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

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

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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 proposing to revise the primary (health-based) and secondary (welfare-based) NAAQS for fine particles (generally referring to particles less than or equal to 2.5 micrometers [µm] in diameter—$PM_{2.5}$) to provide requisite protection of public health and welfare, respectively. 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 several alternative $PM_{2.5}$ standards based on one possible set of selected control strategies for reducing direct PM and PM precursor emissions.

In NAAQS rulemakings, the RIA is prepared for informational purposes only, and the proposed decisions on the PM NAAQS discussed in the proposed rulemaking 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] Benefit and cost estimates provided in the RIA are not additive to benefits and costs from other regulations, and 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.

ES.2 Existing and Alternative PM Air Quality Standards

Currently, 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 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 24-hour standard is set at a level of 35 $\mu g/m^3$, based on the 3-year average of the 98th percentile of 24-hour $PM_{2.5}$ concentration. In the RIA, the current primary $PM_{2.5}$ standard, including both annual and 24-hour averaging times is denoted as 15/35.

In the current PM NAAQS review, the EPA is proposing to revise the level of the primary annual $PM_{2.5}$ standard within the range of 12 to 13 $\mu g/m^3$ in conjunction with retaining the level of the 24-hour standard at 35 $\mu g/m^3$. In order to characterize the costs and benefits, it was

[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.

necessary to identify discrete levels along this continuum. For purposes of this analysis, we identified an annual standard of 12 $\mu g/m^3$ in conjunction with retaining the level of the 24-hour standard at 35 $\mu g/m^3$ (denoted 12/35) and an annual standard of 13 $\mu g/m^3$ in conjunction with retaining the level of the 24-hour standard at 35 $\mu g/m^3$ (denoted as and 13/35).

In addition to 12/35 and 13/35, the RIA also analyzes the benefits and costs of incremental control strategies for two other alternative standards (11/35 and 11/30). The four alternative standards analyzed are as follows:

- A revised annual standard level of 13 $\mu g/m^3$ in conjunction with retaining the 24-hour standard level at 35 $\mu g/m^3$ (13/35)
- A revised annual standard level of 12 $\mu g/m^3$ in conjunction with retaining the 24-hour standard level at 35 $\mu g/m^3$ (12/35)
- A revised annual standard level of 11 $\mu g/m^3$ in conjunction with retaining the 24-hour standard level at 35 $\mu g/m^3$ (11/35)
- A revised annual standard level of 11 $\mu g/m^3$ in conjunction with a revised 24-hour standard level at 30 $\mu g/m^3$ (11/30)

In analyzing the current 15/35 standard (baseline), the EPA determined that all counties would meet the 14/35 standard concurrently with meeting the existing 15/35 standard at no additional cost. Consequently, no incremental costs or benefits are associated with 14/35 and therefore, no need to present an analysis of 14/35 in this RIA.

Currently, the existing secondary $PM_{2.5}$ standards are identical in all respects to the primary standards. In the current PM NAAQS review, the EPA is proposing to add a distinct standard for $PM_{2.5}$ to provide protection from PM-related visibility impairment. Specifically, the EPA is proposing to establish a separate secondary standard defined in terms of a $PM_{2.5}$ visibility index, which would use speciated $PM_{2.5}$ mass concentrations and relative humidity data to calculate $PM_{2.5}$ light extinction, similar to the Regional Haze Program; a 24-hour averaging time; a 90th percentile form; and a level of either 30 deciviews (dv) or 28 dv. Based on the air quality analysis conducted for the primary $PM_{2.5}$ standard, all monitored areas are estimated to be in attainment with both proposed secondary standard levels in 2020, assuming full attainment of the primary $PM_{2.5}$ standard. For the two optional levels proposed for the secondary standard, no additional costs or benefits will be realized beyond those quantified for meeting the primary $PM_{2.5}$ standard in this RIA.

With regard to the primary and secondary standards for particles less than or equal to 10 µm in diameter (PM_{10}), the EPA is proposing to retain 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 (EPA, 1997)[2]. Since the benefit-cost analysis of the alternative PM_{10} standards was conducted when the standard was selected, this RIA does not repeat that analysis here.

ES.2.1 Establishing the Baseline

The RIA is intended to evaluate the costs and benefits of reaching attainment with potential alternative $PM_{2.5}$ standards. In order to develop and evaluate control strategies for attaining a more stringent primary standard, it is important to first estimate $PM_{2.5}$ levels in 2020 given the current NAAQS standards (15/35) and air quality trends. Estimating the 2020 levels is known as the baseline. Establishing this baseline allows us to estimate the incremental costs and benefits of attaining any alternative primary standard.

The baseline includes reductions already achieved as a result of national regulations, reductions expected prior to 2020 from recently promulgated national regulations[3] (i.e., reductions that were not realized before 2005 but are expected prior to attainment of the current PM standard), and reductions from additional controls which the 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 EPA by state and local agencies[4]. Two steps were used to develop the 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 3.2.2 for a more detailed discussion of the rules reflected in the base case emissions inventory.

- Light-Duty Vehicle Tier 2 Rule (U.S. EPA, 1999)

[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.

[3] The recently proposed Boiler MACT and CISWI reconsiderations are not included in the base case. These rules were not yet proposed at the time of this analysis. It is not clear how the geographic scope of this rule will match with the counties analyzed for this RIA—the costs may decrease but the magnitude is uncertain.

[4] The amendments to the Low Emissions Vehicle Program (LEV-III) in California are not included in the base case. This program requires an approval of U.S. EPA via a waiver. At the time of this analysis the waiver had not been submitted.

- 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 Emissions 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)
- Cross-State Air Pollution Rule (U.S. EPA, 2011a)
- Mercury and Air Toxics Standards (U.S. EPA, 2011b)
- Reciprocating Internal Combustion Engines (RICE) NESHAPs (U.S. EPA, 2010)

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 NAAQS RIAs are hypothetical. This analysis presents one scenario that states may employ but does not prescribe how attainment must be achieved.

Second, because the base case reductions alone were not predicted to bring all areas into attainment with the current standard (15/35), please see Figures 4-1 and 4-2 in Chapter 4 of this RIA, the EPA used a hypothetical control strategy to apply additional known controls to illustrate attainment with that standard. To establish the baseline, additional control measures were used in two sectors:[5] Non-Electricity Generating Unit Point Sources (Non-EGUs) and Non-Point Area Sources (Area).

[5] In establishing the baseline, the EPA selected a set of cost-effective controls to simulate attainment of the current $PM_{2.5}$ standard. These control sets are hypothetical because states will ultimately determine controls as part of the SIP process.

For addition details on the baseline, refer to Chapter 4 of this RIA.

ES.2.2 Emission Reduction Estimates by Alternative Standards (2020)

Emission reductions were calculated for the four alternative standards (13/35, 12/35, 11/35, and 11/30) from a baseline of attaining the current standard of 15/35. Emission reductions were calculated for the known control strategy analysis and the extrapolated cost analysis for each alternative standard being analyzed. The EPA estimates the national-scale emission reductions for each of the alternative standards as shown in Table ES-1.

Table ES-1. Emission Reduction Estimates by Standard in 2020 (annual tons/year)[a]

Alternative Standard	$PM_{2.5}$	SO_2	NO_x
13/35	190	0	0
12/35	4,300	970	0
11/35	14,000	19,000	1,500
11/30	22,000	23,000	8,200

[a] Estimates are rounded to two significant figures.

ES.2.3 Health and Welfare Benefits Analysis Approach

The EPA estimated human health (e.g., mortality and morbidity effects) under full attainment of the three alternative combinations of primary $PM_{2.5}$ standards. We considered an array of health impacts attributable to changes in $PM_{2.5}$ exposure. 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 American Cancer Society (ACS) cohort (Krewski et al., 2009), updated health endpoints, new morbidity studies, updated hospital cost-of-illness estimates, and an expanded uncertainty assessment. Each of these updates is fully described in the benefits chapter. Even though the alternative 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 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, welfare benefits are not quantified or monetized in this analysis. Unquantified health benefits are discussed in Chapter 5, and unquantified welfare 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 uncertainty, including from the following key assumptions:

1. We assumed that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This assumption 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 effects estimates by particle type.

2. We assumed that the health impact function for fine particles is linear within the range of ambient concentrations under consideration. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5}$, including both regions that are in attainment with the fine particle standard and those that do not meet the standard down to the lowest modeled concentrations.

As noted in the preamble to the proposed rule, the *Policy Assessment* (U.S. EPA, 2011c) concludes that the range from the 25th to 10th percentiles of the air quality data used the epidemiology studies is a reasonable range below which we have appreciably less confidence in the associations observed in the epidemiological studies. In the RIA accompanying the promulgated PM NAAQS, EPA will characterize the distribution of estimated PM-related health benefits attributable to PM reductions occurring above and below the selected standard. For 12/35, we estimate that 51% and 92% of the estimated avoided premature deaths occur at or above an annual mean $PM_{2.5}$ level of 10 µg/m^3 (the lowest measured level (LML) of the Laden et al. 2006 study) and 5.8 µg/m^3 (the LML of the Krewski et al. 2009 study), respectively. For 13/35, these estimates are 62% and 89%. These are the two source studies for the concentration-response functions used to estimate mortality benefits. The EPA briefly describes the uncertainties in the concentration-response functions below and in considerably more detail in the benefits chapter of this RIA.

Although these concentration benchmark analyses (e.g., 25th percentile, 10th percentile, and LML) provide some insight into the level of uncertainty in the estimated $PM_{2.5}$ mortality benefits, EPA does not view these concentration benchmarks as a concentration threshold. The best estimate of benefits includes estimates below and above these concentration benchmarks, but uncertainty is higher in health benefits estimated at lower concentrations, with the lowest confidence below the LML. Estimated health impacts reflecting air quality improvements both below and above these concentration benchmarks are appropriately included in the total benefits estimate. In other words, our increased confidence in the estimated benefits above these concentration benchmarks should not imply an absence of confidence in the benefits estimated below these concentration benchmarks.

It is important to note that these estimated benefits reflect specific control measures and emission reductions that are needed to lower $PM_{2.5}$ concentrations for monitors projected to exceed the alternative standard analyzed. The result is that air quality will improve in counties that exceed the alternative standards as well as surrounding areas that do not exceed the alternative standards. It is not possible to apply controls that only reduce $PM_{2.5}$ at the monitor without affecting surrounding areas. In order to make a direct comparison between the benefits and costs of these control strategies, it is appropriate to include all the benefits occurring as a result of the control strategies applied.

We estimate benefits using modeled air quality data with 12km grid cells, which is important because the grid cells are smaller than counties and $PM_{2.5}$ concentrations 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 value is above the alternative standard. Thus, emission reductions lead to benefits in grid cells that are below the alternative standards even within a county with a monitor that exceeds the alternative standard. We have not estimated the fraction of benefits that occur only in counties that exceed the alternative standards.

ES.2.4 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. 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.

ES.2.5 Comparison of Benefits and Costs

In the analysis, we estimate the net benefits of the proposed range of annual $PM_{2.5}$ standards of 12/35 to 13/35. For 12/35, net benefits are estimated to be \$2.3 billion to \$5.9 billion at a 3% discount rate and \$2.0 billion to \$5.3 billion at a 7% discount rate in 2020 (2006 dollars).[6] For 13/35, net benefits are estimated to be $85 million to $220 million at the 3% discount rate and $76 million to $200 million at the 7% discount rate.

The EPA estimated the net benefits of the alternative annual $PM_{2.5}$ standard of 11/35 to be $8.9 billion to $23 billion at a 3% discount rate and $8.0 billion to $21 billion at a 7% discount rate in 2020. The EPA estimated the net benefits of the alternative annual $PM_{2.5}$ standard of 11/30 to be $14 billion to $36 billion at a 3% discount rate and $13 billion to $33 billion at a 7% discount rate in 2020. All estimates are in 2006$.[7]

In analyzing the current 15/35 standard (baseline), the EPA determined that all counties would meet the 14/35 standard concurrently with meeting the existing 15/35 standard at no additional cost. No incremental costs or benefits are associated with 14/35 and consequently, there is no analysis 14/35 in this RIA.

We provide these results in Table ES-2 and a regional percentage breakdown of costs and benefits in Table ES-3. In Table ES-4, we provide the avoided health incidences associated with these standard levels.

ES.2.6 Conclusions of the Analysis

The EPA's illustrative analysis has estimated the health and welfare benefits and costs associated with the proposed revised 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.

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[6] Using a 2010$ year increases estimated costs and benefits by approximately 8%. Because of data limitations, we were unable to discount compliance costs for all sectors at 3%. As a result, the net benefit calculations at 3% were computed by subtracting the costs at 7% from the monetized benefits at 3%.

[7] Using a 2010 $ year increases estimated costs and benefits by approximately 8%. Because of data limitations, we were unable to discount compliance costs for all sectors at 3%. As a result, the net benefit calculations at 3% were computed by subtracting the costs at 7% from monetized benefits at 3%.

Table ES-2. Total Monetized Benefits, Total Costs, and Net Benefits in 2020 (millions of 2006$[a])—Full Attainment

Alternative Standard	Total Costs: 3% Discount Rate[c]	Total Costs: 7% Discount Rate	Monetized Benefits[b]: 3% Discount Rate	Monetized Benefits[b]: 7% Discount Rate	Net Benefits[b]: 3% Discount Rate[c]	Net Benefits[b]: 7% Discount Rate
13/35	$2.9	$2.9	$88 to $220	$79 to $200	$85 to $220	$76 to $200
12/35	$69	$69	$2,300 to $5,900	$2,100 to $5,400	$2,300 to $5,900	$2,000 to $5,300
11/35	$270	$270	$9,200 to $23,000	$8,300 to $21,000	$8,900 to $23,000	$8,000 to $21,000
11/30	$390	$390	$14,000 to $36,000	$13,000 to $33,000	$14,000 to $36,000	$13,000 to $33,000

[a] Rounded to two significant figures. Using a 2010$ year increases estimated costs and benefits by approximately 8%.

[b] The reduction in premature deaths each year accounts for over 98% of total monetized benefits. Mortality risk valuation assumes discounting over the Science Advisory Board-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.

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

For the lower end of the proposed standard range of 12/35, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 34 to 86 times at a 3% discount rate and 30 to 78 times at a 7% discount rate. For the upper end of the proposed standard range of 13/35, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 30 to 77 times at a 3% discount rate and 27 to 69 times at a 7% discount rate. For the alternative standards, 11/35 and 11/30, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 34 to 94 times at a 3% discount rate and 30 to 85 times at a 7% discount rate.

Table ES-3. Regional Breakdown of Total Costs and Monetized Benefits Results

	Alternative Combination of Standards							
	13 μg/m³ annual & 35 μg/m³ 24-hour		12 μg/m³ Annual & 35 μg/m³ 24-hr		11 μg/m³ Annual & 35 μg/m³ 24-hr		11 μg/m³ Annual & 30 μg/m³ 24-hr	
Region	Total Costs[a]	Monetized Benefits	Total Costs	Monetized Benefits	Total Costs	Monetized Benefits	Total Costs	Monetized Benefits
East[b]	0%	0%	<1%	27%	18%	53%	18%	43%
California	100%	98%	94%	70%	67%	44%	54%	47%
Rest of West	0%	2%	5%	3%	15%	3%	28%	10%

[a] Costs are discounted at 7%.

[b] 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 12/35 or 13/35.

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

	Alternative Combination of Primary $PM_{2.5}$ Standards			
Health Effect	13/35	12/35	11/35	11/30
Adult Mortality				
Krewski et al. (2009)	11	280	1,100	1,700
Laden et al. (2006) (adult)	27	730	2,900	4,500
Woodruff et al. (1997) (infant)	0	1	3	4
Non-fatal heart attacks (age > 18)				
Peters et al. (2001)	11	320	1,300	1,900
Pooled estimate of 4 studies	1	35	140	210
Hospital admissions—respiratory (all ages)	3	98	430	620
Hospital admissions—cardiovascular (age > 18)	3	95	400	580
Emergency department visits for asthma (age < 18)	6	160	730	1,000
Acute bronchitis (age 8–12)	22	540	2,000	3,100
Lower respiratory symptoms (age 7–14)	290	6,900	25,000	39,000
Upper respiratory symptoms (asthmatics age 9–11)	410	9,800	37,000	56,000
Asthma exacerbation (age 6–18)	410	24,000	89,000	140,000
Lost work days (age 18–65)	1,800	44,000	170,000	260,000
Minor restricted-activity days (age 18–65)	11,000	260,000	1,000,000	1,500,000

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

ES.3 Caveats and Limitations

EPA acknowledges several important limitations of the primary and secondary analysis. These include:

ES.3.1 Benefits Caveats

- $PM_{2.5}$ mortality co-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 twelve 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, 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.

- 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 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.

ES.3.2 Control Strategy and Cost Analysis Caveats and Limitations

Control Technology Data

- Technologies applied may not reflect emerging devices that may be available in future years.
- Control efficiency data depend on equipment being well maintained.
- Area source controls assume a constant estimate of emission reductions, despite variability in extent and scale of application.

Control Strategy Development

- States may develop different control strategies than the ones illustrated.
- Data on baseline controls from current SIPs are lacking.
- Timing of control strategies may be different than envisioned in the RIA.
- Controls are applied within the county with the violating monitor. It is possible that additional known controls could be available in a wider geographical area.
- Unknown controls were needed to reach attainment in several counties. Costs associated with these unknown controls were estimated using a fixed-cost per ton methodology as well as an extrapolated cost methodology.
- Emissions reductions from mobile sources, EGUs, other $PM_{2.5}$ precursors (i.e., ammonia and VOC), and voluntary programs are not reflected in the analyses.

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 not accounted for in the emission reduction estimates.
- Future technology developments in sectors not analyzed here (e.g., EGUs) may be transferrable to non-EGU and area sources, making these sources more viable for achieving future attainment at a lower cost than the cost presented in this analysis.

Engineering Cost Estimates

- Because of data limitations, we were unable to discount compliance costs for all sectors at 3%.
- Estimates of private compliance cost are used as a proxy for social cost.

Unquantified Costs

- A number of costs remain unquantified, including administration costs of federal and state SIP programs, and transactional costs.

ES.3.3 Limitations of the Secondary Standard Analysis

Visibility design values for 2020 were calculated using the CMAQ modeling information and 2004-2006 ambient measurements. To determine the design values for meeting the current primary $PM_{2.5}$ standard and proposed alternative primary standards, we used a methodology, described in Chapter 3, to estimate the small emissions reductions needed from control measures to show attainment and to estimate the costs and benefits of attaining the proposed alternative primary standards. It is not possible to apply this methodology to the visibility design values.[8] As a result, the only analysis available for the proposed alternative secondary standards in 2020 is prior to full attainment of the current primary standard. All monitors analyzed are projected to attain a secondary standard of 30 dv in the 2020 base case. Given the 24-hr design value reductions that were included in simulating attainment of 15/35 in the 2020 base case, we are confident in our conclusion that all monitors will also attain a secondary standard of 28 dv when they attain the current primary standards.[9]

ES.4 Discussion

An extensive body of scientific evidence documented in the *Integrated Science Assessment for Particulate Matter* (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 proposed regulation, the proposed changes 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 U.S. would meet an annual standard of 12 μg/m^3 by 2020 without additional

[8]As described in Chapter 3, we apply a methodology of air quality ratios to estimate the emissions reductions needed to meet the current and proposed alternative levels for the primary standard. While this methodology can estimate how these emissions reductions will affect changes in the future-year annual design value and the corresponding response of the future-year 24-hr design value to changes in the annual design value, it is unable to estimate how each of the $PM_{2.5}$ species will change with these emission reductions. Given that estimating changes in future-year visibility is dependent on the IMPROVE equation and how the $PM_{2.5}$ species are projected to change in time, we are unable to estimate visibility design values for meeting the current and proposed alternative levels for the primary standard.

[9] The projected 2020 base case design values for the secondary standard for the following monitors with id numbers 60658001 (located in Riverside, CA), 60290014 (located in Kern, CA), and 60990005 (located in Stanislaus, CA) are 29 dv, 30 dv, and 29 dv, respectively. The emissions reductions selected for simulating attainment of 15/35 in the 2020 base case resulted in the following reductions in the 24-hr design values for these three monitors: 11.1 μg/m^3, 21.9 μg/m^3 and 5.3 μg/m^3, respectively. We believe that these emissions reductions and 24-hr design value changes for simulating the current primary standard levels of 15/35 will be enough to lower the projected 2020 secondary standard design values for these three monitors to 28 dv or lower.

federal, state, or local PM control programs. This demonstrates the substantial progress that the U.S. 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), the Cross-State Air Pollution Rule (CSAPR), 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, by a ratio of at least 30 to 1.

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 State Implementation Plans (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.

Furthermore, the monetized benefits estimates do not paint a complete picture of the burden of PM to public health. For example, 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. For example, we estimate that SO_2 emissions in the U.S. 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 recent RIAs for the CSAPR (U.S. EPA, 2011a) and 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, 2010). 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. Though 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 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 costs by an even greater margin. Even when considered in light of the quantified and unquantified uncertainties identified in this RIA, we believe that implementing the proposed range of standards would have substantial public health benefits that outweigh the costs.

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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 several particulate matter (PM) National Ambient Air Quality Standards (NAAQS) nationwide. According to the Clean Air Act ("Act"), the 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 new NAAQS and to meet the requirements of Executive Orders 12866 and 13563.

1.2 Background

1.2.1 NAAQS

Two sections of the Clean Air Act ("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 $\mu g/m^3$ and retained the annual standard of 15 $\mu g/m^3$. 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 standard 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.

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

1.3.1 Legislative Roles

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 a new standard. The Act requires the EPA, for each criteria pollutant, to set a standard that protects 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 standard, 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] These documents present guidelines for the EPA to assess the benefits and costs of the selected regulatory option as well as more and less stringent options than those proposed or selected. In concordance with these guidelines, the RIA also analyzes the benefits and costs of alternative combinations of primary $PM_{2.5}$ standards, one combination that is more stringent than the existing standards, but less stringent than the proposed standards and another combination that is more stringent than the proposed standards (see Section 1.4.2).

In the current PM NAAQS review, the EPA is proposing to revise the level of the primary annual $PM_{2.5}$ standard within the range of 12 to 13 $\mu g/m^3$ in conjunction with retaining the level of the 24-hour standard at 35 $\mu g/m^3$ (denoted 12/35 and 13/35). In addition to the range of 12/35 to 13/35, the RIA also analyzes the benefits and costs of incremental control strategies for two other alternative standards (11/35 and 11/30). In analyzing the current 15/35 standard (baseline), the EPA determined that all counties would meet the 14/35 standard concurrently with meeting the existing 15/35 standard at no additional cost. Consequently, no incremental costs or benefits are associated with 14/35; thus, no analysis of 14/35 is presented.

Benefit and cost estimates provided in the RIA are not additive 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.

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 one party's actions impose uncompensated benefits or costs on another party. Environmental problems are a classic case of an externality. Setting primary and secondary air quality standards is one way 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>.

government can address an externality and thereby increase air quality and improve 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.

The illustrative goals of this RIA are somewhat different from other EPA analyses of national rules, or the implementation plans states develop, and 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 RIA

The RIA evaluates the costs and benefits of hypothetical national strategies to attain several alternative PM standards.

1.4.1 Modeling $PM_{2.5}$ Levels in the Future (Analysis Year = 2020)

A national-scale air quality modeling analysis was performed to estimate future-year annual and 24-hour $PM_{2.5}$ concentrations and light extinction for the future year of 2020. Air quality ratios were then developed using model responsiveness to emissions changes between a recent year of air quality, 2005, and a future year of air quality, 2020. The air quality ratios were used to determine potential control scenarios designed to attain the proposed alternative NAAQS, as well as the costs of attaining these levels. These data were then used to estimate how air quality would change under each set of potential control scenarios, and as inputs to the calculation of expected benefits from the alternative NAAQS considered in this assessment.

1.4.2 Existing and Alternative PM Air Quality Standards

Currently two primary $PM_{2.5}$ standards provide public health protection from effects associated with fine particle exposures. The annual standard is set at a level of 15.0 µg/m^3, based on the 3-year average of annual arithmetic mean $PM_{2.5}$ concentrations. The 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 the RIA, the current suite of primary $PM_{2.5}$ standards, including both annual and 24-hour averaging times, is denoted as 15/35.

In the current PM NAAQS review, the EPA is proposing to revise the level of the primary annual $PM_{2.5}$ standard within the range of 12 to 13 µg/m^3 in conjunction with retaining the level of the 24-hour standard at 35 µg/m^3 (denoted 12/35 and 13/35).

In addition to the range of 12/35 to 13/35, the RIA also analyzes the benefits and costs of incremental control strategies for two other alternative standards (11/35 and 11/30). The four alternative standards analyzed are as follows:

- A revised annual standard level of 13 µg/m^3 in conjunction with retaining the 24-hour standard level at 35 µg/m^3 (13/35)
- A revised annual standard level of 12 µg/m^3 in conjunction with retaining the 24-hour standard level at 35 µg/m^3 (12/35)
- A revised annual standard level of 11 µg/m^3 in conjunction with retaining the 24-hour standard level at 35 µg/m^3 (11/35)
- A revised annual standard level of 11 µg/m^3 in conjunction with a revised 24-hour standard level at 30 µg/m^3 (11/30)

In analyzing the current 15/35 standard (baseline), the EPA determined that all counties would meet the 14/35 standard concurrently with meeting the existing 15/35 standard at no additional cost. Consequently, no incremental costs or benefits are associated with 14/35; thus, no analysis of 14/35 is presented in this RIA.

Currently, the existing secondary $PM_{2.5}$ standards are identical in all respects to the primary standards. In the current PM NAAQS review, the EPA is proposing to add a distinct standard for $PM_{2.5}$ to provide protection from PM-related visibility impairment. Specifically, the EPA is proposing to establish a separate secondary standard defined in terms of a $PM_{2.5}$

visibility index, which would use speciated $PM_{2.5}$ mass concentrations and relative humidity data to calculate $PM_{2.5}$ light extinction, similar to the Regional Haze Program; a 24-hour averaging time; a 90th percentile form; and a level of either 30 deciviews (dv) or 28 dv. Based on the air quality analysis conducted for the primary $PM_{2.5}$ standard, all monitored areas are estimated to be in attainment with both proposed secondary standard levels in 2020, assuming full attainment of the primary $PM_{2.5}$ standard. For the two optional levels proposed for the secondary standard, no additional costs or benefits will be realized beyond those quantified for meeting the primary $PM_{2.5}$ standard in this RIA.

With regard to the primary and secondary standards for particles less than or equal to 10 µm in diameter (PM_{10}), the EPA is proposing to retain 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 (EPA, 1997)[2]. Since the benefit cost analysis of the alternative PM_{10} standards was conducted when the standard was selected, this RIA does not repeat that analysis here.

1.4.3 Benefits Analysis Approach

The EPA estimated human health (e.g., mortality and morbidity effects) under full attainment of several alternative PM standards. We considered an array of health impacts attributable to changes in $PM_{2.5}$. Even though the alternative 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 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, many welfare benefits are not quantified or monetized.

1.4.4 Costs Analysis Approach

The EPA estimated total costs under partial and full attainment of several alternative PM 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

[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.

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 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. 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.

1.5 Organization of this Regulatory Impact Analysis

This RIA includes the following 11 chapters:

- *Chapter 1: Introduction and Background*. This chapter introduces the purpose of the RIA.
- *Chapter 2: Defining the $PM_{2.5}$ 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 the health-related benefits of the $PM_{2.5}$-related air quality improvements associated with several alternative standards.
- *Chapter 6*: *Welfare Benefits Analysis Approach*. This chapter quantifies and monetizes selected other 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: Secondary Standards Analysis*. This chapter contains an evaluation of the regulatory impacts associated with a distinct secondary NAAQS for $PM_{2.5}$.

- *Chapter 11*: *Economic Impacts—Employment*. This chapter provides a qualitative discussion of employment impacts of air quality regulations.

CHAPTER 2
DEFINING THE PM AIR QUALITY PROBLEM

2.1 Synopsis

This chapter characterizes the nature, scope and magnitude of the current year PM problem. It includes 1) 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; 2) discussion of what visibility is and how it is calculated from measured concentrations and meteorological values; and 3) current year design values for $PM_{2.5}$ and visibility.

2.2 Particulate Matter (PM) Properties

Particulate matter (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 (EPA, 2006) 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.

[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.

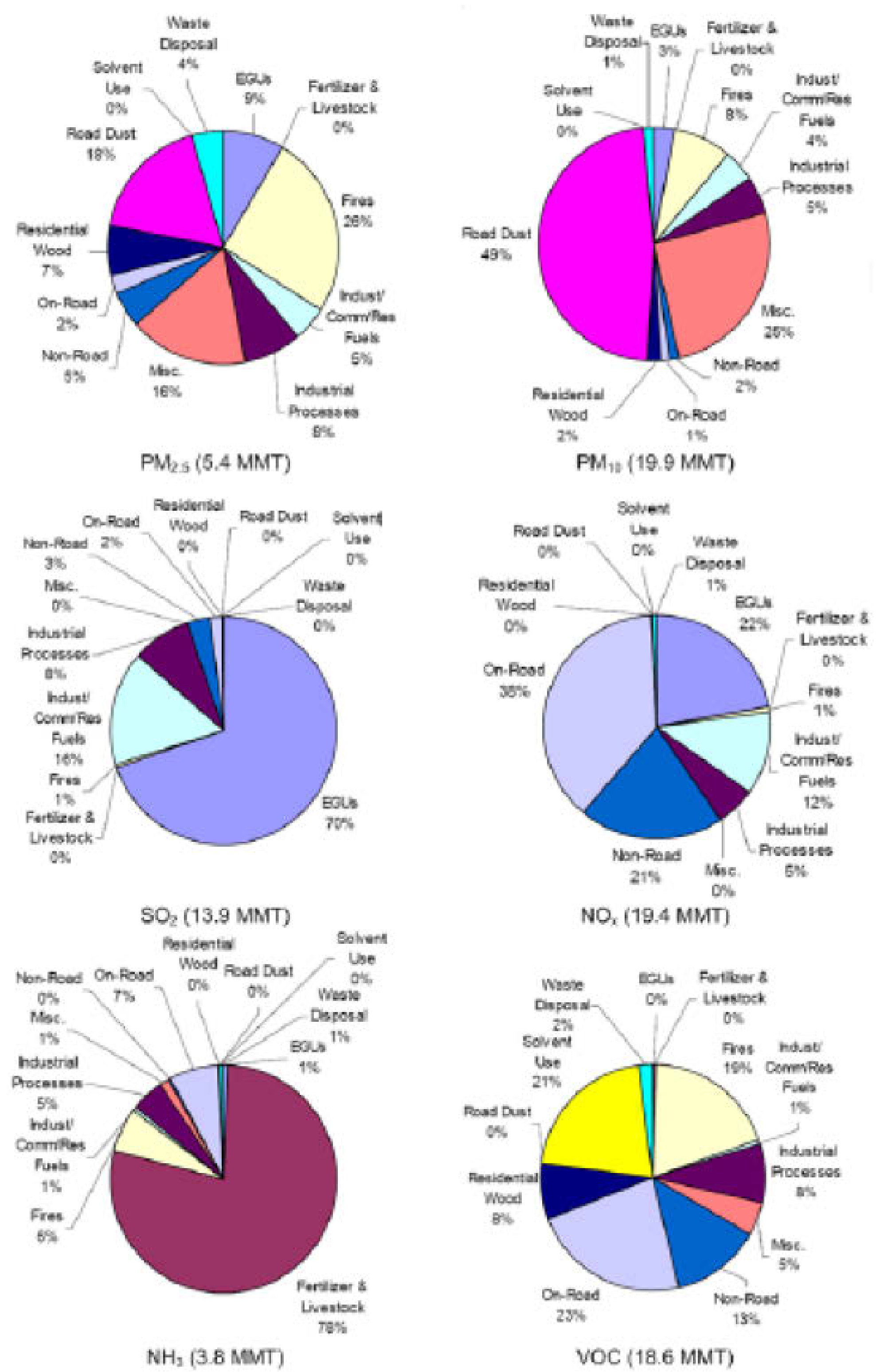

Figure 2-1. Detailed Source Categorization of Anthropogenic Emissions of Primary $PM_{2.5}$, PM_{10} and Gaseous Precursor Species SO_2, NO_x, NH_3 and VOCs for 2002 in Units of Million Metric Tons (MMT). EGUs = Electricity Generating Units

Source: U.S. EPA (2006)

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.

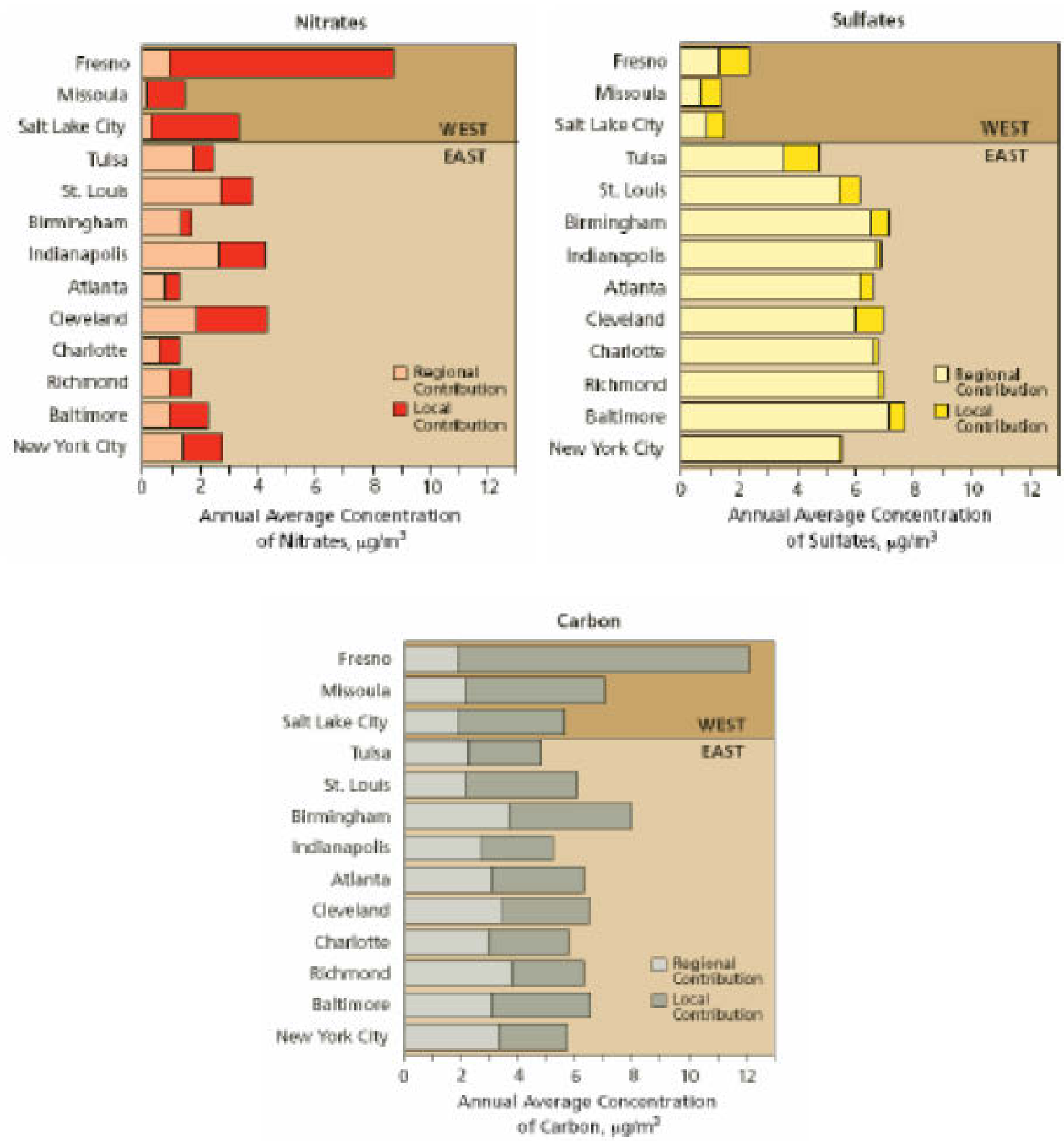

Figure 2-2. Regional and Local Contributions to Annual Average $PM_{2.5}$ by Particulate SO_4^{2-}, Nitrate and Total Carbon (i.e., organic plus EC) for Select Urban Areas Based on Paired 2000-2004 IMPROVE[a] and CSN[b] Monitoring Sites

[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.

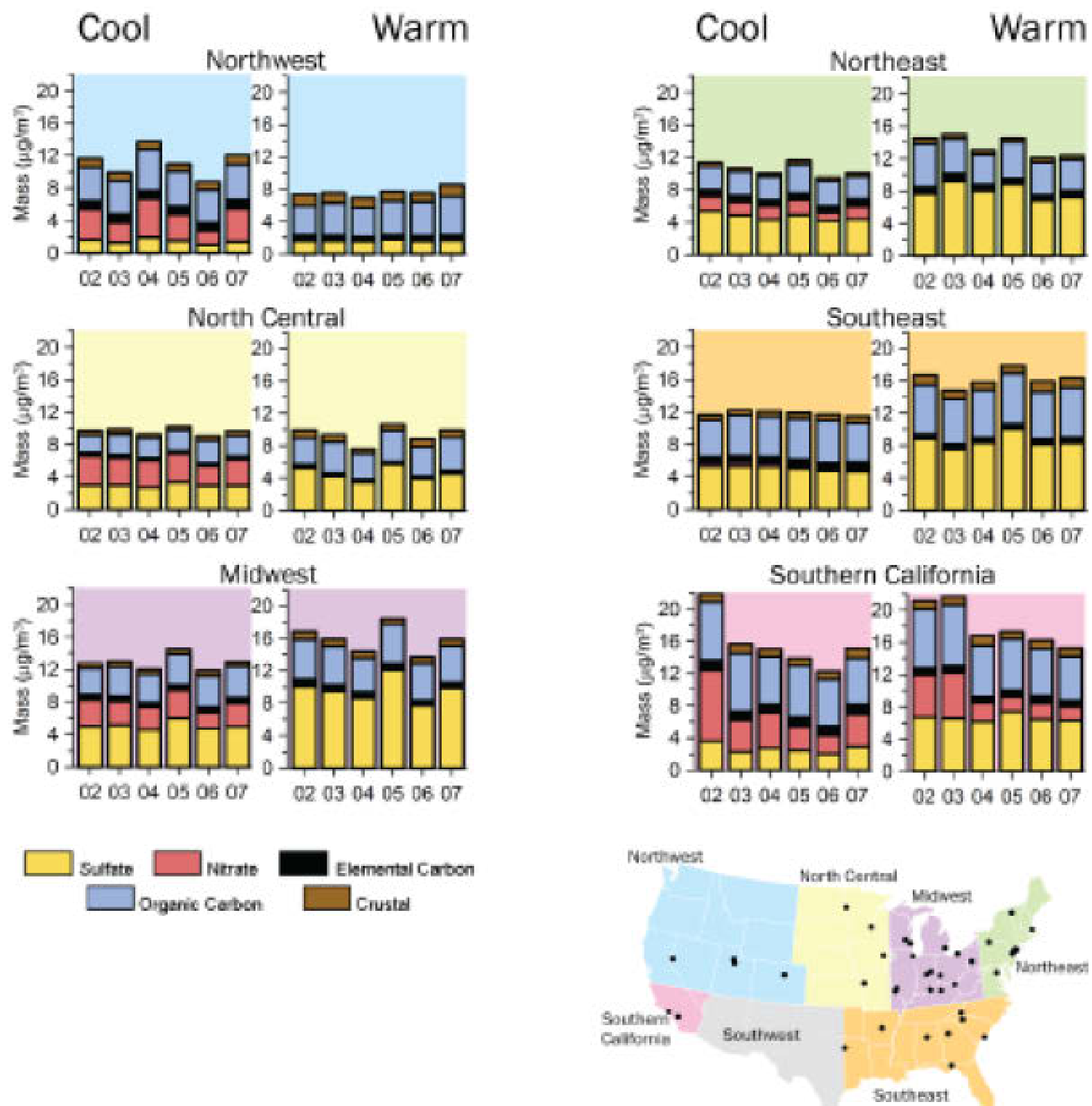

Figure 2-3. Regional and Seasonal Trends in Annual $PM_{2.5}$ Composition from 2002 to 2007 Derived Using the SANDWICH Method. Data from the 42 monitoring locations shown on the map were stratified by region and season including cool months (October–April) and warm months (May–September)

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 collocated 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. EPAs 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.

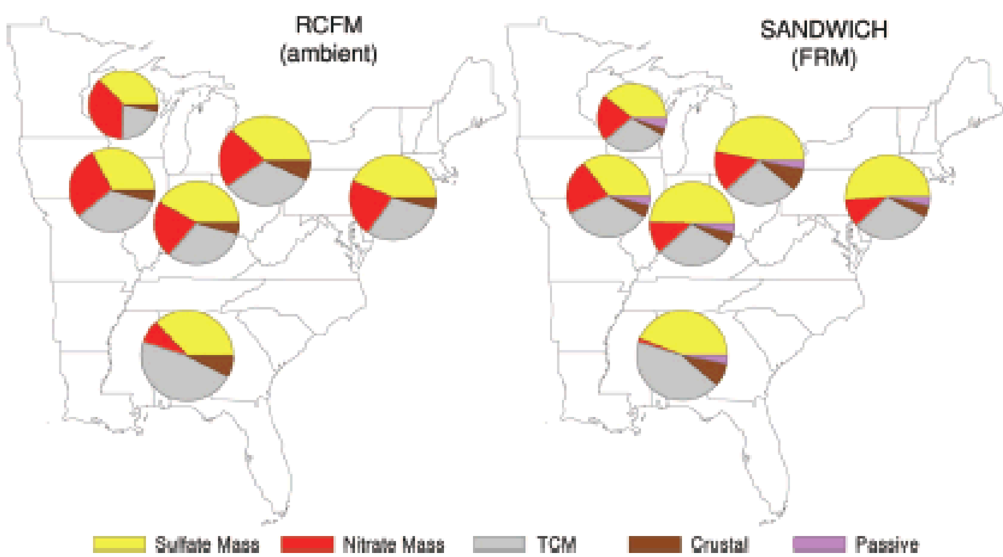

Figure 2-4. RCFM (left) versus SANDWICH (right) Pie Charts Comparing the Ambient and $PM_{2.5}$ FRM Reconstructed Mass Protocols on an Annual Average Basis[a]

[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 2004–2006 Design Values

The annual and 24-hour $PM_{2.5}$ design values were calculated using 2003–2007 FRM 24-hour average $PM_{2.5}$ concentration measurements and consistent with CFR Part 50.[3] Figures 2-5 and 2-6 show the county-level maximum values for both the annual and 24-hour standards, respectively. 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 30 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.

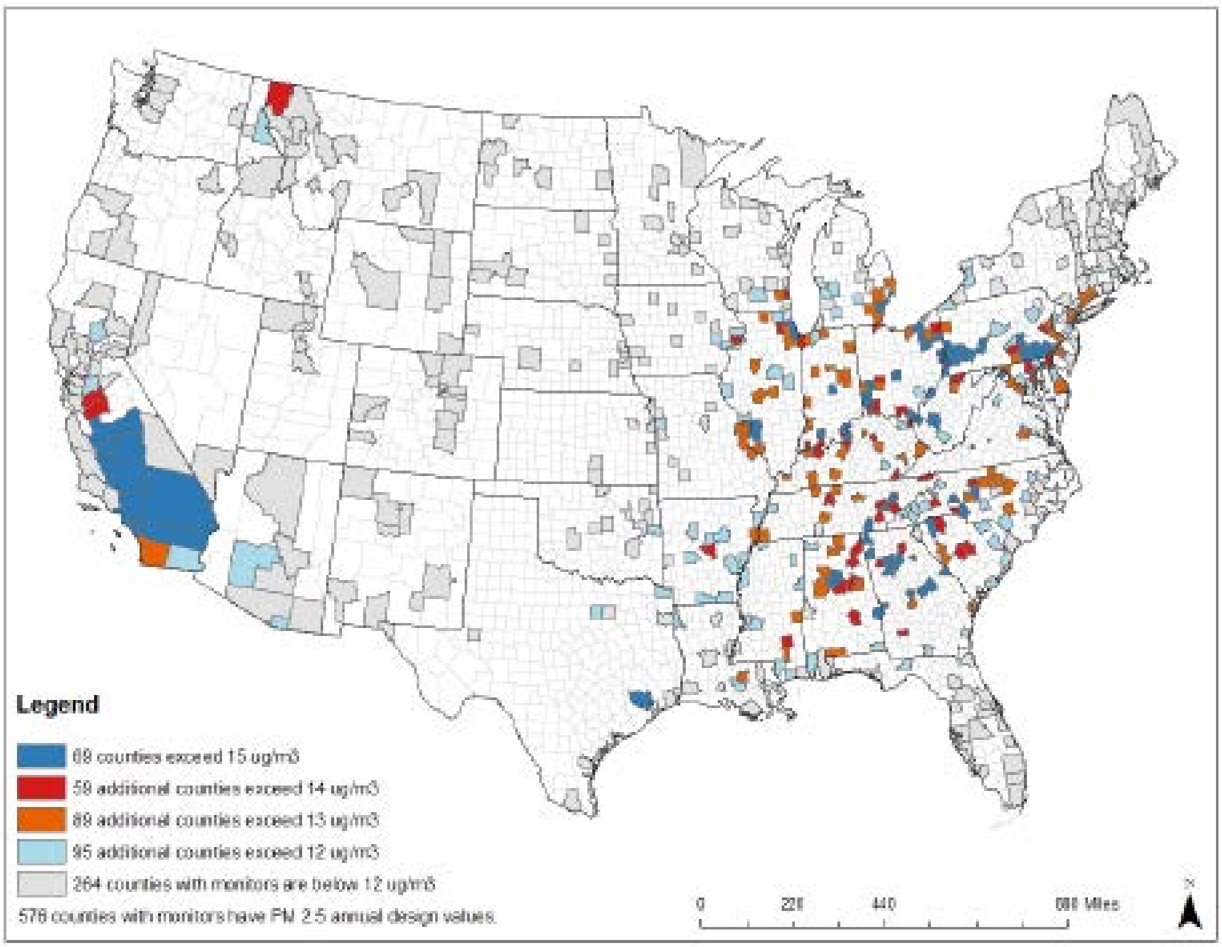

Figure 2-5. Maximum County-level $PM_{2.5}$ Annual Design Values Calculated Using 2003–2007 FRM 24-hr Average $PM_{2.5}$ Measurements

[3] These years of ambient measurements were selected since they frame the air quality model year of 2005. As discussed in Chapter 3, it is most appropriate to select ambient measurement years that include the model year to allow for a more true projection of future year air quality using the air quality model.

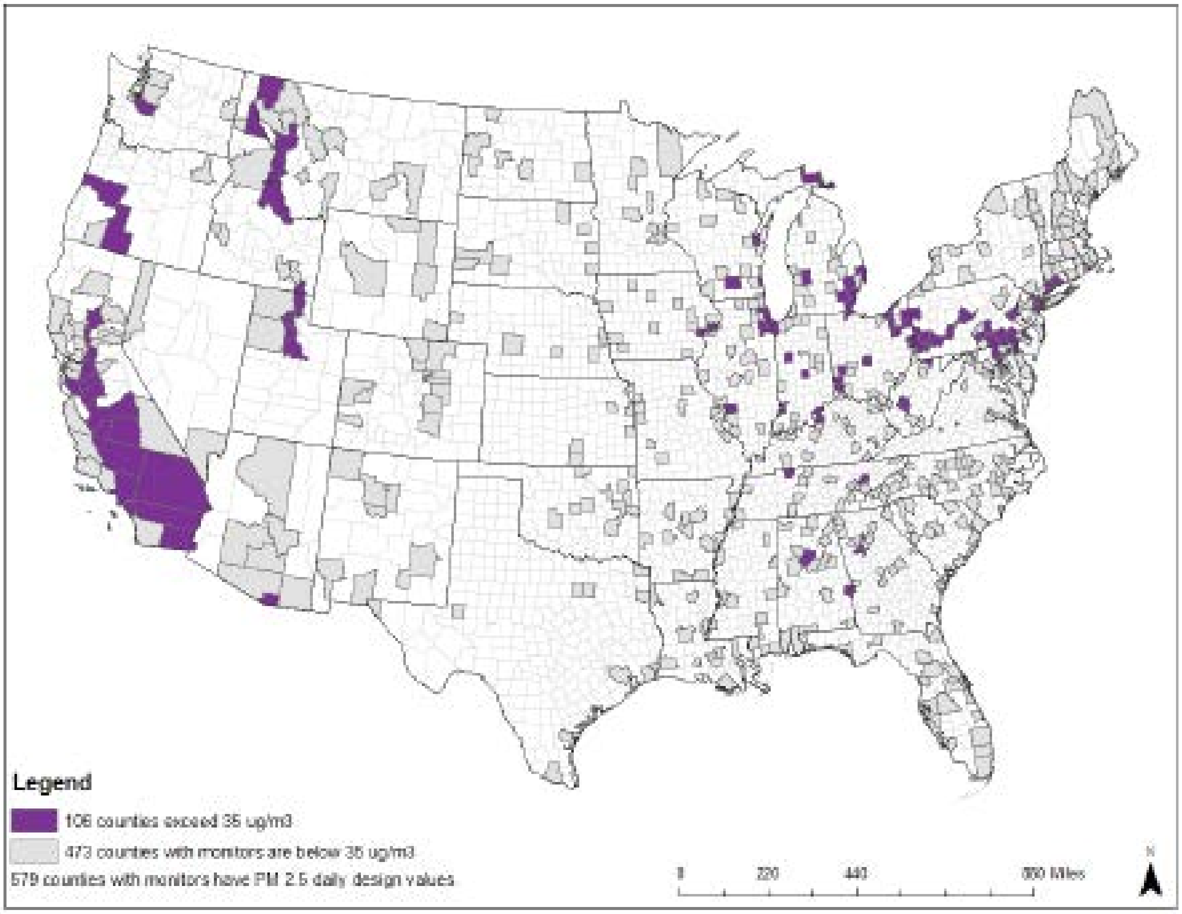

Figure 2-6. Maximum County-level $PM_{2.5}$ 24-hour Design Values Calculated Using 2003–2007 FRM 24-hr Average $PM_{2.5}$ Measurements

2.2.2 Visibility

Air pollution can affect 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 2-7 for an illustration of the important factors affecting visibility.

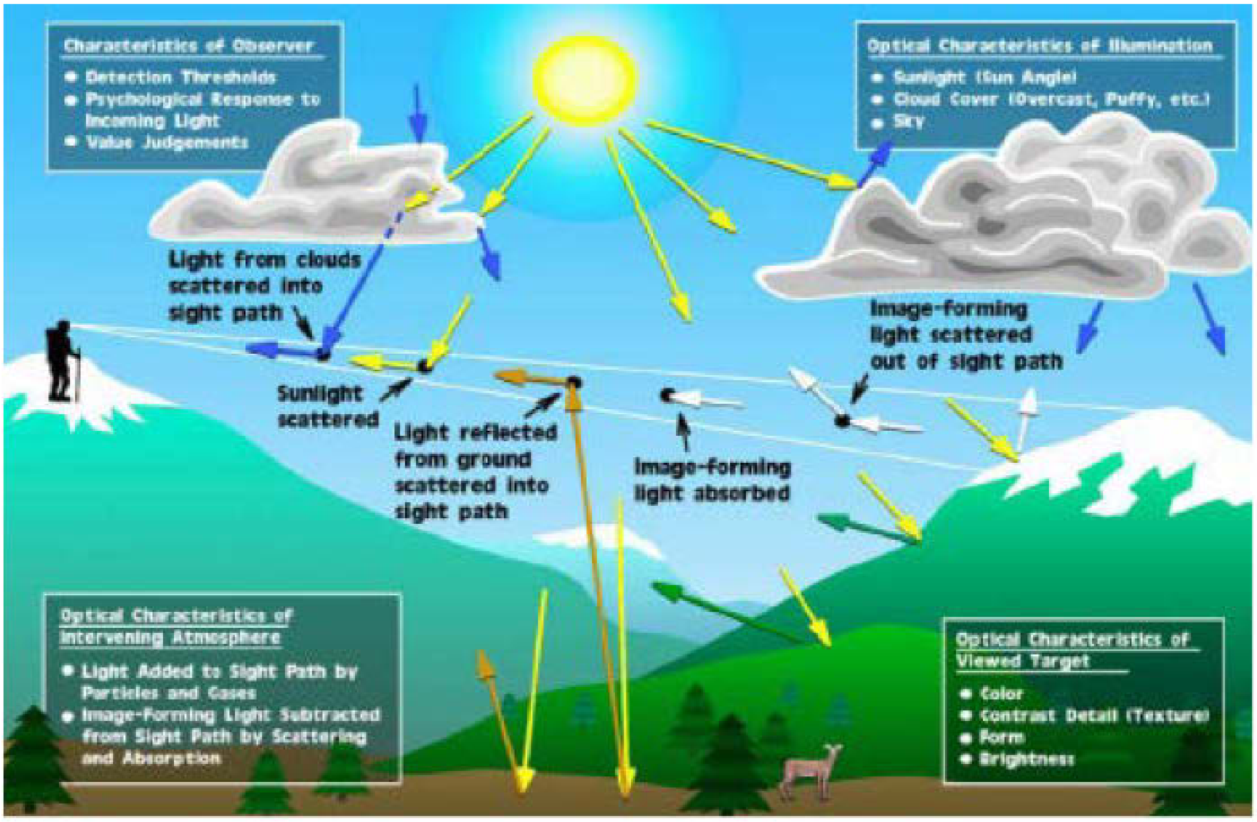

Figure 2-7. Important Factors Involved in Seeing a Scenic Vista

Source: Malm, 1999.

