979, and 1989.” 1970, 1980, and 1990 Censuses of Population and Housing. Income Statistics Branch/HHES Division. Available on the Internet at <http://www.census.gov/hhes/www/income/data/historical/metro/msa1.html>.

U.S. Environmental Protection Agency—Science Advisory Board (U.S. EPA-SAB). 1999*. The Clean Air Act Amendments (CAAA) Section 812 Prospective Study of Costs and Benefits (1999): Advisory by the Advisory Council on Clean Air Compliance Analysis: Costs and Benefits of the CAAA.* EPA-SAB-COUNCIL-ADV-00-002. October. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/E992F918DF32131A852571B9004571F7/$File/coua0002.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2011b*. The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress.* Office of Air and Radiation, Washington, DC. March. Available on the Internet at <http://www.epa.gov/air/sect812/feb11/fullreport.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2000. *Guidelines for Preparing Economic Analyses*. EPA 240-R-00-003. National Center for Environmental Economics, Office of Policy Economics and Innovation. Washington, DC. September. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eed.nsf/webpages/Guidelines.html/$file/cover.pdf>.

APPENDIX 6.B
ILLUSTRATIVE SCENARIO OF RECREATIONAL AND RESIDENTIAL VISIBILITY BENEFITS

6.B.1 Synopsis

In this Appendix, we provide an illustrative example analysis of visibility benefits that applies the visibility benefits methodology described in Chapter 6 and Appendix 6A to a specific modeled scenario. For this illustrative example, we use the 2020 base case simulation and the 2020 control case simulation from the CMAQ model that were used to develop the air quality ratios.[1] In this Appendix, we refer to this specific scenario as the "illustrative scenario," which is not a surrogate of the revised annual primary standard. Because we do not have air quality model runs for the regulatory baseline and the revised or alternative annual standards, we cannot that calculate the visibility co-benefits of attaining the revised annual primary standard. It is important to emphasize that this illustrative scenario does not reflect an emissions control strategy for any specific annual standard level, which is important because light extinction can vary substantially depending on the specific combination of SO_2, NO_X, or directly emitted particles reduced and the magnitude and location of those emission reductions. In addition, the visibility benefits in this chapter cannot be added to the health benefits of the revised or alternative standards. We provide this illustrative scenario to demonstrate the results of applying the methodology for estimating benefits for scenarios with light extinction estimates.

6.B.2 Recreational Benefits Results

The modeling results indicate that visibility would improve in several Class I areas as a result of emission reductions in the illustrative scenario. While we are unable to calculate the specific contribution in this analysis, the emission reductions associated with these emission reductions would help Class 1 areas to meet the goals of the Regional Haze rule. Table 6.B-1 identifies the visibility improvements in the 10 most visited parks for the illustrative scenario using two light extinction metrics: visual range and deciview. The monetized benefits of recreational visibility improvements are provided in Table 6.B-2 for the illustrative scenario.

Because there is considerable uncertainty about the accuracy of the benefit transfer to other regions, we include the estimated recreational visibility benefits for parks in other regions as a sensitivity analysis only. The sensitivity analysis results are not considered part of the total monetized benefits. Table 6.B-3 provides the results of this sensitivity analysis. Figure 6.B-1

[1] See Chapter 4 of this RIA for more information regarding the specific scenario in these modeling simulations including the magnitude, location, and type of emission reductions. See Chapter 3 for more information regarding how these modeling simulations were used to calculate the air quality ratios.

shows how the monetized benefits for recreational visibility are distributed across Class I areas for the illustrative scenario. This sensitivity analysis shows that the benefits in non-studied park regions could be substantial. In addition, in Table 6.B.4, we provide an indication of the potential impact of omitting coarse particles from the light extinction calculation using surrogate coarse particle concentrations from Debell et al. (2006).[2]

Table 6.B-1. Annual Average Visibility Improvements in the Top 10 Most Visited Class I Areas for the Illustrative Scenario in 2020[a,b]

		Illustrative Scenario	
Class I Area	**State**	**Visual Range (m)**	**Deciviews**
Grand Canyon NP	AZ	-	-
Great Smoky Mountains NP	NC/TN	-	-
Yellowstone NP [c]	WY	-	-
Yosemite NP	CA	400	0.1
Sequoia-Kings NP	CA	990	0.3
Glacier NP [c]	MT	-	-
Rocky Mountain NP	CO	-	-
Zion NP	UT	-	-
Grand Teton NP[c]	WY	-	-
Kings Canyon NP	CA	470	0.1

[a] Because these benefits occur within the analysis year, the monetized benefits are the same for all discount rates. Although the light extinction estimates do not reflect coarse particles, the rounded incremental visibility benefits are unaffected.

[b] Visibility measured at the county of the geographic center of park. Top 10 most visited parks ranked by visitor days in 2008 (NPS, 2008).

[c] Not included in the primary benefits because the parks are not located in the studied regions. Benefits for these parks are included in the sensitivity analysis.

[2] See Chapter 6 (Section 6.3.1) for more information regarding the purpose of this sensitivity analysis.

Table 6.B-2. Recreational Visibility Benefits in Studied Regions for the Illustrative Scenario in 2020 (in millions of 2010$)[a]

Studied Park Region	Illustrative Scenario Benefits
Southeast	$4.3
Southwest	-
California	$350
Total	**$350**

[a]Because these benefits occur within the analysis year, the monetized benefits are the same for all discount rates. These benefits reflect the WTP for all U.S. households for parks in these regions. Although the light extinction estimates do not reflect coarse particles, the rounded incremental visibility benefits are unaffected.

Table 6.B-3. Sensitivity Analysis for Recreational Visibility Benefits outside Studied Park Regions for the Illustrative Scenario in 2020 (in millions of 2010$)[a]

Non-Studied Park Region	Illustrative Scenario Benefits
Northwest	-
Northern Rockies	-
Rest of U.S.	-
Total for Non-Studied Parks Regions	-
Total including Studied Park Regions	$350

[a]Because these benefits occur within the analysis year, the monetized benefits are the same for all discount rates. These benefits reflect the WTP for all U.S. households for parks in these regions. These sensitivity analysis results are not considered part of the total monetized benefits. Although the light extinction estimates do not reflect coarse particles, the rounded incremental visibility benefits are unaffected.

Table 6.B-4. Sensitivity Analysis for Incorporating Coarse Particles into Recreational Visibility Benefits for the Illustrative Scenario (in millions of 2010$)[a]

		Illustrative Scenario Benefits
Primary Recreational Benefits (Studied Park Regions)	No coarse particles	$350
	+ 5μg/m³ coarse	$330
	+ 5 μg/m³ coarse with 15 μg/m³ in Southwest	$300
	+8 μg/m³ coarse with 15 μg/m³ in Southwest	$300
Sensitivity Analysis (Other Park Regions)	No coarse particles	-
	+ 5μg/m³ coarse	-
	+ 5 μg/m³ coarse with 15 μg/m³ in Southwest	-
	+8 μg/m³ coarse with 15 μg/m³ in Southwest	-

[a] Because these benefits occur within the analysis year, the monetized benefits are the same for all discount rates. These benefits reflect the WTP for U.S. households who live in these regions for the parks in the study regions (primary benefits) or parks in other regions (sensitivity analysis). The levels of coarse particles represent the full range of possible annual concentrations from a recent report on the IMPROVE monitoring network (Debell et al., 2006). We define Southwest to be the states of California, Nevada, Utah, Arizona, New Mexico, Colorado, and Texas.

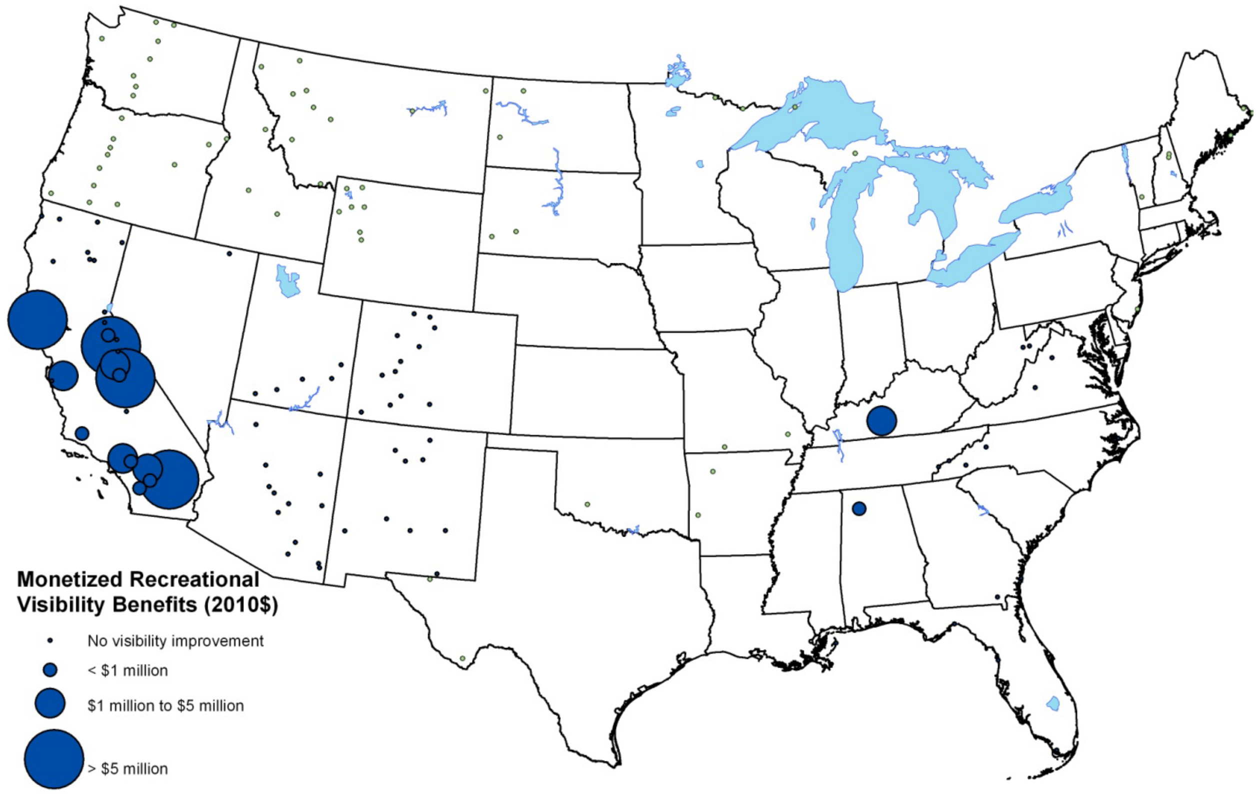


Figure 6.B-1. Recreational Visibility Benefits in Class I Areas for the Illustrative Scenario in 2020 [a]

[a]The size of the circle in this map indicates the magnitude of the recreational benefits for each Class 1 Area. The colors in this map indicate whether the park benefits are included in the primary benefits or in the sensitivity analysis (i.e., non-studied park regions). Blue = primary benefits (studied park regions), Green = sensitivity analysis (non-studied park regions).

6.B.3 Residential Benefits Results

The modeling results indicate that visibility would improve in many of the study areas as a result of the emission reductions associated with emission reductions in the illustrative scenario. Table 6.B-5 shows the monetized residential visibility benefits in the eight study areas. These benefits reflect the value to households living within those MSAs, accounts for inflation, and account for growth in real income since the original WTP estimates were developed.

The results of the sensitivity analysis for the monetized residential visibility benefits in all MSAs for the illustrative scenario are provided in Table 6.B-6. Figure 6.B-2 shows how the sensitivity analysis results are distributed geographically for the illustrative scenario. In addition, Table 6.B-7 provides an indication of the potential impact of omitting coarse particles in the calculation of light extinction.

Table 6.B-5. Monetized Residential Visibility Benefits in Studied Areas in 2020 for the Illustrative Scenario (millions of 2010$, 2020 income)[a]

Study Area	Illustrative Scenario
Atlanta	-
Boston	-
Chicago	$43
Denver	-
Los Angeles	$110
Mobile	-
San Francisco	$39
Washington, DC	-
Total	**$190**

[a]Because these benefits occur within the analysis year, the monetized benefits are the same for all discount rates. Although the light extinction estimates do not reflect coarse particles, the rounded incremental visibility benefits are unaffected.

Table 6.B-6. Sensitivity Analysis for Monetized Residential Visibility Benefits in Other Areas for the Illustrative Scenario in 2020 (in millions of 2010$)[a]

Extent of Benefit Transfer	Illustrative Scenario Benefits
Additional MSAs in East	$140
Additional MSAs in West	$24
Additional MSAs in California	$160
Total	**$330**
Total including Study Areas	**$520**

[a]Because these benefits occur within the analysis year, the monetized benefits are the same for all discount rates. These sensitivity analysis results are not considered part of the total monetized benefits. Although the light extinction estimates do not reflect coarse particles, the rounded incremental visibility benefits are unaffected.

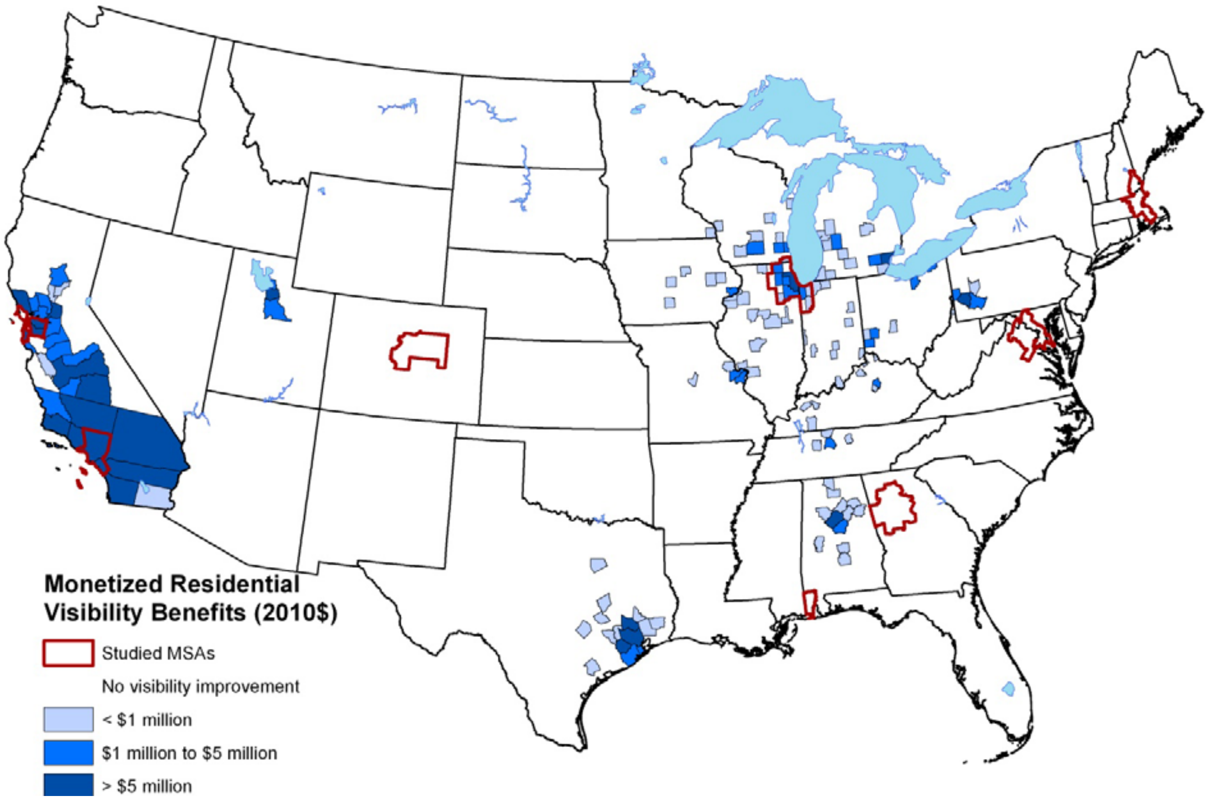


Figure 6.B-2. Residential Visibility Benefits for the Illustrative Scenario in 2020 (2010$)

Table 6.B-7. Sensitivity Analysis for Incorporating Coarse Particles into Residential Visibility Benefits (in millions of 2010$)[a]

		Illustrative Scenario Benefits
Primary Residential Benefits	No coarse particles	$190
	+ 5 μg/m^3 coarse	$180
	+ 5 μg/m^3 coarse with 15 μg/m^3 in Southwest	$150
	+8 μg/m^3 coarse with 15 μg/m^3 in Southwest	$130
Sensitivity Analysis	No coarse particles	$330
	+ 5 μg/m^3 coarse	$310
	+ 5 μg/m^3 coarse with 15 μg/m^3 in Southwest	$250
	+8 μg/m^3 coarse with 15 μg/m^3 in Southwest	$270

[a]Because these benefits occur within the analysis year, the monetized benefits are the same for all discount rates. These benefits reflect the WTP for U.S. households who live in these regions for the parks in the study regions (primary benefits) or parks in other regions (sensitivity analysis). The levels of coarse particles represent the full range of possible annual concentrations from a recent report on the IMPROVE monitoring network (Debell et al., 2006). We define Southwest to be the states of California, Nevada, Utah, Arizona, New Mexico, Colorado, and Texas.