2.2.2.1 Calculating Visibility

Visibility degradation is often directly proportional to decreases in light transmittal in the atmosphere. Scattering and absorption by both gases and particles decrease light transmittance. To quantify changes in visibility, our analysis computes a light-extinction coefficient, based on the work of Sisler (1996), which shows the total fraction of light that is decreased per unit distance. This coefficient accounts for the scattering and absorption of light by both particles and gases, and accounts for the higher extinction efficiency of fine particles compared to coarse particles. Fine particles with significant light-extinction efficiencies include sulfates, nitrates, organic carbon, elemental carbon (soot), and soil (Sisler, 1996).

As described in the Policy Assessment Document (EPA, 2011), the formula for total light extinction (b_{ext}) in units of Mm^{-1} using the original IMPROVE equation is:

$$b\mathrm{ext} = 3 \text{ x } f(\mathrm{RH}) \text{ x } [Sulfate] + 3 \text{ x } f(\mathrm{RH}) \text{ x } [Nitrate] + 4 \text{ x } [Organic\ Mass] + 10 \text{ x } [Elemental\ Carbon] + 1 \text{ x } [Fine\ Soil] + 0.6 \text{ x } [\mathrm{Coarse\ Mass}] + 10 \quad (2.1)$$

where the mass concentrations of the components indicated in brackets are in units of μg/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.

Based upon the light-extinction coefficient, a unitless visibility index, called a "deciview," can also be calculated using Equation (2.2):

$$Deciviews = 10 * \ln\left(\frac{391}{VR}\right) = 10 \ * \ln\left(\frac{\beta_{ext}}{10}\right) \tag{2.2}$$

where VR denotes visual range (in kilometers) and β_{ext} denotes light extinction (in Mm^{-1}). The deciview metric provides a scale for perceived visual changes over the entire range of conditions, from clear to hazy. Under many scenic conditions, the average person can generally perceive a change of one deciview. The higher the deciview value, the worse the visibility. Thus, an improvement in visibility is a decrease in deciview value.

2.2.2.2 Geographical Scale and Variability

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, 2009). 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, 2009). Particulate nitrate is an important contributor to light extinction in California and the upper Midwestern U.S., particularly during winter (U.S. EPA, 2009). 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, 2009). 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, 2009). 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, 2009).

Figure 2-8 shows the average 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); IMPROVE (2010)). 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 overtime due to air pollution regulations such as the Acid Rain Program (U.S. EPA, 2009). For example, Figure 2-8 shows that visual range increased nearly 50% in the eastern U.S. since 1992[4]. While visibility trends have improved in most Class 1 areas[5], the recent data show that these areas continue to suffer from visibility impairment (U.S. EPA, 2009). 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).

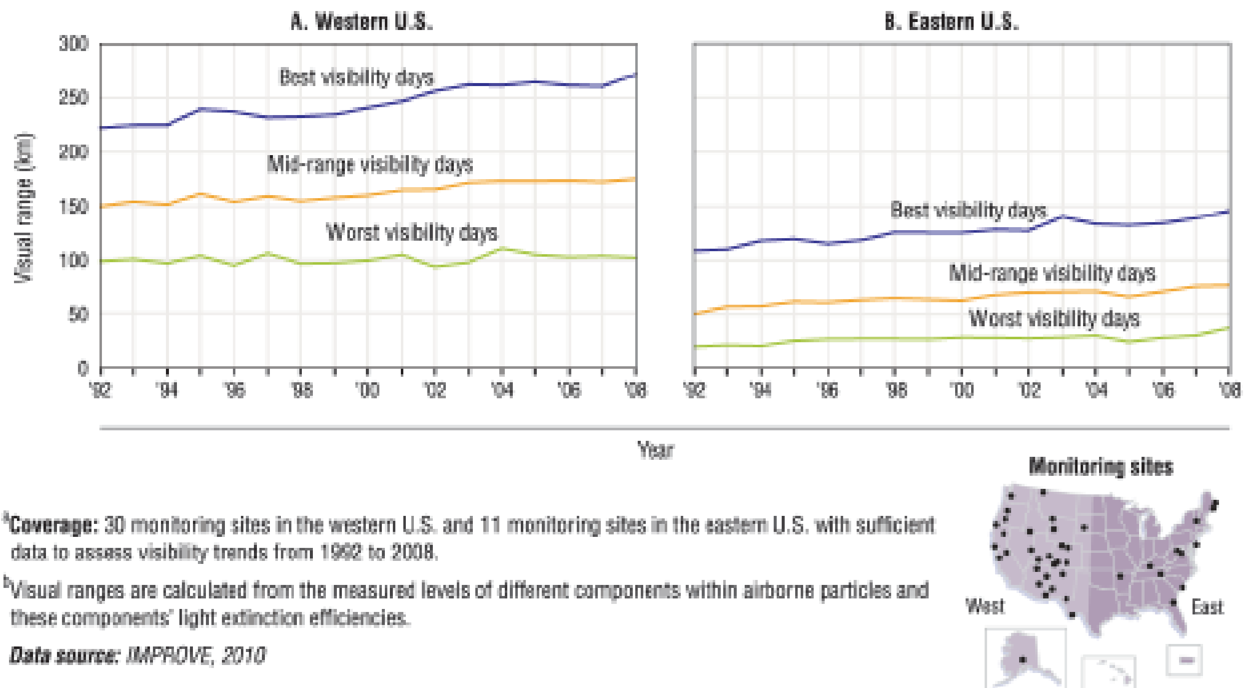

Figure 2-8. Visibility in Selected National Parks and Wilderness Areas in the U.S., 1992–2008[a,b]

Source: U.S. EPA (2008) updated, IMPROVE (2010).

2.2.2.3 2004–2006 Design Values

The secondary $PM_{2.5}$ NAAQS standard consists of three parts: a level, averaging period, and form. EPA proposes using a 3-year average, 90th percentile form for the standard, calculated using 24-hr speciated $PM_{2.5}$ measurements. EPA analyzed two proposed levels of 30

[4] In Figure 2-8, 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).

[5] Class I areas are areas of special national or regional natural, scenic, recreational, or historic value for which the Prevention of Significant Deterioration (PSD) regulations provide special protection.

dv and 28 dv, as well as a more stringent standard of 25 dv. The ambient design values analyzed in this RIA are based on measured 24-hour $PM_{2.5}$ speciation data from 2004–2006[6]. These data were calculated as described in the Policy Assessment Document (EPA, 2011) and provided in Chapter 13. Figure 2-9 shows the county-level maximum design values. 20 counties were above 30 dv and 90 counties were above 28 dv. For the more stringent proposed level, 77 additional counties were above 25 dv. The large majority of these counties are located in the East.

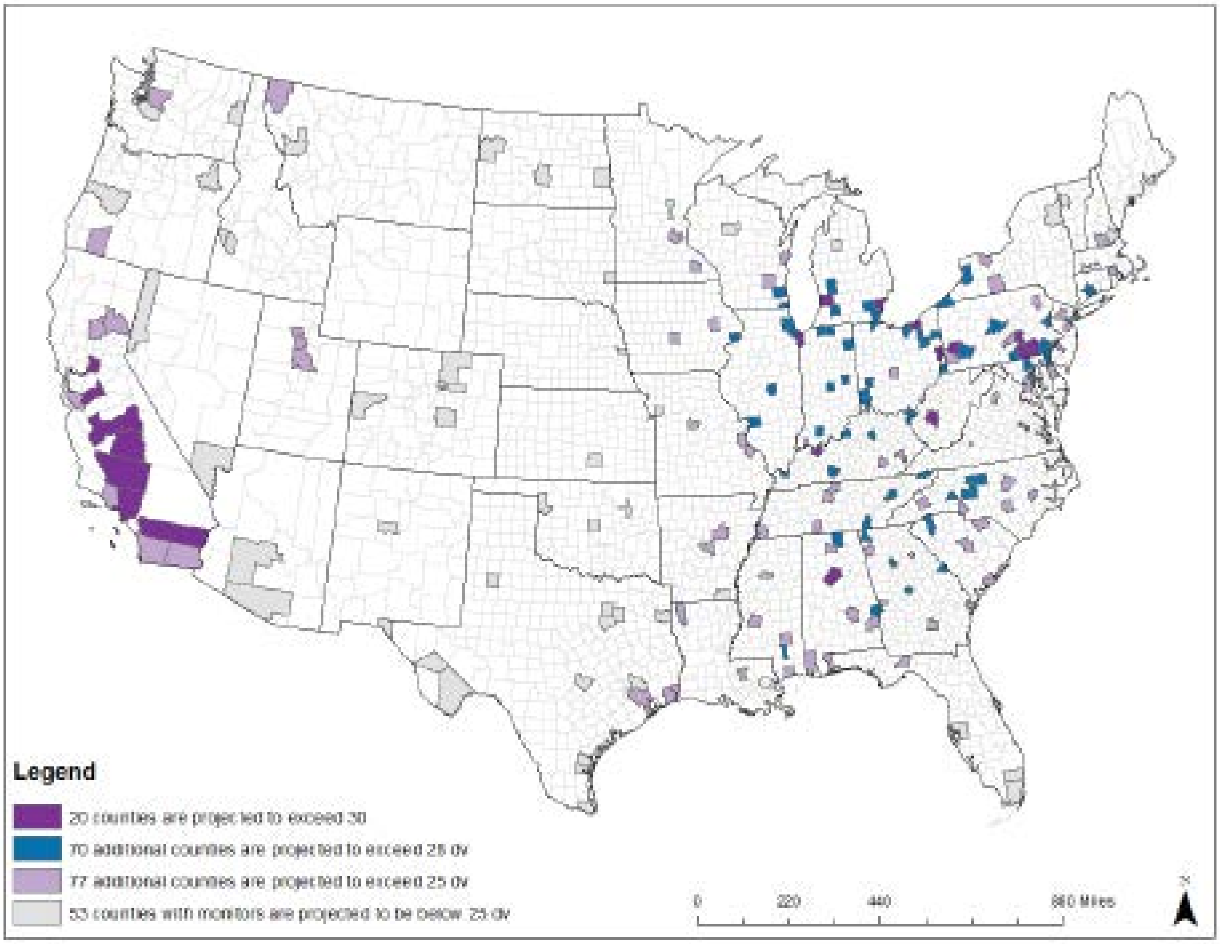

Figure 2-9. Maximum County-level Visibility Design Values Calculated Using 2004–2006 24-hr Average Speciated $PM_{2.5}$ Measured Concentrations

2.3 References

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[6] These years of ambient measurements were selected since they frame the air quality model year of 2005. As discussed in Chapter 3, it is most appropriate to select ambient measurement years that include the model year to allow for a more true projection of future year air quality using the air quality model.

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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 and secondary PM standards proposed in this 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 and light extinction for the future year of 2020.[1] Air quality ratios were then developed using model responsiveness to emissions changes between a recent year of air quality, 2005, and a future year of air quality, 2020. The air quality ratios were used to determine potential control scenarios designed to attain the proposed alternative NAAQS, as well as the costs of attaining these levels. These data were then used to estimate how air quality would change under each set of potential control scenarios, and as inputs to the calculation of expected benefits from the alternative NAAQS considered in this assessment.

3.2.1 Air Quality Modeling Platform

The 2005-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 proposed alternative NAAQS considered in this assessment. In addition to the CMAQ model, the modeling platform includes the emissions, meteorology, and initial and boundary condition data 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

[1] As described in more detail in this chapter, the future-year emissions inventory used in the air quality modeling analysis is a combination of emissions sectors projected to 2017 and 2020. We have chosen to label the future-year of modeling as "2020" because the EGU sector, which is projected to 2020, is of significant importance to the concentrations of $PM_{2.5}$ in the U.S.

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, as described in the Air Quality Modeling Technical Support Document (EPA, 2011b).

3.2.1.1 Air Quality Modeling Domain

Figure 3-1 shows the modeling domains that were used as a part of this analysis. The geographic specifications for these domains are provided in Table 3-1. All three modeling domains contain 14 vertical layers with a top at about 16,200 meters, or 100 millibars (mb). Two domains with 12 km horizontal resolution were used for modeling the 2005 base year and 2020 control strategy scenarios. These domains are labeled as the East and West 12 km domains in Figure 3-1. Simulations for the 36 km domain were only used to provide initial and boundary concentrations for the 12 km domains.

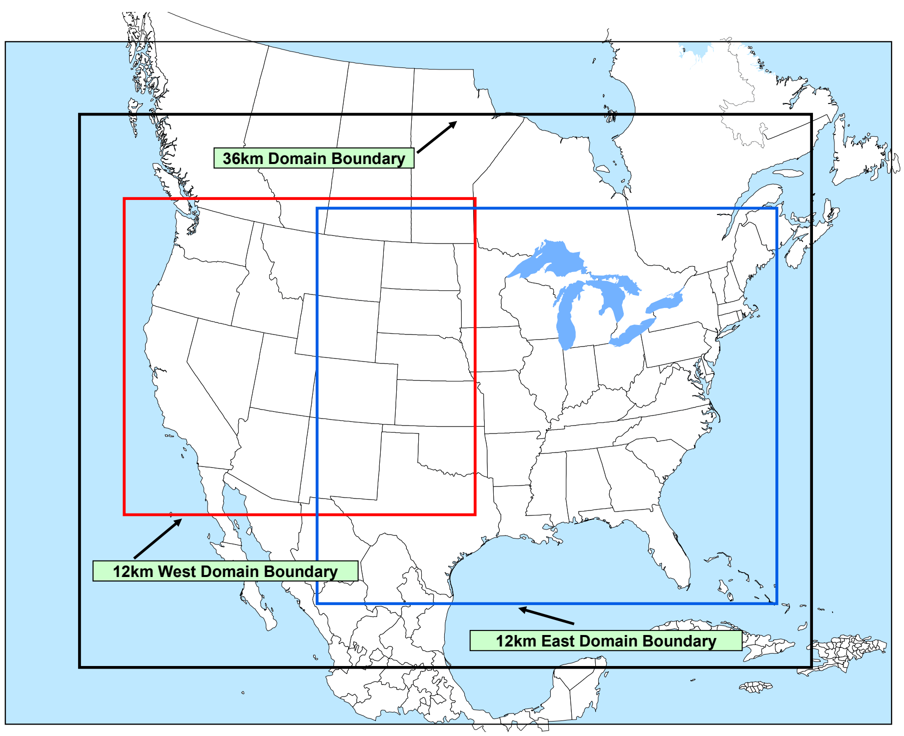


Figure 3-1. Map of the CMAQ Modeling Domains Used for PM NAAQS RIA

Table 3-1. Geographic Specifications of Modeling Domains

36 km Domain (148 x 112 Grid Cells)			12 km East Domain (279 x 240 Grid Cells)			12 km West Domain (213 x 192 Grid Cells)		
	Longitude	Latitude		Longitude	Latitude		Longitude	Latitude
SW	−121.77	18.17	SW	−106.79	24.99	SW	−121.65	28.29
NE	−58.54	52.41	NE	−65.32	47.63	NE	−94.94	51.91

The model produces gridded air quality concentrations on an hourly basis for the entire modeling domain. For this analysis, predictions from the East domain were used to provide data for all areas that are east of approximately 104 degrees longitude. Model predictions from the West domain were used for all areas west of this longitude.

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 2005 base year and the future year of 2020. All other inputs were specified for the 2005 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. Meteorological inputs reflecting 2005 conditions across the contiguous U.S. were derived from Version 5 of the Mesoscale Model (MM5). 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 2005 meteorological modeling are provided in the Air Quality Modeling Technical Support Document: Final EGU NESHAP (EPA, 2011d).

The lateral boundary and initial species concentrations for the CMAQ simulations using a 36 km domain are provided by a three dimensional global atmospheric chemistry and transport model (GEOS-CHEM). The lateral boundary species concentrations varied with height and time (every 3 hours). These data were used in CMAQ for the 36 km domain. Initial and boundary concentrations from the CMAQ 36 km domain were then used to provide initial and boundary concentrations for CMAQ simulations using the East and West 12 km domains. The development of model inputs is discussed in greater detail in the Air Quality Modeling Technical Support Document: Final EGU NESHAP (EPA, 2011d).

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 2005 base year concentrations. This evaluation principally comprises statistical assessments of model predictions versus observations paired in time and space on an hourly, 24-hour, or weekly basis depending on the sampling period of measured data. Details on the evaluation methodology and the calculation of performance statistics are provided in the Air Quality Modeling Technical Support Document: Final EGU NESHAP (EPA, 2011d). Overall, the model performance statistics for sulfate, nitrate, organic carbon, and elemental carbon from the CMAQ 2005 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 2005 modeling platform provide a scientifically credible approach for assessing $PM_{2.5}$ concentrations for the purposes of the RIA.

3.2.2 Emissions Inventory

The future-year base-case inventory, projected from the 2005 Version 4.3 emissions modeling platform, is the starting point for the baseline and control strategy for the Proposed PM NAAQS emissions inventory. The Emissions Modeling for the Final Mercury and Air Toxics Standard (MATS) TSD (EPA, 2011c) describes in detail the development of the 2005 base year inventory, the projection methodology, and the controls applied to create the projected inventory. Note that the referenced Emissions Modeling TSD describes the use of year 2015 emissions for EGUs and 2017 emissions for other sources, while this analysis used 2020 emissions for EGUs.

The 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 environmental rules and regulations, consent decrees and settlements, plant closures, and newly built units for the calendar year 2020. In this analysis, the projected EGU emissions include the Final MATS policy case announced on December 21, 2011 and the Final Cross-State Air Pollution Rule (CSAPR) issued on July 6, 2011. The EGU emissions were developed using version 4.10 Final MATS version of the Integrated Planning

Model (IPM) and documented in detail at http://www.epa.gov/airmarkt/progsregs/epa-ipm/toxics.html. The 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 were selected for illustrating attainment of the current and proposed alternative standard levels, as discussed in Chapter 4, and the EGU emissions are unchanged between the future-year base-case and control strategies.

The mobile source emissions were projected to 2017 using activity data. These emissions represent the effects of the Clean Air Nonroad Diesel Rule, the Light-Duty Vehicle Tier 2 Rule, the Heavy Duty Diesel Rule, and other finalized rules. Table 3-2 provides a comprehensive list of the rules/control strategies and projection assumptions in the projected base-case (i.e., reference case) inventory. A full discussion of the future year base inventory is provided in the Emissions Modeling TSD. The 2017 onroad mobile source emissions were developed by using the MOtor Vehicle Emission Simulator (MOVES)[2] to create emission factors that were then input to the Sparse Matrix Operator Kernel Emissions system (SMOKE). The SMOKE-MOVES Integration Tools combined the county and temperature-specific emission factors with the activity data to compute the actual emissions based on hourly gridded temperature data.

The future year scenarios include the same year 2006 Canada and year 1999 Mexico emissions as the 2005 base case. All 2005 and projected base case emissions inventories are available on the EPA's Emissions Modeling Clearinghouse website at: http://www.epa.gov/ttn/chief/emch/index.html#toxics. The inventories used to support this analysis can be found under ftp://ftp.epa.gov/EmisInventory/2005v4_3/mats.

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

Table 3-2. Control Strategies and Growth Assumptions for Creating 2020 Base Case Emissions Inventories from the 2005 Base Case

Control Strategies and/or Growth Assumptions (Grouped by Affected Pollutants or Standard and Approach Used to Apply to the Inventory)	Pollutants Affected	Approach or Reference:
Non-EGU Point (ptnonipm) Controls		
MACT rules, national, VOC: national applied by SCC, MACT Boat Manufacturing Wood Building Products Surface Coating Generic MACT II: Spandex Production, Ethylene manufacture Large Appliances Miscellaneous Organic NESHAP (MON): Alkyd Resins, Chelating Agents, Explosives, Phthalate Plasticizers, Polyester Resins, Polymerized Vinylidene Chloride Reinforced Plastics Asphalt Processing & Roofing Iron & Steel Foundries Metal: Can, Coil Metal Furniture Miscellaneous Metal Parts & Products Municipal Solid Waste Landfills Paper and Other Web Plastic Parts Plywood and Composite Wood Products Carbon Black Production Cyanide Chemical Manufacturing Friction Products Manufacturing Leather Finishing Operations Miscellaneous Coating Manufacturing Organic Liquids Distribution (Non-Gasoline) Refractory Products Manufacturing Sites Remediation	VOC	EPA, 2007a
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	1
DOJ Settlements: plant SCC controls for: Alcoa, TX Premcor (formerly Motiva), DE	All	2
Refinery Consent Decrees: plant/SCC controls	NO_x, PM, SO_2	3

(continued)

Table 3-2. Control Strategies and Growth Assumptions for Creating 2020 Base Case Emissions Inventories from the 2005 Base Case (continued)

Control Strategies and/or Growth Assumptions (Grouped by Affected Pollutants or Standard and Approach Used to Apply to the Inventory)	Pollutants Affected	Approach or Reference:
Non-EGU Point (ptnonipm) Controls (continued)		
Hazardous Waste Combustion	PM	4
Municipal Waste Combustor Reductions—plant level	PM	5
Hospital/Medical/Infectious Waste Incinerator Regulations	NO_X, PM, SO_2	EPA, 2005
Large Municipal Waste Combustors—growth applied to specific plants	All (including Hg)	5
MACT rules, plant-level, VOC: Auto Plants	VOC	6
MACT rules, plant-level, PM & SO_2: Lime Manufacturing	PM, SO_2	7
MACT rules, plant-level, PM: Taconite Ore	PM	8
Livestock Emissions Growth from year 2002 to year 2017 (some farms in the point inventory)	NH_3, PM	9
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	10; EPA, 2010
New York ozone SIP controls	VOC, NO_X, HAP VOC	11
Additional plant and unit closures provided by state, regional, and the EPA agencies and additional consent decrees. Includes updates from CSAPR comments.	All	12
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	Section 4.2.13.2
Reciprocating Internal Combustion Engines (RICE) NESHAP	NO_X, CO, PM, SO_2	13
Ethanol plants that account for increased ethanol production due to RFS2 mandate	All	14
State fuel sulfur content rules for fuel oil—effective only in Maine, New Jersey, and New York	SO_2	15
Nonpoint (nonpt sector) Projection Approaches		
Municipal Waste Landfills: projection factor of 0.25 applied	All	EPA, 2007a

(continued)

Table 3-2. Control Strategies and Growth Assumptions for Creating 2020 Base Case Emissions Inventories from the 2005 Base Case (continued)

Control Strategies and/or Growth Assumptions (Grouped by Affected Pollutants or Standard and Approach Used to Apply to the Inventory)	Pollutants Affected	Approach or Reference:
Nonpoint (nonpt sector) Projection Approaches (continued)		
Livestock Emissions Growth from year 2002 to 2017	NH_3, PM	9
New York, Connecticut, and Virginia ozone SIP controls	VOC	11, 16
RICE NESHAP	NO_X, CO, VOC, PM, SO_2	13
State fuel sulfur content rules for fuel oil—*effective only in Maine, New Jersey, and New York*	SO_2	15
Residential Wood Combustion Growth and Change-outs from year 2005 to 2017	All	17
Gasoline and diesel fuel Stage II refueling via MOVES2010a month-specific inventories for 2017 with assumed RFS2 and LDGHG fuels	VOC, Benzene, Ethanol	18
Portable Fuel Container Mobile Source Air Toxics Rule 2 (MSAT2) inventory growth and control from year 2005 to 2017	VOC	19
Phase II WRAP 2018 Oil and Gas	VOC, SO_2, NO_X, CO	EM TSD
2008 Oklahoma and Texas Oil and Gas, and apply year 2017 projections for TX, and RICE NESHAP controls to Oklahoma emissions.	VOC, SO_2, NO_X, CO, PM	EM TSD

Approaches/References—Non-EGU Stationary Sources:

1. Appendix B in the MATS Proposal TSD: http://www.epa.gov/ttn/chief/emch/toxics/proposed_toxics_rule_appendices.pdf
2. For Alcoa consent decree, used http://cfpub.epa.gov/compliance/cases/index.cfm; for Motiva: used information sent by State of Delaware
3. Used data provided by the EPA, OAQPS, Sector Policies and Programs Division (SPPD).
4. Obtained from Anne Pope, the US EPA—Hazardous Waste Incinerators criteria and hazardous air pollutant controls carried over from 2002 Platform, v3.1.
5. Used data provided by the EPA, OAQPS SPPD expert.
6. Percent reductions and plants to receive reductions based on recommendations by rule lead engineer, and are consistent with the reference: EPA, 2007a
7. Percent reductions recommended are determined from the existing plant estimated baselines and estimated reductions as shown in the Federal Register Notice for the rule. SO_2 percent reduction are computed by 6,147/30,783 = 20% and PM_{10} and $PM_{2.5}$ reductions are computed by 3,786/13,588 = 28%
8. Same approach as used in the 2006 Clean Air Interstate Rule (CAIR), which estimated reductions of "PM emissions by 10,538 tpy, a reduction of about 62%." Used same list of plants as were identified based on tonnage and SCC from CAIR: http://www.envinfo.com/caain/June04updates/tiop_fr2.pdf

(continued)

Table 3-2. Control Strategies and Growth Assumptions for Creating 2020 Base Case Emissions Inventories from the 2005 Base Case (continued)

Control Strategies and/or Growth Assumptions (Grouped by Affected Pollutants or Standard and Approach Used to Apply to the Inventory)	Pollutants Affected	Approach or Reference:
Nonpoint (nonpt sector) Projection Approaches (continued)		

Approaches/References—Non-EGU Stationary Sources (continued):

9. Except for dairy cows and turkeys (no growth), based on animal population growth estimates from the US Department of Agriculture (USDA) and the Food and Agriculture Policy and Research Institute. See Section 4.2.10.
10. Data files for the cement sector provided by Elineth Torres, the EPA-SPPD, from the analysis done for the Cement NESHAP: The ISIS documentation and analysis for the cement NESHAP/NSPS is in the docket of that rulemaking-docket # EPA-HQ-OAR-2002-005. The Cement NESHAP is in the Federal Register: September 9, 2010 (Volume 75, Number 174, Page 54969-55066
11. New York NO_X and VOC reductions obtained from Appendix J in NY Department of Environmental Conservation Implementation Plan for Ozone (February 2008): http://www.dec.ny.gov/docs/air_pdf/NYMASIP7final.pdf.
12. Appendix D of Cross-State Air Pollution Rule: ftp://ftp.epa.gov/EmisInventory/2005v4_2/transportrulefinal_eitsd_appendices_28jun2011.pdf
13. Appendix F in the Proposed (Mercury and Air) Toxics Rule TSD: http://www.epa.gov/ttn/chief/emch/toxics/proposed_toxics_rule_appendices.pdf
14. The 2008 data used came from Illinois' submittal of 2008 emissions to the NEI.
15. Based on available, enforceable state sulfur rules as of November, 2010: http://www.ilta.org/LegislativeandRegulatory/MVNRLM/NEUSASulfur%20Rules_09.2010.pdf, http://www.mainelegislature.org/legis/bills/bills_124th/billpdfs/SP062701.pdf, http://switchboard.nrdc.org/blogs/rkassel/governor_paterson_signs_new_la.html, http://green.blogs.nytimes.com/2010/07/20/new-york-mandates-cleaner-heating-oil/
16. VOC reductions in Connecticut and Virginia obtained from CSAPR comments.
17. Growth and Decline in woodstove types based on industry trade group data, See Section 4.2.11.
18. MOVES (2010a) results for onroad refueling including activity growth from VMT, Stage II control programs at gasoline stations, and phase in of newer vehicles with onboard Stage II vehicle controls. http://www.epa.gov/otaq/models/moves/index.htm
19. VOC, benzene, and ethanol emissions for 2017 based on MSAT2 rule and ethanol fuel assumptions (EPA, 2007b)

Control Strategies and/or Growth Assumptions	Pollutants Affected	Approach or Reference:
Onroad Mobile and Nonroad Mobile Controls (list includes all key mobile control strategies but is not exhaustive)		
National Onroad Rules: 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 Rule: May, 2010 Corporate Average Fuel Economy standards for 2008–2011	All	1

(continued)

Table 3-2. Control Strategies and Growth Assumptions for Creating 2020 Base Case Emissions Inventories from the 2005 Base Case (continued)

Control Strategies and/or Growth Assumptions (Grouped by Affected Pollutants or Standard and Approach Used to Apply to the Inventory)	Pollutants Affected	Approach or Reference:
Onroad Mobile and Nonroad Mobile Controls **(list includes all key mobile control strategies but is not exhaustive)** **(continued)**		
Local Onroad Programs: National Low Emission Vehicle Program (NLEV): March, 1998 Ozone Transport Commission (OTC) LEV Program: January, 1995	VOC	2
National Nonroad Controls: 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)	All	3,4,5
Aircraft (emissions are in the nonEGU point inventory)**:** Itinerant (ITN) operations at airports to 2017	All	6
Locomotives: 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	All	EPA, 2009; 3; 4; 5

(continued)

Table 3-2. Control Strategies and Growth Assumptions for Creating 2020 Base Case Emissions Inventories from the 2005 Base Case (continued)

Control Strategies and/or Growth Assumptions (Grouped by Affected Pollutants or Standard and Approach Used to Apply to the Inventory)	Pollutants Affected	Approach or Reference:
Onroad Mobile and Nonroad Mobile Controls (list includes all key mobile control strategies but is not exhaustive) (continued)		
Commercial Marine: 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	All	7, 3; EPA, 2009

Approaches/References—Mobile Sources

1. http://epa.gov/otaq/hwy.htm
2. Only for states submitting these inputs: http://www.epa.gov/otaq/lev-nlev.htm
3. http://www.epa.gov/nonroad-diesel/2004fr.htm
4. http://www.epa.gov/cleanschoolbus/
5. http://www.epa.gov/otaq/marinesi.htm
6. Federal Aviation Administration (FAA) Terminal Area Forecast (TAF) System, January 2010: http://www.apo.data.faa.gov/main/taf.asp
7. http://www.epa.gov/otaq/oceanvessels.htm

3.3 Modeling Results and Analyses

The air quality modeling results were used in the RIA to estimate future-year $PM_{2.5}$ concentrations for the 2020 base case and to calculate the air quality ratios that were used to determine potential control scenarios designed to attain the current and proposed alternative NAAQS. These data are then used to estimate the costs and benefits of attaining these current and proposed NAAQS levels. Consistent with EPA guidance (EPA, 2007), the air quality modeling results are applied in a relative sense to estimate 2020 future-year design values for $PM_{2.5}$ and visibility for the base case as described in Sections 3.3.1.1 and 3.3.2.1. Air quality ratios are calculated using the changes in the 2005 and 2020 base case design values and emissions as described in Section 3.3.1.2. The data are then used to estimate the tons of emissions reductions needed to show attainment of the current and alternative NAAQS levels as described in Section 3.3.1.3 and in Chapter 4. Based on the tons of emissions needed in each county, annual standard design values are calculated for attaining the current and alternative standard levels for input into the benefits assessment as described in Section 3.3.1.4. Limitations of this approach are described in Section 3.3.1.5.

Additional data were also processed to calculate visibility design values for the 2020 base case. Details on this post-processing are discussed in Section 3.3.2.

3.3.1 $PM_{2.5}$

As discussed in Chapter 1, this RIA evaluates the costs and benefits of attaining four alterative combinations of standards relative to meeting the current primary $PM_{2.5}$ standards (15/35). The five alterative combinations of standards evaluated are: an annual standard level of 14 $\mu g/m^3$ in conjunction with retaining the current 24-hour standard level at 35 $\mu g/m^3$ (14/35); an annual standard level of 13 $\mu g/m^3$ in conjunction with retaining the current 24-hour standard level at 35 $\mu g/m^3$ (13/35); an annual standard level of 12 $\mu g/m^3$ in conjunction with retaining the current 24-hour standard level at 35 $\mu g/m^3$ (12/35); an annual standard level of 11 $\mu g/m^3$ in conjunction with retaining the current 24-hour standard level at 35 $\mu g/m^3$ (11/35); and an annual standard level of 11 $\mu g/m^3$ in conjunction with a 24-hour standard level of 30 $\mu g/m^3$ (11/30). We modeled to project future-year $PM_{2.5}$ concentrations for a 2020 base case using CMAQ and then estimated the air quality concentrations for meeting 15/35, 14/35, 13/35, 12/35, 11/35 and 11/30 using air quality ratios.

3.3.1.1 Calculating Future-year Design Values for 2020 Base Case

To estimate costs of attaining the alternative NAAQS, we use air quality modeling results to predict the impact of the control strategies on future-year attainment. This is done by using the air quality model results in a relative sense, as recommended by the EPA modeling guidance (EPA, 2007), and estimating future-year $PM_{2.5}$ relative reduction factors (RRFs). RRFs are ratios that are calculated from the changes in $PM_{2.5}$ species concentrations between recent-year and future-year air quality modeling results. RRFs are calculated for each $PM_{2.5}$ component. 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 IMPROVE Network, the Speciated Trends Network (STN), and the Federal Reference Method (FRM) Network.

To more easily apply this methodology, EPA has created software, called Modeled Attainment Test Software (MATS) (Abt, 2010), to calculate future-year $PM_{2.5}$ annual and 24-hour standard design values. For this RIA, the RRFs are based on the changes in modeled concentrations between the 2005 and 2020 base case. Ambient measurements used in MATS are from IMPROVE and STN sites for 2004–2006 and FRM sites for 2003–2007. Output from MATS includes the projected future-year annual and 24-hour standard design values, as well as percentage sulfate, nitrate, ammonium, elemental carbon, organic carbon and crustal matter contributing to the annual and 24-hour standard design values for each site. These data are useful to better understand the PM species contributing to high $PM_{2.5}$ concentrations and to help determine what control measures might be most effective in reducing the future-year design values to the proposed levels. Annual and 24-hour standard design values for 2005 and 2020 base case are discussed in Chapter 4 and shown in Appendix 4.

3.3.1.2 Calculating Future-year Design Values for Meeting the Current Standard and Proposed Alternative Standard Levels

To estimate the tons of emissions reductions needed to reach attainment of the current and proposed alternative standard levels, we calculated air quality ratios based on how modeled concentrations changed with changes in emissions between a recent year of air quality, 2005, and the future year of air quality, 2020. These air quality ratios represent an estimate of how the annual standard design value at a monitor would change in response to emissions reductions of SO_2, NO_x, or direct $PM_{2.5}$. Below are the details of how these air quality ratios were estimated.

To calculate the air quality ratios for changes in response to emissions reductions of SO_2 and NO_x we used the following methodology.

Step 1: The speciated changes in annual standard design values between 2005[3] and 2020 were obtained from the MATS (Abt, 2010) output files. For each monitor, we computed the percent change in the NH_4SO_4 and NH_4NO_3[4] components of the annual standard design value between 2005 and 2020, relative to the 2005 monitor annual standard design value.

Step 2: For NH_4SO_4[1] and NH_4NO_3[3] components, we computed the change in emissions of SO_2 and NO_x used in the air quality modeling for the 2005 and 2020 base case for groups of

[3]As described previously in this section, the "2005" annual design values are based on ambient measurements used in MATS from IMPROVE and STN sites for 2004–2006 and FRM sites for 2003–2007. These years of ambient measurements were selected since they frame the air quality model year of 2005. Because the air quality model is used to predict the change in design values between recent and projected future year air quality, with the modeled RRFs being applied to the recent year measured design values, it is important to select ambient measurement years that include the model year to allow a more true prediction of the future year air quality.

[4]The NH4SO4 and NH4NO3 components are computed using the SO4, NO3, NH4 and water fraction from MATS as described in EPA guidance (EPA, 2007).

adjacent counties. This larger grouping of counties allowed us to better represent the more regional nature of NH_4SO_4 and NH_4NO_3 formation and transport. These groupings of counties were selected by including all counties within a state that bordered a county with a monitor above the current or proposed alternative standard levels. For the state of California, where these groupings could have expanded to include most of the counties within the state if we had used the same selection criteria, we determined smaller groupings to more realistically represent the area around a monitor from which emissions reductions would most influence changes in the design value. This smaller grouping was also done for counties in Utah[5]. These groups are listed in Appendix 4, Table 4.A-5.

Step 3: Using the data from Steps 1 and 2, we computed the percent change in the NH_4SO_4 component of the annual standard design value at each monitor per reduction of 1000 tons of SO_2 emissions in the surrounding counties between the 2005 and 2020 base case air quality and emissions data. Similarly, we computed the percent change in the annual standard NH_4NO_3 component of the design value at each monitor per change in 1000 tons of NOx emissions in the surrounding counties.

Step 4: The data from Step 3 are then used to compute the median value for all monitors within the grouping of counties of the percent change in the NH_4SO_4 and NH_4NO_3 components of the annual standard design value per change in tons of SO_2 and NO_3 emissions, respectively. This gives us an estimate for each grouping of monitors that indicates the response of the sulfate and nitrate components of the annual standard design values to changes in SO_2 and NO_x emissions, relative to the $PM_{2.5}$ speciation at the monitor.

Step 5: The percent change values from Step 4 are then multiplied by the NH_4SO_4 and NH_4NO_3 speciation values at each monitor in the 2020 base case to produce the "air quality ratios." These data give an estimate of how the annual standard design value ($\mu g/m^3$) at a monitor would change if 1000 tons of SO_2 and/or NO_x emissions were reduced in the county in which the monitor is located.

To calculate the air quality ratios for changes in response to emissions reductions of direct $PM_{2.5}$ we follow the following methodology.

[5] To determine the counties for the smaller groupings in California and Utah, we simply grouped counties together geographically with a minimum of two counties allowed in each of the smaller groupings.

Step 1: The speciated changes in annual standard design values between 2005[6] and 2020 were obtained from the MATS (Abt, 2010) output files. For each monitor, we computed the percent change in the direct $PM_{2.5}$[7] components of the annual standard design value between 2005 and 2020, relative to 2005 monitor annual standard design value.

Step 2: We computed the change in emissions of direct $PM_{2.5}$ in the emissions inventory data used in the air quality modeling for the 2005 and 2020 base case for each county.

Step 3: Using the data from Steps 1 and 2, we computed the percent change in the annual standard direct $PM_{2.5}$ component of the design value at each monitor per change in tons of direct $PM_{2.5}$ emissions at the county level.

Step 4: The data from Step 3 are then used to compute the median value of the percent change in the $PM_{2.5}$ component of the annual standard design value per change in tons of direct $PM_{2.5}$ emissions for all monitors within the grouping of counties used to compute the SO_2 and NO_x air quality ratios (see Appendix 4, Table 4.A-5) within a state. We now have an estimate for each grouping of monitors that indicates the response of the direct $PM_{2.5}$ components of the annual standard design values to changes in direct $PM_{2.5}$ emissions, relative to the $PM_{2.5}$ speciation at each monitor.

Step 5: The percent change values from Step 4 are then multiplied by the direct $PM_{2.5}$ speciation values at each monitor in the 2020 base case to produce air quality ratios.

Step 6: The responsiveness of air quality at a specific monitor location to direct $PM_{2.5}$ emission reductions will depend on several factors including the specific meteorology and topography in an area and the nearness of the emissions source to the monitor. Because of the more local influence of changes in directly emitted $PM_{2.5}$ emissions on air quality, a monitor where significant changes in direct $PM_{2.5}$ emissions occurred between 2005 and 2020 due to sources very close to a monitor can result in large non-representative values in Step 5. A large change suggests the monitor is more responsive to $PM_{2.5}$ emissions reductions than it actually

[6]As described previously in this section, the "2005" annual design values are based on ambient measurements used in MATS from IMPROVE and STN sites for 2004–2006 and FRM sites for 2003–2007. These years of ambient measurements were selected since they frame the air quality model year of 2005. Because the air quality model is used to predict the change in design values between recent and projected future year air quality, with the modeled RRFs being applied to the recent year measured design values, it is important to select ambient measurement years that include the model year to allow a more true prediction of the future year air quality.

[7]The direct $PM_{2.5}$ design value component is computed by summing the elemental carbon, organic carbon and crustal portions of the design value.

would be if those reductions were applied further away from the monitor. Given that the air quality ratios must be applicable to multiple monitors across each county and must be applicable for emissions reductions where we may not know the specific source location (e.g., extrapolated emissions reductions), the air quality ratios we employ should not be strongly influenced by very local emissions changes which may have a much larger, non-representative impact on air quality at the nearby monitor. To remedy this and obtain representative values for air quality ratios, we separated all the counties for which $PM_{2.5}$ air quality ratios were computed (as shown in Table 4.A-10) into four areas of the country: East, West, Northern California, and Southern California as shown in Table 3-3[8]. We then computed a single "trimmed" median $PM_{2.5}$ ratio for each of the four areas after removing the highest ten percent of the values in each area. That is, we calculated the median value of all the $PM_{2.5}$ air quality ratios in each area over counties for which air quality ratios had been calculated after the highest ten percent of the values were removed. The resulting $PM_{2.5}$ air quality ratios that are used for all monitors in the four areas are shown in Table 3-3. These data give an estimate of how the annual standard design value (µg/m3) at a monitor would change if 1000 tons of direct $PM_{2.5}$ emissions were reduced in the county in which the monitor is located.

Table 3-3. Area Definitions and $PM_{2.5}$ Air Quality Ratios

Area	States and Counties Included	$PM_{2.5}$ Air Quality Ratio (µg/m^3 Change in Direct $PM_{2.5}$ per 1,000 Tons PM)
East	Alabama, Georgia, Illinois, Michigan, New York, Ohio, Pennsylvania, and Texas (all counties with air quality ratios)	1.238
West	Arizona, Idaho, Montana, Oregon, Utah, and Washington (all counties with air quality ratios)	1.929
Northern California	Butte, Colusa, Contra Costa, Fresno, Inyo, Kern, Kings, Merced, Monterey, Placer, Plumas, Sacramento, San Joaquin, San Luis Obispo, Santa Clara, Solano, Stanislaus, Sutter, Tulare, and Yolo, California	1.879
Southern California	Imperial, Los Angeles, Orange, Riverside, San Bernardino, San Diego, and Ventura, California	0.597

To be able to estimate how the 24-hour standard design value would change in response to a lower annual design value, we computed the ratio of the change in the annual

[8] California was separated into two areas because of the large number of counties analyzed and because of the large differences seen in the $PM_{2.5}$ air quality ratios for the northern versus the southern counties of California.

design value to the change in the 24-hour standard design value between 2005 and 2020 base case for each monitor, and then use these data to compute the average value for all monitors within a state that are included within a grouping shown in Appendix 4, Table 4.A-5. The end result is an estimate for each state of the expected change in the 24-hour standard design value per 1 $\mu g/m^3$ change in the annual standard design value. These values varied from state-to-state but in general, a 1 $\mu g/m^3$ change in the annual standard design value corresponded to a 2–3 $\mu g/m^3$ change in the 24-hour standard design value at the same monitor. Tables of these values are provided with the air quality ratios in Chapter 4.

3.3.1.3 Estimating Emissions Reductions and Costs of Attaining the Current and Proposed Alternative $PM_{2.5}$ Standards

The air quality ratios described in Section 3.3.1.2 are used to determine the most effective control measures for reducing the annual and 24-hour standard design values to meeting the current and proposed alternative standard levels. The total amount of SO_2, NO_x and/or direct $PM_{2.5}$ emissions reduced per county, based on the control measures selected for each strategy, are then used in conjunction with the air quality ratios to estimate how the annual and 24-hour standard design values would change in the counties with emissions reductions. The details of control measure selections and their associated costs are described in Chapter 4.

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

MATS (Abt, 2010) can also provide gridded fields of changes in annual average $PM_{2.5}$ concentrations for the entire CMAQ 12km domain. MATS does this by calculating RRFs at every grid cell within the CMAQ domain for each future-year control scenario, and applying these RRFs to ambient data that have been interpolated to cover all grid cells in the modeling domain. The basic interpolation technique, called Voronoi Neighbor Averaging (VNA), identifies the set of monitors that are nearest to the center of each CMAQ grid cell, and then takes an inverse distance squared weighted average of the monitor concentrations. A "fused" spatial field is then calculated by adjusting the interpolated ambient data (in each grid cell) up or down by a multiplicative factor calculated as the ratio of the modeled concentration at the grid cell divided by the modeled concentration at the nearest neighbor monitor locations (weighted by distance). We use the 2005 and 2020 base case CMAQ modeling outputs, in conjunction with the ambient measurements from the IMPROVE and STN sites for 2004–2006 and FRM sites for 2003–2007, to create a spatial surface for the 2020 base case.

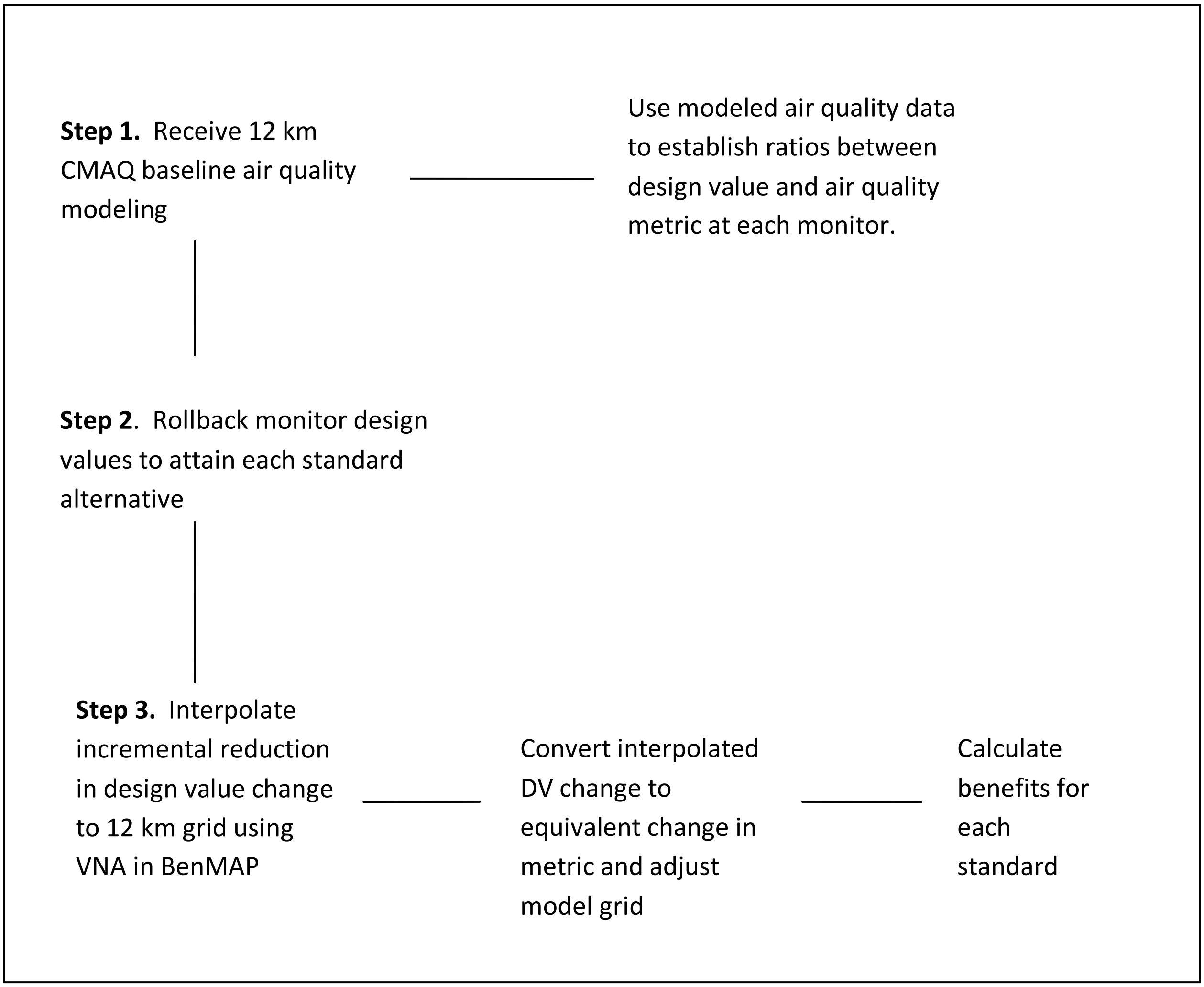
To generate spatial surfaces that represent annual average $PM_{2.5}$ concentrations when attaining the current and proposed alternative standard levels for the benefits analysis, we need to adjust the 2020 base case spatial surface to reflect attainment of the current and alternative standard levels. Given the MATS gridded annual average $PM_{2.5}$ concentrations for the CMAQ 12km domain and projected design values at monitors, the "monitor rollback" approach is used to approximate the air quality change resulting from attaining alternative NAAQS at each design value monitor. Figure 3-2 depicts the rollback process. 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 variant of the VNA interpolation method to adjust the CMAQ-modeled concentrations such that each area attains the standard alternatives. Using the VNA spatial averaging technique, the annual average $PM_{2.5}$ spatial surface is smoothed to minimize sharp gradients in $PM_{2.5}$ concentrations in the spatial fields due to changes in the monitor concentrations.

3.3.1.5 Limitations of Using Air Quality Ratios

There are important limitations to the methodology of calculating and using air quality ratios to predict the response of air quality to emissions changes. The air quality ratios are calculated with results from only two CMAQ model runs and are based on the assumption that the monitor design values would decrease with additional reductions in emissions of SO_2, NO_x and direct $PM_{2.5}$ in the future similar to how the two CMAQ model 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 proposed alternative standards.

Figure 3-2. Diagram of Rollback Method



3.3.2 Visibility

As described in the Policy Assessment Document (EPA, 2011a) and Chapter 2 of this RIA, the formula for total light extinction (b_{ext}) in units of Mm^{-1} using the original IMPROVE equation is:

$$b\text{ext} = 3 \text{ x } f(\text{RH}) \text{ x } [Sulfate] + 3 \text{ x } f(\text{RH}) \text{ x } [Nitrate] + 4 \text{ x } [Organic\ Mass] + 10 \text{ x } [Elemental\ Carbon] + 1 \text{ x } [Fine\ Soil] + 0.6 \text{ x } [\text{Coarse Mass}] + 10 \qquad (3.1)$$

where 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. Using equation (3.1), light extinction (b_{ext}) can then be converted into units of deciviews, a scale frequently used in the scientific and regulatory literature on visibility.

3.3.2.1 Calculating Future-year Visibility Design Values for 2020 Base Case

The visibility design value calculations are based on 24-hour averages and the 90th percentile format. To estimate future-year visibility design values, we use the air quality modeling results in a relative sense, as recommended by the EPA modeling guidance (EPA, 2007), and estimate future-year relative reduction factors (RRFs) for each speciated component of the light extinction (b_{ext}) equation. To be consistent with visibility calculations described in the Policy Assessment Document (EPA, 2011a), which focuses on $PM_{2.5}$ visibility, we do not include coarse mass ($PM_{10\text{-}2.5}$) in the calculation. The steps for projecting the future-year visibility design values are described below.

Step 1: We extract 24-hour averages of sulfate, nitrate, organic mass, elemental carbon and fine soil for the 2020 future-year modeled base case for each CMAQ grid cell in which a STN monitor is located. For the assumption that sulfate is fully neutralized and all nitrate is assumed to be ammonium nitrate, we multiply 24-hour average sulfate mass by 1.375 and nitrate mass by 1.29.

Step 2: For the Regional Haze Program, there exists a gridded file of monthly averaged *f*(RH) climatological mean values.[9] Using these data, we assign a *f*(RH) values to each STN monitor for each season[10] by averaging the 3 monthly *f*(RH) values in each season using the data from the closest available data point.

Step 3: Using the data from Step 1 &2, we calculate b_{ext} for every day in the 2005 modeled base case using equation (3.1) without the coarse mass component.

Step 4: For every season, we extract the top 10% worst modeled visibility days (i.e., top 9 days) in 2005 based on their b_{ext} values. Using the species concentrations for these nine

[9] U.S. EPA, Interpolating Relative Humidity Weighting Factors to Calculate Visibility Impairment and the Effects of IMPROVE Monitor Outliers, prepared by Science Applications International Corporation, Raleigh, NC, EPA Contract No. 68-D-98-113, August 30, 2001.

[10] Each season is defined as Winter (Dec, Jan & Feb), Spring (Mar, Apr & May), Summer (Jun, Jul & Aug) and Fall (Sep, Oct & Nov).

maximum b_{ext} days, we calculate the average species specific b_{ext} value for each season for each monitor location.

Step 5: We repeat step 3 for each of the future-year modeled scenario, and extract the same calendar days that were selected in step 4 for 2005.

Step 6: Using the data from steps 4 and 5, we calculate the species specific Relative Response Factors (RRFs) for each monitor in each season for each of the future-year modeled scenario. This is done by dividing the average speciated b_{ext} value for the top 10% worse visibility days for each season for every monitor in each future-year scenario by the average species specific concentration for same the season for every monitor in 2005. In this way, we will have an RRF for every monitor for each season for each future-year control scenario.

Step 7: The set of seasonal RRFs for each monitor for each future-year control scenario are applied to the corresponding 2004–2006 ambient data[11] and the 90th percentile value is extracted. The end result is a set of visibility design values for each future-year scenario.

3.3.2.2 Calculating Future-year Visibility Design Values for Meeting the Current and Proposed Alternative Standard Levels

It is important to understand how changes in the $PM_{2.5}$ design values to simulate full attainment for each proposed alternative NAAQS will affect visibility design values.

As described in Section 3.3.1.2, we apply a methodology of air quality ratios to estimate the emissions reductions needed to meet the current and proposed alternative levels for the primary standard for $PM_{2.5}$. While this methodology can estimate how the emissions reductions in each control scenario will affect changes in the future-year annual design values, and the corresponding response of the future-year 24-hour design values to these changes in the annual design value, it is unable to estimate how each of the $PM_{2.5}$ species will change with these emission reductions. Given that estimating changes in future-year visibility is dependent on the IMPROVE equation (3.1) and how the $PM_{2.5}$ species are projected to change in time, we are unable to estimate visibility design values for meeting the current and proposed alternative levels for the primary $PM_{2.5}$ standard.

[11] These years of ambient measurements were selected since they frame the air quality model year of 2005. Because the air quality model is used to predict the change in design values between recent and projected future year air quality, with the modeled RRFs being applied to the recent year measured design values, it is important to select ambient measurement years that include the model year to allow a more true prediction of the future year air quality.