6.B.4 References

DeBell, L. J.; Gebhart, K. A.; Hand, J. L.; Malm, W. C.; Pitchford, M. L.; Schichtel, B. A.; White, W. H. 2006. *Spatial and Seasonal Patterns and Temporal Variability of Haze and its Constituents in the United States, Report IV*; Cooperative Institute for Research in the Atmosphere, Nov; ISSN 0737-5352-74. Available on the Internet at <http://vista.cira.colostate.edu/improve/Publications/Reports/2006/2006.htm>.

National Park Service (NPS). 2008. *Statistical Abstract: 2008.* Department of Interior, National Park Service Social Science Program, Public Use Statistics Office, Denver, Colorado. Available on the Internet at <http://www.nature.nps.gov/stats/abstracts/abst2008.pdf>.

CHAPTER 7
ENGINEERING COST ANALYSIS

7.1 Synopsis

This chapter summarizes the data sources and methodology used to estimate the engineering costs of attaining the revised annual standard and two alternative annual standards for the $PM_{2.5}$ primary standard analyzed in this RIA. This chapter provides the estimates of the annual engineering costs for illustrative control strategies designed to demonstrate attainment of the revised annual standard of 12 µg/m^3 in conjunction with retaining the 24-hour standard of 35 µg/$m^{3,}$ as well as control strategies designed to demonstrate attainment of the alternative annual standards of 13 and 11 µg/m^3 in conjunction with retaining the 24-hour standard of 35 µg/m^3 (referred to as 12, 13, and 11). These illustrative control strategies are outlined in Chapter 4. The cost discussion for known controls in Section 7.2.2 is followed by a presentation of estimates for the engineering costs of the additional emissions reductions that are needed beyond the application of known controls to reach full attainment of the alternative standards analyzed; the cost estimates derived from this approach, discussed in Section 7.2.3, are referred to as "extrapolated" costs. By definition, no cost data currently exists for the additional emissions reductions needed beyond known controls. We employ two methodologies for estimating the costs of unidentified future controls, and both approaches assume either that existing technologies can be applied in particular combinations or to specific sources that we currently can't predict or that innovative strategies and new control options make possible the emissions reductions needed for attainment by 2020.

The engineering costs described in this chapter generally include the costs of purchasing, installing, operating, and maintaining the referenced control technologies. For a variety of reasons, actual control costs may vary from the estimates EPA presents. As discussed throughout this document, the technologies and control strategies selected for analysis are illustrative of one way in which nonattainment areas could meet a revised standard. There are numerous ways to construct and evaluate potential control programs that would bring areas into attainment with the revised and alternative standards, and EPA anticipates that local and state governments will consider programs that are best suited for local and regional conditions. Furthermore, based on past experience, EPA believes that it is reasonable to anticipate that the marginal cost of control will decline over time due to technological improvements and more widespread adoption of previously considered niche control technologies. Also, EPA recognizes the extrapolated portion of the engineering cost estimates are uncertain because extrapolated costs do not contain information about which sectors may be affected or which control measures may be employed in the future.

The engineering cost estimates are limited in their scope. This analysis focuses on the emissions reductions needed for attainment of the revised and alternative standards that are described earlier in this RIA. EPA understands that some states will incur costs designing State Implementation Plans (SIPs) and implementing new control strategies to meet the revised standard. However, EPA does not know what specific actions states will take to design their SIPs to meet the revised standards; therefore we do not present estimated costs that government agencies may incur for managing the requirement, implementing these (or other) control strategies, or for offering incentives that may be necessary to encourage the implementation of specific technologies, especially for technologies that are not necessarily market driven. EPA generally estimates state-level administrative costs in an information collection request (ICR) that accompanies the implementation rule or guidance for each NAAQS (as opposed to accompanying the issuance of the NAAQS)

7.2 Estimation of Engineering Control Costs

7.2.1 Data and Methods—Identified Control Costs (non-EGU Point and Area Sources)

After designing the hypothetical control strategy using the methodology discussed in Chapter 4, EPA used the Control Strategy Tool[1] (CoST) to estimate engineering control costs for mobile, non-EGU point and area sources.[2] CoST calculates engineering costs using either: (1) an average annualized cost-per-ton estimate multiplied by the total tons of a pollutant reduced to derive a total cost estimate, or (2) an equation that incorporates key emission source information (e.g., unit capacity and stack flow information).[3] Most control cost information within CoST was developed based on the cost-per-ton approach because estimating engineering costs using an equation requires more data, and these data are sometimes, but not always, available.

The cost equations located in CoST require either unit capacity or stack flow to determine annualized, capital and/or operating and maintenance (O&M) costs. Capital costs

[1] The Control Strategy Tool recently underwent peer review by an ad hoc panel of experts. Responses to the peer review are currently being developed and will be available by final promulgation of this rule.

[2] Area sources are not necessarily non-urban sources.

[3] Annualized costs represent an equal stream of yearly costs over the period the control technology is expected to operate.

are converted to annualized costs using the capital recovery factor (CRF).[4] When the cost equations and input data are available in CoST, the equations are used to calculate total annual control cost (TACC), which is a function of capital costs (CC) and O&M costs. The CRF incorporates the interest rate and equipment life (in years) of the control equipment. Operating costs are calculated as a function of annual O&M and other variable costs. The resulting TACC equation is TACC = (CRF * CC) + O&M.

Engineering costs will differ between different emissions sources based upon quantity of emissions reduced, plant capacity, or stack flow. Engineering costs will also differ in nominal (i.e., not adjusted for inflation) terms by the year the costs are calculated for (i.e., 1999$ versus 2010$).[5] For capital investment, in order to attain standards in 2020 we assume capital investment occurs at the beginning of 2020. We make this simplifying assumption because we do not know what all firms making capital investments will do and when they will do it. For 2020, our estimate of annualized costs includes annualized capital and O&M costs for those controls included in our identified control strategy analysis. Our engineering cost analysis uses the equivalent uniform annual costs (EUAC) method, in which annualized costs are calculated based on the equipment life for the control measure along with the interest rate of 7% incorporated into the CRF. We make no presumption of additional capital investment in years beyond 2020. The EUAC method is discussed in detail in the EPA Air Pollution Control Cost Manual.[6] Applied controls and their respective engineering costs are provided in the PM NAAQS docket.

7.2.2 Identified Control Costs

In this section, we provide engineering cost estimates for the control strategies identified in Chapter 4 that include control technologies on non-EGU point sources and area sources. Engineering costs generally refer to the capital equipment expense, the site preparation costs for the application, and annual operating and maintenance costs. For this analysis, we included known controls for all of the geographic areas likely to exceed the revised

---

[4] The capital recovery factor formula is expressed as [r*(1+r)^n/(1+r)^n -1]*CC. Where *r* is the real rate of interest, *n* is the number of time periods, and *CC* is the capital cost. For more information on this cost methodology and CoST, please refer to the documentation at http://www.epa.gov/ttn/ecas/cost.htm, the EPA Air Pollution Control Cost Manual found at http://epa.gov/ttn/catc/products.html#cccinfo, and EPA's Guidelines for Preparing Economic Analyses, Chapter 6 found at http://yosemite.epa.gov/ee/epa/eed.nsf/webpages/Guidelines.html#download.

[5] The engineering costs will not be any different in real (inflation-adjusted) terms if calculated in 2010 versus other-year dollars, if the other-year dollars are properly adjusted. For this analysis, all costs are reported in real 2010 dollars.

[6] http://epa.gov/ttn/catc/products.html#cccinfo

and/or alternative standards. We included all known controls at an annual cost of $20,000 per ton or less, which included approximately 85% of known controls in the geographic areas likely to exceed the revised and/or alternative standards.[7] We did not include the small number of known controls that had an annual cost of more than $20,000 per ton because either (i) the remaining emissions sources were relatively small sources, and we believe they are already controlled, or (ii) the equations in CoST were not applicable to these remaining emissions sources.

Because we obtain control cost data from many sources, we are not always able to obtain consistent data across original data sources.[8] If disaggregated control cost data is unavailable (i.e., where capital, equipment life value, and O&M costs are not separated out), EPA typically assumes that the estimated control costs are annualized using a 7% discount rate because the majority of the available disaggregated control cost data is calculated using 7%. When disaggregated control cost data is available (i.e., where capital, equipment life value, and O&M costs are explicit) we can recalculate costs using a 3% discount rate. The use of these two discount rates for cost estimation reflects the guidance for RIA preparation found in Circular A-4, issued by OMB in September 2003.[9] In general, we have some disaggregated data available for non-EGU point source controls; we do not have any disaggregated control cost data for area source controls.[10] In this analysis, for the revised annual standard of 12 $\mu g/m^3$ and the alternative standard of 13 $\mu g/m^3$ we did not have any disaggregated known control cost data, and as such we were not able to recalculate known control costs using a 3% discount rate. Because we were not able to recalculate known controls costs using a 3% discount rate, we are not presenting known controls costs for the revised or alternative standards using that discount rate. See Table 7-1 for a summary of sectors and known control costs.

[7] For the **known controls**, for all of the geographic areas likely to exceed the revised and/or alternative standards we include controls at an annual cost of $20,000 per ton or less. To estimate the costs associated with unidentified future controls, or **unknown controls**, we employ a fixed-cost and hybrid methodology. The fixed-cost methodology employs a primary cost estimate of $15,000/ton (2010 dollars), and the hybrid methodology employs an initial, annual cost-per-ton estimate of $15,000/ton (2010 dollars). We explain the choices of these parameters in this Section and Section 7.2.3.

[8] Data sources can include states, as well as technical studies, which do not typically include references to the original data source.

[9] U.S. Office of Management and Budget. Circular A-4, September 17, 2003. Available at http://www.whitehouse.gov/omb/circulars_a004_a-4/.

[10] For area source controls, total annualized costs are assumed to be calculated using a 7% discount rate.

Table 7-1. Summary of Sectors, Emissions Reductions, and Known Annualized Control Costs (millions of 2010$)[a, b]

Revised and Alternative Standard	Emissions Sector	Emissions Reductions[c]	7% Discount Rate
13 µg/m^3	Non-EGU Point Sources	--	—
	Area Sources	53	$0.63
	Total	53	$0.63
12 µg/m^3	Non-EGU Point Sources	380	$0.87
	Area Sources	430	$4.3
	Total	**800**	**$5.1**
11 µg/m^3	Non-EGU Point Sources	23,000	$88
	Area Sources	2,300	$13
	Total	**25,400**	**$100**

[a] All estimates rounded to two significant figures. Estimates may not sum due to rounding convention.
[b] The emissions reductions and total costs are associated with partial attainment.
[c] The emissions reductions for the alternative standard of 11 include $PM_{2.5}$ and SO_2 emissions reductions.

The total annualized cost of control in each sector in the control scenario is summarized by region in Table 7-2. Table 7-2 includes annualized control costs to allow for comparison across regions and between costs and benefits. These numbers reflect the engineering costs annualized at a discount rate of 7%. Engineering cost estimates presented throughout this and subsequent chapters are based on a 7% discount rate.

Table 7-2. Partial Attainment Known Annualized Control Costs in 2020 for Revised and Alternative Standards Analyzed (millions of 2010$)[a,b]

Revised & Alternative Standard	Region	Known Controls
13 µg/m^3	East	—
	West	—
	California	$0.63
	Total	$0.63
12 µg/m^3	East	—
	West	—
	California	$5.1

	Total	$5.1
11 $\mu g/m^3$	East	$96
	West	$0.45
	California	$5.3
	Total	$100

[a] Estimates are rounded to two significant figures. As such, numbers may not sum down columns.

[b] Note that the estimates provided reflect incremental emissions reductions from an analytical baseline that gives "credit" to the San Joaquin Valley and South Coast areas for emissions reductions expected to occur between 2020 and 2025 (when those areas are expected to demonstrate attainment with the revised and/or alternative standards).

The total annualized engineering costs associated with the application of known controls, incremental to the baseline and using a 7% discount rate, are approximately $5.1 million for the revised annual standard of 12, $630,000 for the less stringent alternative annual standard of 13 $\mu g/m^3$, and $100 million for a more stringent alternative annual standard of 11 $\mu g/m^3$.

7.2.3 Extrapolated Costs

This section presents the methodology and results of the extrapolated engineering cost calculations for attainment of the revised annual $PM_{2.5}$ standard of 12 $\mu g/m^3$, as well as alternative annual standards of 13 $\mu g/m^3$and 11 $\mu g/m^3$. All costs presented for the illustrative control strategies are calculated incrementally from the current $PM_{2.5}$ standard of 15/35 $\mu g/m^3$, therefore, any additional emission reductions needed to attain the current 24-hour standard of 35 $\mu g/m^3$ are part of the baseline analysis and not presented here.

As mentioned earlier in this chapter, and as described in more detail in Chapter 4, the application of the known control strategy was not successful in reaching attainment for all areas for these alternative $PM_{2.5}$ standards. Because some areas remained in nonattainment, the engineering costs detailed in Section 7.2.2 represent the costs of partial attainment for the revised annual standard of 12 $\mu g/m^3$ and alternative annual $PM_{2.5}$ standards of 13 $\mu g/m^3$ and 11 $\mu g/m^3$. For the revised standard and each alternative standard and geographic area that cannot reach attainment with known controls, we estimated the additional emissions reductions needed for $PM_{2.5}$ to attain the standard. See Chapter 3, Section 3.3.2 and Tables 3-7, 3-8, and 3-9 for a detailed discussion of how the additional, needed emissions reductions were estimated and for a summary of the needed emissions reductions for the revised annual standard and the alternative annual standards.

To generate estimates of the costs and benefits of meeting the revised and alternative standards, in addition to the application of known controls EPA assumes the application of unidentified future controls that make possible the additional emission reductions needed for attainment in 2020. By definition, no cost data currently exists for unidentified future technologies or innovative strategies. EPA used two methodologies for estimating the costs of unidentified future controls: a fixed-cost methodology and a hybrid methodology. Both approaches assume either that existing technologies can be applied in particular combinations or to specific sources that we currently can't predict or that innovative strategies and new control options make possible the emissions reductions needed for attainment by 2020. However, the two approaches reflect different assumptions about technological progress and innovation in emissions reductions strategies.

7.2.3.1 Fixed-Cost Methodology

The fixed-cost methodology uses a $15,000/ton estimate for each ton of $PM_{2.5}$ reduced; the hybrid methodology is similar to the hybrid methodology used for the 2008 Ozone NAAQS RIA cost analysis and is presented in more detail below. The fixed-cost methodology was preferred by EPA's Science Advisory Board over two other options, including a marginal-cost-based methodology. When reviewing the Office of Air and Radiation's Direct Cost Report and Uncertainty Analysis Plan, the Science Advisory Board noted:

> *When assigning costs to unidentified measures, the Council suggests that a simple, transparent method that is sensitive to the degree of uncertainty about these costs is best. Of the three approaches outlined, assuming a fixed cost/ton appears to be the simplest and most straightforward. Uncertainty might be represented using alternative fixed costs per ton of emissions avoided.*

Note that the choice of $15,000/ton for the fixed-cost methodology was based on the precedent set in the March 2011 final report *The Benefits and Costs of the Clean Air Act from 1990 to 2020.*[11, 12] We also chose $15,000/ton for the national, initial cost-per-ton for use in the

[11] We considered adjusting the $15,000/ton value and reviewed data on inflation between 2006 and 2010. We found that during that period inflation was sufficiently low to not warrant a $/ton value adjustment; any such adjustment would be considered well within the bounds of uncertainty in this analysis. To assess the sensitivity of the results to the value of $15,000/ton, we also include sensitivity analyses at $10,000/ton and $20,000/ton in Appendix 7A. In addition, the extrapolated costs do not rely on specific underlying data sources (e.g., Census data) that periodically change and that would require updating based on those changes. As such, we currently do not have either specific data or a particular rationale to change the $15,000/ton value.