3.3.2.3 Estimating Changes in Visibility for Analyzing Welfare Benefits

The visibility calculations for the welfare benefits assessment are based on 24-hour average light extinction (b_{ext}) values, averaged over the year and converted to units of deciviews. As described in Sections 3.3.2.1 and 3.3.2.2, we calculated the visibility design values for the 2020 base case but were unable to estimate how these visibility design values would change for meeting the current and proposed alternative levels for the primary $PM_{2.5}$ standard. In this same way, we are unable to estimate the light extinction values for meeting the current and proposed alternative levels for the primary $PM_{2.5}$ standard, which are needed to assess welfare benefits.

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CHAPTER 4
CONTROL STRATEGIES

4.1 Synopsis

In order to estimate the costs and benefits of alternative $PM_{2.5}$ standards, the U.S. EPA has analyzed hypothetical control strategies that areas across the country might employ to attain alternative more stringent annual standards of 13, 12, and 11 µg/m^3 in conjunction with retaining the 24-hour standard of 35 µg/m^3, as well as an alternative more stringent annual standard of 11 µg/m^3 in conjunction with an alternative more stringent 24-hour standard of 30 µg/m^3 (referred to as 13/35, 12/35, 11/35, and 11/30). The U.S. EPA also analyzed a 14/35 alternative standard and determined that all counties would meet such a standard concurrent with meeting the existing 15/35 standard at no additional costs and with no additional benefits because of significant air quality improvements from the Mercury and Air Toxics Standards (MATS), the Cross-State Air Pollution Rule (CSAPR), and other Clean Air Act rules as described in Chapter 3, Section 3.2.2. Thus, there is no need to present an analysis of 14/35.

For the purposes of this discussion, it will be helpful to define some terminology. These definitions are specific to this analysis:

- Base Case—Emissions projected to the year 2020 reflecting current state and federal programs, including the Cross-State Air Pollution Rule and the Mercury and Air Toxics Standards. This does not include control programs specifically for the purpose of attaining the current $PM_{2.5}$ standard (15/35).

- Baseline—Emissions projections to the year 2020 reflecting the base case plus additional emission reductions needed to reach attainment of the current $PM_{2.5}$ Standard (15/35).

- Alternative Standard Analysis—Emission reductions and associated hypothetical controls needed to reach attainment of the alternative standards. These reductions and controls are incremental to the baseline.

- Design Value—A metric that is compared to the level of the National Ambient Air Quality Standard (NAAQS) to determine compliance. Design values are typically used to classify nonattainment areas, assess progress towards meeting the NAAQS, and develop control strategies. The design value for the annual $PM_{2.5}$ standard is calculated as the 3-year average of annual means for a single monitoring site or a group of monitoring sites. The design value for the 24-hour standard is calculated as the 3-year average of annual 98th percentile 24-hour average values recorded at each monitoring site.

The U.S. EPA has analyzed the impact that additional emissions controls across numerous sectors would have on predicted ambient $PM_{2.5}$ concentrations incremental to a baseline, which includes the current $PM_{2.5}$ standard as well as other major rules such as CSAPR and MATS. Thus, the analysis for a revised standard 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 achieving emissions reductions to move towards a national attainment of a tighter standard. It is not a recommendation for how a tighter $PM_{2.5}$ standard should be implemented, and states will make all final decisions regarding implementation strategies once a final NAAQS has been established.

In order to analyze these hypothetical control strategies incremental to attainment of the current standard and beyond other existing major rules, the U.S. EPA employed a multi-stage approach. First, the U.S. EPA identified controls to be included in the base case (e.g., reflecting current standard of 15/35) to reflect current state and federal programs. Next additional controls were applied to attain the current $PM_{2.5}$ standard. The current state and federal programs combined with the additional controls needed for attainment of the current $PM_{2.5}$ standard make up the baseline for this analysis. Once the baseline was established, we applied additional known controls within counties containing a monitor predicted to exceed the standard alternatives of 13/35, 12/35, 11/35, and 11/30 so as to bring them into attainment with the various alternatives in 2020.[1] This chapter presents the hypothetical control strategies and the results in 2020 after their application. For most of these alternative standards, application of known control measures did not achieve attainment. In such cases, additional emission reductions beyond the capability of known controls were estimated in order to reach full attainment.

4.2 $PM_{2.5}$ Control Strategy Analysis

4.2.1 Establishing the 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 hypothetical control strategies for attaining a more stringent primary standard, it is important to first estimate $PM_{2.5}$ levels in 2020[2] given the current NAAQS standards (15/35) and trends. This scenario is known as the

[1] Refer to Table 4-2 for details on the number of counties with exceedances and the number of additional counties where reductions were applied.

baseline. Establishing this baseline allows us to estimate the incremental costs and benefits of attaining any alternative primary standard.

The baseline includes reductions already achieved as a result of national regulations, reductions expected prior to 2020 from recently promulgated national regulations[3] (i.e., reductions that were not realized before 2005 but are expected prior to attainment of the current PM standard), 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[4]. Two steps were used to develop the 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 3.2.2 for a more detailed discussion of the rules reflected in the 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)

[3] The recently proposed Boiler MACT and CISWI reconsiderations are not included in the base case. These rules were not yet proposed at the time of this analysis. It is not clear how the geographic scope of this rule will match with the counties analyzed for this RIA—the costs may decrease but the magnitude is uncertain.

[4] The amendments to the Low Emissions Vehicle Program (LEV-III) in California are not included in the base case. This program requires an approval of U.S. EPA via a waiver. At the time of this analysis the waiver had not been submitted.

- 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)
- Cross-State Air Pollution Rule (U.S. EPA, 2011)
- Mercury and Air Toxics Standards (U.S. EPA, 2011)

Note that 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 NAAQS RIAs are hypothetical. Second, because the base case reductions alone were not predicted to bring all areas into attainment with the current standard (2 counties are projected to exceed an alternative standard of 13/35 and 18 counties are projected to exceed an alternative standard of 12/35—see Section 4.2.2.1 for more details), the U.S. EPA used a hypothetical control strategy to apply additional known controls to illustrate attainment with the current $PM_{2.5}$ standard. Additional control measures were used in two sectors to establish the baseline:[5] Non-Electricity Generating Unit Point Sources (Non-EGUs) and Non-Point Area Sources (Area).

The 2020 baseline for this analysis presents one scenario of future year air quality based upon specific control measures, including federal rules such as CSAPR and 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 U.S. EPA plans to issue the final rule no later than December 14, 2012 and intends to complete designations two years following promulgation of the final rule. Under the Clean Air Act, States are required to submit State implementation plans within 3 years of the effective date of the designations. The plans are required to show attainment as expeditiously as practicable but no later than 5 years following the effective date of the designations, with the possibility, in certain cases, of an attainment date up to 10 years from the effective date of the designations, considering the severity of air quality concentrations in the area and the availability and feasibility of emission control measures. Designations will likely be based on air quality data from 2011-2013, but attainment will not occur until 2020 at

[5] In establishing the baseline, the U.S. EPA selected a set of cost-effective controls to simulate attainment of the current PM2.5 standard. These control sets are hypothetical as states will ultimately determine controls as part of the SIP process.

the earliest. Thus EPA's projections for control costs and benefits focus on the year 2020. 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. Refer to Section IX of the proposed $PM_{2.5}$ NAAQS for more details concerning implementation requirements for the proposed NAAQS.

Two maps of the country are presented in Figures 4-1 and 4-2, which show the predicted concentrations for year 2020 for the 575 counties with $PM_{2.5}$ annual design values and 569 counties with 24-hour design values prior to applying controls to meet the current standard of 15/35. Control measures were applied to 14 counties in the baseline analysis to meet the current $PM_{2.5}$ standard.

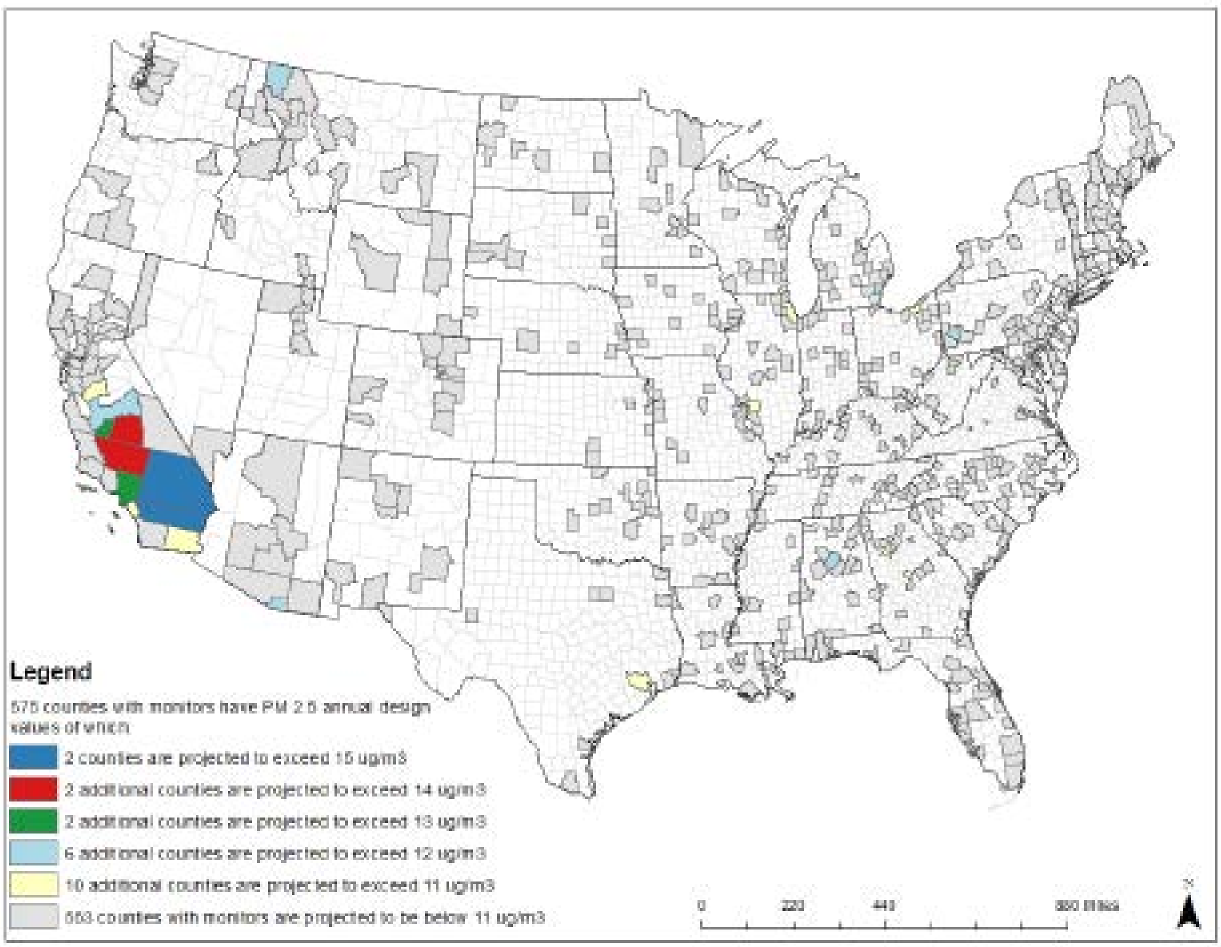

Figure 4-1. Counties Projected to Exceed the Baseline and Analysis Levels of the $PM_{2.5}$ Annual Standard Alternatives in 2020

4.2.1.1 Controls Applied in the Baseline

The purpose of identifying and analyzing hypothetical baseline controls for $PM_{2.5}$ and its precursors is to establish a level of emissions associated with ambient concentrations that would meet the current $PM_{2.5}$ standard. The additional known controls included in the baseline to simulate attainment with 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 $PM_{2.5}$ were determined to be most effective close to the exceeding monitors with NO_x and SO_2 controls supplemented depending upon the monitor speciation data. $PM_{2.5}$ control measures were

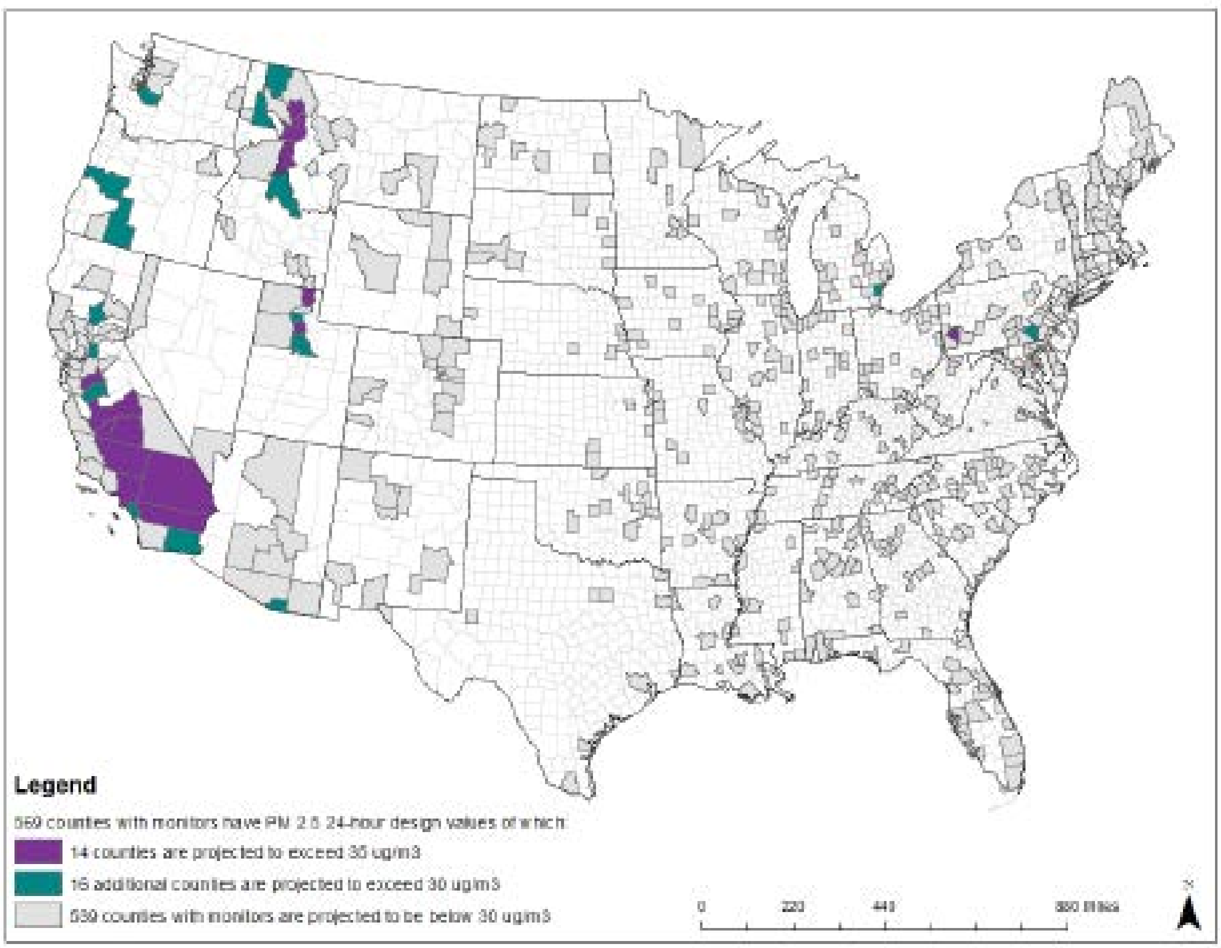

Figure 4-2. Counties Projected to Exceed the Baseline and Analysis Levels of the $PM_{2.5}$ 24-hour Standard Alternatives in 2020

applied in the county containing the exceeding monitor for the non-EGU point and area source emissions. If additional emission control was needed, SO_2 and NO_x control measures were applied within the county exceeding and any contiguous county.[6] Additional control measures were not applied to electric generating units (EGUs) due to the extensive nature of controls resulting from the inclusion of MATS and CSAPR, and additional controls were not applied to mobile sources due to our inability to capture regional reductions using the air quality screening methodology employed for this analysis.

[6] Refer to Table 4-2 for details on the number of counties with exceedances and the number of additional counties where reductions were applied.

Table 4-1. Controls Applied in the Baseline for the Current $PM_{2.5}$ Standard*

Pollutant	Control Measure	15/35	12/35	11/35	11/30
NO_x	Selective Catalytic Reduction (SCR)	x			x
	Non-selective Catalytic Reduction (NSCR)	x		x	x
	Oxy-Firing	x		x	x
	Bio-solid Injection				
	SCR + Steam Injection	x		x	x
	Low NO_x Burners (LNB)	x		x	x
	LNB + SCR	x			x
	LNB + Selective Non-catalytic Reduction (SNCR)	x			
$PM_{2.5}$	Fabric Filters	x	x	x	x
	Dry Electrostatic Precipitators (ESPs)	x			
	Wet ESPs	x		x	x
	Venturi Scrubbers				
SO_2	Spray Dryer Absorber (SDA)	x		x	x
	Flue Gas Desulfurization (FGD)	x		x	x
	Wet FGDs	x		x	x
NO_x	Water Heater + LNB Space Heaters	x		x	x
	Low-NO_x Burners for Residential Natural Gas	x		x	x
$PM_{2.5}$	Fireplace Inserts for Home Heating	x	x	x	x
	Basic Smoke Management Practices and Establishment of Smoke Management Programs for Prescribed Burning and other Open Burning**	x	x	x	x
	Woodstove Advisory Program	x	x	x	x
	ESPs for Commercial Cooking	x		x	x
SO_2	Fuel Switching for Stationary Source Fuel Combustion	x		x	x
	Low Sulfur Home Heating Fuel	x		x	x

* As discussed elsewhere in this chapter, no known controls were applied for 14/35 or 13/35.
**Includes specific practices such as episodic bans on open burning, and substituting chipping for open burning.

4.2.2 Alternative Standard Control Strategies

After establishing the baseline of attaining the current standard of 15/35, additional emission reductions needed to meet four alternative standards 13/35, 12/35, 11/35, and 11/30 were calculated.

4.2.2.1 Counties Exceeding Alternative Standards

Only two counties are projected to exceed an alternative standard of 13/35 using the results from the baseline analysis. These are Riverside County, CA and San Bernardino County, CA. Figures 4-3 through 4-5 show the counties projected to exceed the alternative standards 12/35, 11/35, and 11/30, respectively. Six counties are projected to exceed an alternative standard of 12/35, eighteen counties are projected to exceed an alternative standard of 11/35, and thirty-five counties are projected to exceed an alternative standard of 11/30. For a complete list of monitor values see Appendix 4.A.

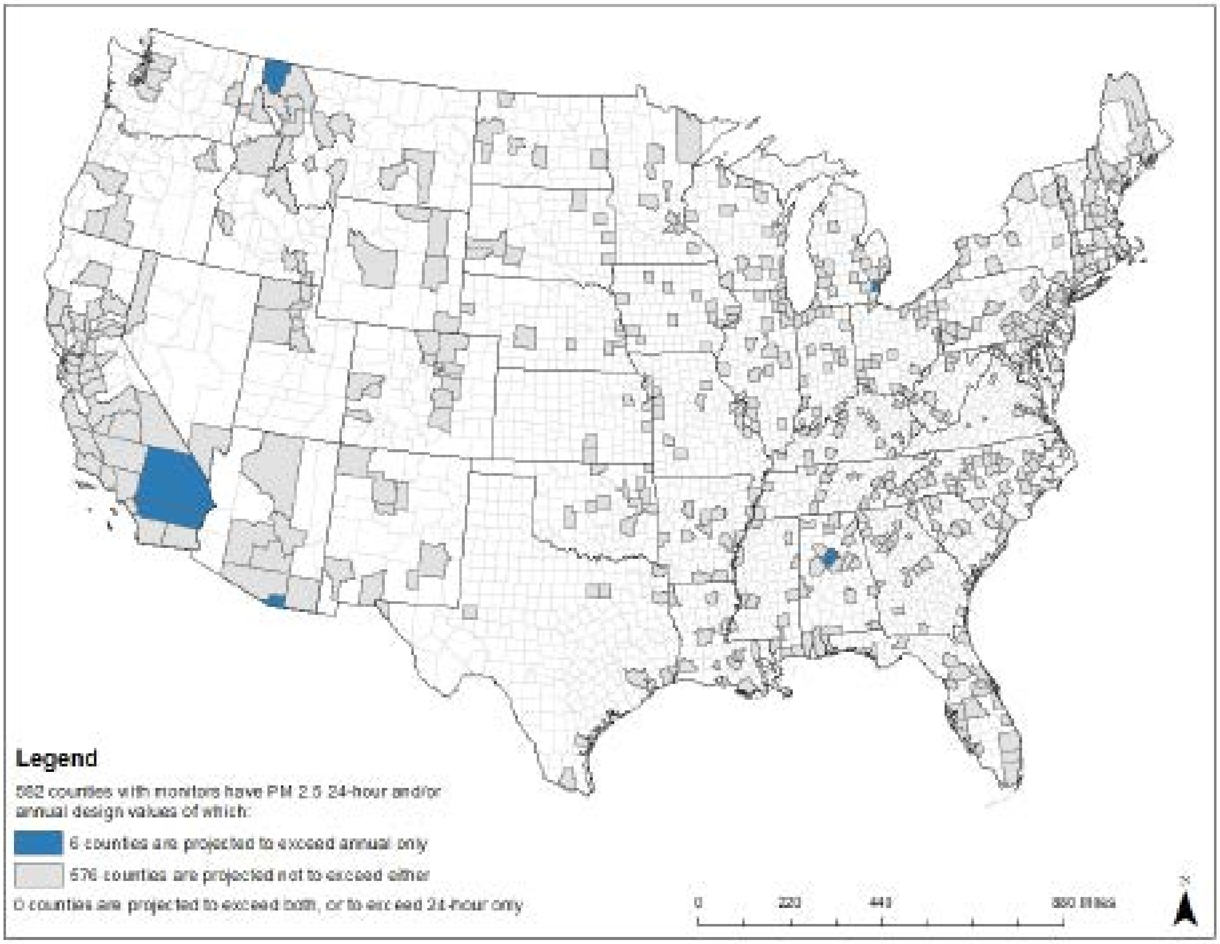

Figure 4-3. Counties Projected to Exceed the 12/35 ug/m^3 Alternative Standard After Meeting the Baseline (Current Standard) in 2020

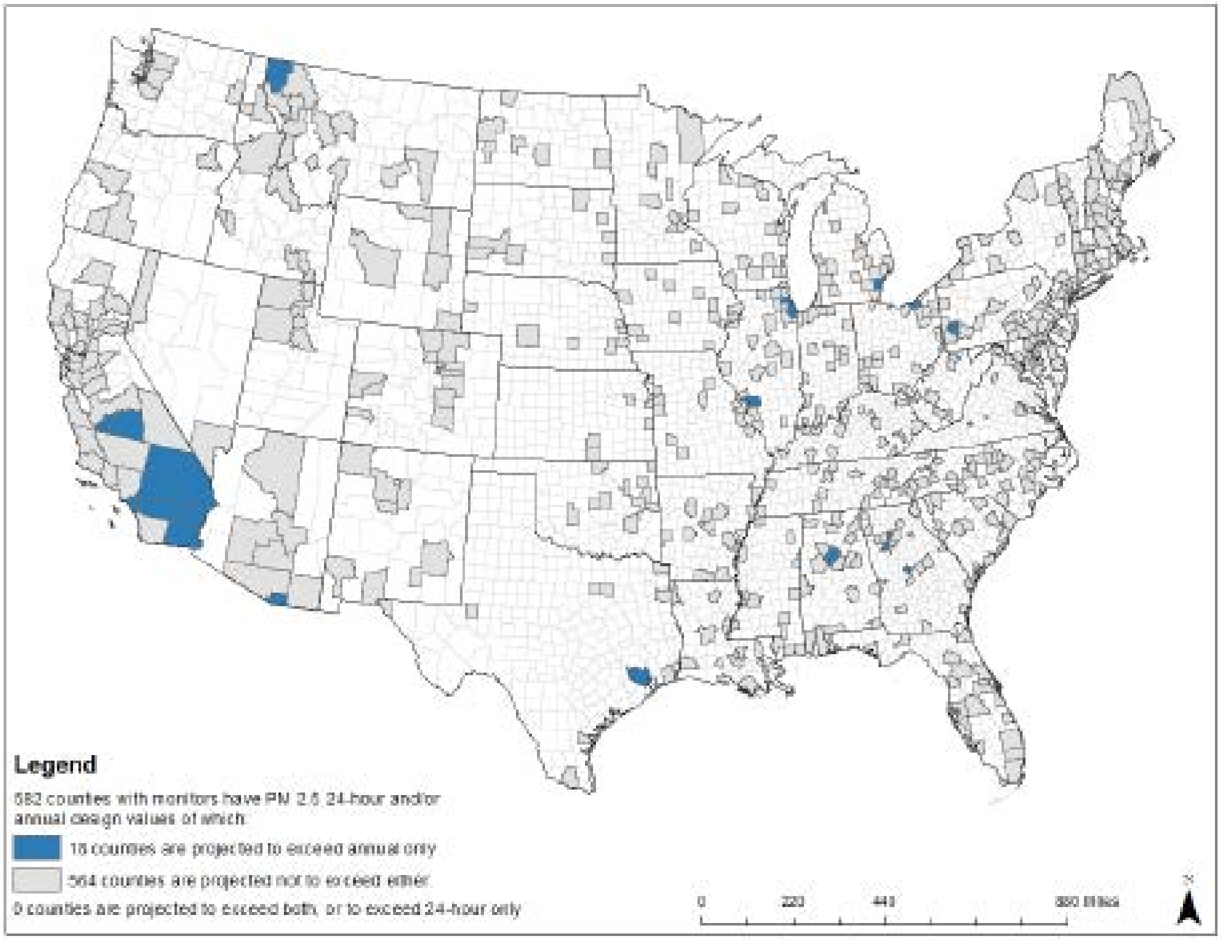

Figure 4-4. Counties Projected to Exceed the 11/35 ug/m^3 Alternative Standard After Meeting the Baseline (Current Standard) in 2020

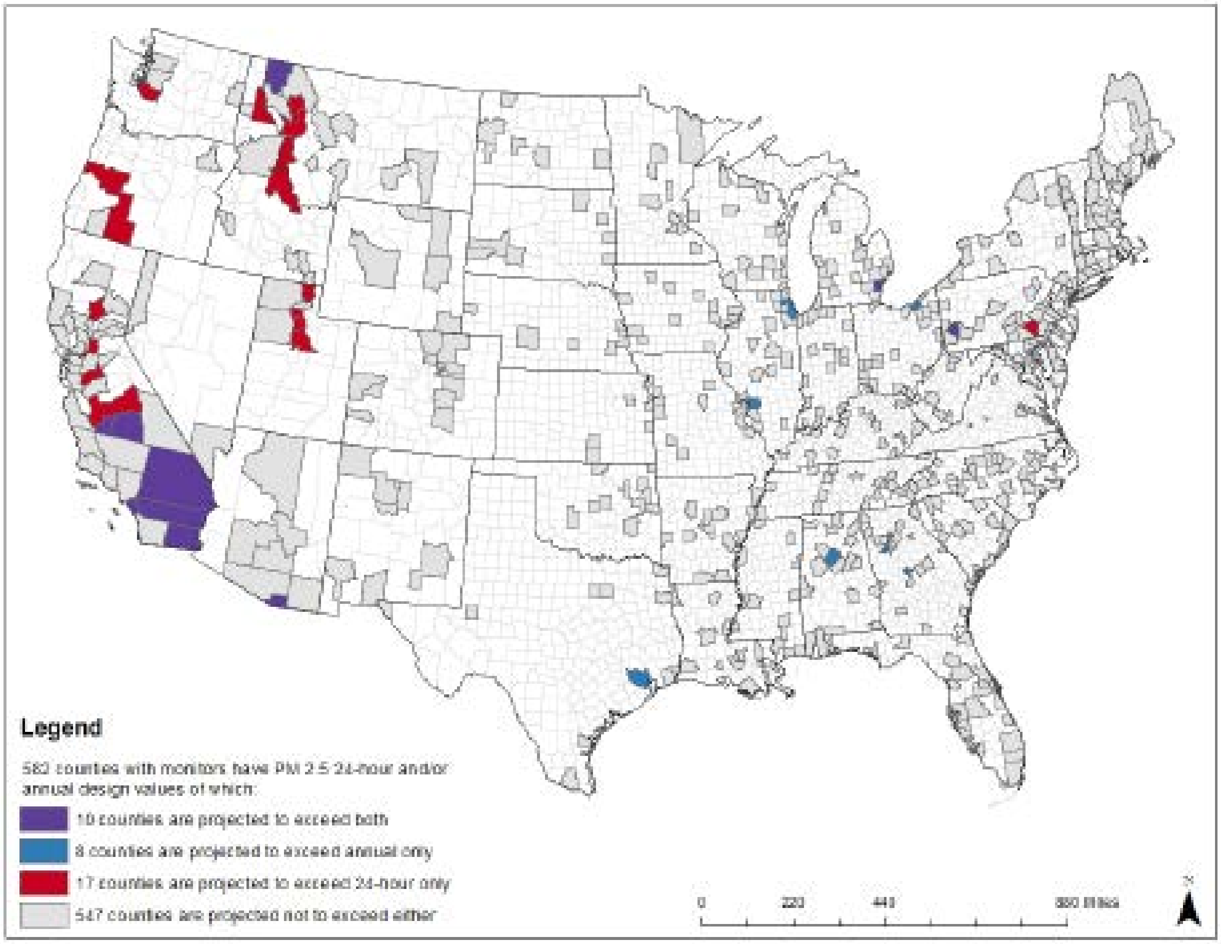

Figure 4-5. Counties Projected to Exceed the 11/30 ug/m^3 Alternative Standard After Meeting the Baseline (Current Standard) in 2020

In developing the control strategies for this RIA, the U.S. EPA first applied known controls to reach attainment. For these control strategies, controls for two sectors were used in developing the control analysis, as discussed previously: non-EGU point and area sources. An approach similar to that taken for the baseline analysis was used in the analysis for the control strategies for the alternative standards. Due to the lack of air quality modeling for the control strategies, county-specific ratios of air quality response to emission reductions were applied based on recent air quality modeling results. A least cost framework was adapted to adjust for the use of the air quality to emissions ratios.

In this analysis, $PM_{2.5}$ controls were applied first because they were more cost-effective and the air quality ratio approach is generally more accurate for $PM_{2.5}$ emission changes than for emission changes from the precursors, SO_2 and NO_x. If additional control was needed SO_2 and NO_x controls were added.

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 in the surrounding 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 surrounding counties to which controls were applied for the alternative standards analyzed. For a complete list of geographic areas for all alternative standards see Appendix 4.A.

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

Alternative Level	Number of Counties with exceedances	Number of Additional Counties where reductions were applied
13/35	2	3
12/35	6	25
11/35	18	86
11/30	35	134

There were some areas where known controls did not achieve enough emission reductions to attain the alternative annual standards in 2020. To complete the analysis, the U.S. EPA then estimated the additional emission reductions required to reach attainment. The methodology used to develop those estimates and those calculations are presented in Section 4.2.3.

4.2.2.2 Non-EGU and Area Controls Applied for Alternative Standards

Non-EGU point and area control measures were identified using the U.S. EPA's Control Strategy Tool[7] (CoST). Many of these controls are summarized in Appendix 4.A.

Area source emissions data are generated at the county level, and therefore controls for this emission sector were applied to the county. Area source controls were applied to NO_x, SO_2, and $PM_{2.5}$. Table 4-1 lists the major controls applied to each sector. The same controls that were applied in the baseline analysis were applied to additional sources and additional counties in the analyses for the alternative standards. Controls for area sources were applied to: home heating, restaurant operations, prescribed burning, and other open burning.

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

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 standalone 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.

To more accurately depict available controls, the U.S. 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. 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.

4.2.2.3 Emission Reductions

Table 4-3 shows the emission reductions from known controls for the alternative standards analyzed.

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

		Emission Reductions in 2020 (annual tons/year)		
Alternative Standard	**Region**	**$PM_{2.5}$**	**SO_2**	**NO_x**
13/35[b]	East	—	—	—
	West	—	—	—
	CA	—	—	—
	Total	**—**	**—**	**—**
12/35[a]	East	670	—	—
	West	60	—	—
	CA		—	—
	Total	**730**	—	**—**
11/35	East	5,000	12,000	850
	West	90	550	620
	CA	800	—	—
	Total	**5,900**	**13,000**	**1,500**
11/30	East	4,700	12,000	850
	West	1,900	3,900	7,400
	CA	3,700	—	—
	Total	**10,000**	**16,000**	**8,200**

[a] Estimates are rounded to two significant figures.

[b] All known controls were applied in the baseline analysis. Thus, no additional known controls were available.

4.2.3 Emission Reductions Needed Beyond Identified Controls

For each alternative standard and geographic area that cannot reach attainment with known controls, we estimated the additional emission reductions needed beyond identified known controls for $PM_{2.5}$ and for the two $PM_{2.5}$ precursors (SO_2 and NO_x) to attain the standard. In Appendix 4.A, we provide estimates of the relationship between additional emission reductions for each pollutant and air quality improvement.

Because three different pollutants affect ambient levels of $PM_{2.5}$ in this analysis there are many different combinations of pollutant reductions that would result in the required air quality improvements. To determine which pollutant reductions to include in our analysis, we employ a least cost approach using what we call the hybrid cost methodology. A detailed discussion of this methodology appears in Chapter 7. In Appendix 7.A, we show cost estimates for each additional emission reduction for each pollutant and geographic area. As expected, the unit costs increase under this methodology as more of a particular pollutant is controlled. The mix of pollutants controlled, however, varies by area because each area has a different combination of known controls applied, varying amounts of additional air quality improvement required, and different amounts of uncontrolled emissions remaining.

The process used to determine the emission reductions needed for each pollutant is described below. First, the U.S. EPA examined the emissions remaining for each geographic area and pollutant (NO_x, $PM_{2.5}$, SO_2). Each pollutant has a marginal cost curve that increases (e.g., the third ton removed costs more than the second ton removed, and so on). Each pollutant has an estimated effectiveness at reducing the ambient concentration of $PM_{2.5}$ per ton of emissions controlled (see Chapter 3 for more details) that varies by geographic area. The U.S. EPA used a least cost methodology to determine the optimum way to reach attainment. The optimization methodology used to estimate the quantity of $PM_{2.5}$ and $PM_{2.5}$ precursors needed for each geographic area is described in more detail in Chapter 7 and Appendix 7.A.

Because the marginal cost equation for each pollutant is expected to be less accurate for the very last portion of a pollutant in an area, and it is unlikely an area would reduce all anthropogenic emissions to zero on one pollutant prior to controlling others, we added the constraint that no more than 90% of the remaining emissions in an area for a given pollutant can be reduced from emission reductions beyond known control measures. This decision was based upon the rationale that no geographic area would be able to eliminate 100% of the emissions of a pollutant given current control measures. This methodology used the marginal cost curves for each of the three pollutants along with the air quality to emissions response ratios to determine the most cost-effective way to achieve the necessary levels of air quality improvement. A detailed discussion of the methodology, including formulas and description of how parameters were estimated, can be found in Chapter 7.

The emission reductions needed beyond known controls are shown in Table 4-4. For a listing of emission reductions needed by county for the unknown controls, see Appendix 4.A. For the alternative standard 13/35 there are only two counties projected to exceed—Riverside County, CA and San Bernardino County, CA. For Riverside County, all known controls were applied in the analysis to illustrate attainment of the baseline (15/35). Thus, no known controls remained for demonstrating attainment of more stringent standards. For the other alternative standards (12/35, 11/35, and 11/30), known controls accounted for over 70% of the needed emission reductions.

The emissions reductions estimated using the hybrid methodology together with reductions associated with known controls form the basis of the cost and benefit estimates. However, a different mix of reductions in SO_2 emissions and $PM_{2.5}$ emissions may have been identified as least cost using a fixed cost per ton approach rather than the hybrid approach.[8]

Using the hybrid methodology, the less expensive pollutant to reduce will be selected until the marginal cost to reduce the next ton exceeds the marginal cost to reduce the next ton of an alternate pollutant. At that point, the methodology chooses a mix of pollutants to achieve the least-cost solution. Since the cost per ton is held constant in the fixed-cost methodology, the least-cost solution would select all available direct $PM_{2.5}$ emissions reductions before selecting SO_2 emissions reductions.[9] Therefore, the hybrid methodology estimates $PM_{2.5}$ emissions reductions lower than or equal to the fixed-cost methodology and SO_2 emission reductions higher than or equal to the fixed-cost methodology.

Even so, for the proposed 12/35 and 13/35 standards, direct $PM_{2.5}$ reductions account for approximately 75%–100% of the reductions, and thus using the fixed cost per ton approach to select the combination of emissions reductions would not have substantially changed the mix of emissions reductions or the outcome of the cost and benefit analyses.

[8] NOx reduction were not selected under either approach as the least cost alternative to achieve the necessary $PM_{2.5}$ reductions.

[9] Because the marginal cost equation for each pollutant is expected to be less accurate for the very last portion of a pollutant in an area, and it is unlikely an area would reduce all anthropogenic emissions to zero on one pollutant prior to controlling others, we included the constraint that no more than 90% of the remaining emissions in an area for a given pollutant can be reduced from emission reductions beyond known control measures.

That said, the hybrid approach still has a number of important uncertainties, and the reliability of the method for extrapolating costs in cases where emissions reductions required go well beyond known controls has not been evaluated. The degree of extrapolation for emissions reductions in California in particular has caused us to rethink the use of the hybrid method in providing a range of cost estimates for the proposed standards, therefore we provide the hybrid approach as a sensitivity analysis in Appendix 7.A. We would like to take comment on analyzing an alternate compliance pathway for California.

Table 4-4. Emission Reductions Needed Beyond Known Control to Reach Alternative Standards in 2020 (annual tons/year)[a]

Alternative Standard	Region	$PM_{2.5}$	SO_2	NO_x
13/35	East	—	—	—
	West	—	—	—
	CA	190	—	—
	Total	**190**	**—**	**—**
12/35[a]	East	—	—	—
	West	210	10	—
	CA	3,400	960	—
	Total	**3,600**	**970**	—
11/35	East	89	—	—
	West	1,100	1,400	—
	CA	6,500	5,500	—
	Total	**7,700**	**6,900**	**—**
11/30	East	1,400	—	—
	West	3,500	1,700	—
	CA	7,200	5,500	—
	Total	**12,000**	**7,200**	**—**

[a] Estimates are rounded to two significant figures.

4.3 Limitations and Uncertainties

The U.S. 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, emission changes, and other impacts of regulatory controls. However, the estimates of emission 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. EPA requests comment on the likelihood that new technologies that control direct $PM_{2.5}$ and its precursors will become available between now and 2020.

A number of limitations and uncertainties are associated with the analysis of emission control measures are listed in Table 4-5. For a complete discussion of the terminology used below please see Chapter 5.5.7.

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
Post-processing of air quality modeled concentrations to estimate future-year visibility design value	Both	High	Medium	Tier 1
"Rollback" methodology for simulating full-attainment	Both	Medium	Medium	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 baseline controls from current SIPs	Both	Medium-high	Medium-high	Tier 0

▪ 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.				
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

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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 area source categories to attain alternative annual and 24-hr air quality 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 area 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 NonEGU 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. Once particles are on the collector plates, they must be removed without re-entraining them

[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.

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.[1]

Table 4.A-1. Example PM Control Measures for NonEGU 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	NonEGUs 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 Area Sources

Specific controls exist for a number of stationary area sources. Area source PM controls at stationary sources include:

- catalytic oxidizers on conveyorized charbroilers at restaurants (up to 80% reduction of PM),
- replacement of older woodstoves with ones compliant with the New Source Performance Standard (NSPS) for residential wood combustion (up to 98% reduction of PM[2]), and
- education and advisory programs to help users to operate woodstoves more efficiently and with fewer emissions (up to 50% reduction of PM)

Another PM area source control measure, diesel particulate filters, can be applied to existing diesel-fueled compression-ignition (C-I) engines to achieve up to a 90% reduction in fine PM. This measure is being applied to new C-I engines as part of a NSPS that was implemented beginning in 2006.

[2] This control measure is largely meant to simulate the effects of a woodstove changeout program as applied to Libby, MT per the efforts of the U.S. EPA and several co-sponsors. For more information, refer to http://www.epa.gov/woodstoves/how-to-guide.html.

Table 4.A-2. Example PM Control Measures for Area 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
Changeout of older woodstoves for new ones by a woodstove changeout campaign or on sale of property, or an education and advisory program for woodstove users	Residential wood combustion sources	46 to near 100	\$1,900
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 area 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 NonEGU 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 nonEGU 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 NonEGU 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 Area Sources

Fuel switching from high to low-sulfur fuels is the predominant control measure available for SO_2 area 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.[3]

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 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.[3]

Table 4.A-4. Example NO_x Control Measures for NonEGU 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.

4.A.2 Projected Monitor Design Values

Table 4.A-5. Area County Definitions for SO_2 and NO_x Emissions Reductions for Control Strategy Analysis

FIPS Code	State Name	County Name	Area Label
1007	Alabama	Bibb	AL
1009	Alabama	Blount	AL
1073	Alabama	Jefferson	AL
1115	Alabama	St. Clair	AL
1117	Alabama	Shelby	AL
1125	Alabama	Tuscaloosa	AL
1127	Alabama	Walker	AL

(continued)

Table 4.A-5. Area County Definitions for SO_2 and NO_x Emissions Reductions for Control Strategy Analysis (continued)

FIPS Code	State Name	County Name	Area Label
42003	Pennsylvania	Allegheny	all
42005	Pennsylvania	Armstrong	all
42007	Pennsylvania	Beaver	all
42019	Pennsylvania	Butler	all
42125	Pennsylvania	Washington	all
42129	Pennsylvania	Westmoreland	all
4003	Arizona	Cochise	AZ
4019	Arizona	Pima	AZ
4023	Arizona	Santa Cruz	AZ
13021	Georgia	Bibb	bib
13079	Georgia	Crawford	bib
13153	Georgia	Houston	bib
13169	Georgia	Jones	bib
13207	Georgia	Monroe	bib
13225	Georgia	Peach	bib
13289	Georgia	Twiggs	bib
6007	California	Butte	but
6011	California	Colusa	but
6021	California	Glenn	but
6063	California	Plumas	but
6103	California	Tehama	but
6115	California	Yuba	but
17031	Illinois	Cook	coo
17043	Illinois	Du Page	coo
17097	Illinois	Lake	coo
17197	Illinois	Will	coo
16005	Idaho	Bannock	fra
16007	Idaho	Bear Lake	fra

(continued)

Table 4.A-5. Area County Definitions for SO_2 and NO_x Emissions Reductions for Control Strategy Analysis (continued)

FIPS Code	State Name	County Name	Area Label
16029	Idaho	Caribou	fra
16041	Idaho	Franklin	fra
16071	Idaho	Oneida	fra
13045	Georgia	Carroll	ful
13057	Georgia	Cherokee	ful
13063	Georgia	Clayton	ful
13067	Georgia	Cobb	ful
13077	Georgia	Coweta	ful
13089	Georgia	De Kalb	ful
13097	Georgia	Douglas	ful
13113	Georgia	Fayette	ful
13121	Georgia	Fulton	ful
13135	Georgia	Gwinnett	ful
13117	Georgia	Forsyth	ful
6025	California	Imperial	imp
6073	California	San Diego	imp
42011	Pennsylvania	Berks	lan
42029	Pennsylvania	Chester	lan
42043	Pennsylvania	Dauphin	lan
42071	Pennsylvania	Lancaster	lan
42075	Pennsylvania	Lebanon	lan
42133	Pennsylvania	York	lan
16023	Idaho	Butte	lem
16033	Idaho	Clark	lem
16037	Idaho	Custer	lem
16049	Idaho	Idaho	lem
16059	Idaho	Lemhi	lem
16085	Idaho	Valley	lem

(continued)

Table 4.A-5. Area County Definitions for SO_2 and NO_x Emissions Reductions for Control Strategy Analysis (continued)

FIPS Code	State Name	County Name	Area Label
17005	Illinois	Bond	mad
17027	Illinois	Clinton	mad
17083	Illinois	Jersey	mad
17117	Illinois	Macoupin	mad
17119	Illinois	Madison	mad
17135	Illinois	Montgomery	mad
17163	Illinois	St Clair	mad
26099	Michigan	Macomb	MI
26115	Michigan	Monroe	MI
26125	Michigan	Oakland	MI
26161	Michigan	Washtenaw	MI
26163	Michigan	Wayne	MI
30001	Montana	Beaverhead	MT
30023	Montana	Deer Lodge	MT
30029	Montana	Flathead	MT
30039	Montana	Granite	MT
30047	Montana	Lake	MT
30053	Montana	Lincoln	MT
30061	Montana	Mineral	MT
30063	Montana	Missoula	MT
30077	Montana	Powell	MT
30081	Montana	Ravalli	MT
30089	Montana	Sanders	MT
36005	New York	Bronx	NY
36047	New York	Kings	NY
36061	New York	New York	NY
36081	New York	Queens	NY
36085	New York	Richmond	NY

(continued)

Table 4.A-5. Area County Definitions for SO_2 and NO_x Emissions Reductions for Control Strategy Analysis (continued)

FIPS Code	State Name	County Name	Area Label
39035	Ohio	Cuyahoga	OH
39055	Ohio	Geauga	OH
39085	Ohio	Lake	OH
39093	Ohio	Lorain	OH
39103	Ohio	Medina	OH
39133	Ohio	Portage	OH
39153	Ohio	Summit	OH
41003	Oregon	Benton	OR
41019	Oregon	Douglas	OR
41029	Oregon	Jackson	OR
41035	Oregon	Klamath	OR
41037	Oregon	Lake	OR
41039	Oregon	Lane	OR
41041	Oregon	Lincoln	OR
41043	Oregon	Linn	OR
41017	Oregon	Deschutes	OR
6003	California	Alpine	sac
6005	California	Amador	sac
6013	California	Contrasta	sac
6017	California	El Dorado	sac
6061	California	Placer	sac
6067	California	Sacramento	sac
6095	California	Solano	sac
6101	California	Sutter	sac
6113	California	Yolo	sac
6019	California	Fresno	san
6027	California	Inyo	san
6029	California	Kern	san

(continued)

Table 4.A-5. Area County Definitions for SO_2 and NO_x Emissions Reductions for Control Strategy Analysis (continued)

FIPS Code	State Name	County Name	Area Label
6031	California	Kings	san
6039	California	Madera	san
6043	California	Mariposa	san
6047	California	Merced	san
6051	California	Mono	san
6053	California	Monterey	san
6069	California	San Benito	san
6077	California	San Joaquin	san
6079	California	San Luis Obispo	san
6085	California	Santa Clara	san
6099	California	Stanislaus	san
6107	California	Tulare	san
6109	California	Tuolumne	san
16009	Idaho	Benewah	sho
16017	Idaho	Bonner	sho
16035	Idaho	Clearwater	sho
16055	Idaho	Kootenai	sho
16057	Idaho	Latah	sho
16079	Idaho	Shoshone	sho
6037	California	Los Angeles	sou
6059	California	Orange	sou
6065	California	Riverside	sou
6071	California	San Bernardino	sou
6111	California	Ventura	sou
48039	Texas	Brazoria	TX
48071	Texas	Chambers	TX
48157	Texas	Fort Bend	TX
48167	Texas	Galveston	TX

(continued)

Table 4.A-5. Area County Definitions for SO_2 and NO_x Emissions Reductions for Control Strategy Analysis (continued)

FIPS Code	State Name	County Name	Area Label
48201	Texas	Harris	TX
48291	Texas	Liberty	TX
48339	Texas	Montgomery	TX
48473	Texas	Waller	TX
49003	Utah	Box Elder	Utah 1
49005	Utah	Cache	Utah 1
49033	Utah	Rich	Utah 1
49057	Utah	Weber	Utah 1
49007	Utah	Carbon	Utah 2
49011	Utah	Davis	Utah 2
49013	Utah	Duchesne	Utah 2
49023	Utah	Juab	Utah 2
49029	Utah	Morgan	Utah 2
49035	Utah	Salt Lake	Utah 2
49039	Utah	Sanpete	Utah 2
49043	Utah	Summit	Utah 2
49045	Utah	Tooele	Utah 2
49049	Utah	Utah	Utah 2
49051	Utah	Wasatch	Utah 2
53033	Washington	King	WA
53035	Washington	Kitsap	WA
53037	Washington	Kittitas	WA
53041	Washington	Lewis	WA
53045	Washington	Mason	WA
53053	Washington	Pierce	WA
53067	Washington	Thurston	WA
53077	Washington	Yakima	WA

Table 4.A-6. Air Quality Ratios for Monitors in Counties with at Least One Monitor with an Annual Design Value (DV) Above 15 or 24-hr Design Value (DV) Above 35 in 2020 Base Case

FIPS Code	High Monitor ID	State Name	County Name	2005 Annual DV	2020 Annual DV	2005 24-hr DV	2020 24-hr DV	24-hr to Annual DV Responsiveness
42003	420030064	Pennsylvania	Allegheny	20.31	12.91	64.2	41.0	3.03
30047	300470028	Montana	Lake	9.00	7.71	43.6	38.3	3.78
30063	300630031	Montana	Missoula	10.52	9.13	44.6	37.4	3.78
30081	300810007	Montana	Ravalli	9.01	7.89	45.1	37.3	3.78
6019	60190008	California	Fresno	16.99	12.71	60.2	41.0	3.45
6029	60290010	California	Kern	18.94	14.34	64.5	45.9	3.45
6031	60310004	California	Kings	17.28	13.24	58.0	42.4	3.45
6099	60990005	California	Stanislaus	14.21	10.85	51.4	37.0	3.45
6107	61072002	California	Tulare	18.51	14.10	56.6	40.1	3.45
6037	60371301	California	Los Angeles	17.66	13.14	48.7	39.7	3.45
6065	60658001	California	Riverside	20.95	16.30	59.1	46.5	3.45
6071	60719004	California	San Bernardino	19.01	14.96	55.5	41.5	3.45
49005	490050004	Utah	Cache	11.56	9.78	56.9	42.6	4.91
49035	490353007	Utah	Salt Lake	12.02	9.72	45.3	36.1	4.91

(continued)

Table 4.A-6. Air Quality Ratios for Monitors in Counties with at Least One Monitor with an Annual Design Value (DV) Above 15 or 24-hr Design Value (DV) Above 35 in 2020 Base Case (continued)

FIPS Code	High Monitor ID	State Name	County Name	NH4SO4 Component of 2005 Annual DV	NH4NO3 Component of 2005 Annual DV	Direct PM2.5 Component of 2005 Annual DV	NH4SO4 Component of 2020 Annual DV	NH4NO3 Component of 2020 Annual DV
42003	420030064	Pennsylvania	Allegheny	9.1830	0.7303	10.4046	4.1977	0.7669
30047	300470028	Montana	Lake	0.9752	0.0936	7.9383	0.9007	0.0730
30063	300630031	Montana	Missoula	1.2692	1.0152	8.2419	1.1767	0.7734
30081	300810007	Montana	Ravalli	0.7614	0.2929	7.9577	0.7034	0.2380
6019	60190008	California	Fresno	2.3402	4.1985	10.4583	1.9907	2.7196
6029	60290010	California	Kern	3.1194	5.8850	9.9388	2.5838	3.7690
6031	60310004	California	Kings	2.7393	4.7392	9.8017	2.2500	2.8483
6099	60990005	California	Stanislaus	2.2879	3.8313	8.0978	1.8519	2.6903
6107	61072002	California	Tulare	2.9375	5.3329	10.2433	2.4607	3.2405
6037	60371301	California	Los Angeles	5.5145	3.5811	8.5679	3.7217	2.9601
6065	60658001	California	Riverside	4.5070	5.2955	11.1521	3.3590	3.9822
6071	60719004	California	San Bernardino	3.7956	3.8907	11.3297	2.8912	2.7994
49005	490050004	Utah	Cache	1.4901	2.1042	7.9703	1.2447	1.5831
49035	490353007	Utah	Salt Lake	1.7974	2.8766	7.3465	1.5642	2.2080

(continued)

Table 4.A-6. Air Quality Ratios for Monitors in Counties with at Least One Monitor with an Annual Design Value (DV) above 15 or 24-hr Design Value (DV) Above 35 in 2020 Base Case (continued)

FIPS Code	High Monitor ID	State Name	County Name	Direct $PM_{2.5}$ Component of 2020 Annual DV	Area Label	Change in Area SO2 Emissions Between 2005 and 2020	Change in Area NO_x Emissions Between 2005 and 2020	Change in County $PM_{2.5}$ Emissions Between 2005 and 2020
42003	420030064	Pennsylvania	Allegheny	7.9528	all	259568	69304	804
30047	300470028	Montana	Lake	6.7370	MT	853	9949	128
30063	300630031	Montana	Missoula	7.1876	MT	853	9949	206
30081	300810007	Montana	Ravalli	6.9574	MT	853	9949	144
6019	60190008	California	Fresno	8.0008	san	5855	106130	1151
6029	60290010	California	Kern	7.9904	san	5855	106130	1241
6031	60310004	California	Kings	8.1443	san	5855	106130	132
6099	60990005	California	Stanislaus	6.3118	san	5855	106130	571
6107	61072002	California	Tulare	8.4003	san	5855	106130	592
6037	60371301	California	Los Angeles	6.4628	sou	20834	230102	5223
6065	60658001	California	Riverside	8.9602	sou	20834	230102	1421
6071	60719004	California	San Bernardino	9.2726	sou	20834	230102	2026
49005	490050004	Utah	Cache	6.9584	Utah 1	639	8192	124
49035	490353007	Utah	Salt Lake	5.9562	Utah 2	8442	33871	799

(continued)

Table 4.A-6. Air Quality Ratios for Monitors in Counties with at Least One Monitor with an Annual Design Value (DV) above 15 or 24-hr Design Value (DV) Above 35 in 2020 Base Case (continued)

FIPS Code	High Monitor ID	State Name	County Name	SO_2 Air Quality Ratio (µg/m^3 change in SO_4 per 1,000 Tons SO_2)	NOx Air Quality Ratio (µg/m^3 Change in NO_3 per 1,000 Tons NO_x)	$PM_{2.5}$ Air Quality Ratio (µg/m^3 Change in Direct $PM_{2.5}$ per 1,000 Tons PM)	24-hr DV Reduction Needed for 35	Annual DV Reduction Associated With the 24-hr DV Meeting 35
42003	420030064	Pennsylvania	Allegheny	0.008	0.000	1.238	5.543	1.829
30047	300470028	Montana	Lake	0.081	0.001	1.929	2.878	0.761
30063	300630031	Montana	Missoula	0.106	0.015	1.929	1.990	0.526
30081	300810007	Montana	Ravalli	0.063	0.005	1.929	1.815	0.480
6019	60190008	California	Fresno	0.062	0.008	1.879	5.548	1.608
6029	60290010	California	Kern	0.080	0.011	1.879	10.442	3.027
6031	60310004	California	Kings	0.070	0.009	1.879	6.939	2.011
6099	60990005	California	Stanislaus	0.057	0.008	1.136	1.578	0.457
6107	61072002	California	Tulare	0.076	0.010	1.879	4.634	1.343
6037	60371301	California	Los Angeles	0.052	0.003	0.597	4.236	1.228
6065	60658001	California	Riverside	0.047	0.004	0.597	11.081	3.212
6071	60719004	California	San Bernardino	0.040	0.003	0.597	6.041	1.751
49005	490050004	Utah	Cache	0.321	0.040	1.929	7.168	1.460
49035	490353007	Utah	Salt Lake	0.028	0.013	1.929	1.240	0.253

Table 4.A-7. Air Quality Ratios for Monitors in Counties With at Least One Monitor With an Annual Design Value (DV) Above 13 in 2020 Baseline (15/35)

FIPS Code	High Monitor ID	State Name	County Name	2005 Annual DV	2020 Annual DV	2005 24-hr DV	2020 24-hr DV	2020 15/35 Annual DV	NH_4SO_4 Component of 2005 Annual DV
6065	60658001	California	Riverside	20.95	16.30	59.1	46.5	13.08	4.5070
6071	60710025	California	San Bernardino	19.67	15.23	51.9	41.2	13.12	4.2309

(continued)

Table 4.A-7. Air Quality Ratios for Monitors in Counties With at Least One Monitor With an Annual Design Value (DV) Above 13 in 2020 Baseline (15/35) (continued)

FIPS Code	High Monitor ID	State Name	County Name	NH_4NO_3 Component of 2005 Annual DV	Direct $PM_{2.5}$ Component of 2005 Annual DV	NH_4SO_4 Component of 2020 Annual DV	NH_4NO_3 Component of 2020 Annual DV	Direct $PM_{2.5}$ Component of 2020 Annual DV	Area Label	Change in Area SO_2 Emissions Between 2005 and 2020
6065	60658001	California	Riverside	5.2955	11.1521	3.3590	3.9822	8.9602	sou	20834
6071	60710025	California	San Bernardino	4.0013	11.4396	3.0564	3.0890	9.0859	sou	20834

(continued)

Table 4.A-7. Air Quality Ratios for Monitors in Counties With at Least One Monitor With an Annual Design Value (DV) Above 13 in 2020 Baseline (15/35) (continued)

FIPS Code	High Monitor ID	State Name	County Name	Change in Area NO_x Emissions Between 2005 and 2020	Change in County $PM_{2.5}$ Emissions Between 2005 and 2020	SO_2 Air Quality Ratio ($\mu g/m^3$ Change in SO_4 per 1,000 Tons SO_2)	NOx Air Quality Ratio ($\mu g/m^3$ Change in NO_3 per 1,000 Tons NO_x)	$PM_{2.5}$ Air Quality Ratio ($\mu g/m^3$ Change in Direct $PM_{2.5}$ per 1,000 Tons PM)	Annual DV Reduction Needed for 12 ($\mu g/m^3$)
6065	60658001	California	Riverside	230102	1421	0.047	0.004	0.597	0.039
6071	60710025	California	San Bernardino	230102	2026	0.043	0.003	0.597	0.077

Table 4.A-8. Air Quality Ratios for Monitors in Counties with at Least One Monitor With an Annual Design Value (DV) Above 12 in 2020 Baseline (15/35)

FIPS Code	High Monitor ID	State Name	County Name	2005 Annual DV	2020 Annual DV	2005 24-hr DV	2020 24-hr DV	2020 15/35 Annual DV	NH_4SO_4 Component of 2005 Annual DV
30053	300530018	Montana	Lincoln	14.93	12.60	42.7	35.3	12.53	1.1342
6065	60658001	California	Riverside	20.95	16.30	59.1	46.5	13.08	4.5070
6071	60710025	California	San Bernardino	19.67	15.23	51.9	41.2	13.12	4.2309
26163	261630033	Michigan	Wayne	17.50	12.35	43.9	31.3	12.35	7.2067
1073	10730023	Alabama	Jefferson	18.57	12.34	44.1	27.9	12.34	7.5801
4023	40230004	Arizona	Santa Cruz	12.94	12.06	36.1	33.8	12.06	1.6891

(continued)

Table 4.A-8. Air Quality Ratios for Monitors in Counties with at Least One Monitor With an Annual Design Value (DV) Above 12 in 2020 Baseline (15/35) (continued)

FIPS Code	High Monitor ID	State Name	County Name	NH_4NO_3 Component of 2005 Annual DV	Direct $PM_{2.5}$ Component of 2005 Annual DV	NH_4SO_4 Component of 2020 Annual DV	NH_4NO_3 Component of 2020 Annual DV	Direct $PM_{2.5}$ Component of 2020 Annual DV	Area Label	Change in Area SO_2 Emissions Between 2005 and 2020
30053	300530018	Montana	Lincoln	0.4047	13.3936	1.0433	0.3556	11.2031	MT	853
6065	60658001	California	Riverside	5.2955	11.1521	3.3590	3.9822	8.9602	sou	20,834
6071	60710025	California	San Bernardino	4.0013	11.4396	3.0564	3.0890	9.0859	sou	20,834
26163	261630033	Michigan	Wayne	2.2041	8.0960	4.2754	1.8639	6.2139	MI	142,340
1073	10730023	Alabama	Jefferson	0.2657	10.7334	3.5837	0.2508	8.5105	AL	243,497
4023	40230004	Arizona	Santa Cruz	0.0149	11.2381	1.5034	0.0121	10.5517	AZ	5,178

(continued)

Table 4.A-8. Air Quality Ratios for Monitors in Counties with at Least One Monitor With an Annual Design Value (DV) Above 12 in 2020 Baseline (15/35) (continued)

FIPS Code	High Monitor ID	State Name	County Name	Change in Area NO_x Emissions Between 2005 and 2020	Change in County $PM_{2.5}$ Emissions Between 2005 and 2020	SO_2 Air Quality Ratio ($\mu g/m^3$ Change in SO_4 per 1,000 Tons SO_2)	NO_x Air Quality Ratio ($\mu g/m^3$ Change in NO_3 per 1,000 Tons NOx)	PM2.5 Air Quality Ratio ($\mu g/m^3$ Change in Direct $PM_{2.5}$ per 1,000 Tons PM)	Annual DV Reduction Needed for 12 ($\mu g/m^3$)
30053	300530018	Montana	Lincoln	9,949	107	0.094	0.007	1.929	0.494
6065	60658001	California	Riverside	230,102	1,421	0.047	0.004	0.597	1.039
6071	60710025	California	San Bernardino	230,102	2,026	0.043	0.003	0.597	0.857
26163	261630033	Michigan	Wayne	142,522	2,846	0.013	0.002	1.238	0.301
1073	10730023	Alabama	Jefferson	61,210	2,902	0.007	0.000	1.238	0.291
4023	40230004	Arizona	Santa Cruz	25,105	48	0.038	0.000	1.929	0.011

Table 4.A-9. Air Quality Ratios for Monitors in Counties With at Least One Monitor With an Annual Design Value (DV) Above 11 in 2020 Baseline (15/35)

FIPS Code	High Monitor ID	State Name	County Name	Area Label	Change in Area SO_2 Emissions Between 2005 and 2020	Change in Area NO_x Emissions Between 2005 and 2020	Change in County $PM_{2.5}$ Emissions Between 2005 and 2020
1073	10730023	Alabama	Jefferson	AL	243,497	61,210	2,902
4023	40230004	Arizona	Santa Cruz	AZ	5,178	25,105	48
13021	130210007	Georgia	Bibb	bib	57,968	23,656	276
17031	170310052	Illinois	Cook	coo	100,781	153,366	5,267
13121	131210039	Georgia	Fulton	ful	91,890	96,664	1,360
6025	60250005	California	Imperial	imp	5,629	51,787	295
17119	171191007	Illinois	Madison	mad	40,890	26,245	700
26163	261630033	Michigan	Wayne	MI	142,340	142,522	2,846
30053	300530018	Montana	Lincoln	MT	853	9,949	107
36061	360610056	New York	New York	NY	32,091	47,190	537
39035	390350038	Ohio	Cuyahoga	OH	115,275	61,836	984
6031	60310004	California	Kings	san	5,855	106,130	132
6107	61072002	California	Tulare	san	5,855	106,130	592
6071	60710025	California	San Bernardino	sou	20,834	230,102	2,026
6059	60590007	California	Orange	sou	20,834	230,102	940
6065	60658001	California	Riverside	sou	20,834	230,102	1,421
48201	482011035	Texas	Harris	TX	81,104	121,308	4,910

(continued)

Table 4.A-9. Air Quality Ratios for Monitors in Counties With at Least One Monitor With an Annual Design Value (DV) Above 11 in 2020 Baseline (15/35) (continued)

FIPS Code	High Monitor ID	State Name	County Name	SO_2 Air Quality Ratio (μg/m^3 Change in SO_4 per 1,000 Tons SO_2)	NO_x Air Quality Ratio (μg/m^3 Change in NO_3 per 1,000 Tons NOx)	$PM_{2.5}$ Air Quality Ratio (μg/m^3 Change in Direct $PM_{2.5}$ per 1,000 Tons PM)	Annual DV Reduction Needed for 11 (μg/m^3)
1073	10730023	Alabama	Jefferson	0.0072	0.0003	1.238	1.291
4023	40230004	Arizona	Santa Cruz	0.0376	0.0001	1.929	1.011
13021	130210007	Georgia	Bibb	0.0306	0.0000	1.238	0.171
17031	170310052	Illinois	Cook	0.0119	0.0000	1.238	0.091
13121	131210039	Georgia	Fulton	0.0180	0.0003	1.238	0.061
6025	60250005	California	Imperial	0.1164	0.0023	0.597	0.171
17119	171191007	Illinois	Madison	0.0404	0.0090	1.238	0.491
26163	261630033	Michigan	Wayne	0.0128	0.0019	1.238	1.301
30053	300530018	Montana	Lincoln	0.0939	0.0071	1.929	1.487
36061	360610056	New York	New York	0.0529	0.0022	1.238	0.121
39035	390350038	Ohio	Cuyahoga	0.0164	0.0048	1.238	0.741
6031	60310004	California	Kings	0.0698	0.0085	1.879	0.180
6107	61072002	California	Tulare	0.0763	0.0097	1.879	0.645
6071	60710025	California	San Bernardino	0.0428	0.0031	0.597	2.077
6059	60590007	California	Orange	0.0435	0.0027	0.597	0.008
6065	60658001	California	Riverside	0.0470	0.0040	0.597	2.039
48201	482011035	Texas	Harris	0.0164	0.0001	1.238	0.561

Table 4.A-10. Air Quality Ratios for Monitors in Counties with at Least One Monitor with an Annual Design Value (DV) Above 11 or 24-hr Design Value (DV) Above 30 in 2020 Baseline (15/35)

FIPS Code	High Monitor ID	State Name	County Name	2005 Annual DV	2020 Annual DV	Area Label	Change in Area SO_2 Emissions Between 2005 and 2020	Change in Area NO_x Emissions Between 2005 and 2020	Change in County $PM_{2.5}$ Emissions Between 2005 and 2020
1073	10730023	Alabama	Jefferson	18.57	12.34	AL	243,497	61,210	2,902
42003	420030064	Pennsylvania	Allegheny	20.31	12.91	all	259,568	69,304	804
4023	40230004	Arizona	Santa Cruz	12.94	12.06	AZ	5,178	25,105	48
13021	130210007	Georgia	Bibb	16.54	11.22	bib	57,968	23,656	276
6007	60070002	California	Butte	12.73	9.56	but	2,320	8,494	693
17031	170310052	Illinois	Cook	15.75	11.14	coo	100,781	153,366	5,267
16041	160410001	Idaho	Franklin	7.7	6.68	fra	228	3,350	40
13121	131210039	Georgia	Fulton	17.43	11.11	ful	91,890	96,664	1,360
6025	60250005	California	Imperial	12.71	11.22	imp	5,629	51,787	295
42071	420710007	Pennsylvania	Lancaster	16.55	10.73	lan	119,209	42,136	8,866
16059	160590004	Idaho	Lemhi	N/A	N/A	lem	169	577	44
17119	171191007	Illinois	Madison	16.72	11.54	mad	40,890	26,245	700
26163	261630033	Michigan	Wayne	17.5	12.35	MI	142,340	142,522	2,846
30047	300470028	Montana	Lake	9	7.71	MT	853	9,949	128
30081	300810007	Montana	Ravalli	9.01	7.89	MT	853	9,949	144
30063	300630031	Montana	Missoula	10.52	9.13	MT	853	9,949	206
30053	300530018	Montana	Lincoln	14.93	12.6	MT	853	9,949	107
36061	360610056	New York	New York	16.18	11.17	NY	32,091	47,190	537

(continued)

Table 4.A-10. Air Quality Ratios for Monitors in Counties with at Least One Monitor with an Annual Design Value (DV) Above 11 or 24-hr Design Value (DV) Above 30 in 2020 Baseline (15/35) (continued)

FIPS Code	High Monitor ID	State Name	County Name	2005 Annual DV	2020 Annual DV	Area Label	Change in Area SO_2 Emissions Between 2005 and 2020	Change in Area NO_x Emissions Between 2005 and 2020	Change in County $PM_{2.5}$ Emissions Between 2005 and 2020
39035	390350038	Ohio	Cuyahoga	17.37	11.79	OH	115,275	61,836	984
41035	410350004	Oregon	Klamath	11.2	8.54	OR	1,489	22,686	769
41039	410392013	Oregon	Lane	11.93	9.43	OR	1,489	22,686	1,666
6067	60670006	California	Sacramento	11.88	8.76	sac	9,448	42,974	1,311
6099	60990005	California	Stanislaus	14.21	10.85	san	5,855	106,130	571
6019	60190008	California	Fresno	16.99	12.71	san	5,855	106,130	1,151
6031	60310004	California	Kings	17.28	13.24	san	5,855	106,130	132
6107	61072002	California	Tulare	18.51	14.1	san	5,855	106,130	592
16079	160790017	Idaho	Shoshone	12.08	10.66	sho	555	5,546	43
6059	60590007	California	Orange	15.75	11.93	sou	20,834	230,102	940
6065	60658001	California	Riverside	20.95	16.3	sou	20,834	230,102	1,421
6071	60710025	California	San Bernardino	19.67	15.23	sou	20,834	230,102	2,026
48201	482011035	Texas	Harris	15.42	11.61	TX	81,104	121,308	4,910
49005	490050004	Utah	Cache	11.56	9.78	Utah 1	639	8,192	124
49035	490350012	Utah	Salt Lake	N/A	N/A	Utah 2	8,442	33,,871	799
49011	490110004	Utah	Davis	10.31	8.58	Utah 2	8,442	33,871	243
49049	490494001	Utah	Utah	10.52	8.8	Utah 2	8,442	33,871	383
53053	530530029	Washington	Pierce	10.55	8.11	WA	11,269	91,530	1,058

(continued)

Table 4.A-10. Air Quality Ratios for Monitors in Counties with at Least One Monitor with an Annual Design Value (DV) Above 11 or 24-hr Design Value (DV) Above 30 in 2020 Baseline (15/35) (continued)

FIPS Code	High Monitor ID	State Name	County Name	SO_2 Air Quality Ratio ($\mu g/m^3$ Change in SO_4 per 1,000 Tons SO_2)	NO_x Air Quality Ratio ($\mu g/m^3$ Change in NO_3 per 1,000 Tons NO_x)	$PM_{2.5}$ Air Quality Ratio (ug/m^3 Change in Direct $PM_{2.5}$ per 1,000 Tons of $PM_{2.5}$)	Annual DV Reduction Needed for 11 (ug/m^3)	24-hr DV Reduction Needed for 30 (ug/m^3)	Annual DV Reduction Corresponding to the 24-hr DV Meeting 30 (ug/m^3)
1073	10730023	Alabama	Jefferson	0.007	0.000	1.238	1.291	0.000	0.000
42003	420030064	Pennsylvania	Allegheny	0.008	0.000	1.238	0.032	5.000	1.650
4023	40230004	Arizona	Santa Cruz	0.038	0.000	1.929	1.011	3.332	1.262
13021	130210007	Georgia	Bibb	0.031	0.000	1.238	0.171	0.000	0.000
6007	60070002	California	Butte	0.079	0.018	1.879	0.000	1.674	0.369
17031	170310052	Illinois	Cook	0.012	0.000	1.238	0.091	0.000	0.000
16041	160410001	Idaho	Franklin	0.609	0.087	1.929	0.000	0.010	0.002
13121	131210039	Georgia	Fulton	0.018	0.000	1.238	0.061	0.000	0.000
6025	60250005	California	Imperial	0.116	0.002	0.597	0.171	1.331	0.386
42071	420710007	Pennsylvania	Lancaster	0.015	0.007	1.238	0.000	0.029	0.010
16059	160590004	Idaho	Lemhi	0.371	0.131	1.929	0.000	0.837	0.192
17119	171191007	Illinois	Madison	0.040	0.009	1.238	0.491	0.000	0.000
26163	261630033	Michigan	Wayne	0.013	0.002	1.238	1.301	0.888	0.299
30047	300470028	Montana	Lake	0.081	0.001	1.929	0.000	5.000	1.323
30081	300810007	Montana	Ravalli	0.063	0.005	1.929	0.000	5.000	1.323
30063	300630031	Montana	Missoula	0.106	0.015	1.929	0.000	3.993	1.056
30053	300530018	Montana	Lincoln	0.094	0.007	1.929	1.487	4.637	1.227
36061	360610056	New York	New York	0.053	0.002	1.238	0.121	0.000	0.000

(continued)

Table 4.A-10. Air Quality Ratios for Monitors in Counties with at Least One Monitor with an Annual Design Value (DV) Above 11 or 24-hr Design Value (DV) Above 30 in 2020 Baseline (15/35) (continued)

FIPS Code	High Monitor ID	State Name	County Name	SO_2 Air Quality Ratio ($\mu g/m^3$ Change in SO_4 per 1,000 Tons SO_2)	NO_x Air Quality Ratio ($\mu g/m^3$ Change in NO_3 per 1,000 Tons NO_x)	$PM_{2.5}$ Air Quality Ratio (ug/m^3 Change in Direct $PM_{2.5}$ per 1,000 Tons of $PM_{2.5}$)	Annual DV Reduction Needed for 11 (ug/m^3)	24-hr DV Reduction Needed for 30 (ug/m^3)	Annual DV Reduction Corresponding to the 24-hr DV Meeting 30 (ug/m^3)
39035	390350038	Ohio	Cuyahoga	0.016	0.005	1.238	0.741	0.000	0.000
41035	410350004	Oregon	Klamath	0.059	0.002	1.929	0.000	0.363	0.082
41039	410392013	Oregon	Lane	0.068	0.003	1.929	0.000	3.520	0.793
6067	60670006	California	Sacramento	0.033	0.007	1.879	0.000	2.438	0.707
6099	60990005	California	Stanislaus	0.057	0.008	1.879	0.000	1.300	0.377
6019	60190008	California	Fresno	0.062	0.008	1.879	0.000	0.402	0.117
6031	60310004	California	Kings	0.070	0.009	1.879	0.180	5.000	1.449
6107	61072002	California	Tulare	0.076	0.010	1.879	0.645	1.333	0.386
16079	160790017	Idaho	Shoshone	0.129	0.014	1.929	0.000	1.475	0.337
6059	60590007	California	Orange	0.043	0.003	0.597	0.008	0.525	0.152
6065	60658001	California	Riverside	0.047	0.004	0.597	2.039	5.000	1.449
6071	60710025	California	San Bernardino	0.043	0.003	0.597	2.077	3.537	1.025
48201	482011035	Texas	Harris	0.016	0.000	1.238	0.561	0.000	0.000
49005	490050004	Utah	Cache	0.321	0.040	1.929	0.000	5.000	1.018
49035	490350012	Utah	Salt Lake	0.026	0.012	1.929	0.000	4.821	0.982
49011	490110004	Utah	Davis	0.028	0.014	1.929	0.000	0.864	0.176
49049	490494001	Utah	Utah	0.024	0.015	1.929	0.000	3.461	0.705
53053	530530029	Washington	Pierce	0.029	0.000	1.929	0.000	0.515	0.146

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Alabama	Baldwin	11.44	7.45	7.45	7.45	7.45	7.45	7.45
Alabama	Clay	13.27	8.42	8.42	8.42	8.42	8.42	8.42
Alabama	Colbert	12.75	8.21	8.21	8.21	8.21	8.21	8.21
Alabama	DeKalb	14.13	8.74	8.74	8.74	8.74	8.74	8.74
Alabama	Escambia	13.19	9.29	9.29	9.29	9.29	9.29	9.29
Alabama	Etowah	14.87	9.38	9.38	9.38	9.38	9.38	9.38
Alabama	Houston	13.22	9.32	9.32	9.32	9.32	9.32	9.32
Alabama	Jefferson	18.57	12.34	12.34	12.34	12.03	11.00	11.00
Alabama	Jefferson	15.46	10.33	10.33	10.33	10.02	8.99	8.99
Alabama	Jefferson	13.52	8.77	8.77	8.77	8.46	7.43	7.43
Alabama	Jefferson	15.89	10.03	10.03	10.03	9.729	8.69	8.69
Alabama	Jefferson	17.15	11.72	11.72	11.72	11.41	10.38	10.38
Alabama	Jefferson	15.1	9.98	9.98	9.98	9.67	8.64	8.64
Alabama	Jefferson	14.42	9.02	9.02	9.02	8.71	7.68	7.68
Alabama	Jefferson	14.53	9.3	9.3	9.3	8.99	7.96	7.96
Alabama	Madison	13.83	8.54	8.54	8.54	8.54	8.54	8.54
Alabama	Mobile	12.9	8.8	8.8	8.8	8.8	8.8	8.8
Alabama	Mobile	12.36	8.33	8.33	8.33	8.33	8.33	8.33
Alabama	Mobile	11.51	7.51	7.51	7.51	7.51	7.51	7.51
Alabama	Montgomery	14.24	9.77	9.77	9.77	9.77	9.77	9.77

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Alabama	Morgan	13.32	8.42	8.42	8.42	8.42	8.42	8.42
Alabama	Russell	15.73	10.63	10.63	10.63	10.63	10.63	10.63
Alabama	Shelby	14.43	9.3	9.3	9.3	9.3	9.3	9.3
Alabama	Sumter	11.92	7.75	7.75	7.75	7.75	7.75	7.75
Alabama	Talladega	14.51	9.08	9.08	9.08	9.08	9.08	9.08
Alabama	Tuscaloosa	13.56	8.78	8.78	8.78	8.78	8.78	8.78
Alabama	Walker	13.86	8.89	8.89	8.89	8.89	8.89	8.89
Arizona	Cochise	7	6.56	6.56	6.56	6.56	6.54	6.54
Arizona	Coconino	6.49	6.02	6.02	6.02	6.02	6.02	6.02
Arizona	Gila	8.94	8.11	8.11	8.11	8.11	8.11	8.11
Arizona	Maricopa	12.17	9.64	9.64	9.64	9.64	9.64	9.64
Arizona	Maricopa	12.59	10.24	10.24	10.24	10.24	10.24	10.24
Arizona	Maricopa	9.97	8.02	8.02	8.02	8.02	8.02	8.02
Arizona	Pima	6.04	5.12	5.12	5.12	5.12	5.09	5.10
Arizona	Pima	5.85	4.95	4.95	4.95	4.95	4.92	4.93
Arizona	Pinal	7.77	6.92	6.92	6.92	6.92	6.92	6.92
Arizona	Pinal	5.71	5.04	5.04	5.04	5.04	5.04	5.04
Arizona	Santa Cruz	12.94	12.06	12.06	12.06	12.02	11.04	10.79
Arkansas	Arkansas	12.45	8.76	8.76	8.76	8.76	8.76	8.76
Arkansas	Ashley	12.83	9.44	9.44	9.44	9.44	9.44	9.44

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Arkansas	Crittenden	13.36	8.63	8.63	8.63	8.63	8.63	8.63
Arkansas	Faulkner	12.79	9.16	9.16	9.16	9.16	9.16	9.16
Arkansas	Garland	12.4	8.79	8.79	8.79	8.79	8.79	8.79
Arkansas	Mississippi	12.61	8.32	8.32	8.32	8.32	8.32	8.32
Arkansas	Phillips	12.1	8.15	8.15	8.15	8.15	8.15	8.15
Arkansas	Polk	11.65	8.35	8.35	8.35	8.35	8.35	8.35
Arkansas	Pope	12.79	9.4	9.4	9.4	9.4	9.4	9.4
Arkansas	Pulaski	13.17	9.11	9.11	9.11	9.11	9.11	9.11
Arkansas	Pulaski	14.05	9.91	9.91	9.91	9.91	9.91	9.91
Arkansas	Pulaski	13.59	9.52	9.52	9.52	9.52	9.52	9.52
Arkansas	Union	12.86	9.3	9.3	9.3	9.3	9.3	9.3
Arkansas	White	12.57	9.13	9.13	9.13	9.13	9.13	9.13
California	Alameda	9.44	7.42	7.42	7.42	7.42	7.42	7.42
California	Alameda	9.34	7.18	7.18	7.18	7.18	7.18	7.18
California	Butte	12.73	9.56	9.56	9.56	9.56	9.56	9.18
California	Calaveras	7.77	6.05	6.05	6.05	6.05	6.05	6.05
California	Colusa	7.39	6.25	6.25	6.25	6.25	6.25	6.25
California	Contra Costa	9.47	7.3	7.3	7.3	7.3	7.3	7.3
California	Fresno	16.99	12.71	9.76	9.76	9.76	9.76	8.00
California	Fresno	16.38	12.33	9.42	9.42	9.42	9.42	7.67

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
California	Fresno	17.17	12.87	9.96	9.96	9.96	9.96	8.21
California	Imperial	12.71	11.22	11.22	11.22	11.22	10.80	10.80
California	Imperial	8.39	7.35	7.35	7.35	7.35	6.93	6.93
California	Imperial	9.2	8.12	8.12	8.12	8.12	7.70	7.70
California	Inyo	5.25	4.8	4.12	4.12	4.12	4.12	3.87
California	Kern	18.94	14.34	7.98	7.98	7.98	7.98	7.29
California	Kern	18.68	14.18	7.84	7.84	7.84	7.84	7.16
California	Kern	19.17	14.72	8.36	8.36	8.36	8.36	7.68
California	Kings	17.28	13.24	11.22	11.22	11.22	11.04	9.33
California	Lake	4.62	3.79	3.79	3.79	3.79	3.79	3.79
California	Los Angeles	17.03	12.88	9.87	9.87	9.82	9.60	9.60
California	Los Angeles	18.19	13.44	10.34	10.34	10.29	10.04	10.04
California	Los Angeles	18	13.19	10.03	10.03	9.984	9.72	9.72
California	Los Angeles	15.35	11.41	8.57	8.57	8.53	8.34	8.34
California	Los Angeles	17.66	13.14	10.07	10.07	10.02	9.79	9.79
California	Los Angeles	17.92	13.16	10.07	10.07	10.02	9.78	9.78
California	Los Angeles	15.36	11.31	8.28	8.28	8.23	8.00	8.00
California	Los Angeles	16.62	12.3	9.35	9.35	9.30	9.09	9.09
California	Los Angeles	15.21	11.24	8.34	8.34	8.29	8.09	8.09
California	Los Angeles	8.42	7.06	4.58	4.58	4.55	4.45	4.45

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
California	Mendocino	6.46	5.14	5.14	5.14	5.14	5.14	5.14
California	Merced	14.78	11.24	10.39	10.39	10.39	10.39	10.10
California	Monterey	6.96	5.39	4.75	4.75	4.75	4.75	4.52
California	Nevada	5.16	3.83	3.83	3.83	3.83	3.83	3.83
California	Nevada	6.71	5.31	5.31	5.31	5.31	5.31	5.31
California	Orange	15.75	11.93	11.05	11.05	11.01	10.75	10.27
California	Orange	11.33	9.43	8.77	8.77	8.74	8.53	8.05
California	Placer	9.8	7.24	7.24	7.24	7.24	7.24	7.24
California	Plumas	9.75	7.8	7.8	7.8	7.8	7.8	7.8
California	Plumas	11.46	8.91	8.91	8.91	8.91	8.91	8.91
California	Riverside	18.91	14.84	11.65	11.61	10.61	9.62	9.62
California	Riverside	10.31	8.66	5.90	5.86	4.88	3.97	3.97
California	Riverside	20.95	16.3	13.08	13.04	12.04	11.04	11.04
California	Sacramento	11.88	8.76	8.76	8.76	8.76	8.76	7.07
California	Sacramento	11.44	8.75	8.75	8.75	8.75	8.75	7.06
California	Sacramento	10.53	8.01	8.01	8.01	8.01	8.01	6.32
California	San Bernardino	19.67	15.23	13.12	13.04	12.04	11.04	11.04
California	San Bernardino	10.29	8.2	6.39	6.32	5.33	4.39	4.39
California	San Bernardino	19.14	14.86	12.67	12.60	11.59	10.58	10.58
California	San Bernardino	10.77	9.16	7.50	7.42	6.44	5.54	5.54

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
California	San Bernardino	19.01	14.96	12.90	12.82	11.83	10.84	10.84
California	San Diego	11.92	8.68	8.68	8.68	8.68	8.68	8.68
California	San Diego	12.27	9.09	9.09	9.09	9.09	9.09	9.09
California	San Diego	10.59	7.64	7.64	7.64	7.64	7.64	7.64
California	San Diego	12.79	9.56	9.56	9.56	9.56	9.56	9.56
California	San Diego	13.46	9.83	9.83	9.83	9.83	9.83	9.83
California	San Francisco	9.62	7.05	7.05	7.05	7.05	7.05	7.05
California	San Joaquin	12.94	9.96	9.07	9.07	9.07	9.07	8.78
California	San Luis Obispo	6.92	5.43	4.76	4.76	4.76	4.76	4.52
California	San Luis Obispo	7.94	6.24	5.47	5.47	5.47	5.47	5.21
California	San Mateo	9.03	6.76	6.76	6.76	6.76	6.76	6.76
California	Santa Barbara	10.37	8.04	8.04	8.04	8.04	8.04	8.04
California	Santa Clara	11.38	8.66	7.75	7.75	7.75	7.75	7.42
California	Santa Clara	10.32	8.03	7.13	7.13	7.13	7.13	6.81
California	Shasta	7.41	5.48	5.48	5.48	5.48	5.48	5.48
California	Solano	9.99	7.7	7.7	7.7	7.7	7.7	7.7
California	Sonoma	8.21	6.15	6.15	6.15	6.15	6.15	6.15
California	Stanislaus	14.21	10.85	9.32	9.32	9.32	9.32	8.43
California	Sutter	9.85	7.53	7.53	7.53	7.53	7.53	7.53
California	Tulare	18.51	14.1	11.69	11.69	11.69	11.04	10.54

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
California	Ventura	10.68	7.84	7.19	7.19	7.16	7.01	7.01
California	Ventura	9.74	7.69	7.04	7.04	7.01	6.86	6.86
California	Ventura	11.68	8.79	8.09	8.09	8.06	7.89	7.89
California	Ventura	10.69	7.64	7.14	7.14	7.12	7.00	7.00
California	Yolo	9.03	7.15	7.15	7.15	7.15	7.15	7.15
Colorado	Adams	10.06	7.58	7.58	7.58	7.58	7.58	7.58
Colorado	Arapahoe	7.96	6.02	6.02	6.02	6.02	6.02	6.02
Colorado	Boulder	8.32	6.68	6.68	6.68	6.68	6.68	6.68
Colorado	Boulder	6.96	5.6	5.6	5.6	5.6	5.6	5.6
Colorado	Delta	7.44	5.74	5.74	5.74	5.74	5.74	5.74
Colorado	Denver	9.37	7.07	7.07	7.07	7.07	7.07	7.07
Colorado	Denver	9.76	7.37	7.37	7.37	7.37	7.37	7.37
Colorado	Elbert	4.4	3.48	3.48	3.48	3.48	3.48	3.48
Colorado	El Paso	6.73	4.85	4.85	4.85	4.85	4.85	4.85
Colorado	El Paso	7.94	5.67	5.67	5.67	5.67	5.67	5.67
Colorado	Larimer	7.33	5.95	5.95	5.95	5.95	5.95	5.95
Colorado	Mesa	9.28	7.34	7.34	7.34	7.34	7.34	7.34
Colorado	Pueblo	7.45	5.73	5.73	5.73	5.73	5.73	5.73
Colorado	San Miguel	4.65	4.09	4.09	4.09	4.09	4.09	4.09
Colorado	Weld	8.19	6.61	6.61	6.61	6.61	6.61	6.61

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Colorado	Weld	8.78	7.06	7.06	7.06	7.06	7.06	7.06
Connecticut	Fairfield	13.21	8.79	8.79	8.79	8.79	8.79	8.79
Connecticut	Fairfield	12.49	8.27	8.27	8.27	8.27	8.27	8.27
Connecticut	Fairfield	12.43	8.15	8.15	8.15	8.15	8.15	8.15
Connecticut	Fairfield	11.48	7.51	7.51	7.51	7.51	7.51	7.51
Connecticut	Hartford	11.03	7.29	7.29	7.29	7.29	7.29	7.29
Connecticut	Litchfield	8.01	5.03	5.03	5.03	5.03	5.03	5.03
Connecticut	New Haven	12.12	7.93	7.93	7.93	7.93	7.93	7.93
Connecticut	New Haven	12.45	8.1	8.1	8.1	8.1	8.1	8.1
Connecticut	New Haven	13.12	8.62	8.62	8.62	8.62	8.62	8.62
Connecticut	New Haven	11.17	7.24	7.24	7.24	7.24	7.24	7.24
Connecticut	New Haven	12.74	8.33	8.33	8.33	8.33	8.33	8.33
Connecticut	New London	10.96	7.25	7.25	7.25	7.25	7.25	7.25
Delaware	Kent	12.61	7.66	7.66	7.66	7.66	7.66	7.66
Delaware	Kent	12.52	7.71	7.71	7.71	7.71	7.71	7.71
Delaware	New Castle	13.73	8.57	8.57	8.57	8.57	8.57	8.57
Delaware	New Castle	12.92	7.93	7.93	7.93	7.93	7.93	7.93
Delaware	New Castle	13.69	8.55	8.55	8.55	8.55	8.55	8.55
Delaware	New Castle	14.87	9.42	9.42	9.42	9.42	9.42	9.42
Delaware	Sussex	13.39	8.13	8.13	8.13	8.13	8.13	8.13

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
District of Columbia	District of Columbia	14.16	8.86	8.86	8.86	8.86	8.86	8.86
District of Columbia	District of Columbia	14.41	8.71	8.71	8.71	8.71	8.71	8.71
District of Columbia	District of Columbia	13.99	8.48	8.48	8.48	8.48	8.48	8.48
Florida	Alachua	9.32	6.19	6.19	6.19	6.19	6.19	6.19
Florida	Alachua	9.59	6.44	6.44	6.44	6.44	6.44	6.44
Florida	Bay	11.46	7.96	7.96	7.96	7.96	7.96	7.96
Florida	Brevard	8.32	5.56	5.56	5.56	5.56	5.56	5.56
Florida	Broward	8.22	5.93	5.93	5.93	5.93	5.93	5.93
Florida	Broward	8.18	5.78	5.78	5.78	5.78	5.78	5.78
Florida	Broward	8.21	5.78	5.78	5.78	5.78	5.78	5.78
Florida	Citrus	9	5.69	5.69	5.69	5.69	5.69	5.69
Florida	Duval	9.9	6.88	6.88	6.88	6.88	6.88	6.88
Florida	Duval	10.44	7.49	7.49	7.49	7.49	7.49	7.49
Florida	Escambia	11.72	8.38	8.38	8.38	8.38	8.38	8.38
Florida	Hillsborough	10.74	7.37	7.37	7.37	7.37	7.37	7.37
Florida	Hillsborough	10.52	7.23	7.23	7.23	7.23	7.23	7.23
Florida	Lee	8.36	5.88	5.88	5.88	5.88	5.88	5.88
Florida	Leon	12.56	9	9	9	9	9	9
Florida	Manatee	8.81	5.64	5.64	5.64	5.64	5.64	5.64
Florida	Marion	10.11	6.93	6.93	6.93	6.93	6.93	6.93

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Florida	Miami-Dade	9.45	6.72	6.72	6.72	6.72	6.72	6.72
Florida	Miami-Dade	8.14	6.42	6.42	6.42	6.42	6.42	6.42
Florida	Orange	9.61	6.43	6.43	6.43	6.43	6.43	6.43
Florida	Orange	9.5	6.28	6.28	6.28	6.28	6.28	6.28
Florida	Palm Beach	7.84	5.83	5.83	5.83	5.83	5.83	5.83
Florida	Palm Beach	7.7	5.69	5.69	5.69	5.69	5.69	5.69
Florida	Pinellas	9.82	6.64	6.64	6.64	6.64	6.64	6.64
Florida	Pinellas	9.52	6.41	6.41	6.41	6.41	6.41	6.41
Florida	Polk	9.53	6.55	6.55	6.55	6.55	6.55	6.55
Florida	St. Lucie	8.34	5.77	5.77	5.77	5.77	5.77	5.77
Florida	Sarasota	8.77	5.77	5.77	5.77	5.77	5.77	5.77
Florida	Seminole	9.51	6.33	6.33	6.33	6.33	6.33	6.33
Florida	Volusia	9.27	6.08	6.08	6.08	6.08	6.08	6.08
Georgia	Bibb	16.54	11.22	11.22	11.22	11.22	11.04	11.04
Georgia	Bibb	13.94	9.1	9.1	9.1	9.1	8.92	8.92
Georgia	Chatham	13.74	9.33	9.33	9.33	9.33	9.33	9.33
Georgia	Chatham	13.93	9.56	9.56	9.56	9.56	9.56	9.56
Georgia	Clarke	14.9	9.69	9.69	9.69	9.69	9.69	9.69
Georgia	Clayton	16.5	10.65	10.65	10.65	10.65	10.65	10.65
Georgia	Cobb	16.15	10.6	10.6	10.6	10.6	10.6	10.6

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Georgia	Cobb	15.42	9.88	9.88	9.88	9.88	9.88	9.88
Georgia	DeKalb	15.48	9.54	9.54	9.54	9.54	9.54	9.54
Georgia	DeKalb	15.37	9.55	9.55	9.55	9.55	9.55	9.55
Georgia	Dougherty	14.46	10.24	10.24	10.24	10.24	10.24	10.24
Georgia	Floyd	16.13	10.63	10.63	10.63	10.63	10.63	10.63
Georgia	Fulton	15.84	9.82	9.82	9.82	9.82	9.82	9.82
Georgia	Fulton	17.43	11.11	11.11	11.11	11.11	10.98	10.98
Georgia	Glynn	12.25	8.74	8.74	8.74	8.74	8.74	8.74
Georgia	Gwinnett	16.07	10.45	10.45	10.45	10.45	10.45	10.45
Georgia	Hall	14.16	9.16	9.16	9.16	9.16	9.16	9.16
Georgia	Houston	14.19	9.34	9.34	9.34	9.34	9.32	9.32
Georgia	Lowndes	12.58	9.26	9.26	9.26	9.26	9.26	9.26
Georgia	Muscogee	14.94	9.94	9.94	9.94	9.94	9.94	9.94
Georgia	Muscogee	15.39	10.4	10.4	10.4	10.4	10.4	10.4
Georgia	Muscogee	14.16	9.58	9.58	9.58	9.58	9.58	9.58
Georgia	Paulding	14.12	8.76	8.76	8.76	8.76	8.76	8.76
Georgia	Richmond	15.61	10.75	10.75	10.75	10.75	10.75	10.75
Georgia	Richmond	15.68	10.84	10.84	10.84	10.84	10.84	10.84
Georgia	Walker	15.49	9.86	9.86	9.86	9.86	9.86	9.86
Georgia	Washington	15.14	10.42	10.42	10.42	10.42	10.42	10.42

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Georgia	Wilkinson	15.27	10.3	10.3	10.3	10.3	10.3	10.3
Idaho	Ada	8.41	7.6	7.6	7.6	7.6	7.6	7.6
Idaho	Bannock	7.66	6.97	6.97	6.97	6.97	6.97	6.97
Idaho	Benewah	9.59	8.59	8.59	8.59	8.59	8.59	8.57
Idaho	Canyon	8.46	7.46	7.46	7.46	7.46	7.46	7.46
Idaho	Franklin	7.7	6.68	6.68	6.68	6.68	6.68	6.68
Idaho	Idaho	9.58	8.8	8.8	8.8	8.8	8.8	8.75
Idaho	Shoshone	12.08	10.66	10.66	10.66	10.66	10.66	10.32
Illinois	Adams	12.5	8.91	8.91	8.91	8.91	8.91	8.91
Illinois	Champaign	12.5	8.39	8.39	8.39	8.39	8.39	8.39
Illinois	Champaign	12.53	8.4	8.4	8.4	8.4	8.4	8.4
Illinois	Cook	15.21	11.11	11.11	11.11	11.11	10.97	10.97
Illinois	Cook	14.81	10.69	10.69	10.69	10.69	10.55	10.55
Illinois	Cook	15.75	11.14	11.14	11.14	11.14	11.00	11.00
Illinois	Cook	15.03	10.54	10.54	10.54	10.54	10.40	10.40
Illinois	Cook	14.89	10.5	10.5	10.5	10.5	10.36	10.36
Illinois	Cook	14.77	10.77	10.77	10.77	10.77	10.63	10.63
Illinois	Cook	15.24	10.78	10.78	10.78	10.78	10.64	10.64
Illinois	Cook	12.78	8.9	8.9	8.9	8.9	8.76	8.76
Illinois	Cook	12.76	8.87	8.87	8.87	8.87	8.73	8.73

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Illinois	Cook	15.48	10.86	10.86	10.86	10.86	10.72	10.72
Illinois	DuPage	13.82	9.84	9.84	9.84	9.84	9.84	9.84
Illinois	Jersey	12.89	8.78	8.78	8.78	8.78	8.51	8.51
Illinois	Kane	13.32	9.51	9.51	9.51	9.51	9.51	9.51
Illinois	Kane	14.34	10.25	10.25	10.25	10.25	10.25	10.25
Illinois	Lake	11.81	8.34	8.34	8.34	8.34	8.34	8.34
Illinois	McHenry	12.4	8.86	8.86	8.86	8.86	8.86	8.86
Illinois	McLean	12.39	8.58	8.58	8.58	8.58	8.58	8.58
Illinois	Macon	13.24	9.21	9.21	9.21	9.21	9.21	9.21
Illinois	Madison	16.72	11.54	11.54	11.54	11.54	10.82	10.82
Illinois	Madison	14.01	9.7	9.7	9.7	9.7	8.98	8.98
Illinois	Madison	14.32	9.97	9.97	9.97	9.97	9.25	9.25
Illinois	Peoria	13.34	9.41	9.41	9.41	9.41	9.41	9.41
Illinois	Randolph	13.11	8.7	8.7	8.7	8.7	8.7	8.7
Illinois	Rock Island	12.01	8.61	8.61	8.61	8.61	8.61	8.61
Illinois	Saint Clair	15.58	10.63	10.63	10.63	10.63	10.34	10.34
Illinois	Saint Clair	14.29	9.66	9.66	9.66	9.66	9.40	9.40
Illinois	Sangamon	13.13	9.39	9.39	9.39	9.39	9.39	9.39
Illinois	Will	13.63	9.69	9.69	9.69	9.69	9.69	9.69
Illinois	Will	11.52	7.96	7.96	7.96	7.96	7.96	7.96

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Illinois	Winnebago	13.57	9.78	9.78	9.78	9.78	9.78	9.78
Indiana	Allen	13.67	9.92	9.92	9.92	9.92	9.92	9.92
Indiana	Allen	13.55	9.84	9.84	9.84	9.84	9.84	9.84
Indiana	Clark	16.44	10.19	10.19	10.19	10.19	10.19	10.19
Indiana	Delaware	13.69	9.11	9.11	9.11	9.11	9.11	9.11
Indiana	Dubois	15.19	9.34	9.34	9.34	9.34	9.34	9.34
Indiana	Floyd	14.85	9	9	9	9	9	9
Indiana	Henry	13.64	9.05	9.05	9.05	9.05	9.05	9.05
Indiana	Howard	13.93	9.61	9.61	9.61	9.61	9.61	9.61
Indiana	Knox	14.03	8.78	8.78	8.78	8.78	8.78	8.78
Indiana	Lake	14.33	10.45	10.45	10.45	10.45	10.45	10.45
Indiana	Lake	13.83	10.06	10.06	10.06	10.06	10.06	10.06
Indiana	Lake	14.02	10.37	10.37	10.37	10.37	10.37	10.37
Indiana	Lake	14.05	10.27	10.27	10.27	10.27	10.27	10.27
Indiana	Lake	13.89	10.11	10.11	10.11	10.11	10.11	10.11
Indiana	LaPorte	12.49	8.9	8.9	8.9	8.9	8.9	8.9
Indiana	LaPorte	12.69	9.03	9.03	9.03	9.03	9.03	9.03
Indiana	Madison	13.97	9.34	9.34	9.34	9.34	9.34	9.34
Indiana	Marion	14.24	9.28	9.28	9.28	9.28	9.28	9.28
Indiana	Marion	15.26	10.16	10.16	10.16	10.16	10.16	10.16

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Indiana	Marion	14.71	9.7	9.7	9.7	9.7	9.7	9.7
Indiana	Marion	16.05	10.81	10.81	10.81	10.81	10.81	10.81
Indiana	Marion	15.9	10.66	10.66	10.66	10.66	10.66	10.66
Indiana	Porter	12.66	8.99	8.99	8.99	8.99	8.99	8.99
Indiana	Porter	13.21	9.42	9.42	9.42	9.42	9.42	9.42
Indiana	St. Joseph	13.29	10.04	10.04	10.04	10.04	10.04	10.04
Indiana	St. Joseph	13.69	10.36	10.36	10.36	10.36	10.36	10.36
Indiana	St. Joseph	12.82	9.65	9.65	9.65	9.65	9.65	9.65
Indiana	Spencer	14.32	8.55	8.55	8.55	8.55	8.55	8.55
Indiana	Tippecanoe	13.7	9.4	9.4	9.4	9.4	9.4	9.4
Indiana	Vanderburgh	14.69	9.82	9.82	9.82	9.82	9.82	9.82
Indiana	Vanderburgh	14.82	9.9	9.9	9.9	9.9	9.9	9.9
Indiana	Vanderburgh	14.99	10.06	10.06	10.06	10.06	10.06	10.06
Indiana	Vigo	13.99	8.95	8.95	8.95	8.95	8.95	8.95
Indiana	Vigo	13.46	8.51	8.51	8.51	8.51	8.51	8.51
Iowa	Black Hawk	11.16	8.15	8.15	8.15	8.15	8.15	8.15
Iowa	Clinton	12.52	9.01	9.01	9.01	9.01	9.01	9.01
Iowa	Johnson	12.08	8.96	8.96	8.96	8.96	8.96	8.96
Iowa	Linn	10.79	7.88	7.88	7.88	7.88	7.88	7.88
Iowa	Montgomery	10.02	7.35	7.35	7.35	7.35	7.35	7.35

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Iowa	Muscatine	12.92	9.47	9.47	9.47	9.47	9.47	9.47
Iowa	Palo Alto	9.53	7.21	7.21	7.21	7.21	7.21	7.21
Iowa	Polk	10.41	7.68	7.68	7.68	7.68	7.68	7.68
Iowa	Polk	9.95	7.34	7.34	7.34	7.34	7.34	7.34
Iowa	Polk	10.64	7.85	7.85	7.85	7.85	7.85	7.85
Iowa	Pottawattamie	11.13	8.42	8.42	8.42	8.42	8.42	8.42
Iowa	Scott	11.86	8.52	8.52	8.52	8.52	8.52	8.52
Iowa	Scott	11.64	8.35	8.35	8.35	8.35	8.35	8.35
Iowa	Scott	14.42	10.62	10.62	10.62	10.62	10.62	10.62
Iowa	Van Buren	10.84	7.98	7.98	7.98	7.98	7.98	7.98
Iowa	Woodbury	10.32	7.93	7.93	7.93	7.93	7.93	7.93
Iowa	Wright	10.37	7.68	7.68	7.68	7.68	7.68	7.68
Kansas	Johnson	10.59	7.68	7.68	7.68	7.68	7.68	7.68
Kansas	Johnson	11.1	8.04	8.04	8.04	8.04	8.04	8.04
Kansas	Johnson	9.68	7	7	7	7	7	7
Kansas	Linn	10.47	7.74	7.74	7.74	7.74	7.74	7.74
Kansas	Sedgwick	10.26	7.55	7.55	7.55	7.55	7.55	7.55
Kansas	Sedgwick	10.29	7.57	7.57	7.57	7.57	7.57	7.57
Kansas	Sedgwick	10.36	7.64	7.64	7.64	7.64	7.64	7.64
Kansas	Shawnee	10.79	8.07	8.07	8.07	8.07	8.07	8.07

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Kansas	Shawnee	10.93	8.26	8.26	8.26	8.26	8.26	8.26
Kansas	Sumner	9.89	7.4	7.4	7.4	7.4	7.4	7.4
Kansas	Wyandotte	12.73	9.34	9.34	9.34	9.34	9.34	9.34
Kansas	Wyandotte	10.93	7.92	7.92	7.92	7.92	7.92	7.92
Kentucky	Bell	14.1	8.59	8.59	8.59	8.59	8.59	8.59
Kentucky	Boyd	14.49	8.77	8.77	8.77	8.77	8.77	8.77
Kentucky	Bullitt	14.92	9.15	9.15	9.15	9.15	9.15	9.15
Kentucky	Campbell	13.67	8.13	8.13	8.13	8.13	8.13	8.13
Kentucky	Carter	12.22	7.08	7.08	7.08	7.08	7.08	7.08
Kentucky	Christian	13.2	8.02	8.02	8.02	8.02	8.02	8.02
Kentucky	Daviess	14.1	8.25	8.25	8.25	8.25	8.25	8.25
Kentucky	Fayette	14.36	8.64	8.64	8.64	8.64	8.64	8.64
Kentucky	Fayette	14.87	9.05	9.05	9.05	9.05	9.05	9.05
Kentucky	Franklin	13.37	7.93	7.93	7.93	7.93	7.93	7.93
Kentucky	Hardin	13.58	8.04	8.04	8.04	8.04	8.04	8.04
Kentucky	Henderson	13.93	8.89	8.89	8.89	8.89	8.89	8.89
Kentucky	Jefferson	15.55	9.43	9.43	9.43	9.43	9.43	9.43
Kentucky	Jefferson	15.35	9.29	9.29	9.29	9.29	9.29	9.29
Kentucky	Jefferson	15.31	9.26	9.26	9.26	9.26	9.26	9.26
Kentucky	Jefferson	14.74	8.84	8.84	8.84	8.84	8.84	8.84

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Kentucky	Kenton	14.39	8.73	8.73	8.73	8.73	8.73	8.73
Kentucky	Laurel	12.55	7.4	7.4	7.4	7.4	7.4	7.4
Kentucky	McCracken	13.41	8.39	8.39	8.39	8.39	8.39	8.39
Kentucky	Madison	13.61	8.01	8.01	8.01	8.01	8.01	8.01
Kentucky	Perry	13.21	7.98	7.98	7.98	7.98	7.98	7.98
Kentucky	Pike	13.49	7.94	7.94	7.94	7.94	7.94	7.94
Kentucky	Warren	13.83	8.29	8.29	8.29	8.29	8.29	8.29
Louisiana	Caddo	12.53	8.76	8.76	8.76	8.76	8.76	8.76
Louisiana	Calcasieu	10.58	7.66	7.66	7.66	7.66	7.66	7.66
Louisiana	Calcasieu	11.07	8.05	8.05	8.05	8.05	8.05	8.05
Louisiana	Concordia	11.42	7.81	7.81	7.81	7.81	7.81	7.81
Louisiana	East Baton Rouge	13.38	9.86	9.86	9.86	9.86	9.86	9.86
Louisiana	East Baton Rouge	12.08	8.74	8.74	8.74	8.74	8.74	8.74
Louisiana	Iberville	12.9	9.44	9.44	9.44	9.44	9.44	9.44
Louisiana	Iberville	11.02	7.68	7.68	7.68	7.68	7.68	7.68
Louisiana	Jefferson	11.52	7.53	7.53	7.53	7.53	7.53	7.53
Louisiana	Lafayette	11.08	7.62	7.62	7.62	7.62	7.62	7.62
Louisiana	Ouachita	11.97	8.63	8.63	8.63	8.63	8.63	8.63
Louisiana	Rapides	11.03	7.64	7.64	7.64	7.64	7.64	7.64
Louisiana	Tangipahoa	12.03	8.14	8.14	8.14	8.14	8.14	8.14