[12] The final report *The Benefits and Costs of the Clean Air Act from 1990 to 2020* includes the following discussion for the rationale for the $15,000 per ton threshold. Controls that are more costly than $15,000 per ton may not be cost effective, and local air quality agencies would likely seek reductions from other unidentified control measures. This is consistent with the practice of the South Coast Air Quality Management District, which

hybrid methodology to facilitate direct comparison with the fixed-cost methodology. [13] In addition, we do not have reason to conclude that the initial cost-per-ton used in the hybrid methodology should be different than the value used in the fixed-cost methodology. In the proposal RIA, we requested comments or suggestions on methodologies for estimating the costs of unspecified future controls to provide illustrative estimates of NAAQS costs. We did not receive any direct comments on methodologies, but we did receive comments from the San Joaquin Valley Air Pollution Control District (SJV APCD) on the magnitude of their potential investments needed to meet the revised annual standard of 12 μg/m^3 relative to our total cost estimates. The SJV APCD commented that expenditures in their jurisdiction alone could likely be more than our total cost estimate. Their comment provides additional context for the need to improve existing or identify new methodologies for estimating the costs of unspecified future controls.

7.2.3.2 Hybrid Cost Methodology

The hybrid methodology generates a total annual cost curve for $PM_{2.5}$ for unknown future controls that might be applied in order to move toward 2020 attainment. The hybrid methodology has the advantage of incorporating information about how significant the needed reductions from unspecified control technologies are relative to the known control measures and matching that information with expected increasing per-ton cost for applying unknown controls.[14] For $PM_{2.5}$ reductions needed in each area, the cost begins with a national constant cost-per-ton for $PM_{2.5}$ and increases as emissions reduction for $PM_{2.5}$ are needed, reflecting the expectation that average per-ton control costs are likely to be higher in areas needing a higher ratio of emission reductions from unknown to known controls. For example, to attain a revised annual standard of 12 μg/m^3, all of the needed emissions reductions for Los Angeles County were from known controls at an average cost of $6,000 per ton; whereas for Riverside County approximately 95 percent of the needed emissions reductions were from unknown future controls at an average cost of $290,000 per ton. For other geographical areas, the average cost

attempts to identify viable alternatives for any control requirements with an estimated cost exceeding $16,500 per ton. When costs are above this threshold, the South Coast Air Quality Management District conducts more detailed cost-effectiveness and economic impact analyses of the controls.

[13] For the **known controls**, for all of the geographic areas likely to exceed the revised and/or alternative standards we include controls at an annual cost of $20,000 per ton or less. To estimate the costs associated with unidentified future controls, or **unknown controls**, we employ a fixed-cost and hybrid methodology. The fixed-cost methodology employs a primary cost estimate of $15,000/ton (2010 dollars), and the hybrid methodology employs an initial, annual cost-per-ton estimate of $15,000/ton (2010 dollars). We explain the choices of these parameters in this Section and Section 7.2.2.

[14] In applying the hybrid methodology, EPA reviewed the data to ensure that the estimated, additional emissions reductions selected for each geographic area were not greater than the remaining uncontrolled emissions in that geographic area. The highest percent selected was 90%.

per ton for unknown controls ranged from \$19,000 to \$28,000 per ton. The incremental improvement in air quality for an unknown control is determined using an area-by-area ratio of air quality improvement to air quality change, which is discussed in more detail in Chapter 3.

EPA developed a model of increasing total annualized costs for controlling $PM_{2.5}$ emissions. The simplest form of $ax^2 + bx + c$ was used where x is the tons of a particular pollutant to be reduced in a particular area and *a, b*, and c are constants. For the hybrid methodology *b* is set to be a national, initial cost-per-ton (N) for unknown controls for $PM_{2.5}$, and c is set to zero because there is no cost to imposing no control. The hybrid methodology has a different *a* for $PM_{2.5}$ for each geographic area. For a particular geographic area *a* is *N/E* where

N	=	national, initial annualized cost/ton (b from above) of \$15,000 per ton.
E	=	by geographic area, is the denominator and represents all particulate emission reductions achieved (from applying known and unknown controls to obtain the 15/35 baseline, as well as known controls to achieve the alternative standard) prior to estimating needed emission reductions from unknown controls to achieve the alternative standard.
U	=	unknown emissions reductions by geographic area and standard.
T	=	cost by geographic area and standard, or $\frac{\mathrm{N}}{\mathrm{E}}\mathrm{U}^2 + \mathrm{NU}$ (i.e., $ax^2 + bx$).

An example of the hybrid methodology is provided below. In this example, in Area B the percentage of total $PM_{2.5}$ reductions needed from unknown controls relative to total emissions reductions needed (e.g., 100/150, or 67%) is larger than the percentage of total $PM_{2.5}$ reductions needed from unknown controls relative to total emissions reductions needed in Area A (e.g., 100/200, or 50%). Because Area B needs a higher portion of emissions reductions from unknown controls, total cost using the hybrid methodology is higher in Area B. This illustration shows that the relative costs of unknown controls reflect the expectation that average per-ton control costs for the same number of unknown tons are likely to be higher in Area B, which needs a higher ratio of emission reductions from unknown to known controls.

Example of Applying Hybrid Methodology

	$PM_{2.5}$ Emissions Reductions Achieved With Known Controls	$PM_{2.5}$ Emissions Reductions Needed From Unknown Controls
Area A	100	100
Area B	50	100

Area A—Cost Using Hybrid Methodology

$$\left(\left(\frac{\$15{,}000}{100\ tons\ reduced}\right) * 100\ tons\ needed^{\,2}\right) + (\$15{,}000 * 100\ tons\ needed) = \$\mathbf{3,000,000}$$

Average cost/ton = $30,000

Area B—Cost Using Hybrid Methodology

$$\left(\left(\frac{\$15{,}000}{50\ tons\ reduced}\right) * 100\ tons\ needed^{\,2}\right) + (\$15{,}000 * 100\ tons\ needed) = \$\mathbf{4,500,000}$$

Average cost/ton = $45,000

7.2.3.3 Fixed-Cost and Hybrid Methodology Extrapolated Cost Estimates

Extrapolated cost estimates are provided using a 7% discount rate because known control measure information is available at 7% for **all** measures applied in this analysis. Table 7-3 provides the extrapolated cost estimates using both the fixed-cost and hybrid methodologies described above. The extrapolated cost estimates range from $48 million (2010$) to $340 million (2010$) for the revised standard of 12 μg/m^3. We included sensitivity analyses using both the alternative fixed cost-per-ton and the hybrid methodologies in Appendix 7.A.

Table 7-3. Extrapolated Costs by Revised and Alternative Standard Analyzed[a,b] (millions of 2010$)

Revised and Alternative Standard	Region	Extrapolated Cost	
		Fixed-Cost Methodology	**Hybrid Methodology[c]**
		7%	**7%**
13 μg/m^3	East	—	—
	West	—	—
	California	$10	$100
	Total	**$10**	**$100**

12 μg/m³	East	—	—
	West	—	—
	California	\$48	\$340
	Total	**\$48**	**\$340**
11 μg/m³	East	\$71	\$650
	West	\$1.3	\$3.3
	California	\$150	\$940
	Total	**\$220**	**\$1,600**

[a] Estimates are rounded to two significant figures.

[b] Note that the estimates provided reflect incremental emissions reductions from an analytical baseline that gives "credit" to the San Joaquin Valley and South Coast areas for emissions reductions expected to occur between 2020 and 2025 (when those areas are expected to demonstrate attainment with the revised and/or alternative standards).

[c] In applying the hybrid methodology, Plumas County, CA and Shoshone County, ID did not have any known PM controls. We took the following approach to estimate prior emissions reductions for these two counties for use in the hybrid methodology cost calculations: for the remaining counties, by county we summed (i) emissions reductions from known controls and (ii) extrapolated emissions reductions to meet the 15/35 baseline and divided each county's sum by that county's base case PM emissions. We selected the overall minimum percentage and for each of the two counties without any known PM controls, we multiplied that overall minimum percentage by the specific county's base case PM emissions.

Of note is the geographic distribution of extrapolated costs. For the revised and alternative standards, the above costs indicate that control measures applied in California represent a significant portion of the extrapolated costs. Using the fixed-cost methodology, for the revised annual standard of 12 μg/m³ and the alternative annual standards of 13 μg/m³ and 11 μg/m³, the California component of the extrapolated cost estimates represents 100%, 100%, and 67%, respectively, of the nationwide extrapolated cost estimates. Using the hybrid methodology, for the revised annual standard of 12 μg/m³and the alternative annual standards of 13 μg/m³ and 11 μg/m³, the California component of the extrapolated cost estimates represents 100%, 100%, and 59%, respectively, of the nationwide extrapolated cost estimates. Because no cost data exists for unknown future controls, it is unclear whether approaches using hypothetical cost curves will be more accurate or less accurate in forecasting total national costs of unknown controls than a fixed-cost methodology that uses a range of national cost-per-ton values.

Estimating engineering costs for emission reductions needed beyond those from known controls to reach attainment in 2020 is inherently a challenging exercise. As described later in this chapter, our experience with Clean Air Act implementation shows that technological advances and development of innovative strategies can reduce emissions and reduce the costs of emerging technologies over time. Technological change may provide new possibilities for controlling emissions as well as reducing the cost and effectiveness of known controls through technological improvements or higher control efficiencies.

7.2.3.4 Interpreting Extrapolated Cost Estimates

The two estimates do not represent lower and upper bound estimates, but simply represent estimates generated by two different methodologies. The fixed-cost methodology assumes that technological change and innovation will result in the availability of additional controls by 2020 that are similar in cost to the higher end of the cost range for current known controls. The hybrid methodology assumes that while additional controls may become available by 2020, they become available at an increasing cost and the increasing cost varies by geographic area and by degree of difficulty associated with obtaining the needed emissions reductions. Without an initial parameter estimate, i.e., $15,000/ton, we are not able to predict which methodology will generate a higher cost estimate; however, with the same initial parameter estimate of $15,000/ton, the hybrid methodology will generate a higher cost estimate.

7.2.4 Total Cost Estimates

In the supporting statement for the Information Collection Request Revision for Particulate Matter 2.5 Ambient Air Monitoring, 40 CFR Part 58 we estimate the incremental cost of relocating 21 existing near-roadway monitors, and those costs are included in the total national cost estimates presented below. The amendments to the ambient air monitoring regulations will revise the network design requirements for $PM_{2.5}$ monitoring sites, resulting in moving 21 monitors to established near-road monitoring stations by January 1, 2015. The incremental cost associated with moving these 21 monitors is a one-time cost of $28,570.[15]

Tables 7-4 and 7-5 present a summary of the total national costs of attaining the revised annual standard of 12 µg/m^3 and the alternative annual standards of 13 µg/m^3 and 11 µg/m^3 in 2020. This summary includes the known and extrapolated costs. As discussed in Section 7.2.2, we were not able to recalculate any known control costs using a 3% discount rate. As such, both

[15] EPA is not increasing the size of the national $PM_{2.5}$ monitoring network; the Agency anticipates that states would be able to relocate existing monitors to meet the near-roadway requirement. For purposes of estimating cost, only 21 monitors will be moved by 2015. Data from these monitors, along with other monitors in the area, could be used to determine whether the area is meeting both the annual and 24-hour standards. However, data from these monitors would not be available in time for use in making initial attainment and nonattainment designations.

known and extrapolated costs were calculated at a 7% discount rate only. The total cost estimates are \$53 million (2010\$) and \$350 million (2010\$) for the revised annual standard of 12 μg/m^3; \$11 million and \$100 million for the alternative annual standard of 13 μg/m^3; and \$320 million and \$1,700 million for the alternative annual standard of 11 μg/m^3. To further evaluate potential costs ranges, we included sensitivity analyses using both the alternative fixed cost-per-ton and the hybrid methodologies in Appendix 7.A. In addition, Appendix 7.A includes costs and information needed to calculate those costs, by county, to meet 12 μg/m^3.

For the revised annual standard of 12 μg/m^3, the total cost estimates are comprised of between 90 and 97 percent extrapolated cost estimates, and the estimated total cost using the hybrid methodology is roughly 6.5 times more than the estimated total cost using the fixed-cost methodology. [16] Because the hybrid methodology reflects increasing marginal costs in areas needing a higher ratio of emissions reductions from unknown to known controls, it could be more representative of total costs. In an effort to consider the potential fitness of the extrapolated cost estimates, we reviewed the South Coast Air Quality Management District's (SCAQMD) 2012 Air Quality Management Plan (AQMP) [17], and we located data on recent emission reduction credit (ERC) transactions in both the SCAQMD and SJV APCD. While this information provides context for the extrapolated cost estimates, the current relationship between available controls and costs to reduce emissions may or may not be applicable in 2020 because of changes in innovation and advances in technology.

The SCAQMD's 2012 AQMP includes information on control measures to meet the current 24-hour standard of 35. This list of control measures includes further $PM_{2.5}$ controls for under-fired charbroilers at a cost per ton reduced of \$15,000. This control cost matches the parameter used in the fixed-cost methodology, as well as the initial value used for the hybrid methodology and is supportive of our selection of that value.

To provide context for the hybrid methodology's increasing per-ton cost format we obtained the California Air Resources Board's 2009 and 2010 *Emission Reduction Offset Transaction Costs, Summary Report* and reviewed the PM_{10} ERC prices in both the SCAQMD and the SJV APCD. To some degree, ERC transaction prices reflect a choice between installing a more stringent control or purchasing ERCs. Between 2009 and 2010 PM_{10} ERC prices in SJV APCD ranged from \$40,000 per ton per year (tpy) to \$70,000/tpy, and PM_{10} ERC prices in the SCAQMD ranged from \$575,000/tpy to more than \$1.9 million/tpy. These prices reflect both

[16] Note that the extrapolated cost estimates do not represent lower and upper bound estimates, but simply represent estimates generated by the fixed-cost and hybrid methodologies.
[17] Available at http://www.aqmd.gov/aqmp/2012aqmp/draft/index.html.

marginal costs that are higher than the fixed-cost estimates and marginal costs that are not inconsistent with the higher cost estimates generated using the hybrid methodology.

Table 7-4. Total Costs by Revised and Alternative Standard Analyzed (millions of 2010$)[a], Fixed-Cost Methodology[b,c]

Revised and Alternative Standard	Region	Known Control Costs	Unknown Control Costs—Fixed-Cost Methodology	Total Costs
13 µg/m^3	East	—	—	—
	West	—	—	—
	California	$0.63	$10	$11
	Total	**$0.63**	**$10**	**$11**
12 µg/m^3	East	—	—	—
	West	—	—	—
	California	$5.1	$48	$53
	Total	**$5.1**	**$48**	**$53**
11 µg/m^3	East	$96	$71	$170
	West	$0.45	$1.3	$1.8
	California	$5.3	$150	$160
	Total	**$100**	**$220**	**$320**

[a] Estimates are rounded to two significant figures. As such, numbers may not sum down columns.

[b] All control costs are presented at a 7% discount rate only.

[c] Note that the estimates provided reflect incremental emissions reductions from an analytical baseline that gives “credit” to the San Joaquin Valley and South Coast areas for emissions reductions expected to occur between 2020 and 2025 (when those areas are expected to demonstrate attainment with the revised and/or alternative standards).

Table 7-5 Total Costs by Revised and Alternative Standard Analyzed (millions of 2010$)[a], Hybrid Methodology[b,c]

Revised and Alternative Standard	Region	Known Control Costs	Unknown Control Costs—Hybrid Methodology	Total Costs
13 $\mu g/m^3$	East	—	—	—
	West	—	—	—
	California	$0.63	$100	$100
	Total	**$0.63**	**$100**	**$100**
12 $\mu g/m^3$	East	—	—	—
	West	—	—	—
	California	$5.1	$340	$350
	Total	**$5.1**	**$340**	**$350**
11 $\mu g/m^3$	East	$96	$650	$750
	West	$0.45	$3.3	$3.8
	California	$5.3	$940	$950
	Total	**$100**	**$1,600**	**$1,700**

[a] Estimates are rounded to two significant figures. As such, numbers may not sum down columns.

[b] All control costs are presented at a 7% discount rate only.

[c] Note that the estimates provided reflect incremental emissions reductions from an analytical baseline that gives "credit" to the San Joaquin Valley and South Coast areas for emissions reductions expected to occur between 2020 and 2025 (when those areas are expected to demonstrate attainment with the revised and/or alternative standards).

7.3 Changes in Regulatory Cost Estimates over Time

There are many examples in which technological innovation and "learning by doing" have made it possible to achieve greater emission reductions than had been feasible earlier, or have reduced the costs of emissions control in relation to original estimates. Studies have concluded that costs of some EPA programs have been less than originally estimated, due in part to EPA's inability to predict and account for future technological innovation in regulatory impact analyses.[18] Additionally, technological change will affect baseline conditions for our analysis. Technical change may lead to potential improvements in the efficiency with which

[18] Harrington et al. (2000) and previous studies cited by Harrington. Harrington, W., R.D. Morgenstern, and P. Nelson. 2000. "On the Accuracy of Regulatory Cost Estimates." *Journal of Policy Analysis and Management* 19(2):297-322.

firms produce goods and services; for example, firms may use less energy to produce the same quantities of output.

Increasing marginal abatement costs could possibly induce the type of innovation that would result in lower costs than estimated in this chapter. By 2020, breakthrough technologies in control equipment could result in a downward shift in the marginal abatement cost curve for such equipment (Figure 7-1)[19] as well as a decrease in its slope, reducing marginal costs per unit of abatement. In addition, elevated abatement costs may result in significant increases in the cost of production and would likely induce production efficiencies, in particular those related to energy inputs, which would lower emissions from the production side.

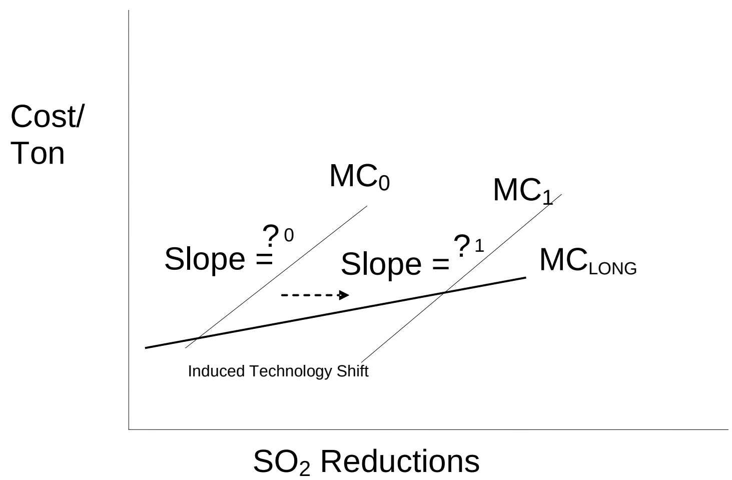


Figure 7-1. Technological Innovation Reflected by Marginal Cost Shift

7.3.1 Examples of Technological Advances in Pollution Control

There are a number of examples of low-emissions technologies and pollution control equipment developed and/or commercialized over the past 15 to 20 years, such as

- Selective catalytic reduction (SCR) and ultra-low NO_x burners for NO_x emissions
- Scrubbers, which achieve 95% and potentially greater SO_2 control on boilers
- Sophisticated new valve seals and leak detection equipment for refineries and chemical plants

[19] Figure 7-1 shows a linear marginal abatement cost curve. It is possible that the shape of the marginal abatement cost curve is non-linear.

- Low- or zero-VOC paints, consumer products and cleaning processes
- Chlorofluorocarbon (CFC) free air conditioners, refrigerators, and solvents
- Water- and powder-based coatings to replace petroleum-based formulations
- Vehicles are much cleaner than believed possible in the late 1980s due to improvements in evaporative controls, catalyst design and fuel control systems for light-duty vehicles; and treatment devices and retrofit technologies for heavy-duty engines
- Idle-reduction technologies for engines, including truck stop electrification efforts
- Market penetration of gas-electric hybrid vehicles, and clean fuels
- The development of retrofit technology to reduce emissions from in-use vehicles and non-road equipment

These technologies were not commercially available two decades ago, and some did not even exist. Yet today, all of these technologies are on the market, and many are widely employed. Several are key components of major pollution control programs.

"Learning by doing" or "learning curve impacts," a distinct concept from technological innovation, has also made it possible to achieve greater emissions reductions than had been feasible earlier or has reduced the costs of emissions control compared to original estimates. Learning curve impacts can be defined generally as the extent to which variable costs (of production and/or pollution control) decline as firms gain experience with a specific technology. Impacts such as these would manifest themselves as a lower expected cost to operate technologies in the future compared to what costs may have been.

The magnitude of learning curve impacts on pollution control costs has been estimated for a variety of sectors as part of the cost analyses done for the Direct Cost Estimates for the Clean Air Act Second Section 812 Prospective Analysis.[20] In that report, learning curve adjustments were included for those sectors and technologies for which learning curve data were available. A typical learning curve adjustment example is to reduce either capital or O&M costs by a certain percentage given a doubling of output from that sector or for that technology. In 1936, T.P. Wright was the first to characterize the relationship between increased productivity and cumulative production. He analyzed man-hours required to

[20] E.H. Pechan and Associates, Inc. and Industrial Economics, Incorporated. Direct Cost Estimates for the Clean Air Act Second Section 812 Prospective Analysis: Final Report, prepared for U.S. EPA, Office of Air and Radiation, February 2011. Available at http://www.epa.gov/oar/sect812/feb11/costfullreport.pdf.

assemble successive airplane bodies. He suggested the relationship is a log linear function, since he observed a constant linear reduction in man-hours every time the total number of airplanes assembled was doubled. The relationship he devised between number assembled and assembly time is called Wright's Equation (Gumerman and Marnay, 2004).[21] This equation, shown below, has been shown to be widely applicable in manufacturing:

$$\text{Wright's Equation: } C_N = C_o * N^b, \tag{7.2}$$

where:

N = cumulative production

C_N = cost to produce Nth unit of capacity

Co = cost to produce the first unit

b = learning parameter = ln (1-LR)/ln(2), where

LR = learning by doing rate, or cost reduction per doubling of capacity or output.

The percentage adjustments to costs can range from 5 to 20%, depending on the sector and technology. Learning curve adjustments were prepared in a memo by IEc supplied to U.S. EPA and applied for the mobile source sector (both onroad and nonroad) and for application of various EGU control technologies within the Draft Direct Cost Report.[22] Advice received from the SAB Advisory Council on Clean Air Compliance Analysis in June 2007 indicated an interest in expanding the treatment of learning curves to those portions of the cost analysis for which no learning curve impact data are currently available. Examples of these sectors are non-EGU point sources and area sources. The memo by IEc outlined various approaches by which learning curve impacts can be addressed for those sectors. The recommended learning curve impact adjustment for virtually every sector considered in the Draft Direct Cost Report is a 10% reduction in O&M costs for two doublings of cumulative output, with proxies such as cumulative fuel sales or cumulative emissions reductions being used when output data was unavailable.

[21] Gumerman, Etan and Marnay, Chris. Learning and Cost Reductions for Generating Technologies in the National Energy Modeling System (NEMS), Ernest Orlando Lawrence Berkeley National Laboratory, University of California at Berkeley, Berkeley, CA. January 2004, LBNL-52559.

[22] Industrial Economics, Inc. Proposed Approach for Expanding the Treatment of Learning Curve Impacts for the Second Section 812 Prospective Analysis: Memorandum, prepared for U.S. EPA, Office of Air and Radiation, August 13, 2007.

For this RIA, we do not have the necessary data for cumulative output, fuel sales, or emission reductions for all sectors included in our analysis in order to properly generate control costs that reflect learning curve impacts. Clearly, the effect of including these impacts would be to lower our estimates of costs for our control strategies in 2020, but we are not able to include such an analysis in this RIA.

7.3.2 Influence on Regulatory Cost Estimates

Studies indicate that it is not uncommon for pre-regulatory cost estimates to be higher than later estimates, in part because of an inability to predict technological advances. Over longer time horizons, the opportunity for technical advances is greater.

7.3.2.1 Multi-Rule Study

Harrington et al. of Resources for the Future (RFF)[23] conducted an analysis of the predicted and actual costs of 28 federal and state rules, including 21 issued by U.S. EPA and the Occupational Safety and Health Administration (OSHA), and found a tendency for predicted costs to overstate actual implementation costs. Costs were considered accurate if they fell within the analysis error bounds or if they fell within 25% (greater or less than) of the predicted amount. They found that predicted total costs were overestimated for 14 of the 28 rules, while total costs were underestimated for only three rules. Differences can result because of quantity differences (e.g., overestimate of pollution reductions) or differences in per-unit costs (e.g., cost per unit of pollution reduction). Per-unit costs of regulations were overestimated in 14 cases, while they were underestimated in six cases. In the case of U.S. EPA rules, the Agency overestimated per-unit costs for five regulations, underestimated them for four regulations (three of these were relatively small pesticide rules), and accurately estimated them for four. Based on an examination of eight rules, "for those rules that employed economic incentive mechanisms, overestimation of per-unit costs seems to be the norm," the study said. In addition, Harrington et al. also states that overestimation of total costs can be due to error in the quantity of emissions reductions achieved, which would also cause the benefits to be overestimated.

It should be noted that many (though not all) of the U.S. EPA rules examined by Harrington et al. had compliance dates of several years, which allowed a limited period for technical innovation. In contrast, the progress demonstration and compliance dates for a

[23] Harrington, W., R.D. Morgenstern, and P. Nelson. 2000. "On the Accuracy of Regulatory Cost Estimates." *Journal of Policy Analysis and Management* 19(2):297-322.

attaining a NAAQS occur over a longer time horizon and could allow for possible technical innovation.

7.3.2.2 Acid Rain SO_2 Trading Program

Recent cost estimates of the Acid Rain SO_2 trading program by RFF and MIT have been as much as 83% lower than originally projected by EPA (see Table 7-6).[24] As noted in the RIA for the Clean Air Interstate Rule, the 1989 *ex ante* numbers associated with the Acid Rain Program were an overestimate in part because of the limitation of economic modeling to predict technological improvement of pollution controls and other compliance options, such as fuel switching. In part, the fuel switching from high-sulfur to low-sulfur coal was spurred by a reduction in rail transportation costs due to deregulation of rail rates during the 1990s. Harrington et al. report that scrubbing turned out to be more efficient (95% removal vs. 80–85% removal) and more reliable (95% vs. 85% reliability) than expected, and that unanticipated opportunities arose to blend low- and high-sulfur coal in older boilers up to a 40/60 mixture, compared with the 5/95 mixture originally estimated.

Table 7-6. Phase 2 Cost Estimates

Phase 2 Cost Estimates	
Ex ante estimates	\$2.7 to \$6.2 billion[a]
Ex post estimates	\$1.0 to \$1.4 billion

[a] 2010 Phase II cost estimate in 1995\$.

7.3.2.3 Chlorofluorocarbon Phase-Out

EPA used a combination of regulatory, market-based (i.e., a cap-and-trade system among manufacturers), and voluntary approaches to phase out the most harmful ozone depleting substances. The phase out was done more efficiently than either EPA or industry originally anticipated. The phase out for Class I substances was implemented 4–6 years faster, included 13 more chemicals, and cost 30% less than was predicted at the time the 1990 Clean Air Act Amendments were enacted.[25]

[24] Carlson, Curtis, Dallas R. Burtraw, Maureen, Cropper, and Karen L. Palmer. 2000. "Sulfur Dioxide Control by Electric Utilities: What Are the Gains from Trade?" *Journal of Political Economy* 108(#6):1292-1326. Ellerman, Denny. January 2003. Ex Post Evaluation of Tradable Permits: The U.S. SO2 Cap-and-Trade Program. Massachusetts Institute of Technology Center for Energy and Environmental Policy Research.

[25] Holmstead, Jeffrey, 2002. "Testimony of Jeffrey Holmstead, Assistant Administrator, Office of Air and Radiation, U.S. Environmental Protection Agency, Before the Subcommittee on Energy and air Quality of the committee on Energy and Commerce, U.S. House of Representatives, May 1, 2002, p. 10.

The Harrington et al. study states, "When the original cost analysis was performed for the CFC phase-out it was not anticipated that the hydrofluorocarbon HFC-134a could be substituted for CFC-12 in refrigeration." However, as Hammit[26] notes "since 1991 most new U.S. automobile air conditioners have contained HFC-134a (a compound for which no commercial production technology was available in 1986) instead of CFC-12" (p. 13). Hammit cites a similar story for HCFRC-141b and 142b, which are currently substituting for CFC-11 in important foam-blowing applications.

7.3.3 Influence of Regulation on Technological Change

We cannot estimate the interplay between EPA regulation and technology improvement but have reason to believe it may be significant. There is emerging research on technology-forcing polices (i.e., where a regulator specifies a policy standard that cannot be met with existing technology or cannot be met with existing technology at an acceptable cost, and over time market demand will provide incentives for industry to develop the appropriate technology). This is illustrated by Gerard and Lave (2005). They demonstrate through a careful review of policy history that the 1970 CAA legislated dramatic improvements in the reduction of emissions for 1975 and 1976 automobiles. Those mandated improvements went beyond the capabilities of existing technologies. But the regulatory pressure "pulled" forth or "forced" catalytic converting technology in 1975.

Popp (2003) and Keohane (2002) have both provided empirical evidence that Title IV led to induced technological change. Popp provides evidence that since Title IV there has been technological innovations that have improved the removal efficiency of scrubbers. Keohane provides evidence that fossil-fuel fired electric utilities that were subject to Title IV were, for a given increase in the cost of switching to low sulfur coal, more likely to install a scrubber.

7.4 Uncertainties and Limitations

EPA based its estimates of emissions control costs on the best available information from available engineering studies of air pollution controls and developed a reliable modeling framework for analyzing the cost, emissions changes, and other impacts of regulatory controls. However, our cost analysis is subject to uncertainties and limitations, which we document on a qualitative basis in Table 7-7 below. For additional discussion of how we assess uncertainty, see Section 5.5.7.

[26] Hammit, J.K. (2000). "Are the costs of proposed environmental regulations overestimated? Evidence from the CFC phase out." *Environmental and Resource Economics*, 16(#3): 281-302.

Table 7-7. Summary of Qualitative Uncertainty for Modeling Elements of PM Engineering Costs

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Costs[a]	Degree of Confidence in Our Analytical Approach[b]	Ability to Assess Uncertainty[c]
Uncertainties Associated with Engineering Costs				
Engineering Cost Estimates ▪ Capital recovery factor estimates (7% and 3%) ▪ Estimates of private compliance cost ▪ Increased advancement in control technologies as well as reduction in costs over time ▪ Cost estimates for PM_{10}	Both	Medium-high	Medium	Tier 2
Unquantified Costs ▪ Costs of federal and state administration of SIP program ▪ Transactional costs	Low	Medium	Medium	Tier 1
Extrapolated Costs	Both	High	Low	Tier 1

[a] Magnitude of Impact
High—If error could influence the total costs by more than 25%
Medium—If error could influence the total costs by 5%–25%
Low—If error could influence the total costs by less than 5%

[b] Degree of Confidence in Our Analytic Approach
High—The current evidence is plentiful and strongly supports the selected approach
Medium—Some evidence exists to support the selected approach, but data gaps are present
Low—Limited data exists to support the selected approach

[c] Ability to Assess Uncertainty (using WHO Uncertainty Framework)
Tier 0—Screening level, generic qualitative characterization
Tier 1—Scenario-specific qualitative characterization
Tier 2—Scenario-specific sensitivity analysis
Tier 3—Scenario-specific probabilistic assessment of individual and combined uncertainty

7.5 References

Carlson, Curtis, Dallas R. Burtraw, Maureen, Cropper, and Karen L. Palmer. 2000. "Sulfur Dioxide Control by Electric Utilities: What Are the Gains from Trade?" *Journal of Political Economy* 108(#6):1292-1326.