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Louisiana	Terrebonne	10.74	7.32	7.32	7.32	7.32	7.32	7.32
Louisiana	West Baton Rouge	13.51	9.96	9.96	9.96	9.96	9.96	9.96
Maine	Androscoggin	9.9	6.85	6.85	6.85	6.85	6.85	6.85
Maine	Aroostook	9.74	8.73	8.73	8.73	8.73	8.73	8.73
Maine	Aroostook	8.27	6.85	6.85	6.85	6.85	6.85	6.85
Maine	Cumberland	11.06	7.51	7.51	7.51	7.51	7.51	7.51
Maine	Cumberland	11.13	7.61	7.61	7.61	7.61	7.61	7.61
Maine	Hancock	5.76	4.24	4.24	4.24	4.24	4.24	4.24
Maine	Kennebec	9.99	7	7	7	7	7	7
Maine	Oxford	10.13	7.65	7.65	7.65	7.65	7.65	7.65
Maine	Penobscot	9.12	6.66	6.66	6.66	6.66	6.66	6.66
Maryland	Anne Arundel	11.91	7.3	7.3	7.3	7.3	7.3	7.3
Maryland	Anne Arundel	14.82	9.65	9.65	9.65	9.65	9.65	9.65
Maryland	Anne Arundel	14.57	9.45	9.45	9.45	9.45	9.45	9.45
Maryland	Baltimore	13.77	8.58	8.58	8.58	8.58	8.58	8.58
Maryland	Baltimore	14.76	9.46	9.46	9.46	9.46	9.46	9.46
Maryland	Cecil	12.68	7.81	7.81	7.81	7.81	7.81	7.81
Maryland	Harford	12.51	7.62	7.62	7.62	7.62	7.62	7.62
Maryland	Montgomery	12.47	7.77	7.77	7.77	7.77	7.77	7.77
Maryland	Prince George's	12.24	7.61	7.61	7.61	7.61	7.61	7.61

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Maryland	Prince George's	13.03	8.06	8.06	8.06	8.06	8.06	8.06
Maryland	Washington	13.7	8.64	8.64	8.64	8.64	8.64	8.64
Maryland	Baltimore (City)	14.12	8.99	8.99	8.99	8.99	8.99	8.99
Maryland	Baltimore (City)	14.38	9.08	9.08	9.08	9.08	9.08	9.08
Maryland	Baltimore (City)	15.76	10.14	10.14	10.14	10.14	10.14	10.14
Maryland	Baltimore (City)	15.63	10.13	10.13	10.13	10.13	10.13	10.13
Massachusetts	Berkshire	10.65	7.34	7.34	7.34	7.34	7.34	7.34
Massachusetts	Bristol	9.58	6.5	6.5	6.5	6.5	6.5	6.5
Massachusetts	Essex	9.03	6.36	6.36	6.36	6.36	6.36	6.36
Massachusetts	Essex	9.1	6.42	6.42	6.42	6.42	6.42	6.42
Massachusetts	Essex	9.58	6.74	6.74	6.74	6.74	6.74	6.74
Massachusetts	Hampden	9.85	6.73	6.73	6.73	6.73	6.73	6.73
Massachusetts	Hampden	12.17	8.29	8.29	8.29	8.29	8.29	8.29
Massachusetts	Hampden	11.85	8.08	8.08	8.08	8.08	8.08	8.08
Massachusetts	Plymouth	9.87	6.87	6.87	6.87	6.87	6.87	6.87
Massachusetts	Suffolk	12.34	8.79	8.79	8.79	8.79	8.79	8.79
Massachusetts	Suffolk	11.86	8.35	8.35	8.35	8.35	8.35	8.35
Massachusetts	Suffolk	10.88	7.7	7.7	7.7	7.7	7.7	7.7
Massachusetts	Suffolk	13.07	9.3	9.3	9.3	9.3	9.3	9.3
Massachusetts	Worcester	10.55	7.19	7.19	7.19	7.19	7.19	7.19

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Massachusetts	Worcester	11.29	7.69	7.69	7.69	7.69	7.69	7.69
Michigan	Allegan	11.84	8.31	8.31	8.31	8.31	8.31	8.31
Michigan	Bay	10.93	7.94	7.94	7.94	7.94	7.94	7.94
Michigan	Berrien	11.72	8.34	8.34	8.34	8.34	8.34	8.34
Michigan	Genesee	11.61	8.28	8.28	8.28	8.28	8.28	8.28
Michigan	Ingham	12.23	8.64	8.64	8.64	8.64	8.64	8.64
Michigan	Kalamazoo	12.84	9.19	9.19	9.19	9.19	9.19	9.19
Michigan	Kent	12.89	9.17	9.17	9.17	9.17	9.17	9.17
Michigan	Macomb	12.7	9.13	9.13	9.13	9.13	9.13	9.13
Michigan	Missaukee	8.26	6.33	6.33	6.33	6.33	6.33	6.33
Michigan	Monroe	13.92	9.46	9.46	9.46	9.46	9.46	9.46
Michigan	Muskegon	11.61	8.39	8.39	8.39	8.39	8.39	8.39
Michigan	Oakland	13.78	9.62	9.62	9.62	9.62	9.62	9.62
Michigan	Ottawa	12.55	8.87	8.87	8.87	8.87	8.87	8.87
Michigan	Saginaw	10.61	7.74	7.74	7.74	7.74	7.74	7.74
Michigan	St. Clair	13.34	9.87	9.87	9.87	9.87	9.87	9.87
Michigan	Washtenaw	12.3	8.58	8.58	8.58	8.58	8.58	8.58
Michigan	Washtenaw	13.88	9.82	9.82	9.82	9.82	9.82	9.82
Michigan	Wayne	14.52	10.33	10.33	10.33	9.802	8.92	8.92
Michigan	Wayne	15.88	11.17	11.17	11.17	10.64	9.76	9.76

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Michigan	Wayne	14.57	10.34	10.34	10.34	9.812	8.93	8.93
Michigan	Wayne	14.32	10.3	10.3	10.3	9.772	8.89	8.89
Michigan	Wayne	13.39	9.28	9.28	9.28	8.75	7.87	7.87
Michigan	Wayne	17.5	12.35	12.35	12.35	11.82	10.94	10.94
Michigan	Wayne	14.67	10.44	10.44	10.44	9.912	9.03	9.03
Minnesota	Cass	5.7	4.79	4.79	4.79	4.79	4.79	4.79
Minnesota	Dakota	9.3	7.09	7.09	7.09	7.09	7.09	7.09
Minnesota	Hennepin	9.76	7.39	7.39	7.39	7.39	7.39	7.39
Minnesota	Hennepin	9.14	6.93	6.93	6.93	6.93	6.93	6.93
Minnesota	Hennepin	9.59	7.25	7.25	7.25	7.25	7.25	7.25
Minnesota	Hennepin	9.54	7.23	7.23	7.23	7.23	7.23	7.23
Minnesota	Hennepin	9.56	7.26	7.26	7.26	7.26	7.26	7.26
Minnesota	Hennepin	9.33	7.07	7.07	7.07	7.07	7.07	7.07
Minnesota	Mille Lacs	6.54	5.3	5.3	5.3	5.3	5.3	5.3
Minnesota	Olmsted	10.13	7.6	7.6	7.6	7.6	7.6	7.6
Minnesota	Ramsey	11.32	8.68	8.68	8.68	8.68	8.68	8.68
Minnesota	Ramsey	11.02	8.34	8.34	8.34	8.34	8.34	8.34
Minnesota	Ramsey	9.63	7.36	7.36	7.36	7.36	7.36	7.36
Minnesota	Saint Louis	6.1	5.04	5.04	5.04	5.04	5.04	5.04
Minnesota	Saint Louis	6.19	4.97	4.97	4.97	4.97	4.97	4.97

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Minnesota	Saint Louis	7.51	5.99	5.99	5.99	5.99	5.99	5.99
Minnesota	Scott	9	6.9	6.9	6.9	6.9	6.9	6.9
Minnesota	Stearns	8.58	6.8	6.8	6.8	6.8	6.8	6.8
Mississippi	Adams	11.29	7.66	7.66	7.66	7.66	7.66	7.66
Mississippi	Bolivar	12.36	8.56	8.56	8.56	8.56	8.56	8.56
Mississippi	DeSoto	12.43	7.95	7.95	7.95	7.95	7.95	7.95
Mississippi	Forrest	13.62	9.03	9.03	9.03	9.03	9.03	9.03
Mississippi	Harrison	12.2	8.16	8.16	8.16	8.16	8.16	8.16
Mississippi	Hinds	12.56	8.43	8.43	8.43	8.43	8.43	8.43
Mississippi	Jackson	12.04	7.9	7.9	7.9	7.9	7.9	7.9
Mississippi	Jones	14.39	9.56	9.56	9.56	9.56	9.56	9.56
Mississippi	Lauderdale	13.07	8.62	8.62	8.62	8.62	8.62	8.62
Mississippi	Lee	12.57	8.06	8.06	8.06	8.06	8.06	8.06
Mississippi	Lowndes	12.79	8.42	8.42	8.42	8.42	8.42	8.42
Mississippi	Pearl River	12.14	8.25	8.25	8.25	8.25	8.25	8.25
Mississippi	Warren	12.32	8.51	8.51	8.51	8.51	8.51	8.51
Missouri	Boone	11.84	8.45	8.45	8.45	8.45	8.45	8.45
Missouri	Buchanan	12.8	9.75	9.75	9.75	9.75	9.75	9.75
Missouri	Cass	10.67	7.76	7.76	7.76	7.76	7.76	7.76
Missouri	Cedar	11.12	7.89	7.89	7.89	7.89	7.89	7.89

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Missouri	Clay	11.03	8.1	8.1	8.1	8.1	8.1	8.1
Missouri	Greene	11.75	8.32	8.32	8.32	8.32	8.32	8.32
Missouri	Jackson	12.78	9.33	9.33	9.33	9.33	9.33	9.33
Missouri	Jefferson	13.79	9.57	9.57	9.57	9.57	9.57	9.57
Missouri	Monroe	10.87	7.59	7.59	7.59	7.59	7.59	7.59
Missouri	Saint Charles	13.29	9.12	9.12	9.12	9.12	9.12	9.12
Missouri	Sainte Genevieve	13.34	9.06	9.06	9.06	9.06	9.06	9.06
Missouri	Saint Louis	13.04	8.9	8.9	8.9	8.9	8.9	8.9
Missouri	Saint Louis	13.46	9.07	9.07	9.07	9.07	9.07	9.07
Missouri	St. Louis City	14.27	9.74	9.74	9.74	9.74	9.74	9.74
Missouri	St. Louis City	14.36	9.71	9.71	9.71	9.71	9.71	9.71
Missouri	St. Louis City	13.44	9.03	9.03	9.03	9.03	9.03	9.03
Missouri	St. Louis City	14.56	9.84	9.84	9.84	9.84	9.84	9.84
Montana	Cascade	5.72	5.01	5.01	5.01	5.01	5.01	5.01
Montana	Flathead	9.99	8.52	8.46	8.46	8.46	8.36	8.33
Montana	Flathead	8.58	7.28	7.23	7.23	7.23	7.13	7.11
Montana	Gallatin	4.38	4.13	4.13	4.13	4.13	4.13	4.13
Montana	Lake	9.06	7.81	7.05	7.05	7.05	6.95	5.74
Montana	Lake	9	7.71	6.94	6.94	6.94	6.83	5.62
Montana	Lewis and Clark	8.2	7.19	7.19	7.19	7.19	7.19	7.19

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Montana	Lincoln	14.93	12.6	12.53	12.53	12.04	11.01	11.04
Montana	Missoula	10.52	9.13	8.33	8.33	8.33	8.19	7.28
Montana	Ravalli	9.01	7.89	7.40	7.40	7.40	7.32	6.08
Montana	Rosebud	6.58	6.1	6.1	6.1	6.1	6.1	6.1
Montana	Sanders	6.75	6.04	6.00	6.00	6.00	5.91	5.89
Montana	Silver Bow	10.14	8.69	8.69	8.69	8.69	8.69	8.69
Montana	Yellowstone	8.14	6.84	6.84	6.84	6.84	6.84	6.84
Nebraska	Cass	9.99	7.48	7.48	7.48	7.48	7.48	7.48
Nebraska	Douglas	9.88	7.41	7.41	7.41	7.41	7.41	7.41
Nebraska	Douglas	9.85	7.4	7.4	7.4	7.4	7.4	7.4
Nebraska	Hall	7.95	6.13	6.13	6.13	6.13	6.13	6.13
Nebraska	Lancaster	8.9	6.53	6.53	6.53	6.53	6.53	6.53
Nebraska	Lincoln	7.57	6.32	6.32	6.32	6.32	6.32	6.32
Nebraska	Sarpy	9.79	7.33	7.33	7.33	7.33	7.33	7.33
Nebraska	Scotts Bluff	6.04	5.17	5.17	5.17	5.17	5.17	5.17
Nebraska	Washington	9.29	7.1	7.1	7.1	7.1	7.1	7.1
Nevada	Clark	4.02	3.64	3.64	3.64	3.64	3.64	3.64
Nevada	Clark	5.75	4.97	4.97	4.97	4.97	4.97	4.97
Nevada	Clark	9.44	8.08	8.08	8.08	8.08	8.08	8.08
Nevada	Clark	3.67	3.3	3.3	3.3	3.3	3.3	3.3

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Nevada	Clark	8.49	7.28	7.28	7.28	7.28	7.28	7.28
Nevada	Washoe	8.11	6.39	6.39	6.39	6.39	6.39	6.39
New Hampshire	Belknap	7.28	5.12	5.12	5.12	5.12	5.12	5.12
New Hampshire	Cheshire	11.53	7.96	7.96	7.96	7.96	7.96	7.96
New Hampshire	Coos	10.24	8.01	8.01	8.01	8.01	8.01	8.01
New Hampshire	Grafton	8.43	6.08	6.08	6.08	6.08	6.08	6.08
New Hampshire	Hillsborough	10.18	7.08	7.08	7.08	7.08	7.08	7.08
New Hampshire	Hillsborough	10.01	7.01	7.01	7.01	7.01	7.01	7.01
New Hampshire	Hillsborough	6.27	4.3	4.3	4.3	4.3	4.3	4.3
New Hampshire	Merrimack	9.72	6.76	6.76	6.76	6.76	6.76	6.76
New Hampshire	Rockingham	9	6.26	6.26	6.26	6.26	6.26	6.26
New Hampshire	Sullivan	9.86	7.01	7.01	7.01	7.01	7.01	7.01
New Jersey	Atlantic	11.47	6.96	6.96	6.96	6.96	6.96	6.96
New Jersey	Bergen	13.09	8.9	8.9	8.9	8.9	8.9	8.9
New Jersey	Camden	13.31	8.49	8.49	8.49	8.49	8.49	8.49
New Jersey	Camden	13.51	8.53	8.53	8.53	8.53	8.53	8.53
New Jersey	Essex	13.27	8.74	8.74	8.74	8.74	8.74	8.74
New Jersey	Gloucester	13.46	8.46	8.46	8.46	8.46	8.46	8.46
New Jersey	Hudson	14.24	9.65	9.65	9.65	9.65	9.65	9.65
New Jersey	Mercer	12.71	8.14	8.14	8.14	8.14	8.14	8.14

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
New Jersey	Mercer	11.14	7	7	7	7	7	7
New Jersey	Middlesex	12.15	7.91	7.91	7.91	7.91	7.91	7.91
New Jersey	Morris	11.5	7.42	7.42	7.42	7.42	7.42	7.42
New Jersey	Morris	10.21	6.56	6.56	6.56	6.56	6.56	6.56
New Jersey	Ocean	10.92	6.85	6.85	6.85	6.85	6.85	6.85
New Jersey	Passaic	12.88	8.59	8.59	8.59	8.59	8.59	8.59
New Jersey	Union	14.94	9.88	9.88	9.88	9.88	9.88	9.88
New Jersey	Union	13.32	8.73	8.73	8.73	8.73	8.73	8.73
New Jersey	Union	13.06	8.45	8.45	8.45	8.45	8.45	8.45
New Jersey	Warren	12.72	8.2	8.2	8.2	8.2	8.2	8.2
New Mexico	Bernalillo	7.03	5.74	5.74	5.74	5.74	5.74	5.74
New Mexico	Bernalillo	6.64	5.41	5.41	5.41	5.41	5.41	5.41
New Mexico	Chaves	6.54	5.68	5.68	5.68	5.68	5.68	5.68
New Mexico	Dona Ana	9.95	8.7	8.7	8.7	8.7	8.7	8.7
New Mexico	Dona Ana	6.31	5.55	5.55	5.55	5.55	5.55	5.55
New Mexico	Grant	5.93	5.5	5.5	5.5	5.5	5.5	5.5
New Mexico	Sandoval	5	4.12	4.12	4.12	4.12	4.12	4.12
New Mexico	Sandoval	7.99	7.15	7.15	7.15	7.15	7.15	7.15
New Mexico	San Juan	5.92	5.23	5.23	5.23	5.23	5.23	5.23
New Mexico	Santa Fe	4.76	4.22	4.22	4.22	4.22	4.22	4.22

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
New York	Albany	11.83	8.43	8.43	8.43	8.43	8.43	8.43
New York	Bronx	15.43	10.87	10.87	10.87	10.87	10.87	10.87
New York	Bronx	13.09	8.68	8.68	8.68	8.68	8.68	8.68
New York	Bronx	13.45	9.41	9.41	9.41	9.41	9.41	9.41
New York	Chautauqua	9.8	6.32	6.32	6.32	6.32	6.32	6.32
New York	Erie	12.62	8.69	8.69	8.69	8.69	8.69	8.69
New York	Erie	12.64	8.65	8.65	8.65	8.65	8.65	8.65
New York	Essex	5.94	4.43	4.43	4.43	4.43	4.43	4.43
New York	Kings	14.2	9.67	9.67	9.67	9.67	9.67	9.67
New York	Monroe	10.64	7.63	7.63	7.63	7.63	7.63	7.63
New York	Nassau	11.66	7.69	7.69	7.69	7.69	7.69	7.69
New York	New York	16.18	11.17	11.17	11.17	11.17	9.88	10.20
New York	New York	14.8	10.06	10.06	10.06	10.06	8.77	9.09
New York	New York	13.61	9.48	9.48	9.48	9.48	8.19	8.51
New York	New York	15.41	10.6	10.6	10.6	10.6	9.31	9.63
New York	Niagara	11.96	8.55	8.55	8.55	8.55	8.55	8.55
New York	Onondaga	10.08	6.89	6.89	6.89	6.89	6.89	6.89
New York	Orange	10.99	7.28	7.28	7.28	7.28	7.28	7.28
New York	Queens	12.18	8.15	8.15	8.15	8.15	8.15	8.15
New York	Richmond	13.31	8.72	8.72	8.72	8.72	8.72	8.72

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
New York	Richmond	11.59	7.58	7.58	7.58	7.58	7.58	7.58
New York	St. Lawrence	7.29	5.6	5.6	5.6	5.6	5.6	5.6
New York	Steuben	9	5.83	5.83	5.83	5.83	5.83	5.83
New York	Suffolk	11.52	7.51	7.51	7.51	7.51	7.51	7.51
New York	Westchester	11.73	7.7	7.7	7.7	7.7	7.7	7.7
North Carolina	Alamance	13.94	8.38	8.38	8.38	8.38	8.38	8.38
North Carolina	Buncombe	12.6	7.71	7.71	7.71	7.71	7.71	7.71
North Carolina	Caswell	13.19	7.74	7.74	7.74	7.74	7.74	7.74
North Carolina	Catawba	15.31	9.17	9.17	9.17	9.17	9.17	9.17
North Carolina	Chatham	11.99	7.07	7.07	7.07	7.07	7.07	7.07
North Carolina	Cumberland	13.73	8.77	8.77	8.77	8.77	8.77	8.77
North Carolina	Davidson	15.17	8.89	8.89	8.89	8.89	8.89	8.89
North Carolina	Duplin	11.3	6.97	6.97	6.97	6.97	6.97	6.97
North Carolina	Durham	13.57	8.31	8.31	8.31	8.31	8.31	8.31
North Carolina	Edgecombe	12.37	7.76	7.76	7.76	7.76	7.76	7.76
North Carolina	Forsyth	14.28	8.28	8.28	8.28	8.28	8.28	8.28
North Carolina	Gaston	14.26	8.4	8.4	8.4	8.4	8.4	8.4
North Carolina	Guilford	13.79	8.17	8.17	8.17	8.17	8.17	8.17
North Carolina	Haywood	12.98	8.54	8.54	8.54	8.54	8.54	8.54
North Carolina	Jackson	12.09	7.42	7.42	7.42	7.42	7.42	7.42

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
North Carolina	Lenoir	11.12	6.88	6.88	6.88	6.88	6.88	6.88
North Carolina	McDowell	14.24	8.92	8.92	8.92	8.92	8.92	8.92
North Carolina	Martin	10.86	6.71	6.71	6.71	6.71	6.71	6.71
North Carolina	Mecklenburg	15.31	9.25	9.25	9.25	9.25	9.25	9.25
North Carolina	Mecklenburg	14.74	8.77	8.77	8.77	8.77	8.77	8.77
North Carolina	Mecklenburg	14.8	8.82	8.82	8.82	8.82	8.82	8.82
North Carolina	Mitchell	12.75	7.61	7.61	7.61	7.61	7.61	7.61
North Carolina	Montgomery	12.35	7.33	7.33	7.33	7.33	7.33	7.33
North Carolina	New Hanover	9.96	6.14	6.14	6.14	6.14	6.14	6.14
North Carolina	Onslow	10.98	6.77	6.77	6.77	6.77	6.77	6.77
North Carolina	Orange	13.12	7.86	7.86	7.86	7.86	7.86	7.86
North Carolina	Pitt	11.59	7.26	7.26	7.26	7.26	7.26	7.26
North Carolina	Robeson	12.78	8.03	8.03	8.03	8.03	8.03	8.03
North Carolina	Rowan	14.02	8.35	8.35	8.35	8.35	8.35	8.35
North Carolina	Swain	12.65	7.76	7.76	7.76	7.76	7.76	7.76
North Carolina	Wake	13.54	8.3	8.3	8.3	8.3	8.3	8.3
North Carolina	Watauga	12.05	6.85	6.85	6.85	6.85	6.85	6.85
North Carolina	Wayne	12.96	8.31	8.31	8.31	8.31	8.31	8.31
North Dakota	Billings	4.61	4.18	4.18	4.18	4.18	4.18	4.18
North Dakota	Burke	5.9	5.51	5.51	5.51	5.51	5.51	5.51

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
North Dakota	Burke	5.78	5.35	5.35	5.35	5.35	5.35	5.35
North Dakota	Burleigh	6.61	5.59	5.59	5.59	5.59	5.59	5.59
North Dakota	Cass	7.72	6.44	6.44	6.44	6.44	6.44	6.44
North Dakota	McKenzie	5.01	4.59	4.59	4.59	4.59	4.59	4.59
North Dakota	Mercer	6.04	5.38	5.38	5.38	5.38	5.38	5.38
Ohio	Athens	12.39	7.12	7.12	7.12	7.12	7.12	7.12
Ohio	Butler	15.74	9.9	9.9	9.9	9.9	9.9	9.9
Ohio	Butler	15.36	10.08	10.08	10.08	10.08	10.08	10.08
Ohio	Butler	14.9	9.63	9.63	9.63	9.63	9.63	9.63
Ohio	Clark	14.64	9.55	9.55	9.55	9.55	9.55	9.55
Ohio	Clermont	14.15	8.58	8.58	8.58	8.58	8.58	8.58
Ohio	Cuyahoga	15.46	10.28	10.28	10.28	10.28	9.53	9.53
Ohio	Cuyahoga	13.76	9.08	9.08	9.08	9.08	8.34	8.34
Ohio	Cuyahoga	17.37	11.79	11.79	11.79	11.79	11.04	11.04
Ohio	Cuyahoga	16.47	11.04	11.04	11.04	11.04	10.29	10.29
Ohio	Cuyahoga	17.11	11.51	11.51	11.51	11.51	10.76	10.76
Ohio	Cuyahoga	15.97	10.64	10.64	10.64	10.64	9.89	9.89
Ohio	Cuyahoga	14.14	9.45	9.45	9.45	9.45	8.71	8.71
Ohio	Franklin	15.27	9.82	9.82	9.82	9.82	9.82	9.82
Ohio	Franklin	15.08	9.7	9.7	9.7	9.7	9.7	9.7

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Ohio	Franklin	14.33	9.18	9.18	9.18	9.18	9.18	9.18
Ohio	Greene	13.36	8.34	8.34	8.34	8.34	8.34	8.34
Ohio	Hamilton	14.84	9.18	9.18	9.18	9.18	9.18	9.18
Ohio	Hamilton	17.29	10.81	10.81	10.81	10.81	10.81	10.81
Ohio	Hamilton	15.5	9.45	9.45	9.45	9.45	9.45	9.45
Ohio	Hamilton	16.85	10.63	10.63	10.63	10.63	10.63	10.63
Ohio	Hamilton	15.55	9.76	9.76	9.76	9.76	9.76	9.76
Ohio	Hamilton	16.17	9.99	9.99	9.99	9.99	9.99	9.99
Ohio	Hamilton	17.54	11.04	11.04	11.04	11.04	11.04	11.04
Ohio	Jefferson	15.41	9.33	9.33	9.33	9.33	9.33	9.33
Ohio	Jefferson	16.51	9.96	9.96	9.96	9.96	9.96	9.96
Ohio	Lake	13.02	8.67	8.67	8.67	8.67	8.60	8.60
Ohio	Lawrence	15.14	9.51	9.51	9.51	9.51	9.51	9.51
Ohio	Lorain	13.87	9.12	9.12	9.12	9.12	9.05	9.05
Ohio	Lorain	12.78	8.73	8.73	8.73	8.73	8.67	8.67
Ohio	Lucas	14.38	9.88	9.88	9.88	9.88	9.88	9.88
Ohio	Lucas	13.95	9.52	9.52	9.52	9.52	9.52	9.52
Ohio	Lucas	14.08	9.73	9.73	9.73	9.73	9.73	9.73
Ohio	Mahoning	14.68	9.47	9.47	9.47	9.47	9.47	9.47
Ohio	Mahoning	15.12	9.9	9.9	9.9	9.9	9.9	9.9

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Ohio	Montgomery	14.58	9.31	9.31	9.31	9.31	9.31	9.31
Ohio	Montgomery	15.54	10	10	10	10	10	10
Ohio	Portage	13.37	8.7	8.7	8.7	8.7	8.63	8.63
Ohio	Preble	13.7	8.8	8.8	8.8	8.8	8.8	8.8
Ohio	Scioto	14.65	8.86	8.86	8.86	8.86	8.86	8.86
Ohio	Stark	16.26	10.41	10.41	10.41	10.41	10.41	10.41
Ohio	Stark	15.23	10.07	10.07	10.07	10.07	10.07	10.07
Ohio	Summit	15.17	10.15	10.15	10.15	10.15	10.08	10.08
Ohio	Summit	14.26	9.54	9.54	9.54	9.54	9.47	9.47
Ohio	Trumbull	14.53	9.53	9.53	9.53	9.53	9.53	9.53
Oklahoma	Caddo	9.22	6.9	6.9	6.9	6.9	6.9	6.9
Oklahoma	Cherokee	11.79	8.59	8.59	8.59	8.59	8.59	8.59
Oklahoma	Kay	10.26	7.84	7.84	7.84	7.84	7.84	7.84
Oklahoma	Lincoln	10.28	7.54	7.54	7.54	7.54	7.54	7.54
Oklahoma	Mayes	11.7	8.64	8.64	8.64	8.64	8.64	8.64
Oklahoma	Mayes	11.44	8.38	8.38	8.38	8.38	8.38	8.38
Oklahoma	Muskogee	11.89	8.85	8.85	8.85	8.85	8.85	8.85
Oklahoma	Oklahoma	10.07	7.25	7.25	7.25	7.25	7.25	7.25
Oklahoma	Oklahoma	9.86	7.08	7.08	7.08	7.08	7.08	7.08
Oklahoma	Ottawa	11.69	8.63	8.63	8.63	8.63	8.63	8.63

(continued)

Table 4.A-11. **Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)**

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Oklahoma	Pittsburg	11.09	8.09	8.09	8.09	8.09	8.09	8.09
Oklahoma	Sequoyah	12.99	9.68	9.68	9.68	9.68	9.68	9.68
Oklahoma	Tulsa	11.52	8.45	8.45	8.45	8.45	8.45	8.45
Oklahoma	Tulsa	11.37	8.41	8.41	8.41	8.41	8.41	8.41
Oregon	Jackson	10.32	7.68	7.68	7.68	7.68	7.68	7.68
Oregon	Jackson	5.41	4.24	4.24	4.24	4.24	4.24	4.24
Oregon	Klamath	11.2	8.54	8.54	8.54	8.54	8.54	8.37
Oregon	Lane	8.64	6.39	6.39	6.39	6.39	6.39	5.35
Oregon	Lane	6.35	4.89	4.89	4.89	4.89	4.89	3.85
Oregon	Lane	7.56	5.75	5.75	5.75	5.75	5.75	4.71
Oregon	Lane	11.93	9.43	9.43	9.43	9.43	9.43	8.39
Oregon	Multnomah	9.13	6.23	6.23	6.23	6.23	6.23	6.23
Oregon	Multnomah	8.35	5.81	5.81	5.81	5.81	5.81	5.81
Oregon	Union	8.35	6.74	6.74	6.74	6.74	6.74	6.74
Pennsylvania	Adams	13.05	8.16	8.16	8.16	8.16	8.16	8.16
Pennsylvania	Allegheny	15.24	9.75	7.92	7.92	7.92	7.89	6.27
Pennsylvania	Allegheny	14.66	9.26	7.43	7.43	7.43	7.40	5.78
Pennsylvania	Allegheny	20.31	12.91	11.08	11.08	11.08	11.04	9.43
Pennsylvania	Allegheny	13.07	7.8	5.98	5.98	5.98	5.95	4.33
Pennsylvania	Allegheny	13.84	8.49	6.66	6.66	6.66	6.63	5.01

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Pennsylvania	Allegheny	15.36	9.76	7.93	7.93	7.93	7.90	6.28
Pennsylvania	Allegheny	15.25	9.33	7.50	7.50	7.50	7.47	5.85
Pennsylvania	Allegheny	16.26	10.06	8.23	8.23	8.23	8.20	6.58
Pennsylvania	Allegheny	15.3	9.43	7.60	7.60	7.60	7.57	5.95
Pennsylvania	Allegheny	14.44	8.84	7.01	7.01	7.01	6.98	5.36
Pennsylvania	Beaver	16.38	10.51	10.46	10.46	10.46	10.46	10.46
Pennsylvania	Berks	15.82	10.58	10.58	10.58	10.58	10.58	10.58
Pennsylvania	Bucks	13.42	8.45	8.45	8.45	8.45	8.45	8.45
Pennsylvania	Cambria	15.4	9.45	9.45	9.45	9.45	9.45	9.45
Pennsylvania	Centre	12.78	8	8	8	8	8	8
Pennsylvania	Chester	15.22	9.64	9.64	9.64	9.64	9.64	9.64
Pennsylvania	Cumberland	14.45	9.32	9.32	9.32	9.32	9.32	9.32
Pennsylvania	Dauphin	15.13	9.54	9.54	9.54	9.54	9.54	9.54
Pennsylvania	Delaware	15.23	9.74	9.74	9.74	9.74	9.74	9.74
Pennsylvania	Erie	12.54	8.39	8.39	8.39	8.39	8.39	8.39
Pennsylvania	Lackawanna	11.73	7.53	7.53	7.53	7.53	7.53	7.53
Pennsylvania	Lancaster	16.55	10.73	10.73	10.73	10.73	10.73	10.71
Pennsylvania	Lehigh	14.5	9.67	9.67	9.67	9.67	9.67	9.67
Pennsylvania	Luzerne	12.76	8.39	8.39	8.39	8.39	8.39	8.39
Pennsylvania	Mercer	13.28	8.47	8.47	8.47	8.47	8.47	8.47

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Pennsylvania	Northampton	13.68	8.96	8.96	8.96	8.96	8.96	8.96
Pennsylvania	Perry	12.81	8.22	8.22	8.22	8.22	8.22	8.22
Pennsylvania	Philadelphia	15.19	9.89	9.89	9.89	9.89	9.89	9.89
Pennsylvania	Washington	15.17	8.95	8.90	8.90	8.90	8.90	8.90
Pennsylvania	Washington	14.92	8.75	8.70	8.70	8.70	8.70	8.70
Pennsylvania	Washington	13.37	8.2	8.15	8.15	8.15	8.15	8.15
Pennsylvania	Westmoreland	15.49	9.26	9.21	9.21	9.21	9.21	9.21
Pennsylvania	York	16.52	10.66	10.66	10.66	10.66	10.66	10.66
Rhode Island	Providence	10.07	6.99	6.99	6.99	6.99	6.99	6.99
Rhode Island	Providence	12.14	8.42	8.42	8.42	8.42	8.42	8.42
Rhode Island	Providence	10.82	7.56	7.56	7.56	7.56	7.56	7.56
Rhode Island	Providence	9.93	6.85	6.85	6.85	6.85	6.85	6.85
South Carolina	Beaufort	11.52	7.25	7.25	7.25	7.25	7.25	7.25
South Carolina	Charleston	12.21	7.86	7.86	7.86	7.86	7.86	7.86
South Carolina	Charleston	11.6	7.13	7.13	7.13	7.13	7.13	7.13
South Carolina	Chesterfield	12.56	7.86	7.86	7.86	7.86	7.86	7.86
South Carolina	Edgefield	13.17	8.5	8.5	8.5	8.5	8.5	8.5
South Carolina	Florence	12.65	8.02	8.02	8.02	8.02	8.02	8.02
South Carolina	Georgetown	12.85	8.38	8.38	8.38	8.38	8.38	8.38
South Carolina	Greenville	15.65	9.7	9.7	9.7	9.7	9.7	9.7

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
South Carolina	Greenville	14.66	8.91	8.91	8.91	8.91	8.91	8.91
South Carolina	Greenwood	13.53	8.41	8.41	8.41	8.41	8.41	8.41
South Carolina	Horry	12.04	7.68	7.68	7.68	7.68	7.68	7.68
South Carolina	Lexington	14.64	9.28	9.28	9.28	9.28	9.28	9.28
South Carolina	Oconee	10.95	6.45	6.45	6.45	6.45	6.45	6.45
South Carolina	Richland	13.59	8.36	8.36	8.36	8.36	8.36	8.36
South Carolina	Richland	14.24	8.89	8.89	8.89	8.89	8.89	8.89
South Carolina	Spartanburg	14.17	8.58	8.58	8.58	8.58	8.58	8.58
South Dakota	Brookings	9.37	7.52	7.52	7.52	7.52	7.52	7.52
South Dakota	Brown	8.42	7.03	7.03	7.03	7.03	7.03	7.03
South Dakota	Codington	10.14	8.35	8.35	8.35	8.35	8.35	8.35
South Dakota	Custer	5.64	5.01	5.01	5.01	5.01	5.01	5.01
South Dakota	Jackson	5.39	4.65	4.65	4.65	4.65	4.65	4.65
South Dakota	Minnehaha	10.18	7.84	7.84	7.84	7.84	7.84	7.84
South Dakota	Minnehaha	9.58	7.38	7.38	7.38	7.38	7.38	7.38
South Dakota	Pennington	7.48	6.56	6.56	6.56	6.56	6.56	6.56
South Dakota	Pennington	8.77	7.71	7.71	7.71	7.71	7.71	7.71
South Dakota	Pennington	7.32	6.44	6.44	6.44	6.44	6.44	6.44
Tennessee	Blount	14.3	9.09	9.09	9.09	9.09	9.09	9.09
Tennessee	Davidson	14.21	8.87	8.87	8.87	8.87	8.87	8.87

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Tennessee	Davidson	13.99	8.63	8.63	8.63	8.63	8.63	8.63
Tennessee	Davidson	12.97	7.85	7.85	7.85	7.85	7.85	7.85
Tennessee	Dyer	12.28	7.8	7.8	7.8	7.8	7.8	7.8
Tennessee	Hamilton	15.67	9.91	9.91	9.91	9.91	9.91	9.91
Tennessee	Hamilton	13.73	8.18	8.18	8.18	8.18	8.18	8.18
Tennessee	Hamilton	15.16	9.38	9.38	9.38	9.38	9.38	9.38
Tennessee	Knox	15.47	9.66	9.66	9.66	9.66	9.66	9.66
Tennessee	Knox	15.64	9.77	9.77	9.77	9.77	9.77	9.77
Tennessee	Knox	15.18	9.24	9.24	9.24	9.24	9.24	9.24
Tennessee	Lawrence	11.69	7.39	7.39	7.39	7.39	7.39	7.39
Tennessee	Loudon	15.49	10.01	10.01	10.01	10.01	10.01	10.01
Tennessee	McMinn	14.29	8.93	8.93	8.93	8.93	8.93	8.93
Tennessee	Maury	13.21	8.39	8.39	8.39	8.39	8.39	8.39
Tennessee	Montgomery	13.8	8.65	8.65	8.65	8.65	8.65	8.65
Tennessee	Putnam	13.37	8.04	8.04	8.04	8.04	8.04	8.04
Tennessee	Roane	14.49	8.94	8.94	8.94	8.94	8.94	8.94
Tennessee	Shelby	13.71	8.73	8.73	8.73	8.73	8.73	8.73
Tennessee	Shelby	13.43	8.47	8.47	8.47	8.47	8.47	8.47
Tennessee	Shelby	13.68	8.6	8.6	8.6	8.6	8.6	8.6
Tennessee	Shelby	12.04	7.53	7.53	7.53	7.53	7.53	7.53

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Tennessee	Sullivan	14.16	9.15	9.15	9.15	9.15	9.15	9.15
Tennessee	Sumner	13.68	8.04	8.04	8.04	8.04	8.04	8.04
Texas	Bowie	12.85	9.21	9.21	9.21	9.21	9.21	9.21
Texas	Dallas	12.77	8.98	8.98	8.98	8.98	8.98	8.98
Texas	Dallas	11.8	8.16	8.16	8.16	8.16	8.16	8.16
Texas	Dallas	11.15	7.54	7.54	7.54	7.54	7.54	7.54
Texas	Ector	7.78	6.52	6.52	6.52	6.52	6.52	6.52
Texas	El Paso	9.09	7.94	7.94	7.94	7.94	7.94	7.94
Texas	Harris	11.77	8.65	8.65	8.65	8.65	7.94	7.94
Texas	Harris	15.42	11.61	11.61	11.61	11.61	10.90	10.90
Texas	Harrison	11.69	7.91	7.91	7.91	7.91	7.91	7.91
Texas	Hidalgo	10.98	9.17	9.17	9.17	9.17	9.17	9.17
Texas	Jefferson	11.56	8.26	8.26	8.26	8.26	8.26	8.26
Texas	Nueces	10.42	7.69	7.69	7.69	7.69	7.69	7.69
Texas	Nueces	9.63	7.04	7.04	7.04	7.04	7.04	7.04
Texas	Orange	11.51	8.37	8.37	8.37	8.37	8.37	8.37
Texas	Tarrant	11.41	7.77	7.77	7.77	7.77	7.77	7.77
Texas	Tarrant	12.23	8.41	8.41	8.41	8.41	8.41	8.41
Utah	Box Elder	8.4	7.16	7.10	7.10	7.10	7.10	7.09
Utah	Cache	11.56	9.78	8.32	8.32	8.32	8.32	7.30

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Utah	Davis	10.31	8.58	8.58	8.58	8.58	8.58	8.21
Utah	Salt Lake	11.68	9.29	9.00	9.00	9.00	9.00	8.01
Utah	Salt Lake	9.21	7.75	7.46	7.46	7.46	7.46	6.47
Utah	Salt Lake	11.3	9.05	8.76	8.76	8.76	8.76	7.77
Utah	Salt Lake	12.02	9.72	9.43	9.43	9.43	9.43	8.43
Utah	Salt Lake	8.33	6.91	6.62	6.62	6.62	6.62	5.66
Utah	Utah	10	8.37	8.37	8.37	8.37	8.37	7.48
Utah	Utah	10.52	8.8	8.8	8.8	8.8	8.8	7.91
Utah	Utah	8.88	7.44	7.44	7.44	7.44	7.44	6.57
Utah	Utah	8.78	7.37	7.37	7.37	7.37	7.37	6.50
Utah	Weber	11.16	9.23	9.16	9.16	9.16	9.16	9.15
Utah	Weber	9.28	7.71	7.64	7.64	7.64	7.64	7.64
Utah	Weber	9.36	7.8	7.73	7.73	7.73	7.73	7.72
Vermont	Addison	8.94	6.75	6.75	6.75	6.75	6.75	6.75
Vermont	Addison	8.91	6.71	6.71	6.71	6.71	6.71	6.71
Vermont	Bennington	8.52	6.02	6.02	6.02	6.02	6.02	6.02
Vermont	Chittenden	9.27	7.16	7.16	7.16	7.16	7.16	7.16
Vermont	Chittenden	10.02	7.76	7.76	7.76	7.76	7.76	7.76
Vermont	Rutland	11.08	8.15	8.15	8.15	8.15	8.15	8.15
Virginia	Arlington	14.27	8.87	8.87	8.87	8.87	8.87	8.87

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Virginia	Charles	12.37	7.45	7.45	7.45	7.45	7.45	7.45
Virginia	Chesterfield	13.44	8.1	8.1	8.1	8.1	8.1	8.1
Virginia	Fairfax	13.33	8.34	8.34	8.34	8.34	8.34	8.34
Virginia	Fairfax	13.62	8.54	8.54	8.54	8.54	8.54	8.54
Virginia	Fairfax	13.88	8.84	8.84	8.84	8.84	8.84	8.84
Virginia	Henrico	13.51	8.09	8.09	8.09	8.09	8.09	8.09
Virginia	Henrico	12.93	7.67	7.67	7.67	7.67	7.67	7.67
Virginia	Loudoun	13.57	8.62	8.62	8.62	8.62	8.62	8.62
Virginia	Page	12.79	7.49	7.49	7.49	7.49	7.49	7.49
Virginia	Bristol City	13.93	8.31	8.31	8.31	8.31	8.31	8.31
Virginia	Hampton City	12.17	7.46	7.46	7.46	7.46	7.46	7.46
Virginia	Lynchburg City	12.84	7.5	7.5	7.5	7.5	7.5	7.5
Virginia	Norfolk City	12.78	7.92	7.92	7.92	7.92	7.92	7.92
Virginia	Roanoke City	14.27	8.62	8.62	8.62	8.62	8.62	8.62
Virginia	Salem City	14.69	9.12	9.12	9.12	9.12	9.12	9.12
Virginia	Virginia Beach City	12.4	7.55	7.55	7.55	7.55	7.55	7.55
Washington	King	9.15	6.95	6.95	6.95	6.95	6.95	6.95
Washington	King	11.24	8.35	8.35	8.35	8.35	8.35	8.35
Washington	King	8.13	6.19	6.19	6.19	6.19	6.19	6.19
Washington	Pierce	10.55	8.11	8.11	8.11	8.11	8.11	7.94

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Washington	Snohomish	9.91	7.79	7.79	7.79	7.79	7.79	7.79
Washington	Spokane	9.97	7.19	7.19	7.19	7.19	7.19	7.19
West Virginia	Berkeley	15.93	10.37	10.37	10.37	10.37	10.37	10.37
West Virginia	Brooke	16.52	10.01	10.01	10.01	10.01	10.01	10.01
West Virginia	Brooke	16.04	9.6	9.6	9.6	9.6	9.6	9.6
West Virginia	Cabell	16.3	10.25	10.25	10.25	10.25	10.25	10.25
West Virginia	Hancock	15.76	9.6	9.6	9.6	9.6	9.6	9.6
West Virginia	Harrison	13.99	8.31	8.31	8.31	8.31	8.31	8.31
West Virginia	Kanawha	15.15	8.89	8.89	8.89	8.89	8.89	8.89
West Virginia	Kanawha	13.17	7.61	7.61	7.61	7.61	7.61	7.61
West Virginia	Kanawha	16.52	9.92	9.92	9.92	9.92	9.92	9.92
West Virginia	Marion	15.03	8.93	8.93	8.93	8.93	8.93	8.93
West Virginia	Marshall	15.19	8.74	8.74	8.74	8.74	8.74	8.74
West Virginia	Monongalia	14.35	8.09	8.09	8.09	8.09	8.09	8.09
West Virginia	Ohio	14.58	8.27	8.27	8.27	8.27	8.27	8.27
West Virginia	Raleigh	12.9	7.38	7.38	7.38	7.38	7.38	7.38
West Virginia	Wood	15.4	9.59	9.59	9.59	9.59	9.59	9.59
Wisconsin	Ashland	6.07	4.85	4.85	4.85	4.85	4.85	4.85
Wisconsin	Brown	11.39	8.48	8.48	8.48	8.48	8.48	8.48
Wisconsin	Dane	12.2	8.59	8.59	8.59	8.59	8.59	8.59

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Wisconsin	Dodge	11.04	7.95	7.95	7.95	7.95	7.95	7.95
Wisconsin	Forest	7.41	5.81	5.81	5.81	5.81	5.81	5.81
Wisconsin	Grant	11.79	8.5	8.5	8.5	8.5	8.5	8.5
Wisconsin	Kenosha	11.98	8.47	8.47	8.47	8.47	8.47	8.47
Wisconsin	Manitowoc	10.2	7.66	7.66	7.66	7.66	7.66	7.66
Wisconsin	Milwaukee	13.32	9.28	9.28	9.28	9.28	9.28	9.28
Wisconsin	Milwaukee	12.88	8.88	8.88	8.88	8.88	8.88	8.88
Wisconsin	Milwaukee	14.08	9.8	9.8	9.8	9.8	9.8	9.8
Wisconsin	Milwaukee	13.68	9.54	9.54	9.54	9.54	9.54	9.54
Wisconsin	Milwaukee	13.54	9.37	9.37	9.37	9.37	9.37	9.37
Wisconsin	Outagamie	10.96	8.11	8.11	8.11	8.11	8.11	8.11
Wisconsin	Ozaukee	11.6	8.37	8.37	8.37	8.37	8.37	8.37
Wisconsin	St. Croix	10.09	7.8	7.8	7.8	7.8	7.8	7.8
Wisconsin	Sauk	10.22	7.28	7.28	7.28	7.28	7.28	7.28
Wisconsin	Taylor	8.24	6.31	6.31	6.31	6.31	6.31	6.31
Wisconsin	Vilas	6.78	5.35	5.35	5.35	5.35	5.35	5.35
Wisconsin	Waukesha	13.91	9.84	9.84	9.84	9.84	9.84	9.84
Wyoming	Campbell	6.29	5.93	5.93	5.93	5.93	5.93	5.93
Wyoming	Campbell	5.11	4.8	4.8	4.8	4.8	4.8	4.8
Wyoming	Campbell	5.26	4.88	4.88	4.88	4.88	4.88	4.88

(continued)

Table 4.A-11. Annual Design Values (DVs) for the 2005 and 2020 Base Case and after Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35), 2020 12/35 and 2020 11/35 (continued)

State Name	County Name	2005 Annual DV	2020 Annual DV	2020 15/35 Annual DV	2020 13/35 Annual DV	2020 12/35 Annual DV	2020 11/35 Annual DV	2020 11/30 Annual DV
Wyoming	Converse	3.58	3.29	3.29	3.29	3.29	3.29	3.29
Wyoming	Fremont	8.17	7.25	7.25	7.25	7.25	7.25	7.25
Wyoming	Laramie	4.48	3.81	3.81	3.81	3.81	3.81	3.81
Wyoming	Sheridan	9.7	8.65	8.65	8.65	8.65	8.65	8.65

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
1003	10030010	Alabama	Baldwin	26.21	16.25	16.25	16.25
1027	10270001	Alabama	Clay	31.88	17.20	17.20	17.20
1033	10331002	Alabama	Colbert	30.43	15.63	15.63	15.63
1049	10491003	Alabama	De Kalb	32.08	17.07	17.07	17.07
1053	10530002	Alabama	Escambia	29.03	18.87	18.87	18.87
1055	10550010	Alabama	Etowah	35.18	19.12	19.12	19.12
1069	10690003	Alabama	Houston	28.66	18.24	18.24	18.24
1073	10730023	Alabama	Jefferson	44.06	27.91	27.91	24.14
1073	10731005	Alabama	Jefferson	34.83	21.49	21.49	17.73
1073	10731009	Alabama	Jefferson	34.5	17.88	17.88	14.12
1073	10731010	Alabama	Jefferson	34.16	18.12	18.12	14.36
1073	10732003	Alabama	Jefferson	40.3	28.55	28.55	24.79
1073	10732006	Alabama	Jefferson	33.17	18.34	18.34	14.58
1073	10735002	Alabama	Jefferson	33.05	17.43	17.43	13.67
1073	10735003	Alabama	Jefferson	35.81	19.46	19.46	15.70
1089	10890014	Alabama	Madison	33.58	17.12	17.12	17.12
1097	10970002	Alabama	Mobile	30.03	18.72	18.72	18.72
1097	10970003	Alabama	Mobile	28.58	17.74	17.74	17.74
1101	11010007	Alabama	Montgomery	32.05	18.56	18.56	18.56
1103	11030011	Alabama	Morgan	31.58	15.05	15.05	15.05
1113	11130001	Alabama	Russell	35.55	23.14	23.14	23.14
1117	11170006	Alabama	Shelby	32.05	18.39	18.39	18.39
1119	11190002	Alabama	Sumter	28.9	16.10	16.10	16.10
1121	11210002	Alabama	Talladega	33.46	18.57	18.57	18.57
1125	11250004	Alabama	Tuscaloosa	29.8	16.98	16.98	16.98
1127	11270002	Alabama	Walker	32.82	17.34	17.34	17.34
4003	40031005	Arizona	Cochise	16.62	15.73	15.73	15.69
4005	40051008	Arizona	Coconino	17.11	16.21	16.21	16.21
4007	40070008	Arizona	Gila	22.12	19.76	19.76	19.76

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
4013	40130019	Arizona	Maricopa	32.8	24.35	24.35	24.35
4013	40134003	Arizona	Maricopa	31.46	24.06	24.06	24.06
4013	40139997	Arizona	Maricopa	26.3	19.01	19.01	19.01
4019	40190011	Arizona	Pima	12.27	9.64	9.64	9.59
4019	40191028	Arizona	Pima	11.34	8.55	8.55	8.50
4021	40210001	Arizona	Pinal	17.55	14.56	14.56	14.56
4021	40213002	Arizona	Pinal	11.85	10.5	10.53	10.53
4023	40230004	Arizona	Santa Cruz	36.08	33.82	33.82	30.49
5001	50010011	Arkansas	Arkansas	29.16	18.21	18.21	18.21
5003	50030005	Arkansas	Ashley	28.91	21.00	21.00	21.00
5035	50350005	Arkansas	Crittenden	35.06	18.60	18.60	18.60
5045	50450002	Arkansas	Faulkner	29.87	19.26	19.26	19.26
5051	50510003	Arkansas	Garland	29.27	18.67	18.67	18.67
5107	51070001	Arkansas	Phillips	29.18	18.10	18.10	18.10
5113	51130002	Arkansas	Polk	26.13	15.71	15.71	15.71
5115	51150003	Arkansas	Pope	28.32	17.96	17.96	17.96
5119	51190007	Arkansas	Pulaski	31.16	19.81	19.81	19.81
5119	51191004	Arkansas	Pulaski	31.93	21.88	21.88	21.88
5119	51191005	Arkansas	Pulaski	31.91	21.73	21.73	21.73
5139	51390006	Arkansas	Union	28.7	19.81	19.81	19.81
5145	51450001	Arkansas	White	29.91	19.33	19.33	19.33
6001	60010007	California	Alameda	32.58	24.22	24.22	24.22
6001	60011001	California	Alameda	29.44	21.44	21.44	21.44
6007	60070002	California	Butte	52.55	32.16	32.16	30.46
6009	60090001	California	Calaveras	20.55	13.86	13.86	13.86
6011	60111002	California	Colusa	26.16	20.47	20.47	20.47
6013	60130002	California	Contra Costa	34.7	25.15	25.15	25.15
6019	60190008	California	Fresno	60.22	41.03	30.89	24.80
6019	60195001	California	Fresno	56.15	38.30	28.29	22.25

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
6019	60195025	California	Fresno	58.83	40.35	30.32	24.28
6025	60250005	California	Imperial	40.21	31.82	31.82	30.40
6025	60250007	California	Imperial	21.63	17.11	17.11	15.70
6025	60251003	California	Imperial	23.32	18.84	18.84	17.42
6027	60271003	California	Inyo	20	17.87	15.53	14.69
6029	60290010	California	Kern	64.54	45.93	24.00	21.62
6029	60290014	California	Kern	61.65	44.08	22.21	19.86
6029	60290016	California	Kern	60.38	44.51	22.60	20.23
6031	60310004	California	Kings	58.06	42.42	35.49	28.94
6033	60333001	California	Lake	12.94	11.83	11.83	11.83
6037	60370002	California	Los Angeles	49.85	35.54	25.18	24.24
6037	60371002	California	Los Angeles	49.7	37.38	26.69	25.66
6037	60371103	California	Los Angeles	50.97	38.39	27.52	26.43
6037	60371201	California	Los Angeles	42.4	31.36	21.57	20.77
6037	60371301	California	Los Angeles	48.71	39.72	29.16	28.17
6037	60371602	California	Los Angeles	50.2	39.16	28.51	27.50
6037	60372005	California	Los Angeles	42.2	28.18	17.75	16.78
6037	60374002	California	Los Angeles	41.42	34.83	24.66	23.78
6037	60374004	California	Los Angeles	39.38	33.49	23.49	22.66
6037	60379033	California	Los Angeles	17.11	13.71	5.166	4.73
6045	60450006	California	Mendocino	15.3	9.415	9.41	9.41
6047	60472510	California	Merced	46.15	31.30	28.37	27.38
6053	60531003	California	Monterey	14.35	10.98	8.783	7.98
6057	60570005	California	Nevada	13.93	11.13	11.13	11.13
6057	60571001	California	Nevada	16.55	12.65	12.65	12.65
6059	60590007	California	Orange	43.76	34.02	31.01	28.31
6059	60592022	California	Orange	33.85	29.68	27.43	24.94
6061	60610006	California	Placer	29.88	20.31	20.31	20.31
6063	60631006	California	Plumas	29.33	22.60	22.60	22.60

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
6063	60631009	California	Plumas	32.44	24.05	24.05	24.05
6065	60651003	California	Riverside	48.88	39.75	28.75	21.74
6065	60652002	California	Riverside	24.22	18.47	8.969	2.31
6065	60658001	California	Riverside	59.13	46.57	35.49	28.45
6067	60670006	California	Sacramento	49.22	32.92	32.92	27.12
6067	60670010	California	Sacramento	41.55	30.10	30.10	24.30
6067	60674001	California	Sacramento	39.55	28.08	28.08	22.27
6071	60710025	California	San Bernardino	51.9	41.28	34.02	26.86
6071	60710306	California	San Bernardino	23.11	16.64	10.43	3.53
6071	60712002	California	San Bernardino	52.85	39.80	32.26	25.03
6071	60718001	California	San Bernardino	37.56	33.65	27.93	21.16
6071	60719004	California	San Bernardino	55.5	41.53	34.44	27.32
6073	60730001	California	San Diego	30.58	21.96	21.96	21.96
6073	60730003	California	San Diego	35.55	25.35	25.35	25.35
6073	60730006	California	San Diego	24.11	17.52	17.52	17.52
6073	60731002	California	San Diego	33.28	23.94	23.94	23.94
6073	60731010	California	San Diego	33.17	23.49	23.49	23.49
6075	60750005	California	San Francisco	30.91	22.10	22.10	22.10
6077	60771002	California	San Joaquin	41.88	29.94	26.89	25.87
6079	60792006	California	San Luis Obispo	15.03	11.58	9.280	8.46
6079	60798001	California	San Luis Obispo	22.58	17.17	14.54	13.63
6081	60811001	California	San Mateo	29.41	21.72	21.72	21.72
6083	60830011	California	Santa Barbara	24.07	16.45	16.45	16.45
6085	60850005	California	Santa Clara	38.61	27.62	24.48	23.36
6085	60852003	California	Santa Clara	35.9	25.34	22.26	21.16
6089	60890004	California	Shasta	20.42	12.79	12.79	12.79
6095	60950004	California	Solano	34.76	25.26	25.26	25.26
6097	60970003	California	Sonoma	29.1	18.67	18.67	18.67
6099	60990005	California	Stanislaus	51.48	37.06	31.79	28.73

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
6101	61010003	California	Sutter	38.55	25.21	25.21	25.21
6107	61072002	California	Tulare	56.63	40.12	31.82	27.84
6111	61110007	California	Ventura	26.43	19.13	16.91	16.28
6111	61110009	California	Ventura	21.53	16.18	13.96	13.32
6111	61112002	California	Ventura	30.3	22.58	20.18	19.50
6111	61113001	California	Ventura	25.4	17.38	15.67	15.19
6113	61131003	California	Yolo	30.38	22.66	22.66	22.66
8001	80010006	Colorado	Adams	25.35	17.94	17.94	17.94
8005	80050005	Colorado	Arapahoe	21.27	15.47	15.47	15.47
8013	80130003	Colorado	Boulder	21.12	16.19	16.19	16.19
8013	80130012	Colorado	Boulder	18.7	14.40	14.40	14.40
8029	80290004	Colorado	Delta	20.76	14.09	14.09	14.09
8031	80310002	Colorado	Denver	26.44	19.47	19.47	19.47
8031	80310023	Colorado	Denver	26.36	19.57	19.57	19.57
8039	80390001	Colorado	Elbert	13.18	10.20	10.20	10.20
8041	80410008	Colorado	El Paso	16.41	10.29	10.29	10.29
8041	80410011	Colorado	El Paso	16.51	10.72	10.72	10.72
8069	80690009	Colorado	Larimer	18.3	13.96	13.96	13.96
8077	80770017	Colorado	Mesa	23.51	17.04	17.04	17.04
8101	81010012	Colorado	Pueblo	15.42	10.93	10.93	10.93
8113	81130004	Colorado	San Miguel	10.11	9.29	9.29	9.29
8123	81230006	Colorado	Weld	22.9	17.47	17.47	17.47
8123	81230008	Colorado	Weld	18.38	14.08	14.08	14.08
9001	90010010	Connecticut	Fairfield	36.27	22.27	22.27	22.27
9001	90011123	Connecticut	Fairfield	32.27	20.46	20.46	20.46
9001	90013005	Connecticut	Fairfield	34.91	19.74	19.74	19.74
9001	90019003	Connecticut	Fairfield	33.66	18.34	18.34	18.34
9003	90031003	Connecticut	Hartford	31.83	17.84	17.84	17.84
9005	90050005	Connecticut	Litchfield	27.16	13.00	13.00	13.00

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
9009	90090026	Connecticut	New Haven	35.65	20.29	20.29	20.29
9009	90090027	Connecticut	New Haven	35.58	20.10	20.10	20.10
9009	90091123	Connecticut	New Haven	38.37	21.78	21.78	21.78
9009	90092008	Connecticut	New Haven	33.68	19.01	19.01	19.01
9009	90092123	Connecticut	New Haven	34.45	20.00	20.00	20.00
9011	90113002	Connecticut	New London	32.03	16.66	16.66	16.66
10001	100010002	Delaware	Kent	32.14	18.35	18.35	18.35
10001	100010003	Delaware	Kent	31.5	17.57	17.57	17.57
10003	100031003	Delaware	New Castle	34.36	21.64	21.64	21.64
10003	100031007	Delaware	New Castle	32.65	17.11	17.11	17.11
10003	100031012	Delaware	New Castle	33.5	22.01	22.01	22.01
10003	100032004	Delaware	New Castle	36.66	22.47	22.47	22.47
10005	100051002	Delaware	Sussex	33.78	19.48	19.48	19.48
11001	110010041	D.C.	Washington	36.35	20.93	20.93	20.93
11001	110010042	D.C.	Washington	34.95	20.29	20.29	20.29
11001	110010043	D.C.	Washington	34.16	20.14	20.14	20.14
12001	120010023	Florida	Alachua	21.35	12.18	12.18	12.18
12001	120010024	Florida	Alachua	20.98	13.56	13.56	13.56
12005	120051004	Florida	Bay	28.08	18.31	18.31	18.31
12009	120090007	Florida	Brevard	20.73	12.88	12.88	12.88
12011	120111002	Florida	Broward	18.34	13.22	13.22	13.22
12011	120112004	Florida	Broward	18.63	13.16	13.16	13.16
12011	120113002	Florida	Broward	15.96	10.92	10.92	10.92
12017	120170005	Florida	Citrus	21.22	11.83	11.83	11.83
12031	120310098	Florida	Duval	23.72	16.13	16.13	16.13
12031	120310099	Florida	Duval	24.35	17.77	17.77	17.77
12033	120330004	Florida	Escambia	28.8	20.86	20.86	20.86
12057	120570030	Florida	Hillsborough	23.44	15.31	15.31	15.31
12057	120573002	Florida	Hillsborough	22.25	13.16	13.16	13.16

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
12071	120710005	Florida	Lee	17.7	12.27	12.27	12.27
12073	120730012	Florida	Leon	27.03	18.52	18.52	18.52
12081	120814012	Florida	Manatee	19.57	11.54	11.54	11.54
12083	120830003	Florida	Marion	22.56	13.49	13.49	13.49
12086	120861016	Florida	Miami-Dade	19.13	11.54	11.54	11.54
12086	120866001	Florida	Miami-Dade	18.6	12.83	12.83	12.83
12095	120952002	Florida	Orange	21.83	12.99	12.99	12.99
12099	120990009	Florida	Palm Beach	17.73	13.63	13.63	13.63
12099	120992005	Florida	Palm Beach	18.22	12.75	12.75	12.75
12103	121030018	Florida	Pinellas	21.73	14.58	14.58	14.58
12103	121031009	Florida	Pinellas	20.8	13.91	13.91	13.91
12105	121056006	Florida	Polk	19.3	12.88	12.88	12.88
12111	121111002	Florida	St Lucie	18.18	11.46	11.46	11.46
12115	121150013	Florida	Sarasota	19.22	12.19	12.19	12.19
12117	121171002	Florida	Seminole	22.08	12.45	12.45	12.45
12127	121275002	Florida	Volusia	22	12.81	12.81	12.81
13021	130210007	Georgia	Bibb	33.56	21.70	21.70	21.27
13021	130210012	Georgia	Bibb	30.74	17.76	17.76	17.33
13051	130510017	Georgia	Chatham	28.45	18.78	18.78	18.78
13051	130510091	Georgia	Chatham	27.9	18.29	18.29	18.29
13063	130630091	Georgia	Clayton	35.88	20.95	20.95	20.95
13067	130670003	Georgia	Cobb	35.04	19.81	19.81	19.81
13067	130670004	Georgia	Cobb	34.12	19.35	19.35	19.35
13089	130890002	Georgia	De Kalb	33.44	18.19	18.19	18.19
13089	130892001	Georgia	De Kalb	33.92	19.54	19.54	19.54
13095	130950007	Georgia	Dougherty	34.15	23.51	23.51	23.51
13115	131150005	Georgia	Floyd	35.12	21.22	21.22	21.22
13121	131210032	Georgia	Fulton	34.13	19.56	19.56	19.56
13121	131210039	Georgia	Fulton	37.66	22.76	22.76	22.45

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
13127	131270006	Georgia	Glynn	26.13	18.04	18.04	18.04
13135	131350002	Georgia	Gwinnett	32.81	18.28	18.28	18.28
13139	131390003	Georgia	Hall	30.11	18.95	18.95	18.95
13153	131530001	Georgia	Houston	29.63	17.62	17.62	17.59
13185	131850003	Georgia	Lowndes	25.68	16.90	16.90	16.90
13215	132150001	Georgia	Muscogee	31.38	21.83	21.83	21.83
13215	132150008	Georgia	Muscogee	34.58	22.33	22.33	22.33
13215	132150011	Georgia	Muscogee	30.25	20.09	20.09	20.09
13223	132230003	Georgia	Paulding	33.02	18.66	18.66	18.66
13245	132450005	Georgia	Richmond	32.7	23.41	23.41	23.41
13245	132450091	Georgia	Richmond	31.97	21.68	21.68	21.68
13295	132950002	Georgia	Walker	30.98	18.36	18.36	18.36
13303	133030001	Georgia	Washington	30.83	18.99	18.99	18.99
13319	133190001	Georgia	Wilkinson	33.16	20.66	20.66	20.66
16001	160010011	Idaho	Ada	28.36	25.01	25.01	25.01
16005	160050015	Idaho	Bannock	27.08	24.18	24.18	24.18
16009	160090010	Idaho	Benewah	32.94	28.46	28.46	28.38
16027	160270004	Idaho	Canyon	31.8	26.29	26.29	26.29
16041	160410001	Idaho	Franklin	36.76	30.49	30.49	30.43
16049	160490003	Idaho	Idaho	28.43	26.37	26.37	26.18
16059	160590004	Idaho	Lemhi	36.53	31.32	31.32	30.48
16077	160770011	Idaho	Power	33.36	29.75	29.75	29.75
16079	160790017	Idaho	Shoshone	38.16	31.96	31.96	30.48
17001	170010006	Illinois	Adams	31.41	18.25	18.25	18.25
17019	170190004	Illinois	Champaign	31.32	18.99	18.99	18.99
17019	170191001	Illinois	Champaign	30.04	19.02	19.02	19.02
17031	170310022	Illinois	Cook	36.61	28.61	28.61	28.25
17031	170310050	Illinois	Cook	36.11	25.08	25.08	24.72
17031	170310052	Illinois	Cook	40.26	26.51	26.51	26.15

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
17031	170310057	Illinois	Cook	37.37	25.25	25.25	24.89
17031	170310076	Illinois	Cook	38.05	25.52	25.52	25.16
17031	170311016	Illinois	Cook	43.03	28.93	28.93	28.56
17031	170312001	Illinois	Cook	37.7	26.75	26.75	26.38
17031	170313103	Illinois	Cook	39.65	27.06	27.06	26.70
17031	170313301	Illinois	Cook	40.22	26.35	26.35	25.98
17031	170314007	Illinois	Cook	34.31	22.60	22.60	22.23
17031	170314201	Illinois	Cook	32	21.74	21.74	21.38
17031	170316005	Illinois	Cook	39.17	29.02	29.02	28.65
17043	170434002	Illinois	Du Page	34.64	25.32	25.32	25.32
17065	170650002	Illinois	Hamilton	31.6	16.98	16.98	16.98
17083	170831001	Illinois	Jersey	32.18	19.67	19.67	18.93
17089	170890003	Illinois	Kane	33.85	23.48	23.48	23.48
17089	170890007	Illinois	Kane	34.83	25.12	25.12	25.12
17097	170971007	Illinois	Lake	33.08	21.19	21.19	21.19
17099	170990007	Illinois	La Salle	28.92	19.62	19.62	19.62
17111	171110001	Illinois	McHenry	31.58	20.88	20.88	20.88
17113	171132003	Illinois	McLean	33.43	20.92	20.92	20.92
17115	171150013	Illinois	Macon	33.25	18.70	18.70	18.70
17119	171190023	Illinois	Madison	37.31	24.71	24.71	22.72
17119	171191007	Illinois	Madison	39.16	25.29	25.29	23.30
17119	171192009	Illinois	Madison	34.97	22.05	22.05	20.06
17119	171193007	Illinois	Madison	34.03	19.78	19.78	17.80
17143	171430037	Illinois	Peoria	32.76	21.05	21.05	21.05
17157	171570001	Illinois	Randolph	28.96	20.19	20.19	20.19
17161	171613002	Illinois	Rock Island	30.9	22.57	22.57	22.57
17163	171630010	Illinois	St Clair	33.7	22.13	22.13	21.35
17163	171634001	Illinois	St Clair	31.91	22.79	22.79	22.07
17167	171670012	Illinois	Sangamon	33.41	21.67	21.67	21.67

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
17197	171971002	Illinois	Will	36.45	24.37	24.37	24.37
17197	171971011	Illinois	Will	30.71	17.71	17.71	17.71
17201	172010013	Illinois	Winnebago	34.73	24.49	24.49	24.49
18003	180030004	Indiana	Allen	33.1	23.14	23.14	23.14
18003	180030014	Indiana	Allen	30.51	20.81	20.81	20.81
18019	180190006	Indiana	Clark	37.57	20.97	20.97	20.97
18035	180350006	Indiana	Delaware	32.07	20.32	20.32	20.32
18037	180372001	Indiana	Dubois	35.36	21.93	21.93	21.93
18039	180390003	Indiana	Elkhart	34.43	25.09	25.09	25.09
18043	180431004	Indiana	Floyd	33.26	17.44	17.44	17.44
18065	180650003	Indiana	Henry	31.86	19.30	19.30	19.30
18067	180670003	Indiana	Howard	32.21	20.20	20.20	20.20
18083	180830004	Indiana	Knox	35.92	21.44	21.44	21.44
18089	180890006	Indiana	Lake	34.97	26.23	26.23	26.23
18089	180890022	Indiana	Lake	38.98	29.59	29.59	29.59
18089	180890026	Indiana	Lake	38.42	27.13	27.13	27.13
18089	180890027	Indiana	Lake	32.63	23.66	23.66	23.66
18089	180890031	Indiana	Lake	34	22.52	22.52	22.52
18089	180891003	Indiana	Lake	32.71	24.90	24.90	24.90
18089	180892004	Indiana	Lake	32.91	26.34	26.34	26.34
18089	180892010	Indiana	Lake	34.23	25.77	25.77	25.77
18091	180910011	Indiana	La Porte	33	21.47	21.47	21.47
18091	180910012	Indiana	La Porte	30.61	21.63	21.63	21.63
18095	180950009	Indiana	Madison	32.82	20.04	20.04	20.04
18097	180970042	Indiana	Marion	34.23	20.92	20.92	20.92
18097	180970043	Indiana	Marion	38.47	23.70	23.70	23.70
18097	180970066	Indiana	Marion	38.31	24.20	24.20	24.20
18097	180970078	Indiana	Marion	36.64	22.66	22.66	22.66
18097	180970079	Indiana	Marion	35.61	21.22	21.22	21.22

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
18097	180970081	Indiana	Marion	38.2	23.59	23.59	23.59
18097	180970083	Indiana	Marion	36.63	23.85	23.85	23.85
18127	181270020	Indiana	Porter	32.96	22.00	22.00	22.00
18127	181270024	Indiana	Porter	31.87	23.05	23.05	23.05
18141	181410014	Indiana	St Joseph	32.45	23.43	23.43	23.43
18141	181411008	Indiana	St Joseph	33.16	24.73	24.73	24.73
18141	181412004	Indiana	St Joseph	30.04	23.49	23.49	23.49
18147	181470009	Indiana	Spencer	32.32	15.43	15.43	15.43
18157	181570008	Indiana	Tippecanoe	35.68	20.81	20.81	20.81
18163	181630006	Indiana	Vanderburgh	34.8	22.81	22.81	22.81
18163	181630012	Indiana	Vanderburgh	33.27	22.48	22.48	22.48
18163	181630016	Indiana	Vanderburgh	32.66	22.91	22.91	22.91
18167	181670018	Indiana	Vigo	34.6	20.64	20.64	20.64
18167	181670023	Indiana	Vigo	34.88	19.44	19.44	19.44
19013	190130008	Iowa	Black Hawk	30.78	21.69	21.69	21.69
19045	190450021	Iowa	Clinton	33.95	23.86	23.86	23.86
19103	191032001	Iowa	Johnson	34.67	24.38	24.38	24.38
19113	191130037	Iowa	Linn	30.6	20.62	20.62	20.62
19137	191370002	Iowa	Montgomery	27.5	17.47	17.47	17.47
19139	191390015	Iowa	Muscatine	36.03	27.11	27.11	27.11
19147	191471002	Iowa	Palo Alto	25.73	18.02	18.02	18.02
19153	191530030	Iowa	Polk	28.41	20.25	20.25	20.25
19153	191532510	Iowa	Polk	27.26	18.56	18.56	18.56
19153	191532520	Iowa	Polk	31.46	22.22	22.22	22.22
19155	191550009	Iowa	Pottawattamie	28.6	21.11	21.11	21.11
19163	191630015	Iowa	Scott	31.01	21.33	21.33	21.33
19163	191630018	Iowa	Scott	32.34	23.19	23.19	23.19
19163	191630019	Iowa	Scott	37.1	25.09	25.09	25.09
19177	191770006	Iowa	Van Buren	28.36	19.89	19.89	19.89

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
19193	191930017	Iowa	Woodbury	26.4	19.58	19.58	19.58
19197	191970004	Iowa	Wright	28.65	19.15	19.15	19.15
20091	200910007	Kansas	Johnson	25.37	17.45	17.45	17.45
20091	200910009	Kansas	Johnson	29.3	22.43	22.43	22.43
20091	200910010	Kansas	Johnson	23.55	15.14	15.14	15.14
20107	201070002	Kansas	Linn	25.38	17.91	17.91	17.91
20173	201730008	Kansas	Sedgwick	23.7	16.04	16.04	16.04
20173	201730009	Kansas	Sedgwick	25.01	17.07	17.07	17.07
20173	201730010	Kansas	Sedgwick	25.37	18.21	18.21	18.21
20177	201770010	Kansas	Shawnee	29.16	21.81	21.81	21.81
20191	201910002	Kansas	Sumner	22.84	16.11	16.11	16.11
20209	202090021	Kansas	Wyandotte	29.58	21.07	21.07	21.07
20209	202090022	Kansas	Wyandotte	26.6	18.33	18.33	18.33
21013	210130002	Kentucky	Bell	29.9	16.93	16.93	16.93
21019	210190017	Kentucky	Boyd	33.15	16.09	16.09	16.09
21029	210290006	Kentucky	Bullitt	34.63	17.50	17.50	17.50
21037	210370003	Kentucky	Campbell	31.2	16.22	16.22	16.22
21043	210430500	Kentucky	Carter	29.91	13.49	13.49	13.49
21047	210470006	Kentucky	Christian	33.6	16.00	16.00	16.00
21059	210590005	Kentucky	Daviess	33.86	16.90	16.90	16.90
21067	210670012	Kentucky	Fayette	31.97	16.44	16.44	16.44
21067	210670014	Kentucky	Fayette	32.23	17.70	17.70	17.70
21073	210730006	Kentucky	Franklin	32.17	17.09	17.09	17.09
21093	210930006	Kentucky	Hardin	32.81	15.86	15.86	15.86
21101	211010014	Kentucky	Henderson	31.85	17.66	17.66	17.66
21111	211110043	Kentucky	Jefferson	35.48	18.39	18.39	18.39
21111	211110044	Kentucky	Jefferson	36.16	19.20	19.20	19.20
21111	211110048	Kentucky	Jefferson	36.44	20.47	20.47	20.47
21111	211110051	Kentucky	Jefferson	32.4	15.08	15.08	15.08

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
21117	211170007	Kentucky	Kenton	34.74	19.01	19.01	19.01
21125	211250004	Kentucky	Laurel	25.16	13.86	13.86	13.86
21145	211451004	Kentucky	McCracken	33.62	17.07	17.07	17.07
21151	211510003	Kentucky	Madison	30.11	15.14	15.14	15.14
21193	211930003	Kentucky	Perry	28.54	13.52	13.52	13.52
21195	211950002	Kentucky	Pike	30.52	15.35	15.35	15.35
21227	212270007	Kentucky	Warren	33.14	16.04	16.04	16.04
22017	220171002	Louisiana	Caddo	27.56	18.95	18.95	18.95
22019	220190009	Louisiana	Calcasieu	24.28	16.84	16.84	16.84
22019	220190010	Louisiana	Calcasieu	26.38	17.48	17.48	17.48
22029	220290003	Louisiana	Concordia	26.16	16.01	16.01	16.01
22033	220330009	Louisiana	East Baton Rouge	29.36	21.12	21.12	21.12
22033	220331001	Louisiana	East Baton Rouge	25.47	17.97	17.97	17.97
22047	220470005	Louisiana	Iberville	28.62	21.52	21.52	21.52
22047	220470009	Louisiana	Iberville	26.14	16.97	16.97	16.97
22051	220511001	Louisiana	Jefferson	27.06	16.37	16.37	16.37
22055	220550006	Louisiana	Lafayette	24.28	15.98	15.98	15.98
22073	220730004	Louisiana	Ouachita	28.91	19.57	19.57	19.57
22079	220790002	Louisiana	Rapides	30.26	18.76	18.76	18.76
22105	221050001	Louisiana	Tangipahoa	29.61	18.23	18.23	18.23
22109	221090001	Louisiana	Terrebonne	26.25	16.19	16.19	16.19
22121	221210001	Louisiana	West Baton Rouge	29.08	20.94	20.94	20.94
23001	230010011	Maine	Androscoggin	26.56	16.78	16.78	16.78
23003	230030013	Maine	Aroostook	24.23	20.18	20.18	20.18
23003	230031011	Maine	Aroostook	22.91	17.01	17.01	17.01
23005	230050015	Maine	Cumberland	27.74	16.79	16.79	16.79
23005	230050027	Maine	Cumberland	29.2	17.48	17.48	17.48
23009	230090103	Maine	Hancock	19.43	11.79	11.79	11.79
23011	230110016	Maine	Kennebec	26.21	15.86	15.86	15.86

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
23017	230172011	Maine	Oxford	28.36	18.96	18.96	18.96
23019	230190002	Maine	Penobscot	22.03	14.30	14.30	14.30
24003	240030014	Maryland	Anne Arundel	33.23	17.56	17.56	17.56
24003	240031003	Maryland	Anne Arundel	35.55	22.09	22.09	22.09
24003	240032002	Maryland	Anne Arundel	36.16	23.69	23.69	23.69
24005	240051007	Maryland	Baltimore	33.33	20.06	20.06	20.06
24005	240053001	Maryland	Baltimore	35.84	21.66	21.66	21.66
24015	240150003	Maryland	Cecil	30.82	19.07	19.07	19.07
24025	240251001	Maryland	Harford	31.21	17.19	17.19	17.19
24031	240313001	Maryland	Montgomery	30.93	17.19	17.19	17.19
24033	240330030	Maryland	Prince Georges	31.73	17.45	17.45	17.45
24033	240338003	Maryland	Prince Georges	33.46	18.03	18.03	18.03
24043	240430009	Maryland	Washington	33.43	20.15	20.15	20.15
24510	245100006	Maryland	Baltimore City	33.38	21.16	21.16	21.16
24510	245100007	Maryland	Baltimore City	34.74	22.38	22.38	22.38
24510	245100008	Maryland	Baltimore City	37.21	24.33	24.33	24.33
24510	245100035	Maryland	Baltimore City	37.75	24.85	24.85	24.85
24510	245100040	Maryland	Baltimore City	39.01	25.23	25.23	25.23
24510	245100049	Maryland	Baltimore City	38.16	26.21	26.21	26.21
25003	250035001	Massachusetts	Berkshire	31.06	19.54	19.54	19.54
25005	250051004	Massachusetts	Bristol	25.07	15.30	15.30	15.30
25009	250092006	Massachusetts	Essex	28.72	18.00	18.00	18.00
25009	250095005	Massachusetts	Essex	26.85	14.98	14.98	14.98
25009	250096001	Massachusetts	Essex	27.8	17.48	17.48	17.48
25013	250130008	Massachusetts	Hampden	27.26	17.08	17.08	17.08
25013	250130016	Massachusetts	Hampden	32.3	20.35	20.35	20.35
25013	250132009	Massachusetts	Hampden	33.13	20.80	20.80	20.80
25023	250230004	Massachusetts	Plymouth	28.48	16.46	16.46	16.46
25025	250250002	Massachusetts	Suffolk	29.45	19.75	19.75	19.75

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
25025	250250027	Massachusetts	Suffolk	29.23	19.20	19.20	19.20
25025	250250042	Massachusetts	Suffolk	28.6	19.01	19.01	19.01
25025	250250043	Massachusetts	Suffolk	32.17	20.80	20.80	20.80
25027	250270016	Massachusetts	Worcester	30.01	17.73	17.73	17.73
25027	250270023	Massachusetts	Worcester	30.66	18.47	18.47	18.47
26005	260050003	Michigan	Allegan	33.82	23.99	23.99	23.99
26017	260170014	Michigan	Bay	31.68	21.73	21.73	21.73
26021	260210014	Michigan	Berrien	31.32	21.21	21.21	21.21
26049	260490021	Michigan	Genesee	30.46	21.92	21.92	21.92
26065	260650012	Michigan	Ingham	31.96	22.40	22.40	22.40
26077	260770008	Michigan	Kalamazoo	31.17	21.05	21.05	21.05
26081	260810020	Michigan	Kent	36.53	23.50	23.50	23.50
26099	260990009	Michigan	Macomb	35.32	27.12	27.12	27.12
26113	261130001	Michigan	Missaukee	24.83	15.51	15.51	15.51
26115	261150005	Michigan	Monroe	38.88	23.36	23.36	23.36
26121	261210040	Michigan	Muskegon	34.71	23.72	23.72	23.72
26125	261250001	Michigan	Oakland	39.94	24.16	24.16	24.16
26139	261390005	Michigan	Ottawa	34.24	25.06	25.06	25.06
26145	261450018	Michigan	Saginaw	30.66	20.77	20.77	20.77
26147	261470005	Michigan	St Clair	39.61	28.92	28.92	28.92
26161	261610005	Michigan	Washtenaw	33.6	22.74	22.74	22.74
26161	261610008	Michigan	Washtenaw	39.46	23.39	23.39	23.39
26163	261630001	Michigan	Wayne	37.83	25.52	25.52	21.34
26163	261630015	Michigan	Wayne	40.12	27.89	27.89	23.71
26163	261630016	Michigan	Wayne	42.92	30.23	30.23	26.06
26163	261630019	Michigan	Wayne	40.92	30.69	30.69	26.51
26163	261630025	Michigan	Wayne	35.18	22.91	22.91	18.73
26163	261630033	Michigan	Wayne	43.88	31.37	31.37	27.19
26163	261630036	Michigan	Wayne	37.16	25.77	25.77	21.59

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
26163	261630039	Michigan	Wayne	37.03	26.93	26.93	22.75
27021	270210001	Minnesota	Cass	18.02	13.84	13.84	13.84
27037	270370470	Minnesota	Dakota	25.42	18.28	18.28	18.28
27053	270530050	Minnesota	Hennepin	27.25	19.26	19.26	19.26
27053	270530961	Minnesota	Hennepin	25.52	17.99	17.99	17.99
27053	270530963	Minnesota	Hennepin	26.07	18.83	18.83	18.83
27053	270530965	Minnesota	Hennepin	24.71	18.35	18.35	18.35
27053	270531007	Minnesota	Hennepin	25.44	17.85	17.85	17.85
27053	270532006	Minnesota	Hennepin	26.76	18.04	18.04	18.04
27095	270953051	Minnesota	Mille Lacs	22.03	16.93	16.93	16.93
27123	271230866	Minnesota	Ramsey	28.04	20.57	20.57	20.57
27123	271230868	Minnesota	Ramsey	28.38	20.59	20.59	20.59
27123	271230871	Minnesota	Ramsey	26.36	19.72	19.72	19.72
27137	271377001	Minnesota	St Louis	20.31	15.63	15.63	15.63
27137	271377550	Minnesota	St Louis	19.51	14.20	14.20	14.20
27137	271377551	Minnesota	St Louis	23.53	16.56	16.56	16.56
27139	271390505	Minnesota	Scott	24.98	17.93	17.93	17.93
28001	280010004	Mississippi	Adams	27.48	16.79	16.79	16.79
28011	280110001	Mississippi	Bolivar	28.98	19.20	19.20	19.20
28033	280330002	Mississippi	De Soto	30.82	15.85	15.85	15.85
28035	280350004	Mississippi	Forrest	30.48	20.78	20.78	20.78
28047	280470008	Mississippi	Harrison	29	18.33	18.33	18.33
28049	280490010	Mississippi	Hinds	28.83	17.03	17.03	17.03
28059	280590006	Mississippi	Jackson	26.96	16.48	16.48	16.48
28067	280670002	Mississippi	Jones	31.21	20.85	20.85	20.85
28081	280810005	Mississippi	Lee	32.18	16.69	16.69	16.69
28087	280870001	Mississippi	Lowndes	32.44	17.28	17.28	17.28
28149	281490004	Mississippi	Warren	30.26	19.28	19.28	19.28
29019	290190004	Missouri	Boone	30.23	19.03	19.03	19.03

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
29021	290210005	Missouri	Buchanan	30.1	21.61	21.61	21.61
29037	290370003	Missouri	Cass	25.61	16.89	16.89	16.89
29039	290390001	Missouri	Cedar	28.7	18.77	18.77	18.77
29047	290470005	Missouri	Clay	28.04	20.00	20.00	20.00
29077	290770032	Missouri	Greene	28.27	18.75	18.75	18.75
29095	290950034	Missouri	Jackson	27.88	20.33	20.33	20.33
29099	290990012	Missouri	Jefferson	33.43	21.12	21.12	21.12
29137	291370001	Missouri	Monroe	27.83	17.98	17.98	17.98
29183	291831002	Missouri	St Charles	33.16	20.01	20.01	20.01
29186	291860006	Missouri	Ste Genevieve	31.44	18.83	18.83	18.83
29189	291890004	Missouri	St Louis	32.03	20.46	20.46	20.46
29189	291892003	Missouri	St Louis	33.21	23.50	23.50	23.50
29510	295100007	Missouri	St Louis City	33.16	20.79	20.79	20.79
29510	295100085	Missouri	St Louis City	33.24	21.03	21.03	21.03
29510	295100086	Missouri	St Louis City	32.5	22.13	22.13	22.13
29510	295100087	Missouri	St Louis City	34.35	22.07	22.07	22.07
30013	300131026	Montana	Cascade	20.15	17.08	17.08	17.08
30029	300290009	Montana	Flathead	27.14	22.76	22.56	22.05
30029	300290047	Montana	Flathead	27.17	24.28	24.11	23.65
30031	300310008	Montana	Gallatin	29.55	26.24	26.24	26.24
30031	300310013	Montana	Gallatin	12.2	11.3	11.39	11.39
30047	300470013	Montana	Lake	27.03	23.75	20.89	15.95
30047	300470028	Montana	Lake	43.66	38.36	35.49	30.49
30049	300490018	Montana	Lewis And Clark	33.53	28.27	28.27	28.27
30053	300530018	Montana	Lincoln	42.71	35.36	35.12	29.50
30063	300630031	Montana	Missoula	44.64	37.47	34.48	30.48
30081	300810007	Montana	Ravalli	45.11	37.30	35.48	30.48
30087	300870307	Montana	Rosebud	19.73	18.34	18.34	18.34
30089	300890007	Montana	Sanders	20.42	18.25	18.10	17.71

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
30093	300930005	Montana	Silver Bow	35	28.15	28.15	28.15
30111	301111065	Montana	Yellowstone	19.38	15.76	15.76	15.76
31025	310250002	Nebraska	Cass	28.3	20.73	20.73	20.73
31055	310550019	Nebraska	Douglas	25.7	19.25	19.25	19.25
31055	310550052	Nebraska	Douglas	25.76	18.99	18.99	18.99
31079	310790004	Nebraska	Hall	19.16	14.31	14.31	14.31
31109	311090022	Nebraska	Lancaster	24.77	17.87	17.87	17.87
31157	311570003	Nebraska	Scotts Bluff	16.66	13.90	13.90	13.90
31177	311770002	Nebraska	Washington	24.01	17.68	17.68	17.68
32003	320030022	Nevada	Clark	9.13	8.18	8.18	8.18
32003	320030298	Nevada	Clark	12.43	10.10	10.10	10.10
32003	320030561	Nevada	Clark	25.26	19.31	19.31	19.31
32003	320031019	Nevada	Clark	8.6	7.53	7.53	7.53
32003	320032002	Nevada	Clark	20.93	16.35	16.35	16.35
32031	320310016	Nevada	Washoe	30.78	20.85	20.85	20.85
33001	330012004	New Hampshire	Belknap	20.55	11.31	11.31	11.31
33005	330050007	New Hampshire	Cheshire	30.23	18.74	18.74	18.74
33007	330070014	New Hampshire	Coos	26.5	17.08	17.08	17.08
33009	330090010	New Hampshire	Grafton	23	14.67	14.67	14.67
33011	330110020	New Hampshire	Hillsborough	28.66	18.87	18.87	18.87
33011	330111015	New Hampshire	Hillsborough	27.33	19.03	19.03	19.03
33011	330115001	New Hampshire	Hillsborough	25.9	12.93	12.93	12.93
33013	330131006	New Hampshire	Merrimack	25.65	15.12	15.12	15.12
33015	330150014	New Hampshire	Rockingham	26.35	15.67	15.67	15.67
33019	330190003	New Hampshire	Sullivan	28.92	16.49	16.49	16.49
34003	340030003	New Jersey	Bergen	37.03	22.46	22.46	22.46
34007	340070003	New Jersey	Camden	36.5	20.93	20.93	20.93
34007	340071007	New Jersey	Camden	37.37	20.89	20.89	20.89
34013	340130015	New Jersey	Essex	38.38	22.59	22.59	22.59

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
34017	340171003	New Jersey	Hudson	39.08	25.73	25.73	25.73
34017	340172002	New Jersey	Hudson	41.43	29.62	29.62	29.62
34021	340210008	New Jersey	Mercer	34.75	18.67	18.67	18.67
34023	340230006	New Jersey	Middlesex	34.82	19.67	19.67	19.67
34027	340270004	New Jersey	Morris	32.32	18.32	18.32	18.32
34027	340273001	New Jersey	Morris	31.5	16.03	16.03	16.03
34029	340292002	New Jersey	Ocean	31.56	16.14	16.14	16.14
34031	340310005	New Jersey	Passaic	36.3	21.13	21.13	21.13
34039	340390004	New Jersey	Union	40.47	24.64	24.64	24.64
34039	340390006	New Jersey	Union	37.35	21.11	21.11	21.11
34039	340392003	New Jersey	Union	36.82	21.32	21.32	21.32
34041	340410006	New Jersey	Warren	34.06	20.46	20.46	20.46
35001	350010023	New Mexico	Bernalillo	18.6	14.61	14.61	14.61
35001	350010024	New Mexico	Bernalillo	16.43	13.12	13.12	13.12
35005	350050005	New Mexico	Chaves	15.68	12.43	12.43	12.43
35013	350130017	New Mexico	Dona Ana	32.95	26.90	26.90	26.90
35013	350130025	New Mexico	Dona Ana	13.8	11.66	11.66	11.66
35017	350171002	New Mexico	Grant	13	12.21	12.21	12.21
35043	350431003	New Mexico	Sandoval	10.3	8.01	8.01	8.01
35043	350439011	New Mexico	Sandoval	15.68	13.73	13.73	13.73
35045	350450006	New Mexico	San Juan	12.4	10.91	10.91	10.91
35049	350490020	New Mexico	Santa Fe	9.78	8.57	8.57	8.57
36001	360010005	New York	Albany	34.26	22.51	22.51	22.51
36005	360050080	New York	Bronx	38.87	26.00	26.00	26.00
36005	360050083	New York	Bronx	34.74	21.32	21.32	21.32
36005	360050110	New York	Bronx	36.11	25.41	25.41	25.41
36013	360130011	New York	Chautauqua	29.15	15.82	15.82	15.82
36029	360290005	New York	Erie	35.35	25.54	25.54	25.54
36029	360291007	New York	Erie	33.61	23.32	23.32	23.32

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
36031	360310003	New York	Essex	22.45	13.77	13.77	13.77
36047	360470122	New York	Kings	36.94	23.04	23.04	23.04
36055	360551007	New York	Monroe	32.2	19.38	19.38	19.38
36059	360590008	New York	Nassau	34.01	19.15	19.15	19.15
36061	360610056	New York	New York	39.7	26.51	26.51	23.61
36061	360610062	New York	New York	38.82	23.85	23.85	20.95
36061	360610079	New York	New York	37.94	25.73	25.73	22.83
36061	360610128	New York	New York	39.45	25.95	25.95	23.05
36063	360632008	New York	Niagara	33.87	21.99	21.99	21.99
36067	360671015	New York	Onondaga	27.35	16.74	16.74	16.74
36071	360710002	New York	Orange	28.92	18.47	18.47	18.47
36081	360810124	New York	Queens	35.56	22.44	22.44	22.44
36085	360850055	New York	Richmond	34.93	21.17	21.17	21.17
36085	360850067	New York	Richmond	32.41	17.57	17.57	17.57
36089	360893001	New York	St Lawrence	22.05	15.22	15.22	15.22
36101	361010003	New York	Steuben	27.81	14.94	14.94	14.94
36103	361030001	New York	Suffolk	34.66	18.09	18.09	18.09
36119	361191002	New York	Westchester	33.51	19.41	19.41	19.41
37001	370010002	North Carolina	Alamance	31.72	18.09	18.09	18.09
37021	370210034	North Carolina	Buncombe	30.05	15.83	15.83	15.83
37033	370330001	North Carolina	Caswell	29.45	16.07	16.07	16.07
37035	370350004	North Carolina	Catawba	34.53	19.24	19.24	19.24
37037	370370004	North Carolina	Chatham	26.94	13.82	13.82	13.82
37051	370510009	North Carolina	Cumberland	30.78	17.31	17.31	17.31
37057	370570002	North Carolina	Davidson	31.35	18.28	18.28	18.28
37061	370610002	North Carolina	Duplin	28.3	15.35	15.35	15.35
37063	370630001	North Carolina	Durham	31.02	16.47	16.47	16.47
37065	370650004	North Carolina	Edgecombe	26.78	16.60	16.60	16.60
37067	370670022	North Carolina	Forsyth	31.92	18.32	18.32	18.32

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
37071	370710016	North Carolina	Gaston	30.86	16.10	16.10	16.10
37081	370810013	North Carolina	Guilford	30.63	17.80	17.80	17.80
37087	370870010	North Carolina	Haywood	27.74	16.38	16.38	16.38
37099	370990006	North Carolina	Jackson	24.96	13.91	13.91	13.91
37107	371070004	North Carolina	Lenoir	25.2	15.55	15.55	15.55
37111	371110004	North Carolina	McDowell	31.55	17.30	17.30	17.30
37117	371170001	North Carolina	Martin	24.83	14.78	14.78	14.78
37119	371190010	North Carolina	Mecklenburg	32.33	18.39	18.39	18.39
37119	371190041	North Carolina	Mecklenburg	31.72	16.71	16.71	16.71
37119	371190042	North Carolina	Mecklenburg	30.7	16.34	16.34	16.34
37121	371210001	North Carolina	Mitchell	30.25	15.29	15.29	15.29
37123	371230001	North Carolina	Montgomery	28.21	15.02	15.02	15.02
37129	371290002	North Carolina	New Hanover	25.4	13.75	13.75	13.75
37133	371330005	North Carolina	Onslow	24.61	14.53	14.53	14.53
37135	371350007	North Carolina	Orange	29.35	15.60	15.60	15.60
37147	371470005	North Carolina	Pitt	26.21	16.20	16.20	16.20
37155	371550005	North Carolina	Robeson	29.92	16.31	16.31	16.31
37159	371590021	North Carolina	Rowan	30.23	17.71	17.71	17.71
37173	371730002	North Carolina	Swain	27.34	15.03	15.03	15.03
37183	371830014	North Carolina	Wake	31.63	16.96	16.96	16.96
37189	371890003	North Carolina	Watauga	30.43	15.96	15.96	15.96
37191	371910005	North Carolina	Wayne	29.72	17.01	17.01	17.01
38007	380070002	North Dakota	Billings	13.07	11.57	11.57	11.57
38013	380130003	North Dakota	Burke	16.73	15.05	15.05	15.05
38015	380150003	North Dakota	Burleigh	17.62	14.39	14.39	14.39
38017	380171004	North Dakota	Cass	21.22	16.05	16.05	16.05
38053	380530002	North Dakota	McKenzie	11.96	10.4	10.41	10.41
38057	380570004	North Dakota	Mercer	16.98	14.36	14.36	14.36
39009	390090003	Ohio	Athens	32.32	15.83	15.83	15.83

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
39017	390170003	Ohio	Butler	39.23	23.53	23.53	23.53
39017	390170016	Ohio	Butler	37.14	19.79	19.79	19.79
39017	390170017	Ohio	Butler	37.93	20.19	20.19	20.19
39017	390171004	Ohio	Butler	37.13	19.29	19.29	19.29
39023	390230005	Ohio	Clark	35.37	19.40	19.40	19.40
39025	390250022	Ohio	Clermont	34.46	17.07	17.07	17.07
39035	390350027	Ohio	Cuyahoga	36.6	24.82	24.82	22.59
39035	390350034	Ohio	Cuyahoga	36.58	21.66	21.66	19.46
39035	390350038	Ohio	Cuyahoga	44.2	29.73	29.73	27.50
39035	390350045	Ohio	Cuyahoga	38.57	23.28	23.28	21.05
39035	390350060	Ohio	Cuyahoga	42.12	26.89	26.89	24.66
39035	390350065	Ohio	Cuyahoga	38.67	22.85	22.85	20.62
39035	390351002	Ohio	Cuyahoga	34.25	20.89	20.89	18.67
39049	390490024	Ohio	Franklin	38.51	21.09	21.09	21.09
39049	390490025	Ohio	Franklin	38.46	20.11	20.11	20.11
39049	390490081	Ohio	Franklin	34.16	18.98	18.98	18.98
39057	390570005	Ohio	Greene	32.21	16.98	16.98	16.98
39061	390610006	Ohio	Hamilton	37.66	17.85	17.85	17.85
39061	390610014	Ohio	Hamilton	38.24	19.50	19.50	19.50
39061	390610040	Ohio	Hamilton	36.73	18.87	18.87	18.87
39061	390610042	Ohio	Hamilton	37.3	20.82	20.82	20.82
39061	390610043	Ohio	Hamilton	35.95	18.63	18.63	18.63
39061	390617001	Ohio	Hamilton	38.81	20.11	20.11	20.11
39061	390618001	Ohio	Hamilton	40.6	22.10	22.10	22.10
39081	390810017	Ohio	Jefferson	40.7	24.05	24.05	24.05
39081	390811001	Ohio	Jefferson	41.96	22.59	22.59	22.59
39085	390851001	Ohio	Lake	37.16	21.08	21.08	20.88
39087	390870010	Ohio	Lawrence	33.77	18.28	18.28	18.28
39093	390933002	Ohio	Lorain	31.56	19.08	19.08	18.91

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
39095	390950024	Ohio	Lucas	36.34	23.56	23.56	23.56
39095	390950025	Ohio	Lucas	35.14	25.95	25.95	25.95
39095	390950026	Ohio	Lucas	34.9	23.59	23.59	23.59
39099	390990005	Ohio	Mahoning	35.16	19.98	19.98	19.98
39099	390990014	Ohio	Mahoning	36.83	21.48	21.48	21.48
39113	391130031	Ohio	Montgomery	35.78	22.68	22.68	22.68
39113	391130032	Ohio	Montgomery	37.8	19.27	19.27	19.27
39133	391330002	Ohio	Portage	34.32	18.83	18.83	18.63
39135	391351001	Ohio	Preble	32.85	17.51	17.51	17.51
39145	391450013	Ohio	Scioto	34.55	18.20	18.20	18.20
39151	391510017	Ohio	Stark	36.9	20.19	20.19	20.19
39153	391530017	Ohio	Summit	38.06	21.46	21.46	21.26
39153	391530023	Ohio	Summit	35.88	20.29	20.29	20.09
39155	391550007	Ohio	Trumbull	36.23	21.39	21.39	21.39
40015	400159008	Oklahoma	Caddo	23.97	16.68	16.68	16.68
40021	400219002	Oklahoma	Cherokee	27.55	20.06	20.06	20.06
40071	400710602	Oklahoma	Kay	31.8	25.60	25.60	25.60
40071	400719010	Oklahoma	Kay	27.93	20.55	20.55	20.55
40081	400819005	Oklahoma	Lincoln	27.83	19.33	19.33	19.33
40097	400970186	Oklahoma	Mayes	28.71	21.86	21.86	21.86
40097	400979014	Oklahoma	Mayes	26.13	18.49	18.49	18.49
40101	401010169	Oklahoma	Muskogee	29.54	20.95	20.95	20.95
40109	401090035	Oklahoma	Oklahoma	23.42	16.46	16.46	16.46
40109	401091037	Oklahoma	Oklahoma	27.12	19.18	19.18	19.18
40115	401159004	Oklahoma	Ottawa	29.14	20.58	20.58	20.58
40121	401210415	Oklahoma	Pittsburg	26.37	18.75	18.75	18.75
40135	401359015	Oklahoma	Sequoyah	31.43	22.98	22.98	22.98
40143	401430110	Oklahoma	Tulsa	28.43	20.69	20.69	20.69
40143	401431127	Oklahoma	Tulsa	30.37	21.77	21.77	21.77

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
41029	410290133	Oregon	Jackson	33.72	23.50	23.50	23.50
41029	410291001	Oregon	Jackson	14.51	10.43	10.43	10.43
41035	410350004	Oregon	Klamath	44.08	30.85	30.85	30.14
41039	410390060	Oregon	Lane	32.55	21.42	21.42	16.81
41039	410391007	Oregon	Lane	15.63	10.16	10.16	5.56
41039	410391009	Oregon	Lane	23.96	16.59	16.59	11.98
41039	410392013	Oregon	Lane	48.95	34.01	34.01	29.40
41051	410510080	Oregon	Multnomah	29.88	19.10	19.10	19.10
41051	410510246	Oregon	Multnomah	23.22	15.24	15.24	15.24
41061	410610119	Oregon	Union	27.38	22.64	22.64	22.64
42001	420010001	Pennsylvania	Adams	34.93	20.05	20.05	20.05
42003	420030008	Pennsylvania	Allegheny	39.44	22.23	16.72	11.72
42003	420030021	Pennsylvania	Allegheny	35.16	19.37	13.86	8.86
42003	420030064	Pennsylvania	Allegheny	64.27	41.03	35.49	30.48
42003	420030067	Pennsylvania	Allegheny	36.48	17.26	11.76	6.76
42003	420030093	Pennsylvania	Allegheny	45.6	24.65	24.65	24.65
42003	420030095	Pennsylvania	Allegheny	38.77	21.02	15.51	10.51
42003	420030116	Pennsylvania	Allegheny	42.56	22.61	17.10	12.10
42003	420030133	Pennsylvania	Allegheny	39.23	24.73	24.73	24.73
42003	420031008	Pennsylvania	Allegheny	41.34	21.40	15.88	10.88
42003	420031301	Pennsylvania	Allegheny	40.3	20.93	15.40	10.40
42003	420033007	Pennsylvania	Allegheny	37.52	21.63	16.12	11.12
42003	420039002	Pennsylvania	Allegheny	37.86	20.14	14.62	9.62
42007	420070014	Pennsylvania	Beaver	43.42	23.46	23.32	23.32
42011	420110011	Pennsylvania	Berks	37.71	27.18	27.18	27.18
42017	420170012	Pennsylvania	Bucks	34.01	20.66	20.66	20.66
42021	420210011	Pennsylvania	Cambria	39.04	19.60	19.60	19.60
42027	420270100	Pennsylvania	Centre	36.28	21.01	21.01	21.01
42029	420290100	Pennsylvania	Chester	36.7	22.40	22.40	22.40

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
42041	420410101	Pennsylvania	Cumberland	38	25.33	25.33	25.33
42043	420430401	Pennsylvania	Dauphin	38.04	26.65	26.65	26.65
42045	420450002	Pennsylvania	Delaware	35.24	21.08	21.08	21.08
42049	420490003	Pennsylvania	Erie	34.46	20.16	20.16	20.16
42069	420692006	Pennsylvania	Lackawanna	31.55	17.61	17.61	17.61
42071	420710007	Pennsylvania	Lancaster	40.83	30.51	30.51	30.47
42077	420770004	Pennsylvania	Lehigh	36.4	24.04	24.04	24.04
42079	420791101	Pennsylvania	Luzerne	32.46	20.14	20.14	20.14
42085	420850100	Pennsylvania	Mercer	36.3	20.84	20.84	20.84
42095	420950025	Pennsylvania	Northampton	36.72	22.79	22.79	22.79
42099	420990301	Pennsylvania	Perry	30.46	20.22	20.22	20.22
42101	421010004	Pennsylvania	Philadelphia	36.53	21.31	21.31	21.31
42101	421010024	Pennsylvania	Philadelphia	35.96	19.61	19.61	19.61
42101	421010047	Pennsylvania	Philadelphia	37.3	21.84	21.84	21.84
42125	421250005	Pennsylvania	Washington	35.52	19.95	19.81	19.81
42125	421250200	Pennsylvania	Washington	33.5	18.59	18.44	18.44
42125	421255001	Pennsylvania	Washington	38.14	17.33	17.20	17.20
42129	421290008	Pennsylvania	Westmoreland	37.12	18.80	18.66	18.66
42133	421330008	Pennsylvania	York	38.24	28.16	28.16	28.16
44007	440070022	Rhode Island	Providence	29.46	17.18	17.18	17.18
44007	440070026	Rhode Island	Providence	30.62	18.87	18.87	18.87
44007	440070028	Rhode Island	Providence	28.1	17.47	17.47	17.47
44007	440071010	Rhode Island	Providence	28.8	17.32	17.32	17.32
45019	450190049	South Carolina	Charleston	27.93	15.52	15.52	15.52
45025	450250001	South Carolina	Chesterfield	28.77	16.09	16.09	16.09
45037	450370001	South Carolina	Edgefield	32.23	17.45	17.45	17.45
45041	450410002	South Carolina	Florence	28.81	16.50	16.50	16.50
45045	450450008	South Carolina	Greenville	31.86	18.71	18.71	18.71
45045	450450009	South Carolina	Greenville	32.55	18.18	18.18	18.18

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
45047	450470003	South Carolina	Greenwood	30.01	16.33	16.33	16.33
45051	450510002	South Carolina	Horry	28.3	16.66	16.66	16.66
45063	450630008	South Carolina	Lexington	32.86	19.02	19.02	19.02
45073	450730001	South Carolina	Oconee	27.98	14.56	14.56	14.56
45079	450790007	South Carolina	Richland	31.38	17.08	17.08	17.08
45079	450790019	South Carolina	Richland	33.2	19.00	19.00	19.00
45083	450830010	South Carolina	Spartanburg	32.46	17.99	17.99	17.99
46011	460110002	South Dakota	Brookings	23.54	17.21	17.21	17.21
46013	460130003	South Dakota	Brown	18.73	14.31	14.31	14.31
46029	460290002	South Dakota	Codington	23.67	17.81	17.81	17.81
46033	460330132	South Dakota	Custer	14.36	12.03	12.03	12.03
46071	460710001	South Dakota	Jackson	12.73	10.27	10.27	10.27
46099	460990006	South Dakota	Minnehaha	24.17	17.48	17.48	17.48
46099	460990007	South Dakota	Minnehaha	23.98	16.91	16.91	16.91
46103	461030016	South Dakota	Pennington	17.2	14.48	14.48	14.48
46103	461030020	South Dakota	Pennington	18.58	16.21	16.21	16.21
46103	461031001	South Dakota	Pennington	15.95	13.29	13.29	13.29
47009	470090011	Tennessee	Blount	32.54	18.75	18.75	18.75
47037	470370023	Tennessee	Davidson	33.5	18.05	18.05	18.05
47037	470370025	Tennessee	Davidson	30.93	16.82	16.82	16.82
47037	470370036	Tennessee	Davidson	32.71	16.15	16.15	16.15
47045	470450004	Tennessee	Dyer	31.92	17.63	17.63	17.63
47065	470650031	Tennessee	Hamilton	33.25	20.49	20.49	20.49
47065	470651011	Tennessee	Hamilton	29.74	14.88	14.88	14.88
47065	470654002	Tennessee	Hamilton	33.53	18.24	18.24	18.24
47093	470930028	Tennessee	Knox	36.66	20.46	20.46	20.46
47093	470931017	Tennessee	Knox	33.46	19.45	19.45	19.45
47099	470990002	Tennessee	Lawrence	28.48	14.91	14.91	14.91
47105	471050108	Tennessee	Loudon	32.2	19.93	19.93	19.93