Ellerman, Denny. January 2003. Ex Post Evaluation of Tradable Permits: The U.S. SO2 Cap-and-Trade Program. Massachusetts Institute of Technology Center for Energy and Environmental Policy Research.

Gerard, D. and L.B. Lave (2005). “Implementing Technology-Forcing Policies: The 1970 Clean Air Act Amendments and the Introduction of Advanced Automotive Emissions Controls in the United States,” Technological Forecasting and Social Change, 72: 761-778.

Gumerman, Etan and Marnay, Chris. Learning and Cost Reductions for Generating Technologies in the National Energy Modeling System (NEMS), Ernest Orlando Lawrence Berkeley National Laboratory, University of California at Berkeley, Berkeley, CA. January 2004, LBNL-52559.

Hammit, J.K. (2000). “Are the costs of proposed environmental regulations overestimated? Evidence from the CFC phase out.” *Environmental and Resource Economics*, 16(#3): 281-302.

Harrington, W., R.D. Morgenstern, and P. Nelson. 2000. “On the Accuracy of Regulatory Cost Estimates.” *Journal of Policy Analysis and Management* 19(2):297-322.

Holmstead, Jeffrey, 2002. “Testimony of Jeffrey Holmstead, Assistant Administrator, Office of Air and Radiation, U.S. Environmental Protection Agency, Before the Subcommittee on Energy and air Quality of the committee on Energy and Commerce, U.S. House of Representatives, May 1, 2002, p. 10.

ICF Consulting. 2005. The Clean Air Act Amendment: Spurring Innovation and Growth While Cleaning the Air. Washington, DC. Available at http://www.icfi.com/Markets/Environment/doc_files/caaa-success.pdf.

Industrial Economics, Inc. Proposed Approach for Expanding the Treatment of Learning Curve Impacts for the Second Section 812 Prospective Analysis: Memorandum, prepared for U.S. EPA, Office of Air and Radiation, August 13, 2007.

Keohane, Nathaniel. 2002. “Environmental Policy and the Choice of Abatement Technique” Evidence from Coal-Fired Power Plants.” February. Yale School of Management.

Office of Management and Budget (OMB). 2003. Circular A-4. Available at http://www.whitehouse.gov/omb/circulars_a004_a-4/

Popp, David. 2003. “Pollution Control Innovations and the Clean Air Act of 1990.” Autumn. Journal of Policy Analysis and Management, 22(4), 641-60.

E.H. Pechan and Associates and Industrial Economics, Direct Cost Estimates for the Clean Air Act Second Section 812 Prospective Analysis: Draft Report, prepared for U.S. EPA, Office of Air and Radiation, February 2007. Available at http://www.epa.gov/oar/sect812/mar07/direct_cost_draft.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2002. EPA Air Pollution Control Cost Manual—Sixth Edition (EPA 452/B-02-001). Office of Air Quality Planning and Standards, Research Triangle Park, NC. Also Available at http://epa.gov/ttn/catc/dir1/c_allchs.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2006. AirControlNET 4.1 Control Measures Documentation Report. Office of Air Quality Planning and Standards, Research Triangle Park, NC. Also Available at http://www.epa.gov/ttnecas1/models/DocumentationReport.pdf.

U.S. Environmental Protection Agency. June 2007. Advisory Council on Clean Air Compliance Analysis (COUNCIL), Council Advisory on OAR's Direct Cost Report and Uncertainty Analysis Plan. Washington, DC.

U.S. Environmental Protection Agency (U.S. EPA). 2010. Guidelines for Preparing Economic Analysis. Office of Policy, National Center for Environmental Economics. Also Available at http://yosemite.epa.gov/ee/epa/eed.nsf/Webpages/Guidelines.html/$file/Guidelines.pdf

U.S. Environmental Protection Agency (U.S. EPA). 2011. The Benefits and Costs of the Clean Air Act from 1990 to 2020. Final Report for the U.S. Environmental Protection Agency, Office of Air and Radiation. Also available at http://www.epa.gov/air/sect812/feb11/fullreport.pdf

APPENDIX 7.A
DATA TO CALCULATE COSTS TO MEET 12 μG/M^3 AND SENSITIVITY ANALYSES OF EXTRAPOLATED COST ESTIMATES

7.A.1 $PM_{2.5}$ Emission Reductions and Costs to Meet 12 μg/m^3

Table7.A.1 below includes costs and information needed to calculate those costs, by county, to meet 12 μg/m^3. The Table includes the $PM_{2.5}$ emissions reductions needed to reach the 15/35 μg/m^{3}level because the hybrid methodology includes a parameter that uses the quantity of prior emissions reductions.[1]

Table 7.A.1 $PM_{2.5}$ Emission Reductions and Costs to Meet 12 μg/m^3

			12 μg/m^3			
			Known Controls		Unknown Controls	
FIPS Code	County	$PM_{2.5}$ Emissions Reductions to Reach 15/35 μg/m^3	Emissions Reductions	Costs	Emissions Reductions	Hybrid Methodology Costs
06037	Los Angeles	--	743	$4.5	--	--
06065	Riverside	--	53	$0.63	980	$290
06025	Imperial	404	--	--	294	$7.6
06029	Kern	1,769	--	--	418	$7.8
06107	Tulare	726	--	--	635	$18
06047	Merced	76	--	--	19	$0.36
06071	San Bernardino	988	--	--	844	$23

[1] EPA developed a model of increasing total annualized costs for controlling $PM_{2.5}$ emissions -- $ax^2 + bx + c$ where x is the tons of a particular pollutant to be reduced in a particular area and a, b, and c are constants. For a particular geographic area a is N/E where (i) N is a national, initial annualized cost/ton of $15,000 per ton and (ii) E is, by geographic area, the denominator and represents all particulate emission reductions achieved (from applying known and unknown controls to obtain the 15/35 baseline, as well as known controls to achieve the alternative standard) prior to estimating needed emission reductions from unknown controls to achieve the alternative standard.

7.A.2 Sensitivity Analyses of Extrapolated Cost Estimates

Because of the uncertainties associated with estimating costs for the $PM_{2.5}$ NAAQS and because a significant portion of the estimated emissions reductions and related costs for attaining the NAAQS come from unknown controls, it is important to test the sensitivity of the assumptions applied to estimate unknown controls. The sensitivity analyses below are included to help characterize the uncertainty for the cost estimates from unknown controls and the responsiveness of the cost estimates to varying parameter estimates and assumptions. *Note that the tables below include cost estimates associated with unknown controls and not total cost estimates.*

While there are many approaches to sensitivity analysis, we selected analyses below, keeping emissions estimates constant, to show variability in the cost estimates and remain consistent with the benefits analysis. *Note that the extrapolated cost estimates are provided using a 7 percent discount rate because known control measure information is available at 7 percent for* ***all*** *measures applied in this analysis.*

7.A.2.1 Sensitivity Analysis of Fixed-Cost Methodology

Table 7.A.2 below presents the sensitivity analysis of the fixed-cost methodology and includes, by region and revised and alternative standard, the primary cost estimate of $15,000/ton. The Table also includes, by region and revised and alternative standard, cost estimates using $10,000/ton and $20,000/ton. For the revised standard of 12/35, the total cost estimate associated with unknown control costs ranges from $32 million to $64 million, depending on the fixed-cost-per-ton assumed.

Table 7.A.2 Sensitivity Analysis of Fixed-Cost Methodology for Unknown Controls by Revised and Alternative Standard Analyzed (millions of 2010$)[a]

		Extrapolated Costs		
		$10,000/ton	**$15,000/ton**	**$20,000/ton**
Revised & Alternative Standard	**Region**	**7%**	**7%**	**7%**
13 μg/m^3	East	—	—	—
	West	—	—	—
	California	$6.7	$10	$13
	Total	**$6.7**	**$10**	**$13**
12 μg/m^3	East	—	—	—
	West	—	—	—

	California	$32	$48	$64
	Total	**$32**	**$48**	**$64**
11 µg/m^3	East	$48	$71	$95
	West	$0.86	$1.3	$1.7
	California	$97	$150	$190
	Total	**$150**	**$220**	**$290**

[a] Estimates are rounded to two significant figures.

7.A.2.2 Sensitivity Analysis of Alternative Hybrid Methodology

Table 7.A.3 below presents the sensitivity analysis of the alternative hybrid methodology. To be consistent with the sensitivity analysis of the fixed-cost methodology, the Table also includes, by region and revised and alternative standard, cost estimates using alternate parameter estimates for the initial cost per ton. For the revised standard of 12/35, the total cost estimate associated with unknown controls ranges from $230 million to $460 million.

Table 7.A.3 Sensitivity Analysis of Hybrid Methodology for Unknown Controls by Revised and Alternative Standard Analyzed (millions of 2010$)[a]

		Extrapolated Costs		
Revised & Alternative Standard	**Region**	**$10,000/ton** 7%	**$15,000/ton** 7%	**$20,000/ton** 7%
13 µg/m^3	East	—	—	—
	West	—	—	—
	California	$69	$100	$140
	Total	**$69**	**$100**	**$140**
12 µg/m^3	East	—	—	—
	West	—	—	—
	California	$230	$340	$460
	Total	**$230**	**$340**	**$460**
11 µg/m^3	East	$430	$650	$870
	West	$2.2	$3.3	$4.4
	California	$630	$940	$1,300
	Total	**$1,100**	**$1,600**	**$2,100**

[a] Estimates are rounded to two significant figures.

CHAPTER 8
COMPARISON OF BENEFITS AND COSTS

8.1 Synopsis

This chapter compares estimates of the benefits with costs and summarizes the net benefits of the revised annual standard of 12 µg/m^3 and the alternative annual standards of 13 µg/m^3 and 11 µg/m^3 relative to the analytical baseline that includes recently promulgated national regulations and additional emissions reductions needed to attain the existing 15/35 µg/m^3 standards, as well as adjustments to NO_x emissions in the San Joaquin and South Coast areas.

8.2 Comparison of Benefits and Costs

The EPA's illustrative analysis has estimated the health and welfare benefits and costs associated with the revised annual PM NAAQS. The results in Table 8-1 for 2020 suggest there will be significant health and welfare benefits and these benefits will outweigh the costs associated with the illustrative control strategies in 2020. In the analysis, we estimate the net benefits of the revised annual $PM_{2.5}$ standard of 12 µg/m^3 and alternative annual standards of 13 µg/m^3 and 11 µg/m^3, incremental to the 2020 analytical baseline. For the revised annual standard of 12 µg/m^3, net benefits are estimated to be $3.7 billion to $9 billion at a 3% discount rate and $3.3 billion to $8.1 billion at a 7% discount rate in 2020 (2010 dollars). For an alternative annual standard of 13 µg/m^3, net benefits are estimated to be $1.2 billion to $2.9 billion at the 3% discount rate and $1.1 billion to $2.6 billion at the 7% discount rate. Net benefits of an alternative annual $PM_{2.5}$ standard of 11 µg/$m^3$are estimated to be $11 billion to $29 billion at a 3% discount rate and $10 billion to $26 billion at a 7% discount rate in 2020.

For the revised annual standard of 12 µg/m^3, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 11 to 154 times at a 7% discount rate. For the alternative annual standard of 13 µg/m^3, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 11 to 246 times at a 7% discount rate. For the alternative annual standards of 11 µg/m^3, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 7 to 81 times at a 7% discount rate.

Table 8-1. Total Monetized Benefits, Total Costs, and Net Benefits in 2020 (millions of 2010$)—Full Attainment[a]

	Total Costs[b]		Monetized Benefits[d]		Net Benefits	
Revised Annual Standard	3% Discount Rate[c]	7% Discount Rate	3% Discount Rate	7% Discount Rate	3% Discount Rate[b]	7% Discount Rate
12	**$53 to $350**	**$53 to 350**	**$4,000 to $9,100**	**$3,600 to $8,200**	**$3,700 to $9,000**	**$3,300 to $8,100**
Alternative Standards						
13	$11 to $100	$11 to $100	$1,300 to $2,900	$1,200 to $2,600	$1,200 to $2,900	$1,100 to $2,600
11	$320 to $1,700	$320 to $1,700	$13,000 to $29,000	$12,000 to $26,000	$11,000 to $29,000	$10,000 to $26,000

[a] These estimates reflect incremental emissions reductions from an analytical baseline that gives an "adjustment " to the San Joaquin and South Coast areas in California for NO_x emissions reductions expected to occur between 2020 and 2025, when those areas are expected to demonstrate attainment with the revised standards. Full benefits of the revised standards in those two areas will not be realized until 2025.

[b] The two cost estimates do not represent lower- and upper-bound estimates but represent estimates generated by two different methodologies. The lower estimate is generated using the fixed-cost methodology, which assumes that technological change and innovation will result in the availability of additional controls by 2020 that are similar in cost to the higher end of the cost range for current, known controls. The higher estimate is generated using the hybrid methodology, which assumes that while additional controls may become available by 2020, they become available at an increasing cost and the increasing cost varies by geographic area and by degree of difficulty associated with obtaining the needed emissions reductions.

[c] Due to data limitations, we were unable to discount compliance costs for all sectors at 3%. See Chapter 7, Section 7.2.2 for additional details on the data limitations. As a result, the net benefit calculations at 3% were computed by subtracting the costs at 7% from the monetized benefits at 3%.

[d] The reduction in premature deaths each year accounts for over 90% of total monetized benefits. Mortality risk valuation assumes discounting over the SAB-recommended 20-year segmented lag structure. Not all possible benefits or disbenefits are quantified and monetized in this analysis. B is the sum of all unquantified benefits. Data limitations prevented us from quantifying these endpoints, and as such, these benefits are inherently more uncertain than those benefits that we were able to quantify. The range of benefits reflects the range of the central estimates from two mortality cohort studies (i.e., Krewski et al. [2009] to Lepeule et al. [2012]).

Figure 8-1 demonstrates the size of the benefits relative to costs for the revised annual standards of 12 $\mu g/m^3$ at the 7% discount rate. This figure shows benefits for two different mortality studies and costs using two methods for extrapolating costs to emissions reductions associated with unknown controls.

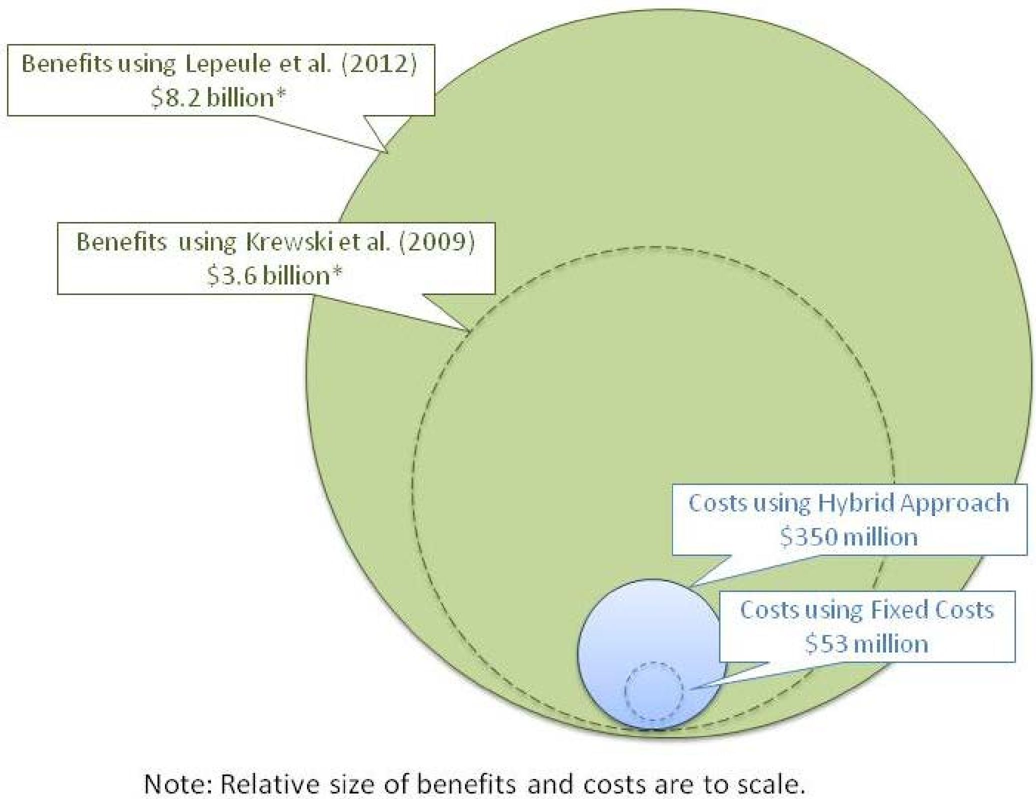


Figure 8-1. Monetized Benefit to Cost Comparison for the Revised Annual Standard of 12 μg/m^3 in 2020 (7% Discount Rate)

Note: Relative size of benefits and costs are to scale.

Figure 8-2 displays the range of net benefits for the selected standards using the two epidemiology functions and 12 expert elicitation functions for PM-related premature mortality that the EPA employs in its analysis of benefits. As shown in the figure, the benefits exceed costs in every combination analyzed.

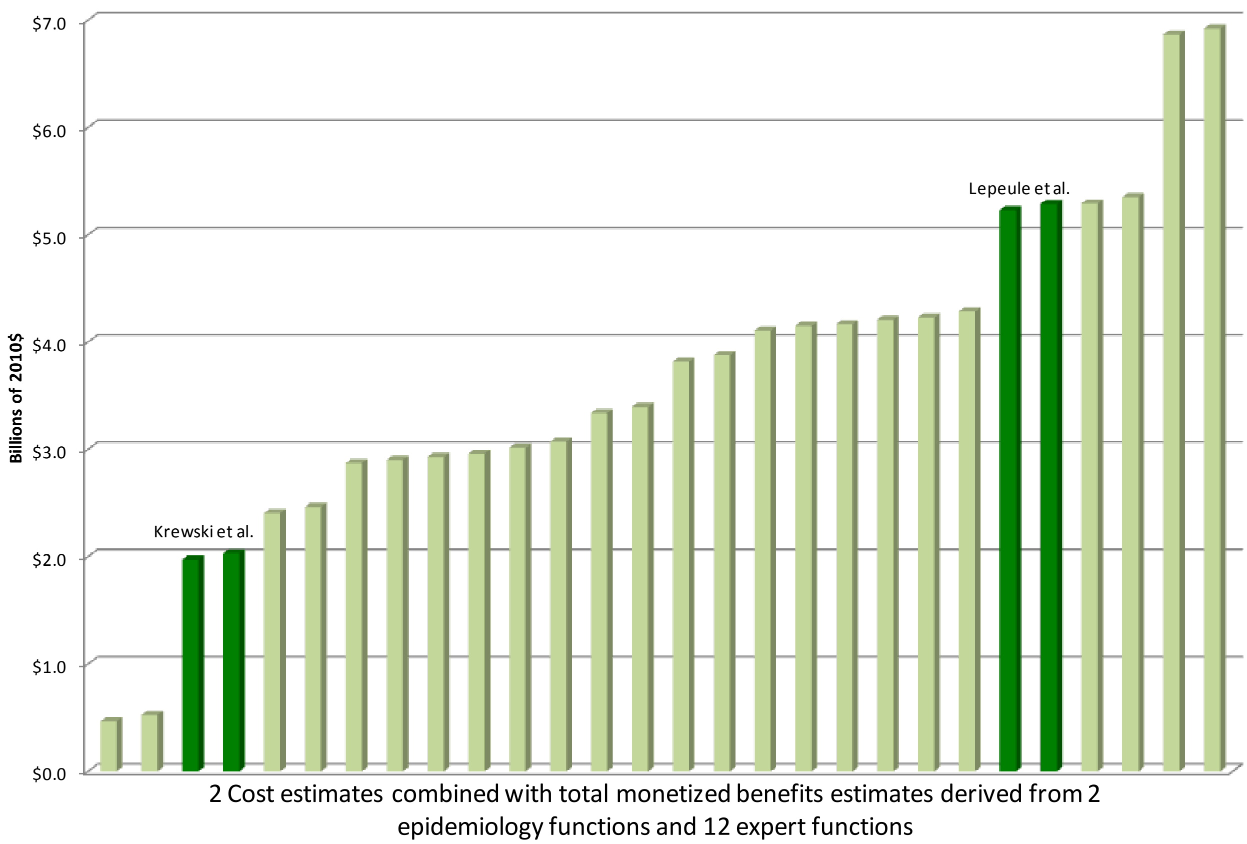


Figure 8-2. Net Benefits for Revised Annual Standard of 12 µg/m³ in 2020 at a 7% Discount Rate

Due to data and methodology limitations, the EPA was unable to quantify some health benefits associated with exposure to $PM_{2.5}$, as well as the additional co-benefits from improvements in welfare effects associated with emission reductions to attain the primary standard, such as visibility. Tables 8-2 and 8-3 summarize the human health and welfare categories contained within the core benefits estimate as well as those categories that are unquantified in the core benefits estimate. Because the illustrative emission reduction strategy for the revised annual standard at 12 µg/m^3 consisted of only directly emitted $PM_{2.5}$, these tables are limited to only those categories associated with emission reductions of directly emitted $PM_{2.5}$.[1] It is important to emphasize that the list of unquantified benefit categories is not exhaustive, nor is quantification of each effect complete.

[1] For the unquantified benefits associated with emission reductions of NO_x and SO_2, please see Chapters 5 and 6.

Table 8-2. Human Health Effects of Ambient $PM_{2.5}$

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Improved Human Health				
Reduced incidence of premature mortality from exposure to $PM_{2.5}$	Adult premature mortality based on cohort study estimates and expert elicitation estimates (age >25 or age >30)	✓	✓	Section 5.6
	Infant mortality (age <1)	✓	✓	Section 5.6
	Non-fatal heart attacks (age > 18)	✓	✓	Section 5.6
	Hospital admissions—respiratory (all ages)	✓	✓	Section 5.6
	Hospital admissions—cardiovascular (age >20)	✓	✓	Section 5.6
	Emergency department visits for asthma (all ages)	✓	✓	Section 5.6
	Acute bronchitis (age 8–12)	✓	✓	Section 5.6
	Lower respiratory symptoms (age 7–14)	✓	✓	Section 5.6
	Upper respiratory symptoms (asthmatics age 9–11)	✓	✓	Section 5.6
	Asthma exacerbation (asthmatics age 6–18)	✓	✓	Section 5.6
	Lost work days (age 18–65)	✓	✓	Section 5.6
	Minor restricted-activity days (age 18–65)	✓	✓	Section 5.6
	Chronic bronchitis (age >26)	—[a]	—[a]	Section 5.6
	Emergency department visits for cardiovascular effects (all ages)	—[a]	—[a]	Section 5.6
	Strokes and cerebrovascular disease (age 50–79)	—[a]	—[a]	Section 5.6
	Other cardiovascular effects (e.g., other ages)	—	—	PM ISA[b]
	Other respiratory effects (e.g., pulmonary function, non-asthma ER visits, non-bronchitis chronic diseases, other ages and populations)	—	—	PM ISA[b]
	Reproductive and developmental effects (e.g., low birth weight, pre-term births, etc.)	—	—	PM ISA[b,c]
	Cancer, mutagenicity, and genotoxicity effects	—	—	PM ISA[b,c]

[a]We quantify these benefits in a sensitivity analysis, but not in the core analysis.

[b]We assess these benefits qualitatively because we do not have sufficient confidence in available data or methods.

[c]We assess these benefits qualitatively because current evidence is only suggestive of causality or there are other significant concerns over the strength of the association.

Table 8-3. Welfare Co-Benefits of $PM_{2.5}$

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Improved Environment				
Reduced visibility impairment	Visibility in Class I areas in SE, SW, and CA regions	[a]	[a]	Section 6.3, Appendix 6b
	Visibility in Class I areas in other regions	—	[a]	Section 6.3, Appendix 6b
	Visibility in 8 cities	—	[a]	Section 6.3, Appendix 6b
	Visibility in other residential areas	—	[a]	Section 6.3, Appendix 6b
Reduced climate effects	Climate impacts from PM	—	—	Section 6.5, PM ISA[b]
Reduced effects on materials	Household soiling	—	—	Section 6.4, PM ISA[b]
	Materials damage (e.g., corrosion, increased wear)	—	—	Section 6.4, PM ISA[c]
Reduced effects from PM deposition (metals and organics)	Effects on Individual organisms and ecosystems	—	—	Section 6.6.1, PM ISA[b]

[a] We quantify these co-benefits in an illustrative analysis, but these results of that illustrative scenario are not an estimate of the co-benefits for the revised primary standard.

[b] We assess these benefits qualitatively because we do not have sufficient confidence in available data or methods.

8.3 Discussion and Conclusions

An extensive body of scientific evidence documented in PM ISA indicates that $PM_{2.5}$ can penetrate deep into the lungs and cause serious health effects, including premature death and other non-fatal illnesses (U.S. EPA, 2009). As described in the preamble to the rule, the revisions to the standards are based on an integrative assessment of an extensive body of new scientific evidence (U.S. EPA, 2009). Health studies published since the PM ISA (e.g., Pope et al. [2009]) confirm that recent levels of $PM_{2.5}$ have had a significant impact on public health. Based on the air quality analysis in this RIA, the EPA projects that nearly all counties with $PM_{2.5}$ monitors in the United States would meet an annual standard of 12 $\mu g/m^3$ without additional Federal, State, or local PM control programs. This demonstrates the substantial progress that the United States has made in reducing air pollution emissions over the last several decades.

Regulations such as the EPA's recent Mercury and Air Toxics Standards (MATS) and other Federal programs such as diesel standards will provide substantial improvements in regional concentrations of $PM_{2.5}$. Our analysis shows a few areas would still need additional emissions reductions to address local sources of air pollution, including ports and uncontrolled industrial emissions. For this reason, we have designed the RIA analysis to focus on local controls in these few areas. We estimate that these additional local controls would yield benefits well in excess of costs.

The setting of a NAAQS does not compel specific pollution reductions and as such does not directly result in costs or benefits. For this reason, NAAQS RIAs are merely illustrative. The NAAQS RIAs illustrate the potential costs and benefits of additional steps States could take to attain a revised air quality standard nationwide beyond rules already on the books. We base our illustrative estimates on an array of emission control strategies for different sources. The costs and benefits identified in this RIA will not be realized until specific controls are mandated by SIPs or other Federal regulations. In short, NAAQS RIAs hypothesize, but do not prescribe, the control strategies that States may choose to enact when implementing a revised NAAQS.

It is important to emphasize that the EPA does not "double count" the costs or the benefits of our rules. Emission reductions achieved under rules that require specific actions from sources—such as MATS—are in the baseline of this NAAQS analysis, as are emission reductions needed to meet the current NAAQS. For this reason, the cost and benefits estimates provided in this RIA and all other NAAQS RIAs should not be added to the estimates for implementation rules.

In calculating the costs, the EPA assumed the application of a significant number of unidentified future controls that would make possible the additional emissions reductions needed for attainment in 2020. EPA used two methodologies—the fixed-cost and hybrid methodologies—for estimating the costs of unidentified future controls, and both approaches assume either that existing technologies can be applied in particular combinations or to specific sources that we currently can't predict or that innovative strategies and new control options make possible the emissions reductions needed for attainment by 2020. Estimates generated by the two approaches do not represent lower- and upper-bound estimates but simply represent estimates generated by two different methodologies. The fixed-cost methodology implicitly assumes that technological change and innovation will result in the availability of additional controls by 2020 that are similar in cost to the higher end of the cost range for current controls. The hybrid methodology implicitly assumes that while additional controls become available by 2020, they become available at an increasing cost and the increasing cost varies by geographic area and by degree of difficulty associated with obtaining the needed emissions reductions.

For the revised annual standard of 12 μg/m^3, the total cost estimates comprise between 90 and 97% extrapolated cost estimates, and the estimated total cost using the hybrid methodology is roughly 6.5 times more than the estimated total cost using the fixed-cost methodology. Because the hybrid methodology reflects increasing marginal costs in areas needing a higher ratio of emissions reductions from unknown to known controls, it could be more representative of total costs. In an effort to consider the potential fitness of the extrapolated cost estimates, we reviewed the South Coast Air Quality Management District's (SCAQMD) 2012 Air Quality Management Plan (AQMP), and we located data on recent emission reduction credit (ERC) transactions in both the SCAQMD and San Joaquin Valley Air Pollution Control District (SJV APCD). While this information provides context for the extrapolated cost estimates, the current relationship between available controls and costs to reduce emissions may or may not be applicable in 2020 because of changes in innovation and advances in technology.

The SCAQMD's 2012 AQMP includes information on control measures to meet the current 24-hour standard of 35 μg/m^3, including further $PM_{2.5}$ controls for under-fired charbroilers at a cost per ton reduced of $15,000. This control cost matches the parameter used in the fixed-cost methodology, as well as the initial value used for the hybrid methodology and is supportive of our selection of that value. In addition, the California Air Resources Board's 2009 and 2010 *Emission Reduction Offset Transaction Costs, Summary Report* included PM_{10} ERC prices in both the SCAQMD and the SJV APCD. To some degree, ERC transaction prices reflect a choice between installing a more stringent control and purchasing ERCs. Between 2009 and 2010 PM_{10} ERC prices in SJV APCD ranged from $40,000 per ton per year (tpy) to $70,000/tpy, and PM_{10} ERC prices in the SCAQMD ranged from $575,000/tpy to more than $1.9 million/tpy. These prices reflect both marginal costs that are higher than the fixed-cost estimates and marginal costs that are not inconsistent with the higher cost estimates generated using the hybrid methodology. For further discussion of the total cost estimates, refer to Section 7.2.4 in Chapter 7 of this RIA.

Furthermore, the monetized benefits estimates presented in this RIA are not intended to capture the full burden of PM to public health but rather represent the incremental benefits expected upon attaining the revised annual primary standard of 12 μg/m^3. In comparison, modeling by Fann et al. (2012) estimated that 2005 levels of air pollution were responsible for between 130,000 and 320,000 $PM_{2.5}$-related deaths, or between 6.1% and 15% of total deaths from all causes in the continental United States. The monetized benefits associated with attaining the proposed range of standards appear modest when viewed within the context of the potential overall public health burden of $PM_{2.5}$ and ozone air pollution estimated by Fann et al. (2012), but this is primarily because regulations already on the books will make great strides toward reducing future levels of PM. One important distinction between the total public health burden estimated for 2005 air pollution levels and the estimated benefits in this RIA is that ambient levels of $PM_{2.5}$ will have improved substantially by 2020, due to major emissions reductions resulting from implementation of Federal regulations. For example, we estimate that SO_2 emissions (an important $PM_{2.5}$ precursor) in the United States would fall from 14 million tons in 2005 to less than 5 million tons by 2020 (a reduction of 66%). For this reason, States will only need to achieve small air quality improvements to reach the proposed PM standards. As shown in the recent RIA for MATS (U.S. EPA, 2011b), implementing other Federal and State air quality actions will address a substantial fraction of the total public health burden of $PM_{2.5}$ and ozone air pollution.

The NAAQS are not set at levels that eliminate the risk of air pollution. Instead, the Administrator sets the NAAQS at a level requisite to protect public health with an adequate margin of safety, taking into consideration effects on susceptible populations based on the scientific literature. The risk analysis prepared in support of this PM NAAQS reported risks below these levels, while acknowledging that the confidence in those effect estimates is higher at levels closer to the standard (U.S. EPA, 2010c). While benefits occurring below the standard are assumed to be more uncertain than those occurring above the standard, the EPA considers these to be legitimate components of the total benefits estimate. Although there are greater uncertainties at lower $PM_{2.5}$ concentrations, there is no evidence of a population-level threshold in $PM_{2.5}$-related health effects in the epidemiology literature.