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
47107	471071002	Tennessee	Mc Minn	32.73	17.70	17.70	17.70
47119	471192007	Tennessee	Maury	30.96	16.91	16.91	16.91
47125	471251009	Tennessee	Montgomery	36.3	17.88	17.88	17.88
47141	471410001	Tennessee	Putnam	32.66	16.31	16.31	16.31
47145	471450004	Tennessee	Roane	30.24	15.93	15.93	15.93
47157	471570014	Tennessee	Shelby	32.25	16.94	16.94	16.94
47157	471570038	Tennessee	Shelby	32.52	16.25	16.25	16.25
47157	471570047	Tennessee	Shelby	33.5	16.90	16.90	16.90
47157	471571004	Tennessee	Shelby	29.88	15.72	15.72	15.72
47163	471631007	Tennessee	Sullivan	31.13	18.99	18.99	18.99
47165	471650007	Tennessee	Sumner	33.66	15.20	15.20	15.20
48037	480370004	Texas	Bowie	29.42	19.29	19.29	19.29
48113	481130050	Texas	Dallas	27.44	17.73	17.73	17.73
48113	481130069	Texas	Dallas	25.7	16.73	16.73	16.73
48113	481130087	Texas	Dallas	24.21	15.08	15.08	15.08
48135	481350003	Texas	Ector	17.81	13.75	13.75	13.75
48141	481410037	Texas	El Paso	22.93	19.47	19.47	19.47
48201	482011035	Texas	Harris	30.81	21.23	21.23	19.46
48203	482030002	Texas	Harrison	25.95	17.31	17.31	17.31
48215	482150043	Texas	Hidalgo	26.42	22.24	22.24	22.24
48355	483550032	Texas	Nueces	27.55	18.66	18.66	18.66
48355	483550034	Texas	Nueces	20.74	12.40	12.40	12.40
48361	483611001	Texas	Orange	27.78	18.57	18.57	18.57
48439	484391002	Texas	Tarrant	25.34	16.26	16.26	16.26
48439	484391006	Texas	Tarrant	25.76	16.82	16.82	16.82
49003	490030003	Utah	Box Elder	33.2	27.74	27.46	27.44
49005	490050004	Utah	Cache	56.95	42.65	35.48	30.48
49011	490110004	Utah	Davis	38.95	31.35	31.35	29.56
49035	490350003	Utah	Salt Lake	47.36	33.48	32.06	27.20

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
49035	490350012	Utah	Salt Lake	50.14	36.73	35.31	30.48
49035	490351001	Utah	Salt Lake	37.73	30.29	28.87	24.04
49035	490353006	Utah	Salt Lake	47.84	34.97	33.55	28.70
49035	490353007	Utah	Salt Lake	45.38	36.10	34.68	29.80
49035	490353008	Utah	Salt Lake	30.07	25.11	23.69	19.01
49045	490450003	Utah	Tooele	30.53	26.09	26.09	25.32
49049	490490002	Utah	Utah	38.18	29.25	29.25	24.92
49049	490494001	Utah	Utah	44	33.95	33.95	29.61
49049	490495008	Utah	Utah	35.9	27.60	27.60	23.36
49049	490495010	Utah	Utah	35.93	28.13	28.13	23.86
49057	490570002	Utah	Weber	38.58	30.01	29.68	29.66
49057	490570007	Utah	Weber	33.6	26.35	26.03	26.01
49057	490571003	Utah	Weber	36.16	28.34	28.02	27.99
50001	500010002	Vermont	Addison	28.2	17.10	17.10	17.10
50001	500010003	Vermont	Addison	31.73	18.53	18.53	18.53
50003	500030004	Vermont	Bennington	26.47	15.89	15.89	15.89
50007	500070012	Vermont	Chittenden	29.84	18.90	18.90	18.90
50007	500070014	Vermont	Chittenden	30.13	21.58	21.58	21.58
50021	500210002	Vermont	Rutland	30.6	22.73	22.73	22.73
51013	510130020	Virginia	Arlington	34.18	18.75	18.75	18.75
51036	510360002	Virginia	Charles City	31.76	16.76	16.76	16.76
51041	510410003	Virginia	Chesterfield	31.25	15.30	15.30	15.30
51059	510590030	Virginia	Fairfax	34.47	18.50	18.50	18.50
51059	510591005	Virginia	Fairfax	33.72	18.14	18.14	18.14
51059	510595001	Virginia	Fairfax	33.31	19.36	19.36	19.36
51087	510870014	Virginia	Henrico	31.95	16.57	16.57	16.57
51087	510870015	Virginia	Henrico	29.18	14.29	14.29	14.29
51107	511071005	Virginia	Loudoun	34.45	19.24	19.24	19.24
51139	511390004	Virginia	Page	30.06	16.65	16.65	16.65

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
51520	515200006	Virginia	Bristol City	30.24	16.20	16.20	16.20
51650	516500004	Virginia	Hampton City	29.01	15.90	15.90	15.90
51680	516800015	Virginia	Lynchburg City	30.71	15.77	15.77	15.77
51710	517100024	Virginia	Norfolk City	29.66	16.97	16.97	16.97
51770	517700014	Virginia	Roanoke City	32.7	18.00	18.00	18.00
51775	517750010	Virginia	Salem City	34.06	19.37	19.37	19.37
53033	530330024	Washington	King	28.78	20.60	20.60	20.60
53033	530330057	Washington	King	29.16	20.74	20.74	20.74
53033	530330080	Washington	King	22.03	16.10	16.10	16.10
53053	530530029	Washington	Pierce	41.82	31.00	31.00	30.41
53061	530611007	Washington	Snohomish	34.36	26.99	26.99	26.99
53063	530630016	Washington	Spokane	29.7	19.15	19.15	19.15
53063	530630047	Washington	Spokane	29.86	18.60	18.60	18.60
54003	540030003	West Virginia	Berkeley	34.51	23.43	23.43	23.43
54009	540090005	West Virginia	Brooke	39.43	22.36	22.36	22.36
54009	540090011	West Virginia	Brooke	43.9	25.30	25.30	25.30
54011	540110006	West Virginia	Cabell	35.1	18.06	18.06	18.06
54029	540291004	West Virginia	Hancock	40.64	20.54	20.54	20.54
54033	540330003	West Virginia	Harrison	33.53	15.75	15.75	15.75
54039	540390010	West Virginia	Kanawha	34.73	16.61	16.61	16.61
54039	540390011	West Virginia	Kanawha	33.1	16.24	16.24	16.24
54039	540391005	West Virginia	Kanawha	36.98	18.25	18.25	18.25
54049	540490006	West Virginia	Marion	33.68	15.56	15.56	15.56
54051	540511002	West Virginia	Marshall	33.98	16.98	16.98	16.98
54061	540610003	West Virginia	Monongalia	35.65	14.68	14.68	14.68
54069	540690010	West Virginia	Ohio	32	16.50	16.50	16.50
54081	540810002	West Virginia	Raleigh	30.67	14.42	14.42	14.42
54089	540890001	West Virginia	Summers	31.26	14.30	14.30	14.30
54107	541071002	West Virginia	Wood	35.44	17.75	17.75	17.75

(continued)

Table 4.A-12. 24-hr Design Values (DVs) for the 2005 and 2020 Base Case and After Meeting the Current and Proposed Alternative Standard Levels: 2020 Baseline (15/35) and 2020 11/30 (continued)

FIPS	Monitor ID	State Name	County Name	2005 24-hr DV	2020 24-hr DV	2020 15/35 24-hr DV	2020 11/30 24-hr DV
55003	550030010	Wisconsin	Ashland	18.61	12.48	12.48	12.48
55009	550090005	Wisconsin	Brown	36.56	24.89	24.89	24.89
55009	550090009	Wisconsin	Brown	35.86	25.53	25.53	25.53
55025	550250047	Wisconsin	Dane	35.57	24.20	24.20	24.20
55027	550270007	Wisconsin	Dodge	31.82	21.63	21.63	21.63
55041	550410007	Wisconsin	Forest	25.26	17.13	17.13	17.13
55043	550430009	Wisconsin	Grant	34.35	24.95	24.95	24.95
55059	550590019	Wisconsin	Kenosha	32.78	22.88	22.88	22.88
55071	550710007	Wisconsin	Manitowoc	29.7	21.11	21.11	21.11
55079	550790010	Wisconsin	Milwaukee	38.67	26.12	26.12	26.12
55079	550790026	Wisconsin	Milwaukee	37.38	24.90	24.90	24.90
55079	550790043	Wisconsin	Milwaukee	39.92	26.08	26.08	26.08
55079	550790059	Wisconsin	Milwaukee	35.56	24.18	24.18	24.18
55079	550790099	Wisconsin	Milwaukee	37.78	25.75	25.75	25.75
55087	550870009	Wisconsin	Outagamie	32.87	23.37	23.37	23.37
55089	550890009	Wisconsin	Ozaukee	32.53	22.69	22.69	22.69
55109	551091002	Wisconsin	St Croix	26.66	19.57	19.57	19.57
55111	551110007	Wisconsin	Sauk	28.63	21.31	21.31	21.31
55119	551198001	Wisconsin	Taylor	25.38	18.12	18.12	18.12
55125	551250001	Wisconsin	Vilas	22.61	16.14	16.14	16.14
55133	551330027	Wisconsin	Waukesha	35.48	24.68	24.68	24.68
56005	560050877	Wyoming	Campbell	18.63	17.12	17.12	17.12
56005	560050892	Wyoming	Campbell	12.55	12.19	12.19	12.19
56005	560050899	Wyoming	Campbell	12.66	12.19	12.19	12.19
56009	560090819	Wyoming	Converse	10	9.33	9.33	9.33
56013	560131003	Wyoming	Fremont	29.8	23.81	23.81	23.81
56021	560210001	Wyoming	Laramie	11.93	10.0	10.07	10.07
56033	560330002	Wyoming	Sheridan	30.86	27.17	27.17	27.17

4.A.3 References

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CHAPTER 5
HUMAN HEALTH BENEFITS ANALYSIS APPROACH AND RESULTS

5.1 Synopsis

This chapter presents the estimated human health benefits for the proposed 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 attain alternative $PM_{2.5}$ NAAQS levels in 2020.

These benefits are relative to a 2020 baseline reflecting attainment of the current primary $PM_{2.5}$ standards (i.e., annual standard at 15 µg/m^3 and 24-hour standard of 35 µg/m^3, referred to as "15/35") that includes promulgated national regulations and illustrative emission controls to simulate attainment with 15/35. We project $PM_{2.5}$ levels in certain areas would exceed 13/35, 12/35, 11/35, and 11/30 after illustrative controls to simulate attainment with 15/35. In analyzing the current 15/35 standard (baseline), EPA determined that all counties would meet the 14/35 standard concurrently with meeting the existing 15/35 standard at no additional cost. Consequently, there are no incremental costs or benefits for 14/35, and no need to present an analysis of 14/35. Table 5-1 summarizes the total monetized benefits of these alternative $PM_{2.5}$ standards in 2020. These estimates reflect the sum of the estimated $PM_{2.5}$ mortality impacts identified and the value of all morbidity impacts.

Table 5-1. Estimated Monetized Benefits of the Proposed and Alternative Combinations of $PM_{2.5}$ Standards in 2020, Incremental to Attainment of 15/35 (millions of 2006$)[a]

Benefits Estimate	13 µg/m^3 Annual & 35 µg/m^3 24-Hour	12 µg/m^3 Annual & 35 µg/m^3 24-Hour	11 µg/m^3 Annual & 35 µg/m^3 24-Hour	11 µg/m^3 Annual & 30 µg/m^3 24-Hour
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	$88 + B	$2,300 +B	$9,2000+B	$14,000 +B
7% discount rate	$79 + B	$2,100 +B	$8,3000 +B	$13,000 +B
Economic value of avoided $PM_{2.5}$-related morbidities and premature deaths using $PM_{2.5}$ mortality estimate from Laden et al. (2006)				
3% discount rate	$220 + B	$5,900 +B	$23,000 +B	$36,000 +B
7% discount rate	$200 + B	$5,400+B	$21,000 +B	$33,000 +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 and welfare benefits.

For annual standards at 12 µg/m^3 and 13 µg/m^3, the majority of benefits (i.e., 70% and 98%, respectively) occur in California because this highly populated area is where the most air quality improvement beyond 15/35 is needed to reach these levels. In addition, several recent rules such as the Mercury and Air Toxics Standard (MATS) and the Cross-State Air Pollution Rule (CSAPR) will have substantially reduced $PM_{2.5}$ levels by 2020 in the East, thus few additional controls would be needed to reach 12/35 or 13/35 in the East.

In general, we have greater confidence in risk estimates based on $PM_{2.5}$ concentrations where the bulk of the data reside and somewhat less confidence where data density is lower. As noted in the preamble to the proposed rule, the range from the 25th to 10th percentiles of the air quality data used the epidemiology studies is a reasonable range below which we have appreciably less confidence in 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 used to estimate mortality benefits.

In addition to $PM_{2.5}$ benefits, implementation of emissions controls to attain 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 simulate 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 benefits from improvements in welfare effects, such as ecosystem effects and visibility. We describe the unquantified health benefits in this chapter and the unquantified welfare benefits in Chapter 6.

5.2 Overview

This chapter contains a subset of the estimated health benefits of the proposed and alternative $PM_{2.5}$ standards in 2020 that 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* (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 main 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. In order to identify the most meaningful human health and environmental co-benefits, 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 Integrated Science Assessment (ISA). This 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, some effects that were identified in previous lists of unquantified benefits in other RIAs have been dropped (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.

As described in Chapter 1, 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 control strategies for different sources, incremental to implementation of existing regulations and controls needed to attain current standards. In short, NAAQS RIAs hypothesize, but do not predict, the control 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, 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.

Table 5-2. Human Health Effects of Pollutants Potentially Affected by Attainment of 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
Reduced incidence of morbidity from exposure to $PM_{2.5}$	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)	—[4]	—[4]	Section 5.6
	Emergency department visits for cardiovascular effects (all ages)	—[4]	—[4]	Section 5.6
	Strokes and cerebrovascular disease (age 50–79)	—[4]	—[4]	Section 5.6
	Other cardiovascular effects (e.g., other ages)	—	—	PM ISA[2]
	Other respiratory effects (e.g., pulmonary function, non-asthma ER visits, non-bronchitis chronic diseases, other ages and populations)	—	—	PM ISA[2]
	Reproductive and developmental effects (e.g., low birth weight, pre-term births, etc)	—	—	PM ISA[2,3]
	Cancer, mutagenicity, and genotoxicity effects	—	—	PM ISA[2,3]

(continued)

Table 5-2. Human Health Effects of Pollutants Potentially Affected by Attainment of 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 CD, Draft Ozone ISA[1]
	Premature mortality based on long-term study estimates (age 30–99)	—	—	Ozone CD, Draft Ozone ISA[1]
	Hospital admissions—respiratory causes (age > 65)	—	—	Ozone CD, Draft ISA[1]
	Hospital admissions—respiratory causes (age <2)	—	—	Ozone CD, Draft ISA[1]
	Emergency department visits for asthma (all ages)	—	—	Ozone CD, Draft ISA[1]
	Minor restricted-activity days (age 18–65)	—	—	Ozone CD, Draft ISA[1]
	School absence days (age 5–17)	—	—	Ozone CD, Draft ISA[1]
	Decreased outdoor worker productivity (age 18–65)	—	—	Ozone CD, Draft ISA[1]
	Other respiratory effects (e.g., premature aging of lungs)	—	—	Ozone CD, Draft ISA[2]
	Cardiovascular and nervous system effects	—	—	Ozone CD, Draft ISA[3]
	Reproductive and developmental effects	—	—	Ozone CD, Draft ISA[3]
Reduced incidence of morbidity from exposure to NO_2	Asthma hospital admissions (all ages)	—	—	NO_2 ISA[1]
	Chronic lung disease hospital admissions (age > 65)	—	—	NO_2 ISA[1]
	Respiratory emergency department visits (all ages)	—	—	NO_2 ISA[1]
	Asthma exacerbation (asthmatics age 4–18)	—	—	NO_2 ISA[1]
	Acute respiratory symptoms (age 7–14)	—	—	NO_2 ISA[1]
	Premature mortality	—	—	NO_2 ISA[2,3]
	Other respiratory effects (e.g., airway hyperresponsiveness and inflammation, lung function, other ages and populations)	—	—	NO_2 ISA[2,3]

(continued)

Table 5-2. Human Health Effects of Pollutants Potentially Affected by Attainment of 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[1]
	Asthma emergency department visits (all ages)	—	—	SO_2 ISA[1]
	Asthma exacerbation (asthmatics age 4–12)	—	—	SO_2 ISA[1]
	Acute respiratory symptoms (age 7–14)	—	—	SO_2 ISA[1]
	Premature mortality	—	—	SO_2 ISA[2,3]
	Other respiratory effects (e.g., airway hyperresponsiveness and inflammation, lung function, other ages and populations)	—	—	SO_2 ISA[2,3]
Reduced incidence of morbidity from exposure to methylmercury (through role of sulfate in methylation)	Neurologic effects—IQ loss	—	—	IRIS; NRC, 2000[1]
	Other neurologic effects (e.g., developmental delays, memory, behavior)	—	—	IRIS; NRC, 2000[2]
	Cardiovascular effects	—	—	IRIS; NRC, 2000[2,3]
	Genotoxic, immunologic, and other toxic effects	—	—	IRIS; NRC, 2000[2,3]

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

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

[3] 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.

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

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 most recent major benefits analysis, documented in the benefits chapter of the RIA accompanying the final MATS(U.S. EPA, 2011d). Below we note the aspects of this analysis that differ from the MATS RIA:

1. *Incorporation of the newest American Cancer Society (ACS) mortality study*. In 2009, the Health Effects Institute published an extended analysis of the ACS cohort (Krewski et al., 2009). Compared to the study it replaces (Pope et al., 2002), this new analysis incorporates a number of methodological improvements that we describe in detail below. The all-cause $PM_{2.5}$ mortality risk estimate drawn from Krewski et al.

(2009) is identical to the Pope et al. (2002) risk estimate applied in recent EPA analyses of long-term $PM_{2.5}$ mortality but has narrower confidence intervals.

2. *Updated health endpoints*. We have moved the quantification of chronic bronchitis from our main analysis to a sensitivity analysis. This change is consistent with the findings of the Integrated Science Assessment (ISA) that the evidence for an association between long-term exposure to $PM_{2.5}$ and respiratory effects is more tenuous (U.S. EPA, 2009). We also add two new sensitivity analyses, including an assessment of PM-related cerebrovascular disease and cardiovascular emergency department visits. The incorporation of these two new endpoints follows the findings of the PM ISA that recent studies have strengthened the relationship between $PM_{2.5}$ exposure and cardiovascular outcomes (U.S. EPA, 2009b).

3. *Incorporation of new morbidity studies*. Since the publication of the 2004 *Criteria Document for Particulate Matter* (U.S. EPA, 2004) the epidemiological literature has produced a significant number of new studies examining the association between short-term $PM_{2.5}$ exposure and acute myocardial infarctions, respiratory and cardiovascular hospitalizations, respiratory emergency department visits, acute respiratory symptoms and exacerbation of asthma. Upon careful evaluation of this new literature we have incorporated new studies into our health impact assessment; in many cases we have replaced older single-city time-series studies with newer multi-city time-series analyses.

4. *Updated hospital cost-of-illness (COI), including median wage data*. In previous versions of BenMAP, estimates of hospital charges and lengths of hospital stays were based on discharge statistics provided by the Agency for Healthcare Research and Quality's Healthcare Utilization Project National Inpatient Sample (NIS) database for 2000 (AHRQ, 2000). The newest version of BenMAP (version 4.0.51) used in this analysis updated this information to use the 2007 database. The data source for the updated median annual income is the 2007 American Community Survey (ACS, 2007).

5. *Expanded uncertainty assessment*. We have incorporated a more comprehensive assessment of the various uncertain parameters and assumptions within the benefits analysis.

While the list above identifies the major changes implemented since the MATS RIA, EPA has 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 removed the threshold assumption 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 main benefits estimates, while still including a range of sensitivity estimates based on 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.

5.4 Human Health Benefits Analysis Methods

We follow a “damage-function” approach in calculating total benefits of the modeled changes in environmental quality. This approach estimates changes in individual health and welfare 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 and welfare 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 health and welfare 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 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 *B*enefits *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). 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[1] 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:

$$\Delta y = y_o \cdot \left(e^{\beta \cdot \Delta x} - 1\right) \cdot Pop \qquad (5\text{-}1)$$

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.

[1] 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.

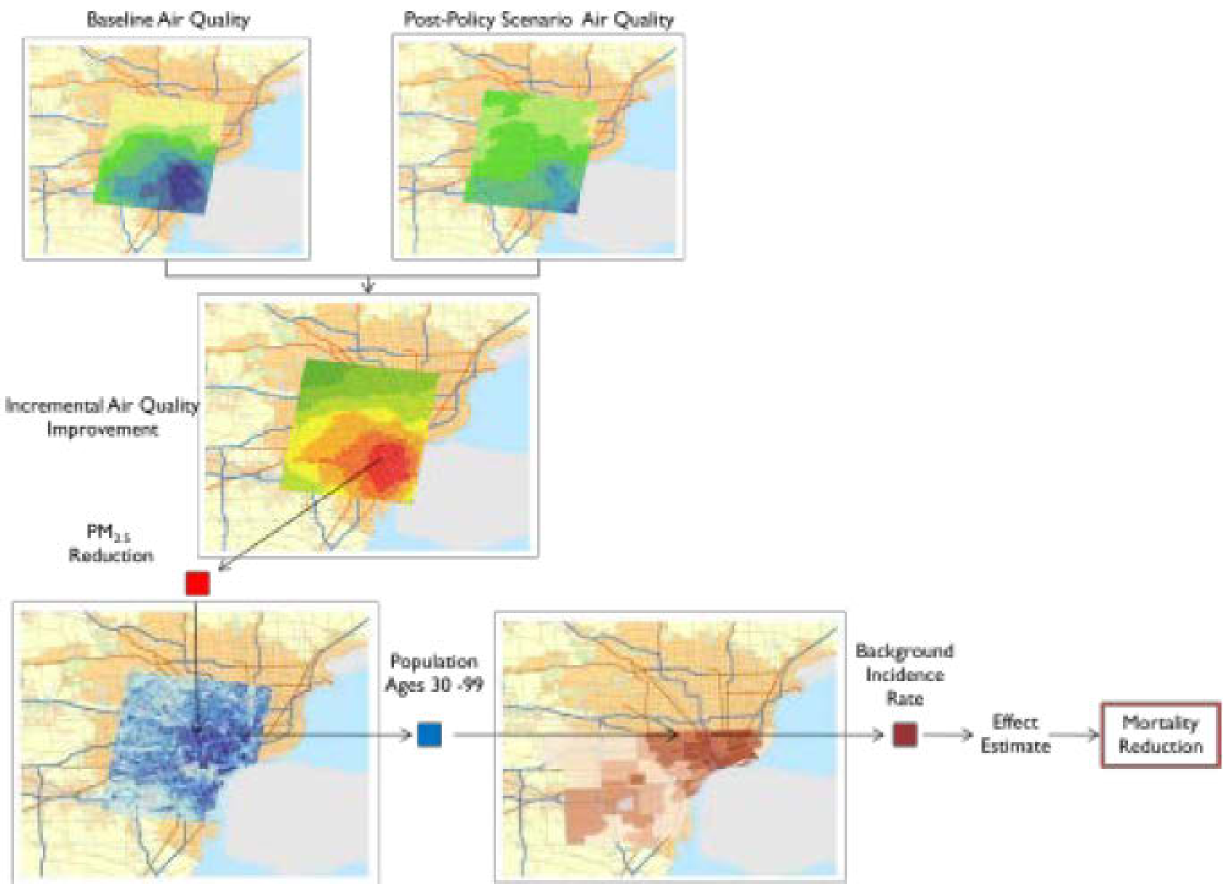

Figure 5-1. Illustration of BenMAP Approach

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. However, epidemiological studies generally provide estimates of the relative risks of a particular health effect avoided due to a reduction in air pollution. A convenient way to use these data in a consistent framework is to convert probabilities to units of avoided statistical incidences. 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 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.

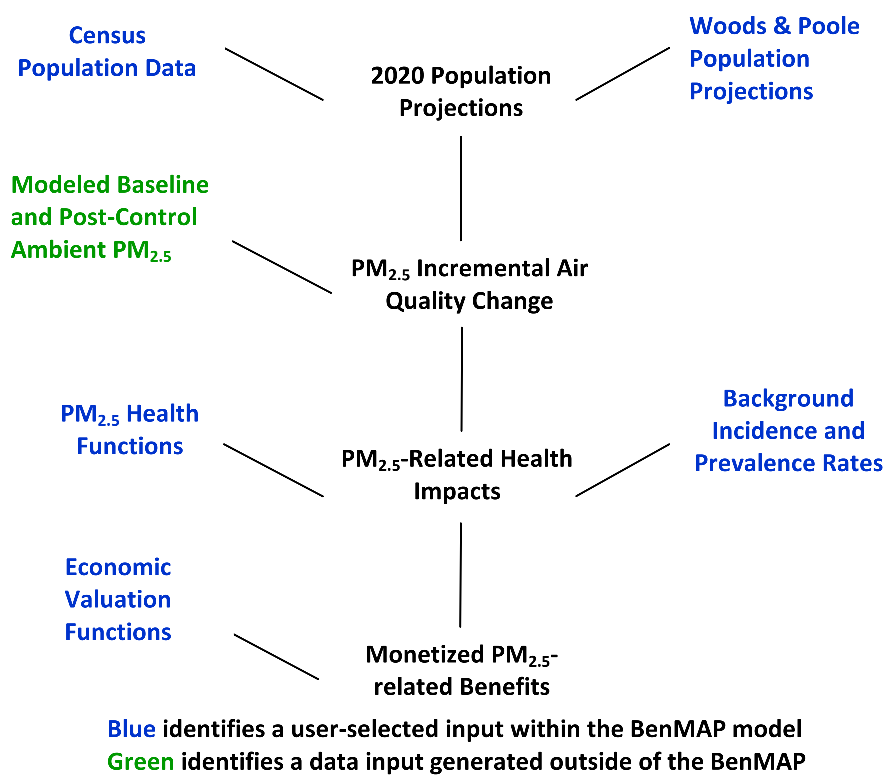

Figure 5-2. Data Inputs and Outputs for the BenMAP Model

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 EPA's approach, the National Research Council (NRC) (2002, 2008), which is part of the National Academies of Science, concluded that 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, 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.

To characterize uncertainty and variability, we follow an approach that combines elements from two recent EPA analyses (U.S. EPA, 2010b; U.S. EPA, 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. While data limitations prevent us from treating each source of uncertainty quantitatively and from reaching a full-probabilistic simulation of our results, 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 characterize avoided $PM_{2.5}$-related deaths relative the the 10^{th} and 25^{th} percentiles of the 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 Laden et al. (2006) 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, EPA conducted an expert elicitation in 2006 (Roman et al., 2008; IEc, 2006).[2] 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, 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 EPA's independent Science Advisory Board raised concerns about this presentation (U.S. EPA-SAB, 2008). Therefore, we began to present the cohort-based studies (Krewski et al., 2009; Laden et al., 2006) 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

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

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.

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 four such analyses for the proposed 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 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. Compared the valuation of hospitalizations and work loss days using the 2000 AHRQ database to the 2007 database.

5.5.5 Distributional Assessment

In the Appendix to this chapter, we characterize the distribution of $PM_{2.5}$-related benefits based on the geographic distribution of race and education in areas where the selected control strategies would reduce $PM_{2.5}$ concentrations. In this assessment, we aim to answer two key questions:

1. What is 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 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?[3]

This assessment is generally consistent with the distributional assessments performed in support of the CSAPR (EPA, 2011c) and the MATS (EPA, 2011c), with one key difference. The environmental justice analyses accompanying the CSAPR and MATS RIAs applied CMAQ-modeled $PM_{2.5}$ predictions that represent the change in air quality after the implementation of each rule. By contrast, this RIA aims 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 EPA modeled here. Alternative attainment 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 et al., 2012b).

5.5.6 Influence Analysis—Quantitative Assessment of Uncertainty

In the past few years, EPA has initiated several projects to improve the characterization of uncertainty for benefits analysis. In particular, 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. While 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.

[3] 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.

- 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 attainment strategies for each of the alternative 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 EPA's independent Science Advisory Board (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 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 down to the lowest air quality levels modeled in this analysis. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5,}$ including both regions that are in attainment with the fine particle standard and those that do not meet the standard down to the lowest modeled concentrations.

3. 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 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 EPA from quantifying or monetizing the benefits from several important health benefit categories in this RIA, including 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. 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 5C. Because this approach reflects a new application for regulatory benefits analysis, we request comments on this general approach to characterizing uncertainty as well as the specific uncertainty assessments.

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 (U.S. EPA, 2011d).

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 projections based on economic forecasting models developed by Woods and Poole, Inc. (Woods and Poole, 2007). The Woods and Poole (WP) database contains county-level projections of population by age, sex, and race out to 2030. 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 2020 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 have projected mortality rates such that future mortality rates are consistent with our projections of population growth (Abt Associates, 2011).

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 also 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.

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

Endpoint	Parameter	Rates: Value	Rates: Source
Mortality	Daily or annual mortality rate projected to 2015[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 [d]	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 dyspnea	 • 0.076 • 0.067 • 0.037	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 [c]	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)
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);

(continued)

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

Endpoint	Parameter	Rates	
		Value	Source
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] Lower respiratory symptoms are defined as two or more of the following: cough, chest pain, phlegm, and wheeze.

[d] 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.

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 5-4. 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 EPA's Integrated Science Assessments (which replace previous Criteria Documents), with input and advice from the EPA Science Advisory Board—Health Effects Subcommittee (SAB-HES), a scientific review panel specifically established to provide advice on the use of the scientific literature in developing benefits analyses for air pollution regulations. 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 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, 2011). 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. 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, 2011).

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.

The specific studies from which effect estimates for the main 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 incorporated into the central benefits estimate. 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 5-5. 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.
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.

(continued)

Table 5-6. Health Endpoints and Epidemiological Studies Used to Quantify Health Impacts in the Main 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)	>29 years	RR = 1.06 (1.04 – 1.06) per 10 $\mu g/m^3$
	Laden et al. (2006)	>24 years	RR = 1.16 (1.07 – 1.26) per 10 $\mu g/m^3$
Premature mortality, total exposures	$PM_{2.5}$ Expert Elicitation (Roman et al., 2008)	>24 years	Varies by expert
Premature mortality—all-cause	Woodruff et al. (1997)	Infant (<1 year)	OR = 1.04 (1.02 – 1.07) per 10 $\mu g/m^3$
Chronic Illness			
Nonfatal heart attacks	Peters et al. (2001)	Adults (>18 years)	OR = 1.62 (1.13 – 2.34) per 20 $\mu g/m^3$
	Pooled estimate:		
	Pope et al. (2006)		β = 0.00481 (0.00199)
	Sullivan et al. (2005)		β = 0.00198 (0.00224)
	Zanobetti et al. (2009)		β = 0.00225 (0.000591)
	Zanobetti and Schwartz (2006)		β = 0.0053 (0.00221)
Hospital Admissions			
Respiratory	Zanobetti et al. (2009)—ICD 460-519 (All respiratory)	>64 years	β=0.00207 (0.00446)
	Moolgavkar (2000)—ICD 490–496 (Chronic lung disease)	18–64 years	1.02 (1.01—1.03) per 36 $\mu g/m^3$
	Babin et al. (2007)—ICD 493 (asthma)	<19	β=0.002 (0.004337)
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 $\mu g/m^3$
Asthma-related ER visits	*Pooled estimate:* Mar et al. (2010)	All ages	RR = 1.04 (1.01 – 1.07) per 7 $\mu g/m^3$
	Slaughter et al. (2005)		RR = 1.03 (0.98 – 1.09) per 10 $\mu g/m^3$

(continued)

Endpoint	Study	Study Population	Risk Estimate (95th Percentile Confidence Interval)[a]
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
Other Health Endpoints (continued)			
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) per 30 µg/m^3
			OR = 1.06 (1.01 – 1.11) per 30 µg/m^3
			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 10 µg/m^3
			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)

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

[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 Science Advisory Board Health Effects Subcommittee (SAB-HES), we extended the applied population to 6 to 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

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

5.6.3.1 $PM_{2.5}$ Premature Mortality Effect Coefficients

Both long- and short-term exposures to ambient levels of $PM_{2.5}$ air pollution have been associated with increased risk of premature mortality. 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.

Although a number of uncertainties remain to be addressed by continued research (NRC, 2002), a substantial body of published scientific literature documents the correlation between elevated $PM_{2.5}$ concentrations and increased mortality rates (U.S. EPA, 2009b). 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. Researchers have found statistically significant associations between $PM_{2.5}$ and premature mortality using both types of studies. In general, the effect estimates based on the cohort studies are larger than those derived from time-series studies. 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). This section discusses some of the issues surrounding the estimation of $PM_{2.5}$-related premature mortality. To demonstrate the sensitivity of the benefits estimates to various concentration-response estimates in the epidemiological literature, we present benefits estimates using several relative risk estimates from the largest long-term epidemiological studies as well as a U.S. EPA-sponsored expert elicitation (Roman et al. 2008). The epidemiological studies from which these estimates are drawn are described below. The $PM_{2.5}$ expert elicitation and the derivation of effect estimates from the expert elicitation results are described in the 2006 $PM_{2.5}$ NAAQS RIA (U.S. EPA, 2006a) and Roman et al. (2008). In the interest of brevity, we do not repeat those details here. [4]

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[4] In summary, the goal of the 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 questions requiring qualitative responses probed experts' beliefs concerning key evidence and critical sources of uncertainty and enabled them to establish a conceptual basis supporting their quantitative judgments. The results of the full-scale study consist of twelve individual distributions for the coefficient or slope of the C-R function relating changes in annual average $PM_{2.5}$ exposures

Over a dozen epidemiological studies demonstrate significant associations between various measures of long-term exposure to $PM_{2.5}$ and mortality, beginning with Lave and Seskin (1977). Most of the published studies found positive (but not always statistically significant) associations with available PM indices such as total suspended particles (TSP). However, exploration of alternative model specifications sometimes raised questions about causal relationships (e.g., Lipfert et al., 1989). These early "ecological cross-sectional" studies (Lave and Seskin, 1977; Ozkaynak and Thurston, 1987) were criticized for a number of methodological limitations, particularly for inadequate control at the individual level for variables that are potentially important in causing mortality, such as wealth, smoking, and diet.

Over the last two decades, several studies using "prospective cohort" designs have been published that are consistent with the earlier body of literature. These "prospective cohort" studies reflect a significant improvement over the earlier work because they include individual level information with respect to health status and residence. Two prospective cohort groups, often referred to as the Harvard "Six Cities Study" (Dockery et al., 1993; Laden et al., 2006) 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. A third major data set comes from the California-based 7th Day Adventist Study (e.g., Abbey et al., 1999), which reported associations between long-term $PM_{2.5}$ exposure and mortality in men. Results from this cohort, however, have been inconsistent, the air quality results are not geographically representative of most of the United States, and the lifestyle of the population is not reflective of much of the U.S. population. Analysis is also available for a cohort of adult male veterans diagnosed with hypertension (Lipfert et al., 2000, 2003, 2006). The characteristics of this group also differ from the cohorts in the Six Cities and ACS studies as well as the 7th Day Adventist study with respect to income, race, health status, and smoking status. Unlike previous long-term analyses, this study found some associations between mortality and ozone but found inconsistent results for PM indicators.

Given their consistent results and broad geographic coverage, and importance in informing the NAAQS development process, the Six Cities and ACS data have been particularly important in benefits analyses. 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

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to annual, adult all-cause mortality. The results have not been combined in order to preserve the breadth and diversity of opinion on the expert panel. Roman et al (2006).

extensive reexamination and reanalysis by an independent team of scientific experts commissioned by the Health Effect Institute (HEI) (Krewski et al., 2000). The final results of the reanalysis were then independently peer reviewed by a Special Panel of the HEI Health Review Committee. The results of these reanalyses confirmed and expanded the conclusions of the original investigators. While the HEI reexamination lends credibility to the original studies, it also highlights sensitivities concerning the relative impact of various pollutants, such as SO_2, the potential role of education in mediating the association between pollution and mortality, and the influence of spatial correlation modeling. Further confirmation and extension of the findings of the 1993 Six Cities Study and the 1995 ACS study were recently completed using more recent air quality and a longer follow-up period for the ACS cohort was published over the past several years (Pope et al., 2002, 2004; Laden et al., 2006; Krewski et al., 2009). The follow up to the Harvard Six Cities Study both confirmed the effect size from the first analysis and provided additional confirmation that reductions in $PM_{2.5}$ are associated with reductions in the risk of premature death. This additional evidence stems from the observed reductions in $PM_{2.5}$ in each city during the extended follow-up period. Laden et al. (2006) found that mortality rates consistently went down at a rate proportional to the observed reductions in $PM_{2.5}$.

A number of additional analyses have been conducted on the ACS cohort data (Pope et al., 2009). These studies have continued to find a strong significant relationship between $PM_{2.5}$ and mortality outcomes and life expectancy. Specifically, much of the recent research has suggested a stronger relationship between cardiovascular mortality and lung cancer mortality with $PM_{2.5}$, and a less significant relationship between respiratory-related mortality and $PM_{2.5}$. 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 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 EPA benefits analyses.

In developing and improving the methods for estimating and valuing the potential reductions in mortality risk over the years, EPA consulted with the Health Effects Subcommittee of the Science Advisory Board (SAB-HES). That panel recommended using long-term prospective cohort studies in estimating mortality risk reduction (U.S. EPA-SAB, 1999). This recommendation has been confirmed by a report from the National Research Council, which 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). 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). More specifically, the SAB recommended emphasis on the ACS study because it includes a much larger sample size and longer exposure interval and covers more locations (e.g., 50 cities compared to the Six Cities Study) than other studies of its kind. Because of the refinements in the extended follow-up analysis, the SAB-HES recommended using the Pope et al. (2002) study as the basis for the main mortality estimate for adults and suggests that alternate estimates of mortality generated using other cohort and time-series studies could be included as part of the sensitivity analysis (U.S. EPA-SAB, 2004a). 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 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), which was twice reviewed by Clean Air Scientific Review Committee (CASAC) (U.S. EPA-SAB, 2009b, 2009c), 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 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).

As both the ACS and Six Cities studies have inherent strengths and weaknesses and the expert elicitation results encompass within their range the estimates from both the earlier ACS Study (Pope et al. 2002) and from the Laden et al. (2006) study, we present benefits estimates using RR estimates from the Krewski et al. (2009) (RR=1.06, 95% confidence intervals 1.04–1.08 per 10μg/m^3 increase in $PM_{2.5}$) and Laden et al. (2006; RR=1.16, 95% confidence intervals 1.07–1.26 per 10 μg/m^3 increase in $PM_{2.5}$) studies. For the ACS Study (Krewski et al., 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 co-variates, consistent with the *Quantitative Health Risk Assessment for Particulate Matter* (U.S. EPA, 2010b). Unlike the Pope et al. (2002) study, Krewski et al. (2009) do not report a risk estimate based on an average between the initial monitoring period (1979–1983) and the follow-up period (1999–2000). When considering each time period from which we could select risk coefficients, we elected to use the estimate based on the 1999–2000 air quality monitoring period because it reflected more recent population exposures, a larger number of urban areas (116 vs. 58) and a larger population cohort (488,000 vs. 343,000). The relative risk estimate (1.06 per 10µg/m^3 increase in $PM_{2.5}$) is identical to the risk estimate drawn from the Pope et al. (2002) study, though the confidence interval around the Krewski et al. (2009) risk estimate is tighter.

Presenting results using both ACS and Six Cities is consistent with other recent RIAs (e.g., U.S. EPA, 2006a, 2010c, 2011c, 2011d). EPA's independent SAB also supported using these two cohorts for benefits, concluding that "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).

In addition to the ACS and Six Cities cohorts described above, 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 thus not summarized here). We describe the newer multi-state studies below.[5,6] Table 5-8 provides the effect estimates from each of these cohort studies (new and included in the PM ISA) for all-cause, cardiovascular, cardiopulmonary, and ischemic heart disease (IHD) mortality as well as the lowest measured air quality level (LML) in the study.

Puett et al. (2009) examined the risk of all-cause mortality and fatal congestive heart disease among a cohort of about 66,000 female nurses in 13 northeastern and midwestern states (i.e., the Nurses' Health Study cohort) resulting from long-term exposure to $PM_{2.5}$. Consistent with findings from previous cohort studies, the researchers found significant associations between long-term $PM_{2.5}$ exposure and all-cause mortality and fatal coronary heart disease. Puett et al. (2011) examined the risk of all-cause and cardiovascular mortality among a cohort of 17,000 male health professionals with high socioeconomic status in 13 northeastern and midwestern states. The researchers found no association between long-term $PM_{2.5}$

[5] It is important to note that these newer studies 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.

[6] In this chapter, we only describe multi-state cohort studies. There are additional cohorts that focus on single cities, such as Gan et al. (2012) that we have not included. In Appendix 5B, we provide additional information regarding cohort studies in California, which is the only state for which we identified single state cohorts.

exposure and mortality, concluding that additional research is needed to determine whether men with higher socioeconomic status are less susceptible to cardiovascular outcomes associated with long-term particle exposure.

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

Study	Cohort (age)	LML ($\mu g/m^3$)	Hazard Ratios per 10 $\mu g/m^3$ Change in $PM_{2.5}$ (95th percentile confidence intervals)			
			All Causes	Cardiovascular	Cardiopulmonary	IHD
Pope et al. (2002)	ACS (age >30)	7.5	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	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	1.15 (1.05–1.25)	N/A	N/A	N/A
Miller et al. (2007)[b]	WHI (age 50–79)	3.4	N/A	1.76 (1.25–2.47)	N/A	2.21 (1.17–4.16)
Eftim et al. (2008)	Medicare (age > 65)	6	1.21 (1.15–1.27)	N/A	N/A	N/A
Zeger et al. (2008)[c]	Medicare (age > 65)	<9.8	1.068 (1.049–1.087)	N/A	N/A	N/A
Krewski et al. (2009)[d]	ACS (age >30)	5.8	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	1.26 (1.02–1.54)	N/A	N/A	2.02 (1.07–3.78)
Crouse et al. (2011)[d, f]	Canadian census	1.9	1.06 (1.01–1.10)	N/A	N/A	N/A
Puett et al. (2011)[e]	Health Professionals (age 40–75)	<14.4	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	1.14 (1.07–1.22)	1.26 (1.14–1.40)	N/A	N/A

[a] 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] Men with high socioeconomic status only.

[f] Canadian population.

Crouse et al. (2012) found elevated risks of non-accidental and ischemic heart disease mortality associated with long-term exposure to $PM_{2.5}$ from the period of 1991 to 2001 among a cohort of Canadian adults aged 25 and older. This study used a combination of monitored air quality data and remote-sensing (i.e., satellite) data to assign $PM_{2.5}$ concentrations to the population cohort. Notably, the median annual mean $PM_{2.5}$ levels observed, or modeled, in this study were 7.4 µg/m^3 and the minimum value was 1.9 µg/m^3. The authors note that these air quality values are significantly lower than those observed in either the ACS or Six Cities studies, which provides further evidence that PM effects may occur at very low annual mean $PM_{2.5}$ levels.

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.

As further described in the mortality valuation discussion, we assume that there is a “cessation” lag between PM exposures and the total realization of changes in health effects. While the structure of the lag is uncertain, most of the premature deaths occur within the first couple years after the change in exposure (U.S. EPA, 2004c; Schwartz et al, 2008). Changes in the cessation lag assumptions do not change the total number of estimated deaths but rather the timing of those deaths.

In addition to the adult mortality studies described above, several studies provide evidence for an association between PM exposure and respiratory inflammation and infection leading to premature mortality in children under 5 years of age. Specifically, 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. Recently, a study by Chay and Greenstone (2003) found that reductions in TSP caused by the recession of 1981– 1982 were related to 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 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 recommends that 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). A more recent study by Woodruff et al. (2006) continues to find associations between $PM_{2.5}$ and infant mortality. The study also found the most significant relationships with respiratory-related causes of death. We have not yet sought comment from the SAB on this more recent study and as such continue to rely on the earlier 1997 analysis.

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; 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, based on Rosamond et al. (1999), we apply an adjustment factor of 0.93 in the concentration-response function to reflect the probability of survival 28 days after the heart attack.

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) and Slaughter et al. (2005). Both studies examine populations 0 to 99 in Washington State. 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.[7] 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.

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 study by Zanobetti et al. (2009) provides an effect estimate for total respiratory hospital admissions (defined as ICD codes 460–519). 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.

[7] 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.

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,[8] 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). 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).

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. Incidences of upper respiratory symptoms in asthmatic children aged 9 to 11 are estimated using an effect estimate developed from Pope et al. (1991).

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.

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

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

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 main estimate (U.S. EPA-SAB, 2004a). While 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 main 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), which 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 a health impact assessment would be to match the baseline incidence rate correctly.

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. 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 treat these ED visits as a sensitivity analysis.

5.6.4 Unquantified Human Health Benefits

Implementing the illustrative control strategy described in Chapter 4 would reduce emissions of directly emitted particles, SO_2, and NO_x. 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 ozone exposure, SO_2 exposure, NO_2 exposure or methylmercury exposure due to the absence of air quality modeling data for these pollutants in this analysis. Although we applied the rollback method to simulate the impact of attaining alternative combination of standard levels 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 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). The extent to which ozone, SO_2, NOx, and/or methylmercury would be reduced would depend on the specific control strategy used to reduce $PM_{2.5}$ in a given area.

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). When adequate data and resources are available, 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 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. These cost-of-illness (COI) estimates generally 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 out to 2020 using projections provided by Standard and Poor's. 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. Because real income has grown since the studies were conducted, people's willingness to pay for reductions in the risk of premature death and disease likely has grown as well. We do not have data to adjust the COI estimates for projections of medical costs in the future, which leads to an inherent though unavoidable inconsistency between COI- and WTP-based estimates. For these two reasons, these cost-of-illness estimates may underestimate the economic value of avoided health impacts. The discussion below provides additional details on valuing $PM_{2.5}$-related related endpoints.

Table 5-9. Unit Values for Economic Valuation of Health Endpoints (2006$)[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)	$6,300,000	$8,900,000	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 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). Lost earnings: Cropper and Krupnick (1990). Present discounted value of 5 years of lost earnings in 2000$: age of onset: at 3% / at 7% 25–44: $9,000 / $8,000 45–54: $13,000 / $12,000 55–65: $77,000 / $69,000 Direct medical expenses (2000$): An average of: 1. Wittels et al. (1990) ($100,000—no discounting) 2. Russell et al. (1998), 5-year period ($22,000 at 3% discount rate; $21,000at 7% discount rate)
3% discount rate			
Age 0–24	$85,000	$85,000	
Age 25–44	$96,000	$96,000	
Age 45–54	$100,000	$100,000	
Age 55–65	$180,000	$180,000	
Age 66 and over	$85,000	$85,000	
7% discount rate			
Age 0–24	$84,000	$84,000	
Age 25–44	$94,000	$94,000	
Age 45–54	$98,000	$98,000	
Age 55–64	$170,000	$170,000	
Age 65 and over	$84,000	$84,000	
Hospital Admissions			
Chronic Lung Disease (18-64)	$19,000	$19,000	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).

(continued)

Table 5-9. Unit Values for Economic Valuation of Health Endpoints (2006$)[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 (continued)			
Asthma Admissions (0-64)	$14,000	$14,000	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	$37,000	$37,000	
Age 65-99	$35,000	$35,000	
All respiratory (ages 65+)	$32,000	$32,000	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	$370	$370	No distributional information available. Simple average of two unit COI values (2000$): (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 (2006$)[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)	$25	$31	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)	$16	$19	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	$43	$53	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 (2006$)[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	$360	$440	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 = $140)	Variable (U.S. median = $140)	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)	$51	$63	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, 2011).

5.6.6.1 Mortality Valuation

Following the advice of the EEAC of the SAB, EPA currently uses the value of statistical life (VSL) approach in calculating the main 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 the Science Advisory Board (SAB) or other peer-review group.

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 EPA and the SAB on combining estimates from the various data sources. In addition, the Agency consulted several times with the Science Advisory Board Environmental Economics Advisory Committee (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)[1] while the Agency continues its efforts to update its guidance on this issue. This approach calculates a mean value across VSL estimates

[1] 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.

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 $6.3 million (2000$).[2] 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 EPA's independent Science Advisory Board). 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. 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, we use a single VSL for all reductions in mortality risk.

Although there are several differences between the labor market studies 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, 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.

The SAB-EEAC has reviewed many potential VSL adjustments and the state of the economics literature. The SAB-EEAC advised 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 main estimate of the

[2] In this analysis, we adjust the VSL to account for a different currency year (2006$) and to account for income growth to 2020. After applying these adjustments to the $6.3 million value, the VSL is $8.9M.

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, 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 5B of the 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%.[3] 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 EPA’s *Guidelines for Preparing Economic Analyses* (U.S. EPA, 2010e).

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

*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 of the CSAPR. In addition, in prior analyses, EPA has identified valuation of mortality-related benefits as the largest contributor to the range of uncertainty in monetized

[3] 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.

benefits (Mansfield et al., 2009).[4] 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 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.

[4] 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.

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. 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. 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). This worker must have either a higher workplace risk than the average worker, a lower risk tolerance than the average worker, or both. However, the risk estimate used in hedonic studies is generally based on average risk, so the VSL may be upwardly biased 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.

5.6.6.2 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):

Table 5-11. Alternative Direct Medical Cost of Illness Estimates for Nonfatal Heart Attacks[a]

Study	Direct Medical Costs (2006$)	Over an x-Year Period, for x =
Wittels et al. (1990)	$140,000[b]	5
Russell et al. (1998)	$30,000[c]	5
Eisenstein et al. (2001)	$64,000[c]	10
Russell et al. (1998)	$38,000[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, 2011).

[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.

- 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.

- Russell et al. (1998) estimated first-year direct medical costs of treating nonfatal myocardial infarction of $16,000 (in rounded 1995$) and $1,100 annually thereafter for a 10-year period.

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 $66,000, 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 2006$)[a]

Age Group	Opportunity Cost	Medical Cost [b]	Total Cost
0–24	$0	$85,000	$85,000
25–44	$11,000[c]	$85,000	$96,000
45–54	$16,000[c]	$85,000	$100,000
55–65	$91,000[c]	$85,000	$180,000
> 65	$0	$85,000	$85,000

[a] All estimates rounded to two significant digits. Unrounded estimates in 2000$ are available in appendix J of the BenMAP user manual (Abt Associates, 2011).

[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.

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 2006$ using the CPI-U "all items" (Abt Associates, 2011). The resulting national average lost daily wage is $134. The total cost-of-illness estimate for an ICD code-specific hospital stay lasting *n* days, then, was the mean hospital charge plus $134 multiplied by *n.* In general, the mean length of stay has decreased since the 2000 database used in previous version of BenMAP while the mean hospital charge has increased. We provide the rounded unit values in 2000$ 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 (2000$)	Mean Length of Stay (days)	Total Cost of Illness (unit value in 2000$)
HA, All Cardiovascular	390–429	18	64	$27,000	4.1	$27,000
HA, All Cardiovascular	390–429	65	99	$25,000	4.9	$25,000
HA, All Respiratory	460–519	65	99	$21,000	6.1	$21,000
HA, Asthma	493	0	64	$9,700	3.0	$10,000
HA, Chronic Lung Disease	490–496	18	64	$13,000	3.9	$13,000

* All estimates rounded to two significant digits. Unrounded estimates in 2000$ are available in Appendix J of the BenMAP user manual (Abt Associates, 2011).

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 $186.5 million (1987$). The average cost per visit that year was $155; in 2006$, that cost was $401 (using the CPI-U for medical care to adjust to 2006$). 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 $368.

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 $62 (2006$). 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[5] 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, 2000). An 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" and that "[t]he same increase should be assumed for the 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, 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

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[5] Income elasticity is a common economic measure equal to the percentage change in WTP for a 1% change in income.

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). 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. 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. We also expect that the WTP for improved visibility in Class I areas 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. The elasticity values used to adjust estimates of benefits in 2020 are presented in Table 5-14.

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.[6] We used projections of real GDP (in chained 1996 dollars) provided by Standard and Poor's (2000) for the years 2010 to 2020.[7]

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. Note that, for premature mortality, we applied the income adjustment factor to the present discounted value of the stream of avoided mortalities occurring over the lag period. Because of a lack of data on the dependence of COI and income and a lack of data on projected growth in average wages, no adjustments are made to benefits based on the COI approach or to work loss days and worker productivity. This assumption leads us to 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).

[6] 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.

[7] 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 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 Attaining Alternative Combinations of 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 non-monetized health and welfare benefits; this B represents both uncertainty and a bias in this analysis, as it reflects those benefits categories that we are unable quantify 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 5-24 present the benefits results for the alternative combinations of primary $PM_{2.5}$ standards. In analyzing the current 15/35 standard (baseline), EPA determined that all counties would meet the 14/35 standard concurrently with meeting the existing 15/35 standard at no additional cost. Consequently, there are no incremental costs or benefits for 14/35, and no need to present an analysis of 14/35. Figure 5-3 graphically displays the total monetized benefits of the proposed range of primary standard combinations (12/35 and 13/35) 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 12/35 and 13/35.

Table 5-16. Population-Weighted Air Quality Change for Adults (30+) for Alternative Standards Relative to 15/35

Standard	Population-Weighted Air Quality Change
13/35	0.0008 $\mu g/m^3$
12/35	0.0206 $\mu g/m^3$
11/35	0.0807 $\mu g/m^3$
11/30	0.1228 $\mu g/m^3$

5.7.2 Uncertainty in Benefits Results

Health benefits account for between 97 and 99% of total benefits depending on the $PM_{2.5}$ mortality estimates used, 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 100 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.