Lastly, the EPA recognizes that there are uncertainties in both the cost and benefit estimates provided in this RIA. The EPA was unable to monetize fully all of the benefits associated with reaching these standards in this RIA, including other health effects of PM, visibility effects, ecosystem effects, and climate effects. If the EPA were able to monetize all of the benefits, the benefits would exceed the estimated costs by an even greater margin.

8.4 References

Abt Associates, Inc. 2012. *BenMAP User's Manual Appendices*. Prepared for U.S. Environmental Protection Agency Office of Air Quality Planning and Standards. Research Triangle Park, NC. September.

Fann, N., A. Lamson, K. Wesson, D. Risley, S.C. Anenberg, and B.J. Hubbell. 2012. "Estimating the National Public Health Burden Associated with Exposure to Ambient $PM_{2.5}$ and Ozone. Risk Analysis." *Risk Analysis* 32(1): 81-95.

Krewski, D., R.T. Burnett, M.S. Goldbert, K. Hoover, J. Siemiatycki, M. Jerrett, M. Abrahamowicz, and W.H. White. 2009. "Reanalysis of the Harvard Six Cities Study and the American Cancer Society Study of Particulate Air Pollution and Mortality." Special Report to the Health Effects Institute. Cambridge MA. July.

Lepeule, J., F. Laden, D. Dockery, and J. Schwartz. 2012. "Chronic Exposure to Fine Particles and Mortality: An Extended Follow-Up of the Harvard Six Cities Study from 1974 to 2009." *Environ Health Perspect.* In press. Available at: http://dx.doi.org/10.1289/ehp.1104660.

Pope, C.A. III, E. Majid, and D. Dockery. 2009. "Fine Particle Air Pollution and Life Expectancy in the United States." *New England Journal of Medicine* 360: 376–386.

U.S. Environmental Protection Agency (U.S. EPA). 2006. *Regulatory Impact Analysis, 2006 National Ambient Air Quality Standards for Particulate Matter, Chapter 5*. Office of Air Quality Planning and Standards, Research Triangle Park, NC. October. Available at: http://www.epa.gov/ttn/ecas/regdata/RIAs/Chapter%205—Benefits.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2009. *Integrated Science Assessment for Particulate Matter (Final Report)*. EPA-600-R-08-139F. National Center for Environmental Assessment—RTP Division. December. Available at: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=216546.

U.S. Environmental Protection Agency (U.S. EPA). 2010*. Quantitative Health Risk Assessment for Particulate Matter—Final Repor*t. EPA-452/R-10-005. Office of Air Quality Planning and Standards, Research Triangle Park, NC. September. Available on the Internet at <http://www.epa.gov/ttnnaaqs/standards/pm/data/PM_RA_FINAL_June_2010.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2011a. *Regulatory Impact Analysis for the Federal Implementation Plans to Reduce Interstate Transport of Fine Particulate Matter and Ozone in 27 States; Correction of SIP Approvals for 22 States*. June. Available at: http://www.epa.gov/airtransport/pdfs/FinalRIA.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011b. *Regulatory Impact Analysis for the Final Mercury and Air Toxics Standards.* EPA-452/R-11-011. December. Available at: http://www.epa.gov/ttn/ecas/regdata/RIAs/matsriafinal.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011c. *The Benefits and Costs of the Clean Air Act 1990 to 2020: EPA Report to Congress*. Office of Air and Radiation, Office of Policy, Washington, DC. March. Available at: http://www.epa.gov/oar/sect812/feb11/fullreport.pdf.

U.S. Environmental Protection Agency (U.S. EPA). 2011d. *Policy Assessment for the Review of the Particulate Matter National Ambient Air Quality Standards*. EPA-452/D-11-003. April. Available at: http://www.epa.gov/ttnnaaqs/standards/pm/s_pm_2007_pa.html.

CHAPTER 9
STATUTORY AND EXECUTIVE ORDER REVIEWS

9.1 Synopsis

This chapter summarizes the Statutory and Executive Order (EO) impact analyses relevant for the PM NAAQS Regulatory Impact Analysis. For each EO and Statutory requirement we describe both the requirements and the way in which our analysis addresses these requirements.

9.2 Executive Order 12866: Regulatory Planning and Review

Under section 3(f)(1) of Executive Order 12866 (58 FR 51735, October 4, 1993), this action is an "economically significant regulatory action” because it is likely to have an annual effect on the economy of $100 million or more. The $100 million threshold can be triggered by either costs or benefits, or a combination of them. Accordingly, the EPA submitted this action to the Office of Management and Budget (OMB) for review under Executive Orders 12866 and 13563 (76 FR 3821, January 21, 2011), and any changes made in response to OMB recommendations have been documented in the docket for this action.

9.3 Paperwork Reduction Act

This action does not impose an information collection burden under the provisions of the Paperwork Reduction Act, 44 U.S.S. 3501 et seq. Burden is defined at 5 CFR 1320.3(b). There are no information collection requirements directly associated with revisions to a NAAQS under section 109 of the CAA.

9.4 Regulatory Flexibility Act

The Regulatory Flexibility Act (RFA) generally requires an agency to prepare a regulatory flexibility analysis of any rule subject to notice and comment rulemaking requirements under the Administrative Procedure Act or any other statute unless the agency certifies that the rule will not have a significant economic impact on a substantial number of small entities. Small entities include small businesses, small organizations, and small governmental jurisdictions.

For purposes of assessing the impacts of this rule on small entities, small entity is defined as: (1) a small business that is a small industrial entity as defined by the Small Business Administration’s (SBA) regulations at 13 CFR 121.201; (2) a small governmental jurisdiction that is a government of a city, county, town, school district or special district with a population of less than 50,000; and (3) a small organization that is any not-for-profit enterprise which is independently owned and operated and is not dominant in its field.

After considering the economic impacts of this final rule on small entities, I certify that this action will not have a significant economic impact on a substantial number of small entities. This final rule will not impose any requirements on small entities. Rather, this rule establishes national standards for allowable concentrations of particulate matter in ambient air as required by section 109 of the CAA. See also American Trucking Associations v. EPA. 175 F.3d at 1044-45 (NAAQS do not have significant impacts upon small entities because NAAQS themselves impose no regulations upon small entities).

9.5 Unfunded Mandates Reform Act

This action contains no Federal mandates under the provisions of Title II of the Unfunded Mandates Reform Act of 1995 (UMRA), 2 U.S.C. 1531-1538 for state, local, or tribal governments or the private sector. The action imposes no enforceable duty on any state, local or tribal governments or the private sector. Therefore, this action is not subject to the requirements of sections 202 or 205 of the UMRA.

This action is also not subject to the requirements section 205 of the UMRA because it contains no regulatory requirements that might significantly or uniquely affect small governments. This action imposes no new expenditure or enforceable duty on any state, local, or tribal governments or the private sector, and the EPA has determined that this rule contains no regulatory requirements that might significantly or uniquely affect small governments.

Furthermore, in setting a NAAQS, the EPA cannot consider the economic or technological feasibility of attaining ambient air quality standards although such factors may be considered to a degree in the development of state plans to implement the standards. See also American Trucking Associations v. EPA, 175 F. 3d at 1043 (noting that because the EPA is precluded from considering costs of implementation in establishing NAAQS, preparation of a Regulatory Impact Analysis pursuant to the Unfunded Mandates Reform Act would not furnish any information which the court could consider in reviewing the NAAQS). The EPA acknowledges, however, that any corresponding revisions to associated SIP requirements and air quality surveillance requirements, 40 CFR part 51 and 40 CFR part 58, respectively, might result in such effects. Accordingly, the EPA will address, as appropriate, unfunded mandates if and when it proposes any revisions to 40 CFR parts 51 or 58.

9.6 Executive Order 13132: Federalism

This action does not have federalism implications. It will not have substantial direct effects on the states, on the relationship between the national government and the states, or on the distribution of power and responsibilities among the various levels of government, as specified in Executive Order 13132. The rule does not alter the relationship between the Federal government and the states regarding the establishment and implementation of air quality improvement programs as codified in the CAA. Under section 109 of the CAA, the EPA is mandated to establish and review NAAQS; however, CAA section 116 preserves the rights of states to establish more stringent requirements if deemed necessary by a state. Furthermore, this final rule does not impact CAA section 107 which establishes that the states have primary responsibility for implementation of the NAAQS. Finally, as noted in section D (above) on UMRA, this rule does not impose significant costs on state, local, or Tribal governments or the private sector. Thus, Executive Order 13132 does not apply to this action.

However, as also noted in section D (above) on UMRA, the EPA recognizes that states will have a substantial interest in this rule and any corresponding revisions to associated air quality surveillance requirements, 40 CFR part 58.

9.7 Executive Order 13175: Consultation and Coordination with Indian Tribal Governments

Executive Order 13175, entitled ‘‘Consultation and Coordination with Indian Tribal Governments’’ (65 FR 67249, November 9, 2000), requires the EPA to develop an accountable process to ensure ‘‘meaningful and timely input by tribal officials in the development of regulatory policies that have tribal implications.” This rule concerns the establishment of national standards to address the health and welfare effects of particulate matter. Historically, the EPA’s definition of “tribal implications” has been limited to situations in which it can be shown that a rule has impacts on the tribes’ ability to govern or implications for tribal sovereignty. Based on this historic definition, this action does not have Tribal implications, as specified in Executive Order 13175 (65 FR 67249, November 9, 2000), i.e. because it does not have a substantial direct effect on one or more Indian tribes, since tribes are not obligated to adopt or implement any NAAQS. Nevertheless, we were aware that many tribes would be interest in this rule and we undertook a number of outreach activities to inform tribes about the PM NAAQS review and offered to two consultations with tribes.

Although Executive Order 13175 does not apply to this rule, the EPA undertook a consultation process including: prior to proposal on March 29,2012 we sent letters to tribal leadership inviting consultation on the rule and then sent a second round of letters offering consultation after the proposal was issued on June 29, 2012. We conducted outreach and information calls to tribal environmental staff on May 9, 2012; June 15, 2012; and August 1, 2012. We also participated on the National Tribal Air Association call on June 28, 2012.

As a result we received comments from the National Tribal Air Association, the Southern Ute Mountain Ute Tribe, and the Navajo Nation EPA.

9.8 Executive Order 13045: Protection of Children from Environmental Health and Safety Risks

This action is subject to Executive Order 13045 (62 FR 19885, April 23, 1997) because it is an economically significant regulatory action as defined by Executive Order 12866, and the EPA believes that the environmental health or safety risk addressed by this action may have a disproportionate effect on children. Accordingly, we have evaluated the environmental health or safety effects of PM exposures on children. The protection offered by these standards may be especially important for children because childhood represents a lifestage associated with increased susceptibility to PM-related health effects. Because children have been identified as an at-risk population, we have carefully evaluated the environmental health effects of exposure to PM pollution among children. Discussions of the results of the evaluation of the scientific evidence and policy considerations pertaining to children are contained in sections III.B, III.D, III.E, IV.B, and IV.C of the rule's preamble.

9.9 Executive Order 13211: Actions that Significantly Affect Energy Supply, Distribution or Use

This action is not a "significant energy action" as defined in Executive Order 13211, (66 FR 28355, May 22, 2001) because it is not likely to have a significant adverse effect on the supply, distribution, or use of energy. The purpose of this action concerns the review of the NAAQS for PM. The action does not prescribe specific pollution control strategies by which these ambient standards will be met. Such strategies are developed by states on a case-by-case basis, and the EPA cannot predict whether the control options selected by states will include regulations on energy suppliers, distributors, or users.

9.10 National Technology Transfer and Advancement Act

Section 12(d) of the National Technology Transfer and Advancement Act of 1995 (NTTAA), Public Law 104–113, section 12(d) (15 U.S.C. 272 note) directs the EPA to use voluntary consensus standards in its regulatory activities unless to do so would be inconsistent with applicable law or otherwise impractical. Voluntary consensus standards are technical standards (e.g., materials specifications, test methods, sampling procedures, and business practices) that are developed or adopted by voluntary consensus standards bodies. The NTTAA directs the EPA to provide Congress, through OMB, explanations when the Agency decides not to use available and applicable voluntary consensus standards.

This final rulemaking involves technical standards for environmental monitoring and measurement. Specifically, the EPA proposes to retain the indicators for fine (PM2.5) and coarse (PM10) particles. The indicator for fine particles is measured using the Reference Method for the Determination of Fine Particulate Matter as PM2.5 in the Atmosphere (appendix L to 40 CFR part 50), which is known as the PM2.5 FRM, and the indicator for coarse particles is measured using the Reference Method for the Determination of Particulate Matter as PM10 in the Atmosphere (appendix J to 40 CFR part 50), which is known as the PM10 FRM.

To the extent feasible, the EPA employs a Performance-Based Measurement System (PBMS), which does not require the use of specific, prescribed analytic methods. The PBMS is defined as a set of processes wherein the data quality needs, mandates or limitations of a program or project are specified, and serve as criteria for selecting appropriate methods to meet those needs in a cost-effective manner. It is intended to be more flexible and cost effective for the regulated community; it is also intended to encourage innovation in analytical technology and improved data quality. Though the FRM defines the particular specifications for ambient monitors, there is some variability with regard to how monitors measure PM, depending on the type and size of PM and environmental conditions. Therefore, it is not practically possible to fully define the FRM in performance terms to account for this variability. Nevertheless, our approach in the past has resulted in multiple brands of monitors being approved as FRM for PM, and we expect this to continue. Also, the FRMs described in 40 CFR part 50 and the equivalency criteria described in 40 CFR part 53, constitute a performance-based measurement system for PM, since methods that meet the field testing and performance criteria can be approved as FEMs. Since finalized in 2006 (71 FR, 61236, October 17, 2006) the new field and performance criteria for approval of PM2.5 continuous FEMs has resulted in the approval of six approved FEMs. In summary, for measurement of PM2.5 and PM10, the EPA relies on both FRMs and FEMs, with FEMs relying on a PBMS approach for their approval. The EPA is not precluding the use of any other method, whether it constitutes a voluntary consensus standard or not, as long as it meets the specified performance criteria.

9.11 Executive Order 12898: Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations

Executive Order 12898 (59 FR 7629, February 16, 1994) establishes federal executive policy on environmental justice. Its main provision directs federal agencies, to the greatest extent practicable and permitted by law, to make environmental justice part of their mission by identifying and addressing, as appropriate, disproportionately high and adverse human health or environmental effects of their programs, policies, and activities on minority populations and low-income populations in the United States.

The EPA maintains an ongoing commitment to ensure environmental justice for all people, regardless of race, color, national origin, or income. Ensuring environmental justice means not only protecting human health and the environment for everyone, but also ensuring that all people are treated fairly and are given the opportunity to participate meaningfully in the development, implementation, and enforcement of environmental laws, regulations, and policies.

The EPA has identified potential disproportionately high and adverse effects on minority and/or low-income populations related to PM2.5 exposures. In addition, the EPA has identified persons from lower socioeconomic strata as an at-risk population for PM-related health effects. As a result, the EPA has carefully evaluated the potential impacts on low-income and minority populations as discussed in section III.E.3.a of the rule's preamble. Based on this evaluation and consideration of public comments on the proposal, the EPA is eliminating the spatial averaging provisions as part of the form of the annual standard to avoid potential disproportionate impacts on at-risk populations. The Agency expects this final rule will lead to the establishment of uniform NAAQS for PM. The Integrated Science Assessment and Policy Assessment contain the evaluation of the scientific evidence and policy considerations that pertain to these populations. These documents are available as described in the Supplementary Information section of the rule's preamble and copies of all documents have been placed in the public docket for this action.

CHAPTER 10
QUALITATIVE DISCUSSION OF EMPLOYMENT IMPACTS OF AIR QUALITY REGULATIONS

10.1 Introduction

Executive Order 13563 states that federal agencies should consider the effect of regulations on employment. According to the Executive Order, "our regulatory system must protect public health, welfare, safety, and our environment while promoting economic growth, innovation, competitiveness, and job creation. It must be based on the best available science" (Executive Order 13563, 2011). Although a stand-alone analysis of employment impacts is not typically included in a standard cost-benefit analysis,[1] employment impacts are currently of particular concern due to recent economic conditions reflecting relatively high levels of unemployment. This chapter provides a context for considering the potential influence of environmental regulation on growth and job shifts in the U.S. economy. Section 10.2 addresses the particular influence of this proposed rule on employment. Section 10.3 presents a descriptive overview of the peer-reviewed literature relevant to evaluating the effect of air quality regulation on employment. Finally, in Section 10.4, we offer several conclusions.

10.2 Influence of NAAQS Controls on Employment

Peer-reviewed econometric studies that estimate the impact of air quality regulation on net overall employment and within the regulated sector converge on the finding that any net employment effects, whether positive or negative, have been small. This finding holds for even major nationwide environmental regulations. Therefore, given the overall small effect environmental regulations have been shown to have on net employment in the regulated sectors, we do not expect them to have a significant impact on the overall economy.

Estimating specific employment impacts from a new NAAQS standard is particularly challenging for two reasons. First, the NAAQS targets a level of public health protection that individual areas have flexibility to meet in a variety of ways, and the primary regulatory activity and implementation occur at the state or local level. Under these circumstances, states and localities are given considerable flexibility in choosing which strategies to adopt to meet the NAAQS target. State and local officials can consider a variety of economic impacts including employment impacts of various control strategies, as well as other factors, when designing their state implementation plans (SIPs). This makes it challenging to predict how specific sectors will be impacted and how those impacts vary across regions of the country. Analyses in the RIA are based on a particular NAAQS compliance scenario that reflects assumptions about control

[1] This is the case except to the extent that labor costs are part of total costs in a cost-benefit analysis.

measures applied across all sectors and locations, specific control strategies adopted by the states, and associated extrapolated costs. EPA believes this compliance scenario supports reasonably illustrative quantitative estimates of the potential overall economic effects of the revised NAAQS. However, EPA does not consider this illustrative, aggregate compliance scenario to be sufficiently certain and precise to support quantitative projections of outcomes in particular locations, sectors, or markets, including labor markets, in light of the scarcity of applicable studies that can be used to generate such estimates. Therefore, this RIA does not include quantitative projections of aggregate shifts in employment.

Second, we anticipate that national employment levels will be changing during the period that the NAAQS is being implemented, a period that may be greater than 10 years for some areas, following designations of nonattainment. Although current unemployment rates remain high relative to historical averages largely due to the sharp increase in unemployment that began in early 2008 (U.S. Department of Labor, Bureau of Labor Statistics, 2012a), current data suggest unemployment rates have been declining in recent months (U.S. Department of Labor, Bureau of Labor Statistics, 2012b). Policies to meet the NAAQS in all areas will not go into effect for several years. By this time, we anticipate the economy will have had a chance to recover toward higher employment levels that more closely approximate full employment. In addition, over a period of 10 years or longer, potentially significant changes in technology, growth and distribution of economic activities, and other key determinants of local and national labor market conditions further complicate projections of future employment and the potential incremental effect of regulatory programs.

Although a quantitative assessment of employment consequences of today's proposed revision to the national ambient PM standards remains beyond the reach of available data and modeling tools, EPA is in the process of supporting the development of tools and research that could assist in the future. In the interim, some insights on the potentially relevant consequences of revising ambient air pollution standards can be gained by considering currently available literature, including its limitations. In light of these challenges, Section 10.3 focuses on qualitative insights from currently available peer-reviewed literature on the impact of air quality regulations in general.

10.3 The Current State of Knowledge Based on the Peer-Reviewed Literature

There is limited peer-reviewed econometric literature estimating employment effects of environmental regulations. We present an overview here, highlighting studies with particular relevance for NAAQS. Determining the direction of employment effects in the regulated industries is challenging because of competing effects. Complying with the new or more stringent regulation requires additional inputs, including labor, and may alter the relative proportions of labor and capital used by regulated firms in their production processes.

When the economy is at full employment, an environmental regulation is unlikely to have a considerable impact on net employment in the long run. Instead, labor would primarily be reallocated from one productive use to another (e.g., from producing electricity or steel to producing pollution abatement equipment). Theory supports the argument that, in the case of full employment, the net national employment effects from environmental regulation are likely to be small and transitory (e.g., as workers move from one job to another). There is reason to believe that when the economy is operating at less than full employment environmental regulation could result in a short-run net increase in employment.[2] Several empirical studies suggest that net employment impacts may be positive but small even in the regulated sector. Taken together, the peer-reviewed literature does not contain evidence that environmental regulation would have a notable impact on net employment across the whole economy.

This discussion focuses on both short- and long-term employment impacts in the regulated industries, as well as on the environmental protection sector for construction of needed pollution control equipment prior to the compliance date of the regulation. EPA is committed to using the best available science and the relevant theoretical and empirical literature in this assessment and is pursuing efforts to support new research in this field.

10.3.1 Immediate and Short-Run Employment Impacts

Environmental regulations are typically phased in to allow firms time to invest in the necessary technology and process changes to meet the new standards. Whatever effects a regulation will have on employment in the regulated sector will typically occur only after a regulation takes effect or in the long term, as new technologies are introduced. However, the environmental protection sector (pollution control equipment) often sees immediate employment effects. When a regulation is promulgated, the first response of industry is to order pollution control equipment and services to comply with the regulation when it becomes effective. This can produce a short-term increase in labor demand for specialized workers within the environmental protection sector related to design, construction, installation, and operation of the new pollution control equipment required by the regulation (see Schmalansee and Stavins, 2011; Bezdek, Wendling, and Diperna, 2008).

---

[2] See Schmalansee and Stavin (2011).

As the NAAQS are implemented, it is possible that the regulated sector will experience short-run changes in employment. Because it is the states' responsibility to design their SIPs over the next few years, we cannot assess the short-term effects of those SIPs on the regulated sector with sufficient precision to quantify the resulting incremental effects on employment. However, as previously noted, even in a full employment case, there may be transitory effects as workers change jobs. Some workers may need to retrain or relocate in anticipation of the new requirements or require time to search for new jobs, while shortages in some sectors or regions could bid up wages to attract workers.

It is important to recognize that these adjustment costs can entail local labor disruptions, and, although the net change in the national workforce might be small, gross reductions in employment can still have negative impacts on individuals and communities. The peer-reviewed literature that is currently available is focused on medium- and long-term employment impacts and does not offer much insight into the short-term balance between increased employment in the environmental protection sector and possible decreased employment in some regulated sectors.

10.3.2 Long-Term Employment Impacts on the Regulated Industry

Determining the direction of net employment effects in regulated industries is challenging because of competing effects. Morgenstern, Pizer, and Shih (2002) discuss how environmental regulations can be understood as requiring regulated firms to add a new output (environmental quality) to their product mix. Although legally compelled to produce this new output, regulated firms have to finance this additional production input with the proceeds of sales of their other (market) products. The current literature on employment impacts of air quality regulations can be disaggregated into two types of approaches or models: 1) structural and 2) reduced-form models. Two papers that present a formal structural model of the underlying profit-maximizing/cost-minimizing problem of the firm are Berman and Bui (2001) and Morgenstern, Pizer, and Shih (2002). Berman and Bui (2001) developed an innovative approach to estimating the effect of environmental regulations designed to meet a NAAQS (e.g., ozone and NO_X) requirement in California on employment. Berman and Bui's model allows environmental regulation to operate via two separate mechanisms: 1) the output elasticity of labor demand and 2) the effect of pollution abatement activities on demand for variable factors, combined with the marginal rates of technical substitution between abatement activity and variable factors, including labor. Berman and Bui show how Neoclassical economic theory predicts that the output effect is, in most cases, negative, while the direction of the second, composite effect is indeterminate, making the overall net effect ambiguous.

Morgenstern, Pizer, and Shih (2002) developed a similar structural model to Berman and Bui's (2001) model. Their model focuses on three mechanisms whereby environmental regulation may impact employment in regulated industries. The first mechanism is the demand, or output, effect, where new compliance costs increase the cost of production, raising prices and thereby reducing consumer demand, which, in turn, reduces labor demand. The second mechanism is the cost effect, which increases the demand for inputs, including labor, because more inputs are now required to produce the same amount of output. Finally, the factor-shift effect notes how regulated firms' production technologies may be more or less labor intensive after complying with the regulation (i.e., more/less labor is required relative to capital per dollar of output), implying an ambiguous overall net effect on labor demand. Conceptually, this theoretical approach, which is very similar to Berman and Bui's approach, could be applied to NAAQS. However, Morgenstern et al.'s empirical approach uses pollution abatement expenditures for only four highly polluting/regulated sectors (pulp and paper, plastics, steel, and petroleum refining) to estimate effects on net employment; therefore, their empirical results are not directly applicable to the full range of manufacturing and nonmanufacturing industries affected by NAAQS. Regardless, their work represents one of the most rigorous attempts to quantify the net employment impacts of regulation on the regulated sector. Morgenstern et al. conclude from their empirical results that increased pollution abatement expenditures generally have *not* caused a significant change in net employment in those four sectors. More specifically, their results suggest that, on average across the industries studied, each additional $1 million ($1987) spent on pollution abatement resulted in a (statistically insignificant) net increase of 1.5 jobs.

Berman and Bui (2001) use their model to empirically examine how an increase in local air quality regulation that reduces NO_X emissions as a precursor to ozone and PM_{10} affects manufacturing employment in the South Coast Air Quality Management District (SCAQMD), which incorporates Los Angeles and its suburbs. During the time frame of their study, 1979 to 1992, the SCAQMD enacted some of the country's most stringent air quality regulations. Using SCAQMD's local air quality regulations, which are more stringent than federal and state regulations, Berman and Bui identify the effect of environmental regulations on net employment in the regulated sectors.[3] They compare changes in employment in affected plants to those in other plants in the same industries but in regions not subject to the local regulations. The authors find that "while regulations do impose large costs, they have a limited effect on employment"—even when exit and dissuaded entry effects are considered (Berman

[3] Note, like Morgenstern, Pizer, and Shih (2002), this study does not estimate the number of jobs created in the environmental protection sector.

and Bui, 2001, p. 269). Their conclusion is that local air quality regulation “probably increased labor demand slightly” but that “the employment effects of both compliance and increased stringency are *fairly precisely estimated zeros* [emphasis added], even when exit and dissuaded entry effects are included” (Berman and Bui, 2001, p. 269). In their view, the limited effects likely arose because 1) the regulations applied disproportionately to capital-intensive plants with relatively little employment, 2) the plants sold to local markets where competitors were subject to the same regulations (so that sales were relatively unaffected), and 3) abatement inputs served as complements to employment. Although Berman and Bui focus on more sectors than Morgenstern et al. and focus specifically on air regulations, the study only examined impacts in Southern California and impacts may differ in other nonattainment areas.

Other studies, including Henderson (1996), Becker and Henderson (2000), Greenstone (2002), and List et al. (2003), have taken a reduced-form approach to ask a related but quite different question regarding the impact of environmental regulation on economic activity. All of these studies examined the effect of attainment status, with respect to NAAQS, on various forms of economic activity (e.g., employment growth, plant openings and closings, investment). Polluting plants already located in and new polluting plants wanting to open in nonattainment counties (counties not in compliance with one or more NAAQSs) are likely to face more stringent air pollution regulations to help bring them into compliance. Thus, the stringency in environmental regulations may vary spatially, which may affect the spatial distribution of economic activity but not necessarily the overall level of economic activity. These studies find limited evidence that employment grows more slowly, investment is lower, or fewer new polluting plants open in nonattainment areas relative to attainment areas. However, this evidence does not mean that there is less aggregate economic activity as a result of environmental regulation nor does it provide evidence regarding absolute growth rates; it simply suggests that the relative growth rate of some sectors may differ between attainment and nonattainment areas. The approach used in all of these other studies is not capable of estimating net employment effects as would be necessary for a national rulemaking, only certain aspects of gross labor flows in selected areas.

10.4 Conclusion

The long-term effects of a regulation on the environmental protection sector (which provides goods and services to the regulated sector) are difficult to assess. Employment in the industry supplying pollution control equipment is likely to increase with the increased demand from the regulated industry for the equipment.[4] According to U.S. Department of Commerce

[4] See Bezdek, Wendling, and Diperna (2008), for example, and U.S. Department of Commerce (2010).

(2010) data, by 2008, there were 119,000 environmental technology (ET) firms generating approximately $300 billion in revenues domestically (2% of national gross domestic product), producing $43.8 billion in exports (2% of total exports), and supporting nearly 1.7 million jobs (0.93% of total jobs). Air pollution control accounted for 18% of the domestic ET market and 16% of exports. Small and medium-size companies represent 99% of private ET firms, producing 20% of total revenue. The remaining 1% of companies are large companies supplying 49% of ET revenue (OEEI, 2010).[5]

As described above, deriving estimates of how regulations will impact economy-wide net employment is a difficult task, especially in the case of setting a new NAAQS, given that economic theory predicts that the net effect of an environmental regulation on regulated sectors and the overall economy is indeterminate (not necessarily positive or negative). Peer-reviewed econometric studies that use a structural approach, applicable to overall net effects in the regulated sectors, converge on the finding that any net employment effects of environmental regulation in general, whether positive or negative, have been small and have not affected employment in the national economy in a significant way.

10.5 References

Becker, R. and V. Henderson (2000). "Effects of Air Quality Regulations on Polluting Industries." *Journal of Political Economy 108*(2): 379-421.

Berman, E. and L. T. M. Bui (2001). "Environmental Regulation and Labor Demand: Evidence from the South Coast Air Basin." *Journal of Public Economics 79*(2): 265-295.

Bezdek, R. H., R. M. Wendling, and P. Diperna (2008). "Environmental Protection, the Economy, and Jobs: National and Regional Analyses." *Journal of Environmental Management 86*(1): 63-79.

Executive Order 13563 (January 21, 2011). "Improving Regulation and Regulatory Review. Section 1. General Principles of Regulation." *Federal Register 76*(14): 3821-3823.

Greenstone, M. (2002). "The Impacts of Environmental Regulations on Industrial Activity: Evidence from the 1970 and 1977 Clean Air Act Amendments and the Census of Manufactures." *Journal of Political Economy 110*(6): 1175-1219.

Henderson, V. (1996). "Effects of Air Quality Regulation." *The American Economic Review 86*(4): 789-813.

[5] To calculate the percentages, total national 2008 GDP ($14,369.1 billion), exports ($1,842.68 billion), and employment (181.75 million employees) were obtained from Bureau of Economic Analysis, U.S. Census Bureau, and Woods & Poole, respectively.

List, J. A., D. L. Millimet, P. G. Fredriksson, and W. W. McHone (2003). “Effects of Environmental Regulations on Manufacturing Plant Births: Evidence from a Propensity Score Matching Estimator.” *The Review of Economics and Statistics 85*(4): 944-952.

Morgenstern, R. D., W. A. Pizer, and J-S. Shih (2002). “Jobs Versus the Environment: An Industry-Level Perspective.” *Journal of Environmental Economics and Management 43*(3): 412-436.

Schmalansee, R. and R. Stavins (2011). “A Guide to Economic and Policy Analysis for the Transport Rule.” White Paper. Boston, MA: Exelon Corp.

U.S. Department of Commerce, International Trade Administration (2010, April). Environmental Industries Fact Sheet, using 2008 data from Environmental Business International, Inc. http://web.ita.doc.gov/ete/eteinfo.nsf/068f3801d047f26e85256883006ffa54/4878b7e2fc08ac6d85256883006c452c?OpenDocument.

U.S. Department of Labor, Bureau of Labor Statistics (2012a). Databases, Tables & Calculators by Subject. http://data.bls.gov/timeseries/LNS14000000. Accessed April 16, 2012.

U.S. Department of Labor, Bureau of Labor Statistics (2012b, April 6). “The Employment Situation—March 2012.” News Release. http://www.bls.gov/news.release/pdf/empsit.pdf. Accessed April 16, 2012.

United States Environmental Protection Agency	Office of Air Quality Planning and Standards Health and Environmental Impacts Division Research Triangle Park, NC	Publication No. EPA-452/R-12-005 December 2012