- Implementation of this new air quality standard is expected to reduce emissions of directly emitted $PM_{2.5}$, SO_2, and NO_x. 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.

- We assume that the health impact function for fine particles is linear within the range of ambient concentrations under consideration. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5}$, including both regions that are in attainment with fine particle standard and those that do not meet the standard down to the lowest modeled concentrations.

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.

Table 5-17. Estimated number of Avoided $PM_{2.5}$ Health Impacts for Alternative Combinations of Primary $PM_{2.5}$ Standards (Incremental to Attaining Current Suite of Primary $PM_{2.5}$ Standards)[a]

Health Effect	Alterative Combination of Standards (95th percentile confidence interval) 13 µg/m³ Annual & 35 µg/m³ 24-hr[b]	12 µg/m³ Annual & 35 µg/m³ 24-hr	11 µg/m³ Annual & 35 µg/m³ 24-hr	11 µg/m³ Annual & 30 µg/m³ 24-hr
Non-fatal heart attacks				
Peters et al. (2001) (age >18)	11 (6–19)	320 (80–550)	1,300 (390–2,100)	1,900 (590–3,200)
Pooled estimate of 4 studies (age >18)	1 (1–3)	35 (15–92)	140 (64–330)	210 (98–510)
Hospital admissions—respiratory (all ages)	3 (2–5)	98 (51–150)	430 (240–620)	620 (350–0,890)
Hospital admissions—cardiovascular (age > 18)	3 (2–6)	95 (43–170)	400 (190–0,700)	580 (280–1,000)
Emergency department visits for asthma (age < 18)[b]	6 (2–13)	160 (-29–340)	730 (-56–1,500)	1,000 (-79–2,100)
Acute bronchitis (ages 8–12)[b]	22 (7–48)	540 (-120–1,200)	2,000 (-260–4,200)	3,100 (-400–6,400)
Lower respiratory symptoms (ages 7–14)	290 (180–450)	6,900 (2,700–11,000)	25,000 (11,000–40,000)	39,000 (17,000–61,000)
Upper respiratory symptoms (asthmatics ages 9–11)	410 (220–710)	9,800 (1,800–18,000)	37,000 (9,200–64,000)	56,000 (14,000–98,000)
Asthma exacerbation (asthmatics ages 6–18)	410 (150–860)	24,000 (0–180,000)	89,000 (1,900–570,000)	140,000 (2,900–870,000)
Lost work days (ages 18–65)	1,800 (1700–2100)	44,000 (38,000–51,000)	170,000 (150,000–200,000)	260,000 (220,000–290,000)
Minor restricted-activity days (ages 18–65)	11,000 (9,500–12,000)	260,000 (210,000–310,000)	1,000,000 (840,000–1,200,000)	1,500,000 (1,300,000–1,800,000)

[a] All incidence estimates are rounded to whole numbers with a maximum of two significant digits.

[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 5-18. Estimated Number of Avoided $PM_{2.5}$-Related Deaths for Alternative Combinations of Primary $PM_{2.5}$ Standards (Incremental to Attaining Current Suite of Primary $PM_{2.5}$ Standards)[a]

	Alterative Combination of Standards (95[th] percentile confidence interval)			
	13 µg/m^3 Annual & 35 µg/m^3 24-hr	12 µg/m^3 Annual & 35 µg/m^3 24-hr	11 µg/m^3 Annual & 35 µg/m^3 24-hr	11 µg/m^3 Annual & 30 µg/m^3 24-hr
Mortality impact functions derived from the epidemiology literature				
Krewski et al. (2009)	11 (9–14)	280 (190–370)	1,100 (790–1,400)	1,700 (1,200–2,300)
Laden et al. (2006)	27 (19–41)	730 (330–1100)	2,900 (1,400–4,300)	4,500 (2,200–6,700)
Woodruff et al. (1997) (infant mortality)	0 (0–0)	1 (0–1)	3 (1–5)	4 (2–7)
Mortality impact functions derived from the $PM_{2.5}$ Expert Elicitation (Roman et al., 2006)				
Expert A	28 (11–55)	740 (55–1,500)	2,900 (400–5,700)	4,500 (620–8,900)
Expert B	22 (4–47)	590 (27–1,300)	2,300 (160–5,000)	3,600 (220–7,700)
Expert C	22 (12–37)	580 (140–1000)	2,300 (670–3,900)	3,600 (1100–6,100)
Expert D	15 (0–26)	410 (0–700)	1,600 (0–2,700)	2,500 (0–4,200)
Expert E	36 (23–55)	950 (370–1,500)	3,700 (1,600–5,700)	5,900 (2,600–9,000)
Expert F	20 (14–28)	530 (310–770)	2,100 (1,300–3,000)	3,300 (2,000–4,700)
Expert G	13 (0–24)	340 (0–640)	1,300 (0–2,500)	2,100 (0–3,800)
Expert H	16 (0–38)	420 (0–1100)	1,700 (0–4,000)	2,600 (0–6,200)
Expert I	22 (0–38)	570 (0–1000)	2,300 (0–3,900)	3,500 (0–6,100)
Expert J	18 (6–39)	470 (17–1100)	1,800 (170–4,000)	2,900 (270–6,300)
Expert K	3 (0–12)	72 (0–330)	270 (0–1,200)	420 (0–1,900)
Expert L	16 (0–29)	400 (0–790)	1,600 (0–3,000)	2,400 (0–4,700)

[a] All incidence estimates are rounded to whole numbers with a maximum of two significant digits.

Table 5-19. Estimated Monetized $PM_{2.5}$ Health Impacts for Alternative Combinations of Primary $PM_{2.5}$ Standards (Incremental to Attaining Current Suite of Primary $PM_{2.5}$ Standards) (Millions of 2006$, 3% discount rate)[a]

	Alterative Combination of Standards (95th percentile confidence interval)			
Health Effect	13 µg/m³ Annual & 35 µg/m³ 24-hr	12 µg/m³ Annual & 35 µg/m³ 24-hr	11 µg/m³ Annual & 35 µg/m³ 24-hr	11 µg/m³ Annual & 30 µg/m³ 24-hr
Non-fatal heart attacks				
Peters et al. (2001) (age >18)	$1.1 ($0.19–$2.9)	$33 ($5.5–$82)	$130 ($23–$330)	$200 ($34–$500)
Pooled estimate of 4 studies (age >18)	$0.13 ($0.029–$0.43)	$3.7 ($0.85–$12)	$15 ($3.5–$49)	$23 ($5.3–$75)
Hospital admissions—respiratory (all ages)	$0.081 ($0.050–$0.11)	$2.4 ($1.5–$3.3)	$10 ($6–$15)	$15 ($9–$21)
Hospital admissions—cardiovascular (age > 18)	$0.11 ($0.058–$0.19)	$3.2 ($1.7–$5.4)	$13 ($7–$23)	$20 ($10–$33)
Emergency department visits for asthma (age < 18)	$0.0023 ($0.000065–$0.0050)	$0.058 ($0.0016–$0.13)	$0.27 ($0.008–$0.58)	$0.38 ($0.011–$0.81)
Acute bronchitis (ages 8–12)[b]	$0.010 (-$0.00045–$0.028)	$0.24 (-$0.011–$0.66)	$0.89 (-$0.040–$2.4)	$1.40 (-$0.062–$3.7)
Lower respiratory symptoms (ages 7–14)	$0.0055 ($0.0018–$0.011)	$0.13 ($0.044–$0.27)	$0.49 ($0.16–$1.0)	$0.76 ($0.25–$1.5)
Upper respiratory symptoms (asthmatics ages 9–11)	$0.012 ($0.0028–$0.030)	$0.30 ($0.067–$0.74)	$1.1 ($0.25–$2.7)	$1.7 ($0.38–$4.2)
Asthma exacerbation (asthmatics ages 6–18)	$0.022 ($0.0019–$0.058)	$1.3 ($0.047–$9.0)	$4.7 ($0.18–$34)	$7.2 ($0.27–$51)
Lost work days (ages 18–65)	$0.27 ($0.24–$0.31)	$6.7 ($5.8–$7.5)	$26 ($22–$29)	$39 ($34–$44)
Minor restricted-activity days (ages 18–65)	$0.67 ($0.35–$1.0)	$16.0 ($8.7–$25)	$64 ($34–$96)	$96 ($51–$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 benefits noted in Chapter 6.

[b] 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. Estimated Monetized $PM_{2.5}$ Health Impacts for Alternative Combinations of Primary $PM_{2.5}$ Standards (Incremental to Attaining Current Suite of Primary $PM_{2.5}$ Standards) (Millions of 2006$, 7% discount rate)[a]

	Alterative Combination of Standards (95th percentile confidence interval)			
Health Effect	13 μg/m³ Annual & 35 μg/m³ 24-hr	12 μg/m³ Annual & 35 μg/m³ 24-hr	11 μg/m³ Annual & 35 μg/m³ 24-hr	11 μg/m³ Annual & 30 μg/m³ 24-hr
Non-fatal heart attacks				
Peters et al. (2001) (age >18)	$1.1 ($0.180–$2.9)	$32 ($5.1–$81)	$130 ($21–$320)	$190 ($32–$490)
Pooled estimate of 4 studies (age >18)	$0.12 ($0.027–$0.42)	$3.6 ($0.79–$12)	$14 ($3.2–$48)	$22 ($4.9–$74)
Hospital admissions—respiratory (all ages)	$0.081 ($0.050–$0.11)	$2.4 ($1.5–$3.3)	$10 ($6–$15)	$15 ($9–$21)
Hospital admissions—cardiovascular (age > 18)	$0.11 ($0.058–$0.19)	$3.2 ($1.7–$5.4)	$13 ($7.0–$23)	$20 ($10–$33)
Emergency department visits for asthma (age < 18)	$0.0023 ($0.000065–$0.0050)	$0.058 ($0.0016–$0.13)	$0.27 ($0.008–$0.58)	$0.38 ($0.011–$0.81)
Acute bronchitis (ages 8–12)[b]	$0.010 (-$0.00045–$0.028)	$0.24 (-$0.011–$0.66)	$0.89 (-$0.040–$2.4)	$1.40 (-$0.062–$3.7)
Lower respiratory symptoms (ages 7–14)	$0.0055 ($0.0018–$0.011)	$0.13 ($0.044–$0.27)	$0.49 ($0.16–$1.0)	$0.76 ($0.25–$1.5)
Upper respiratory symptoms (asthmatics ages 9–11)	$0.012 ($0.0028–$0.030)	$0.30 ($0.067–$0.74)	$1.1 ($0.25–$2.7)	$1.7 ($0.38–$4.2)
Asthma exacerbation (asthmatics ages 6–18)	$0.022 ($0.0019–$0.058)	$1.3 ($0.047–$9.0)	$4.7 ($0.18–$34)	$7.2 ($0.27–$51)
Lost work days (ages 18–65)	$0.27 ($0.24–$0.31)	$6.7 ($5.8–$7.5)	$26 ($22–$29)	$39 ($34–$44)
Minor restricted-activity days (ages 18–65)	$0.67 ($0.35–$1.0)	$16.0 ($8.7–$25)	$64 ($34–$96)	$96 ($51–$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 benefits noted in Chapter 6.

[b] 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. Estimated Monetized $PM_{2.5}$-Related Deaths for Alternative Combinations of Primary $PM_{2.5}$ Standards (Incremental to Attaining Current Suite of Primary $PM_{2.5}$ Standards)(Millions of 2006$, 3% discount rate)[a]

Health Effect	Alterative Combination of Standards (95th percentile confidence interval)			
	13 µg/m³ Annual & 35 µg/m³ 24-hr	12 µg/m³ Annual & 35 µg/m³ 24-hr	11 µg/m³ Annual & 35 µg/m³ 24-hr	11 µg/m³ Annual & 30 µg/m³ 24-hr
Mortality impact functions derived from the epidemiology literature				
Krewski et al. (2009)	$86 ($8.0–$240)	$2,300 ($210–$6,300)	$9,000 ($840–$25,000)	$14,000 ($1,300–$39,000)
Laden et al. (2006)	$220 ($19–$640)	$5,900 ($520–$17,000)	$23,000 ($2,000–$67,000)	$36,000 ($3,200–$100,000)
Woodruff et al. (1997) (infant mortality)	$0.28 ($0.023–$0.82)	$6.9 ($0.58–$20)	$26 ($2.2–$78)	$39 ($3.3–$110)
Mortality impact functions derived from the $PM_{2.5}$ Expert Elicitation (Roman et al., 2008)				
Expert A	$220 ($13–$760)	$5,900 ($330–$20,000)	$23,000 ($1,300–$78,000)	$37,000 ($2,100–$120,000)
Expert B	$180 ($6.0–$700)	$4,800 ($150–$19,000)	$19,000 ($0,560–$73,000)	$29,000 ($0,780–$110,000)
Expert C	$180 ($13–$540)	$4,700 ($350–$14,000)	$18,000 ($1,400–$57,000)	$29,000 ($2,200–$88,000)
Expert D	$120 ($3.5–$390)	$3,300 ($94–$10,000)	$13,000 ($370–$41,000)	$20,000 ($580–$63,000)
Expert E	$290 ($25–$860)	$7,700 ($650–$23,000)	$30,000 ($2,600–$89,000)	$47,000 ($4,000–$140,000)
Expert F	$160 ($14.0–$460)	$4,300 ($390–$12,000)	$17,000 ($1,600–$49,000)	$26,000 ($2,400–$76,000)
Expert G	$100 ($0–$370)	$2,700 ($0–$9,800)	$11,000 ($0–$39,000)	$17,000 ($0–$61,000)
Expert H	$130 ($0–$510)	$3,400 ($0–$13,000)	$13,000 ($0–$53,000)	$21,000 ($0–$82,000)
Expert I	$170 ($6.2–$560)	$4,600 ($170–$15,000)	$18,000 ($650–$58,000)	$28,000 ($1,000–$90,000)
Expert J	$140 ($6.7–$510)	$3,800 ($180–$13,000)	$15,000 ($700–$53,000)	$23,000 ($1100–$82,000)
Expert K	$23 ($0–$140)	$580 ($0–$3,700)	$2,200 ($0–$14,000)	$3,300 ($0–$22,000)
Expert L	$130 ($0.63–$450)	$3,300 ($14–$12,000)	$13,000 ($48–$47,000)	$19,000 ($53–$72,000)

[a] Rounded to two significant figures. Estimates do not include unquantified health benefits noted in Table 5-2 or Section 5.6.5 or welfare benefits noted in Chapter 6.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.

Table 5-22. Estimated Monetized $PM_{2.5}$-Related Deaths for Alternative Combinations of Primary $PM_{2.5}$ Standards (Incremental to Attaining Current Suite of Primary $PM_{2.5}$ Standards) (Millions of 2006$, 7% discount rate)[a]

Health Effect	Alterative Combination of Standards (95th percentile confidence interval) 13 µg/m³ Annual & 35 µg/m³ 24-hr	12 µg/m³ Annual & 35 µg/m³ 24-hr	11 µg/m³ Annual & 35 µg/m³ 24-hr	11 µg/m³ Annual & 30 µg/m³ 24-hr
Mortality impact functions derived from the epidemiology literature				
Krewski et al. (2009)	$77 ($7.2–$210)	$2,100 ($190–$5,600)	$8,100 ($750–$22,000)	$13,000 ($1,200–$35,000)
Laden et al. (2006)	$200 ($18–$580)	$5,300 ($460–$15,000)	$21,000 ($1,800–$60,000)	$32,000 ($2,900–$94,000)
Woodruff et al. (1997) (infant mortality)	$0.28 ($0.023–$0.82)	$6.9 ($0.58–$20)	$26 ($2.2–$78)	$39 ($3.3–$110)
Mortality impact functions derived from the $PM_{2.5}$ Expert Elicitation (Roman et al., 2008)				
Expert A	$200 ($11–$680)	$5,300 ($300–$18,000)	$21,000 ($1,200–$71,000)	$33,000 ($1,900–$110,000)
Expert B	$160 ($5.4–$630)	$4,300 ($130–$17,000)	$17,000 ($500–$66,000)	$26,000 ($0,700–$100,000)
Expert C	$160 ($12–$490)	$4,200 ($320–$13,000)	$17,000 ($1,300–$51,000)	$26,000 ($2,000–$80,000)
Expert D	$110 ($3.2–$350)	$3,000 ($85–$9,300)	$12,000 ($330–$37,000)	$18,000 ($520–$57,000)
Expert E	$260 ($22–$770)	$6,900 ($590–$20,000)	$27,000 ($2,300–$80,000)	$43,000 ($3,600–$130,000)
Expert F	$140 ($13–$410)	$3,900 ($350–$11,000)	$15,000 ($1,400–$44,000)	$24,000 ($2,200–$68,000)
Expert G	$93 ($0–$330)	$2,500 ($0–$8,900)	$9,700 ($0–$35,000)	$15,000 ($0–$55,000)
Expert H	$120 ($0–$460)	$3,100 ($0–$12,000)	$12,000 ($0–$47,000)	$19,000 ($0–$74,000)
Expert I	$160 ($5.6–$500)	$4,200 ($150–$13,000)	$16,000 ($590–$52,000)	$26,000 ($920–$81,000)
Expert J	$130 ($6.0–$460)	$3,400 ($160–$12,000)	$13,000 ($630–$47,000)	$21,000 ($980–$74,000)
Expert K	$20 ($0–$130)	$520 ($0–$3,400)	$2,000 ($0–$13,000)	$3,000 ($0–$20,000)
Expert L	$110 ($0.57–$400)	$2,900 ($12–$11,000)	$11,000 ($43–$42,000)	$17,000 ($48–$65,000)

[a] Rounded to two significant figures. Estimates do not include unquantified health benefits noted in Table 5-2 or Section 5.6.5 or welfare benefits noted in Chapter 6.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.

Table 5-23. Total Estimated Monetized Benefits of the for Alternative Combinations of Primary $PM_{2.5}$ Standards (Incremental to Attaining Current Suite of Primary $PM_{2.5}$ Standards) (millions of 2006$)[a]

Benefits Estimate	13 µg/m^3 annual & 35 µg/m^3 24-hour	12 µg/m^3 annual & 35 µg/m^3 24-hour	11 µg/m^3 annual & 35 µg/m^3 24-hour	11 µg/m^3 Annual & 30 µg/m^3 24-hr
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	$88 + B	$2,300 +B	$9,200+B	$14,000 +B
7% discount rate	$79 + B	$2,100 +B	$8,300 +B	$13,000 +B
Economic value of avoided $PM_{2.5}$-related morbidities and premature deaths using $PM_{2.5}$ mortality estimate from Laden et al. (2006)				
3% discount rate	$220 + B	$5,900 +B	$23,000 +B	$36,000 +B
7% discount rate	$200 + B	$5,400 +B	$21,000 +B	$33,000 +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 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.

Table 5-24. Regional Breakdown of Monetized Benefits Results

Region	Alterative Combination of Standards			
	13 µg/m^3 annual & 35 µg/m^3 24-hour	12 µg/m^3 Annual & 35 µg/m^3 24-hr	11 µg/m^3 Annual & 35 µg/m^3 24-hr	11 µg/m^3 Annual & 30 µg/m^3 24-hr
East[a]	0%	27%	53%	43%
California[b]	98%	70%	44%	47%
Rest of West	2%	3%	3%	10%

[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 12/35 or 13/35.

[b] For 12/35 and 13/35, the majority of benefits (occur in California because this highly populated area is where the most air quality improvement beyond 15/35 is needed to reach these levels.

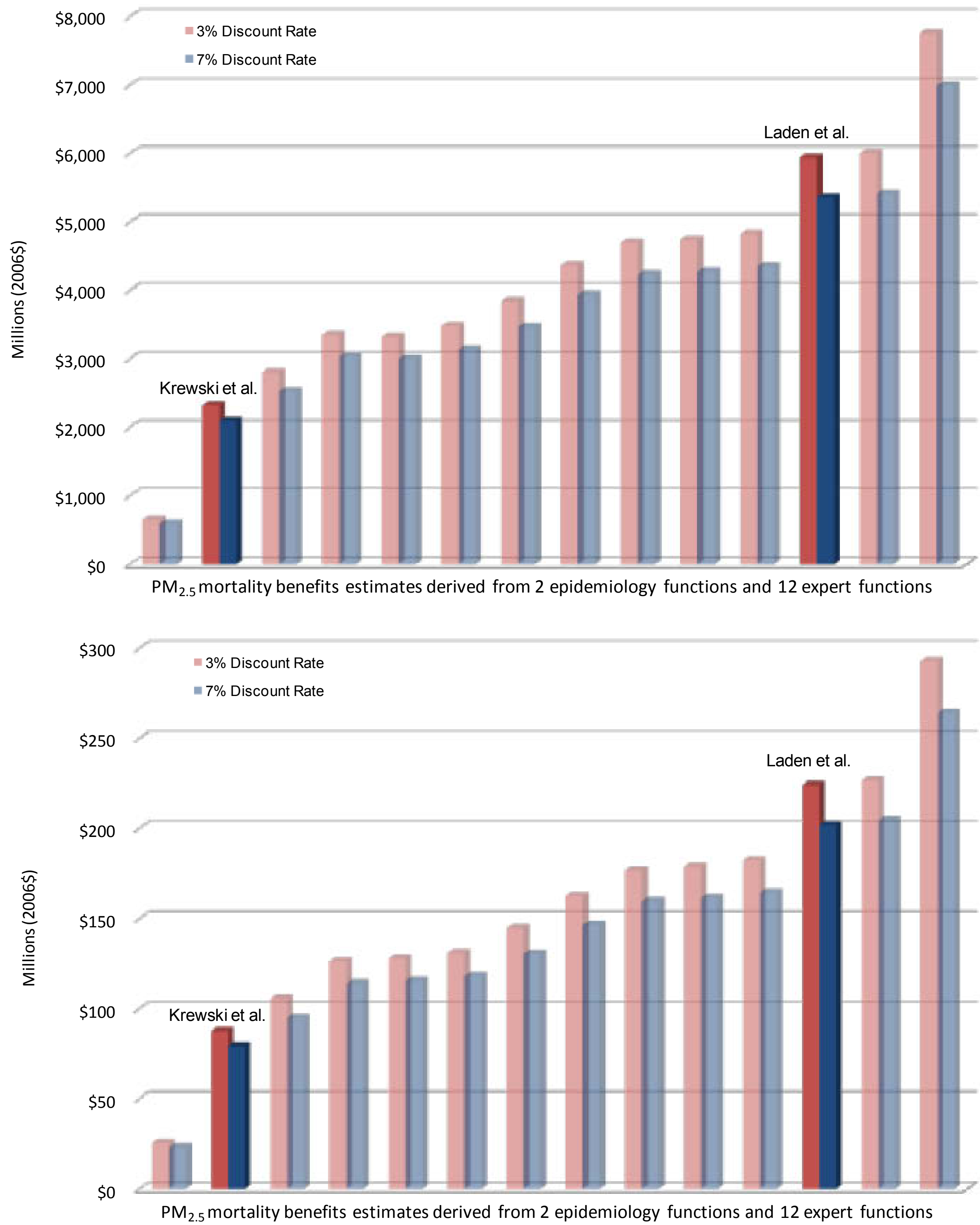


Figure 5-3. Estimated $PM_{2.5}$-Related Premature Mortalities Avoided According to Epidemiology or Expert-Derived $PM_{2.5}$ Mortality Risk Estimate for 12/35 and 13/35

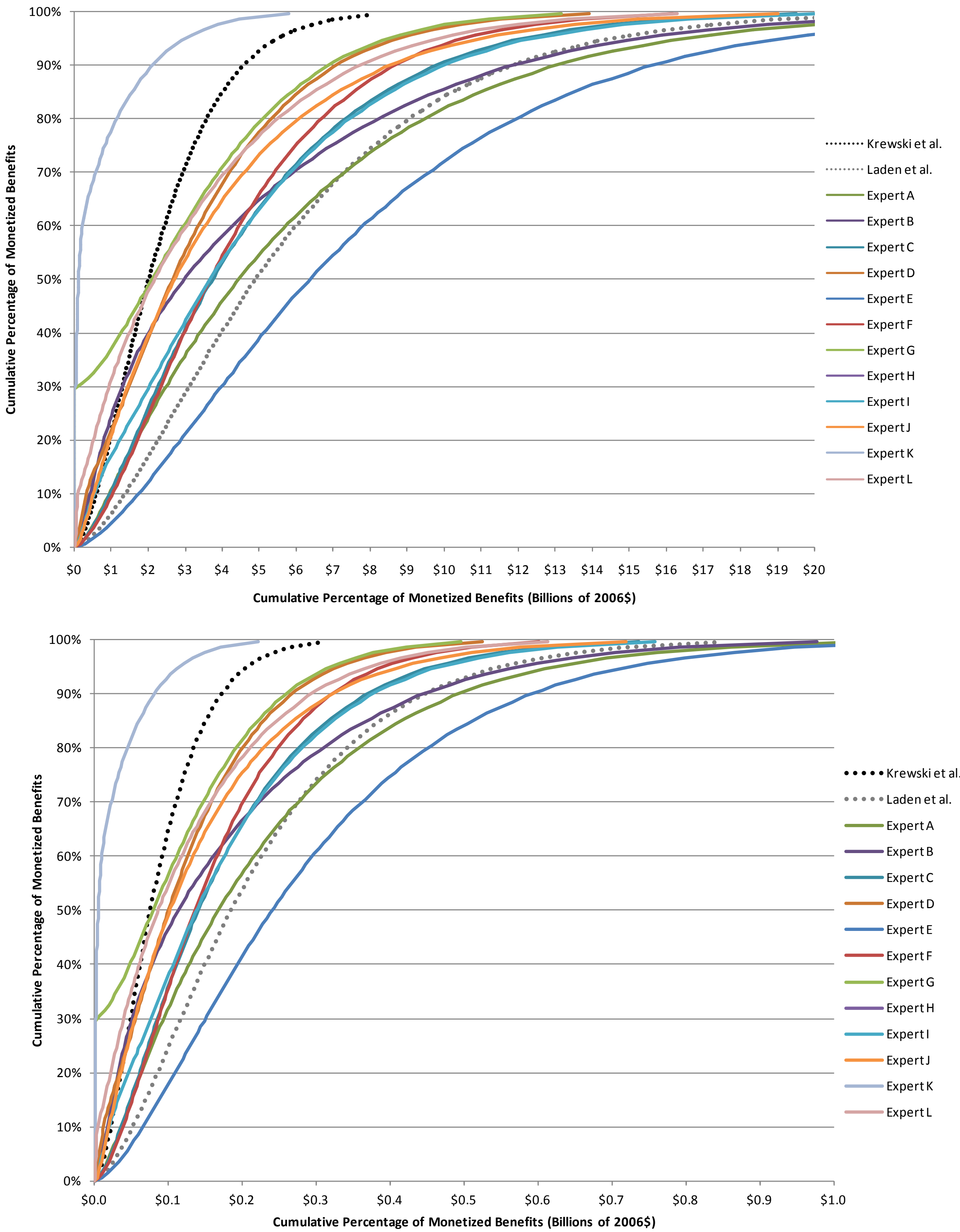


Figure 5-4. Total Monetized Benefits Using 2 Epidemiology-Derived and 12-Expert Derived Relationships Between $PM_{2.5}$ and Premature Mortality for 12/35 and 13/35

5.7.3 Estimated Life Years Gained Attributable to Reduced $PM_{2.5}$ Exposure and Percent of Total Mortality

In their 2008 review of 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 previously 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 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 estimate the number of life years gained and the reduction in the percentage of all deaths attributable to $PM_{2.5}$ resulting from attainment of the alternative combinations of primary $PM_{2.5}$ standards. 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 HES (U.S. EPA-SAB, 2010a).

Because changes in life years and changes in life expectancy at birth are frequently conflated, it is important to distinguish these two very different metrics. 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). 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. 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). If a 60-year old individual dies, we estimate about 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).

In this analysis, we use the same general approach as Hubbell (2006) and Fann et al. (2012a) for estimating potential life years gained by reducing exposure to $PM_{2.5}$ in adult populations. 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 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, finding that reducing exposure to air pollution increased 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 inconsistent because they are reporting different metrics. Because life expectancy is an average of the entire population (including those who will not die from PM exposure as well as those who will), average life expectancy changes associated with PM exposure will 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 calculate the potential distribution of life years gained for populations of different ages, we use standard life tables available from the CDC (2003) and the following formula:

$$Total\ Life\ Years = \sum_{i=1}^{n} LE_i \times M_i \qquad (5\text{-}2)$$

where LE_i is the remaining life expectancy for age interval i, M_i is the change in number of deaths in age interval i, and n is the number of age intervals. We binned the life year results by age range and calculated the average per life lost.

When we estimate the number of avoided premature deaths attributed to changes in $PM_{2.5}$ exposure in 2020, we apply risk coefficients estimated for all adult populations in conjunction with age-specific mortality rates. That is, we apply risk coefficients that do not vary by age, but use baseline mortality rates 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. 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. On average, we estimate that the average individual who would otherwise have died prematurely from PM exposure would gain 16 additional years of life.

When calculating changes in life years, we assume that the life expectancy at birth of those dying from $PM_{2.5}$ exposure is the same as the general population. In reference to the most recent Six Cities extended analysis by Laden et al. (2006), Krewski et al. (2009) notes that "[w]hether $PM_{2.5}$ exposure was modeled as the annual average in the year of death or as the average over the entire follow-up period, it had similar effects on mortality. The results from the study suggest that since $PM_{2.5}$ exposure may affect sensitive individuals with preexisting conditions and play a role in the development of chronic disease, as exposure declines so may the excess mortality related to it." For this reason, we believe that this is a reasonable assumption.

In addition, this analysis includes an estimate of the percentage of all-cause mortality attributed to reduced $PM_{2.5}$ exposure in 2020 as a result of the illustrative control strategies. The percentage of premature $PM_{2.5}$-related mortality is calculated by dividing the number of excess deaths estimated for each alternative standard by the total number of deaths in each county. We have also binned these results by age range.

Tables 5-25 and 5-26 summarize the estimated number of life years gained and the reduction in the percentage of all-cause mortality attributable to reduced $PM_{2.5}$ exposure in 2020 by age range for 12/35. Figure 5-5 bins the potential life years gained and avoided premature deaths into age ranges for 12/35 for comparison. The number of life years gained and avoided mortalities would be similar across various combinations of standards on a relative 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. While the structure of the lag is uncertain, some studies suggest that most of the premature deaths are avoided within the first three years after the change in exposure, while others are unable to find 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 of exposure or cumulative lifetime exposure.

Table 5-25. Sum of Life Years Gained by Age Range from Changes in $PM_{2.5}$ Exposure in 2020 for 12/35[a,b]

Age Range[b]	Krewski et al. (2009) Risk Coefficient [c]	Laden et al. (2006) Risk Coefficient
25–29	—	420
30–34	140	370
35–44	410	1,000
45–54	690	1,800
55–64	1,100	2,700
65–74	1,100	2,800
75–84	740	1,900
85–99	350	880
Total life years gained	4,500	12,000
Average life years gained per individual	**16.0**	**16.4**

[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.

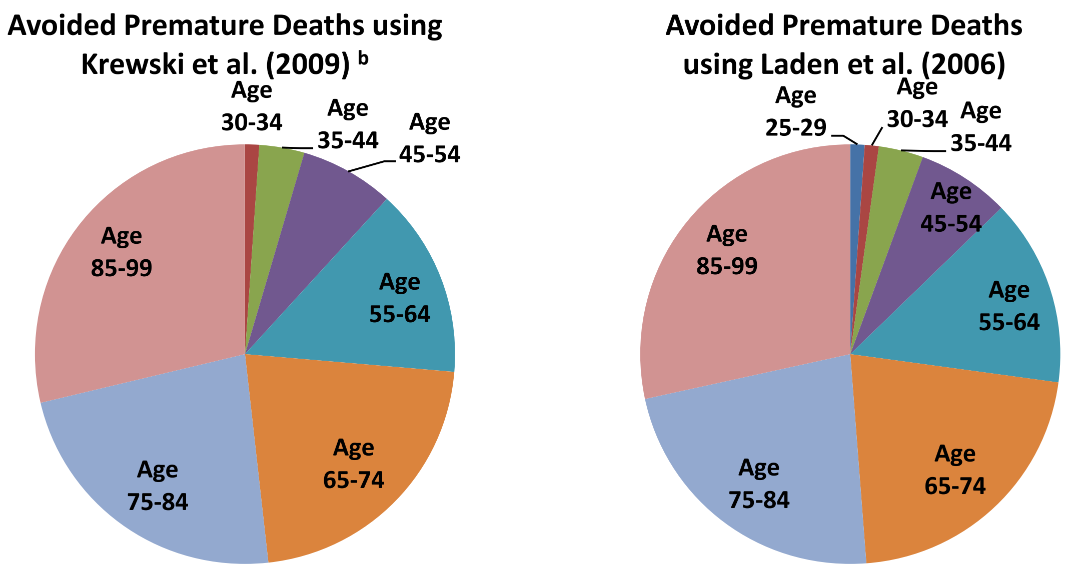


Figure 5-5a. Distribution of Estimated Avoided Premature Deaths by the Age at which these Populations were Exposed in 2020 for 12/35 [a]

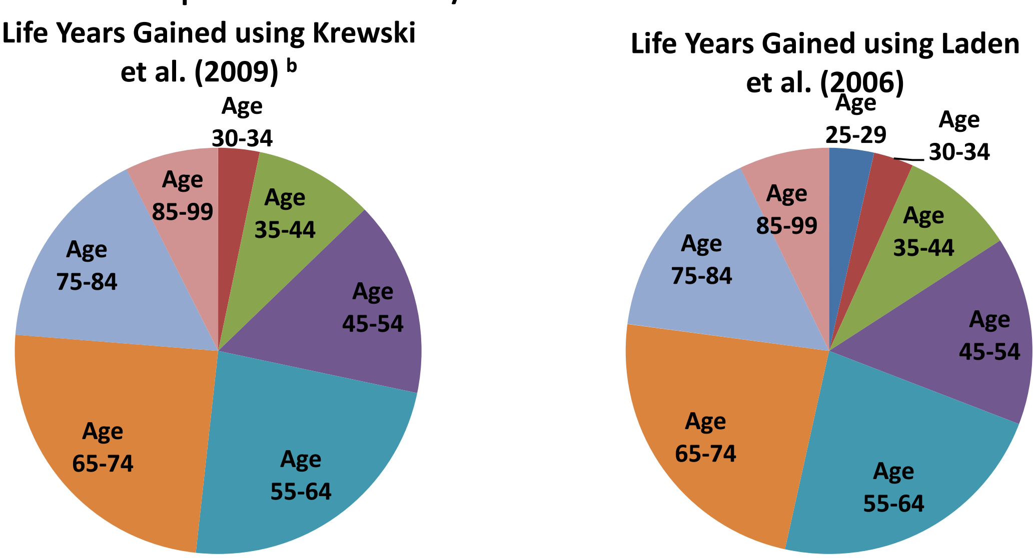


Figure 5-5b. Distribution of Estimated Life Years Gained by the Age at which these Populations were Exposed in 2020 for 12/35 [a]

[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.

Table 5-26. Estimated Reduction the Percentage of All-Cause Mortality Attributable to $PM_{2.5}$ for 12/35 from Changes in $PM_{2.5}$ Exposure in 2020[a,b]

Age Range[b]	Krewski et al. (2009) Risk Coefficient[c]	Laden et al. (2006) Risk Coefficient
25–29	—	0.036%
30–34	0.014%	0.035%
35–44	0.013%	0.033%
45–54	0.013%	0.034%
55–64	0.012%	0.031%
65–74	0.011%	0.028%
75–84	0.010%	0.026%
85–99	0.010%	0.025%

[a] Rounded to two significant figures. Results would be similar for other standard levels on a percentage basis.

[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.

5.7.4 Analysis of Mortality Impacts at Various Concentration Benchmarks

In this analysis, we estimate the number of avoided $PM_{2.5}$-related deaths occurring down to various $PM_{2.5}$ concentration benchmarks, including the Lowest Measured Level (LML) of each long-term $PM_{2.5}$ mortality study. We include this sensitivity analysis because assessments quantifying $PM_{2.5}$ related health impacts generally find that cases of avoided mortality represent the majority of the monetized benefits. This analysis is one of several sensitivities that 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).

Based on our review of the current body of scientific literature, EPA estimated PM-related mortality without applying an assumed concentration threshold. The PM ISA (U.S. EPA, 2009b), which was reviewed by EPA's Clean Air Scientific Advisory Committee (U.S. EPA-SAB, 2009a; U.S. EPA-SAB, 2009b), concluded that the scientific literature consistently finds that a no-threshold log-linear model most adequately portrays the PM-mortality concentration-response relationship while also recognizing potential uncertainty about the exact shape of the concentration-response function.[8] Consistent with this finding, we incorporated a LML assessment, which is a method 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 EPA's *Policy Assessment for Particulate Matter* (U.S. EPA, 2011b).

This approach summarizes the distribution of avoided $PM_{2.5}$-related mortality impacts according to the baseline (i.e., pre-rule) annual mean $PM_{2.5}$ levels at which populations are exposed and the minimum observed air quality level of each long-term cohort study we use to quantify mortality impacts. In general, our confidence in the estimated number of premature deaths diminishes as we estimate these impacts in locations where $PM_{2.5}$ levels are below this level. This interpretation is consistent with the *Policy Assessment* (U.S. EPA, 2011b) and advice from CASAC during their peer review (U.S. EPA-SAB, 2010d). In general, we have greater confidence in risk estimates based on $PM_{2.5}$ concentrations where the bulk of the data reside and somewhat less confidence where data density is lower. However, there are uncertainties inherent in identifying any particular point at which our confidence in reported associations becomes appreciably less.

As noted in the preamble to the proposed rule, the *Policy Assessment* (U.S. EPA, 2011b) concludes that the range from the 25th to 10th percentiles of the air quality data used in the epidemiology studies is a reasonable range below which we have appreciably less confidence in the associations observed in the epidemiological studies.

Although these types of concentration benchmark analyses (e.g., 25th percentile, 10th percentile, and LML) provide some insight into the level of uncertainty in the estimated $PM_{2.5}$ mortality benefits, EPA does not view these concentration benchmarks as a concentration threshold. The central benefits estimates reported in this RIA reflect a full range of modeled air quality concentrations. In reviewing the *Policy Assessment*, 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 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

[8] 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).

estimates below and above these concentration benchmarks but uncertainty is higher in 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 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 Laden et al. (2006) 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 of these studies in Tables 5-27 and 5-28, and we graphically display the distribution of $PM_{2.5}$-related mortality impacts for 12/35 and 13/35 in Figures 5-6 and 5-7. When interpreting these LML graphs, it is important to understand that the plots illustrate the avoided $PM_{2.5}$ deaths estimated to occur from $PM_{2.5}$ reductions in the baseline air quality simulation in which we assume that 15/35 is already met. When simulating attainment with alternative standards, we do not adjust the $PM_{2.5}$ concentration in every 12km grid cell to equal the alternative standard. Instead, we adjust the design value at the monitor to equal the alternative standard and simulate how that adjustment would be reflected in the surrounding grid cells. As such, there may be a small number of grid cells with concentrations greater than 15 $\mu g/m^3$ in the baseline even though all monitors meet an annual standard at 15 $\mu g/m^3$. Specifically, there is one grid cell in San Bernardino County with a baseline concentration of 16.4 $\mu g/m^3$, which falls in the 16 to 17 $\mu g/m^3$ bin. This one grid cell is highly populated and has a relatively high percentage of the avoided premature mortalities because this area received the most air quality improvement from the control strategies to reach 12/35 and 13/35. In addition, several recent rules such as the Mercury and Air Toxics Standard (MATS) and the Cross-State Air Pollution Rule (CSAPR) will have substantially reduced $PM_{2.5}$ levels by 2020 in the East, thus few additional controls would be needed to reach 12/35 or 13/35 in the East.

It is important to note that these estimated benefits reflect specific control measures and emission reductions that are needed to lower $PM_{2.5}$ concentrations for monitors projected to exceed the alternative standard analyzed. The result is that air quality will improve in counties that exceed the alternative standards as well as surrounding areas that do not exceed the alternative standards. It is not possible to apply controls that only reduce $PM_{2.5}$ at the monitor without affecting surrounding areas. In order to make a direct comparison between the benefits and costs of these control strategies, it is appropriate to include all the benefits occurring as a result of the control strategies applied.

We estimate benefits using modeled air quality data with 12km grid cells, which is important because the grid cells are smaller than counties and $PM_{2.5}$ concentrations 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 value is above the alternative standard. Thus, emission reductions lead to benefits in grid cells that are below the alternative standards even within a county with a monitor that exceeds the alternative standard. We have not estimated the fraction of benefits that occur only in counties that exceed the alternative standards.

Table 5-27. Estimated Reduction in Incidence of Adult Premature Mortality Occurring Above and Below the Lowest Measured Levels in the Underlying Epidemiology Studies for 12/35 and 13/35[a]

Study and Lowest Measured Level (LML)	Total Reduced Mortality Incidence	Allocation of Reduced Mortality Incidence	
		Below LML	At or Above LML
12/35			
Krewski et al. (2009) 5.8 µg/m^3	280	23	260
Laden et al. (2006) 10 µg/m^3	730	360	370
13/35			
Krewski et al. (2009) 5.8 µg/m^3	11	1	9
Laden et al. (2006) 10 µg/m^3	27	10	17

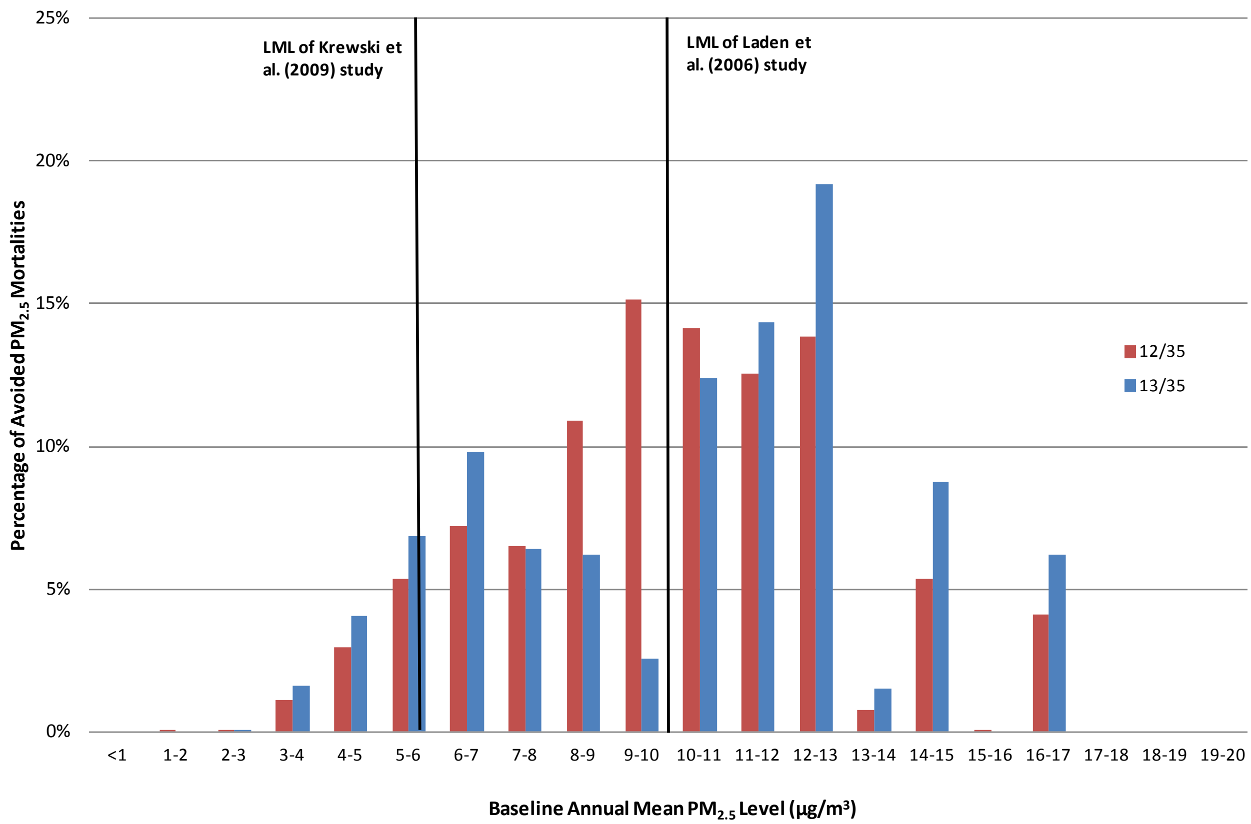
[a] Mortality incidence estimates are rounded to whole numbers and two significant digits, so estimates may not sum across columns. It is important to emphasize that although we have lower levels of confidence in levels below the LML for each study, the scientific evidence does not support the existence of a level below which health effects from exposure to $PM_{2.5}$ do not occur.

Table 5-28. Percentage of Avoided Premature Deaths Occurring At or Above the Lowest Measured Levels in the Underlying Epidemiology Studies for each Alternative Combination of Primary $PM_{2.5}$ Standards[a]

Study and Lowest Measured Level (LML)	13/35	12/35	11/35	11/30
Krewski et al. (2009) 5.8 µg/m^3	89%	92%	95%	93%
Laden et al. (2006) 10 µg/m^3	62%	51%	46%	32%

[a] 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.

While the LML of each study is important to consider when characterizing and interpreting the overall level $PM_{2.5}$-related co-benefits, as discussed earlier in this chapter, EPA believes that both 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 Laden et al. (2006) 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.



Of total $PM_{2.5}$-Related deaths avoided for 12/35:

92% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Krewski et al. (2009) study.

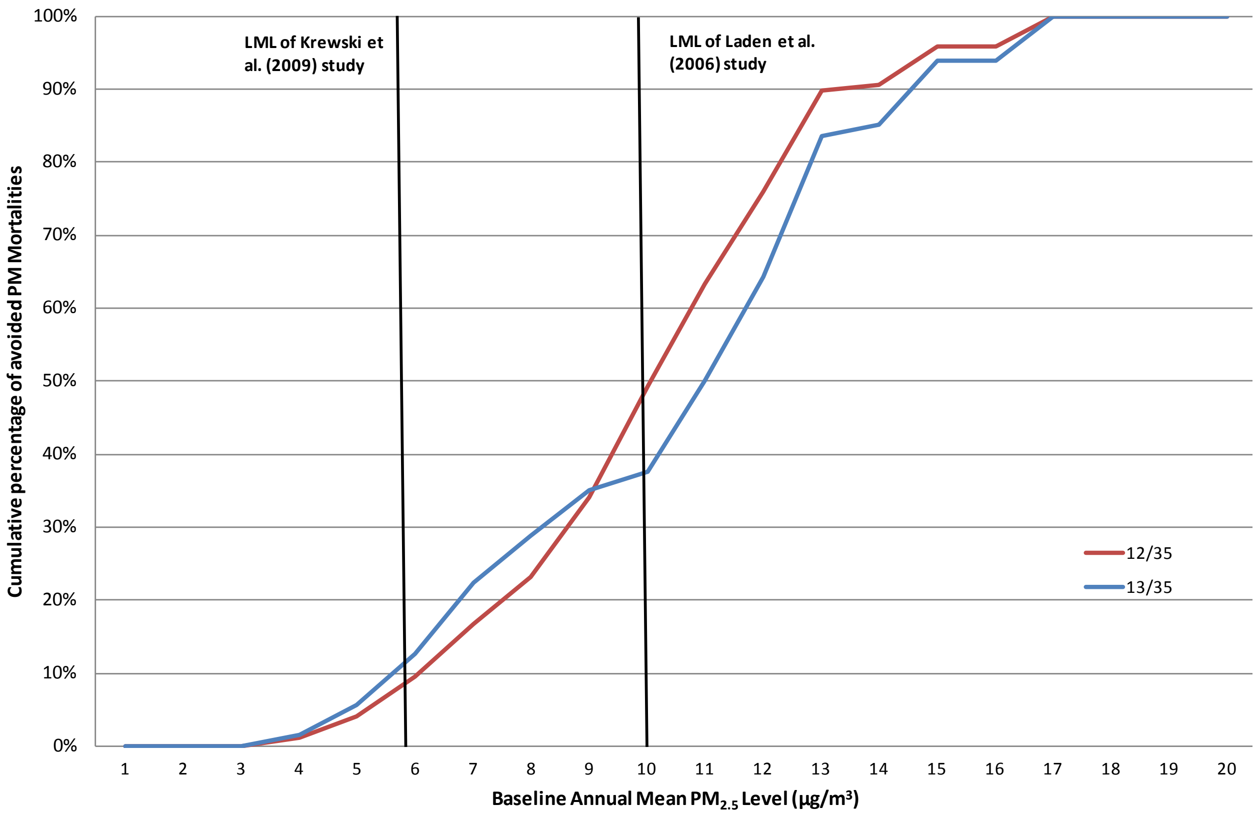
51% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Laden et al. (2006) study.

Of total $PM_{2.5}$-Related deaths avoided for 13/35:

89% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Krewski et al. (2009) study.

62% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Laden et al. (2006) study.

Figure 5-6. Number of Premature $PM_{2.5}$-related Deaths Avoided for 12/35 and 13/35 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 12/35:

92% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Krewski et al. (2009) study.
51% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Laden et al. (2006) study.

Of total $PM_{2.5}$-Related deaths avoided for 13/35:

89% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Krewski et al. (2009) study.
62% occur among populations exposed to $PM_{2.5}$ levels at or above the LML of the Laden et al. (2006) study.

Figure 5-7. Number of Premature $PM_{2.5}$-related Deaths Avoided for 12/35 According to the Baseline Level of $PM_{2.5}$ and the Lowest Measured Air Quality Levels of Each Mortality Study

5.7.5 Additional Sensitivity Analyses

The details of these sensitivity analyses are provided in appendix 5B, 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 cost-of-illness functions for hospital admissions, the rounded estimates of total monetized benefits do not change, but the monetary value of respiratory hospital admissions increases 3.4% and cardiovascular hospital admissions increase 2.1%. These results on a percentage basis would be similar for alternative combinations of standards.

5.8 Discussion

This analysis demonstrates the potential for significant health benefits of the illustrative emission controls applied to simulate attainment with the alternative combination of primary $PM_{2.5}$ standards. We estimate that by 2020 the proposed standards 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, though the quantification of those endpoints in this RIA is incomplete. Even considering the quantified and unquantified uncertainties identified in this chapter, we believe that the implementing the proposed standard would have substantial public health benefits that outweigh the costs.

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 a 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 attaining a revised air quality standard nationwide based on an array of emission control strategies for different sources, incremental to implementation of existing regulations and controls needed to attain current standards. In short, NAAQS RIAs hypothesize, but do not predict, the control 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, 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, 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 at 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, 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.

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APPENDIX 5.A
DISTRIBUTION OF THE $PM_{2.5}$-RELATED BENEFITS

5.A.1 Overview

EPA is developing new approaches and metrics to improve its characterization of the impacts of EPA rules on different populations. This analysis reflects one such approach, which attempts to answer two questions regarding the distribution of $PM_{2.5}$-related benefits resulting from the illustrative attainment of a more stringent National Ambient Air Quality Standards (NAAQS) for particulate matter (PM):

1. What is the baseline distribution of $PM_{2.5}$-related mortality risk according to the race, income and education of the population living within areas projected to exceed alternative combinations of primary $PM_{2.5}$ standards?

2. How would air quality improvements within these counties change the distribution of risk among populations of different races—particularly those populations at greatest risk in the baseline?[1]

There are important methodological differences between this distributional analysis and the Environmental Justice analyses accompanying the Regulatory Impact Analyses (RIAs) for the Cross-State Air Pollution Rule and the Mercury and Air Toxics Standard that are worth noting here. These latter two RIAs applied photochemical modeling to characterize the change in population exposure to $PM_{2.5}$ after the implementation of well-characterized emission controls on Electricity Generating Units. By contrast, this RIA aims to illustrate the potential benefits and costs of attaining alternative primary $PM_{2.5}$ standards. For this reason, similar to the main benefits analysis in this RIA, we performed monitor rollbacks to just attain the alternative combinations of primary standards following the approach described in Chapter 2 of this RIA.

A limitation of this approach to characterizing improvements in $PM_{2.5}$ air quality is that populations in each projected nonattainment area share the exposure reductions equally; this is because simple rollbacks do not reflect the spatial heterogeneity in $PM_{2.5}$ changes one would expect from a modeled attainment strategy. However, as EPA demonstrated in the Detroit multi-pollutant pilot project, states can design attainment strategies to maximize air quality improvements among those populations at greatest risk of air pollution health impacts—which both maximizes overall benefits while lowering the level of risk inequality (Fann et al., 2012a).

[1] 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.

In this analysis we estimated that in 2020, prior to the attainment of a more stringent PM standard, the level of $PM_{2.5}$ mortality risk is not distributed equally among populations of different levels of educational attainment—though the level of mortality risk appears to be shared fairly equally among populations of different races. We find that attaining a more stringent alternative annual PM NAAQS level of 12 µg/m^3 in conjunction with a 24-hour standard of 35 µg/m^3 (as an illustrative example) would provide air quality improvements, and lower $PM_{2.5}$-related mortality risk, by a fairly consistent margin among minority populations. We note that while the methods used for this analysis have been employed in recent EPA Regulatory Impact Assessments (U.S. EPA, 2011) and are drawn from techniques described in the peer reviewed literature (Fann et al., 2012b) EPA will continue to modify these approaches based on evaluation of the methods.

5.A.2 Methodology

As a first step, we identify the counties exceeding an annual standard of 12 µg/m^3 in conjunction with a 24-hour standard of 35 µg/m^3 in 2020, using the results of the baseline CMAQ air quality modeling. This air quality modeling simulation projects $PM_{2.5}$ levels after the incorporation of all "on the books" rules (i.e., those promulgated at the time the air quality modeling was performed), but does not reflect the illustrative attainment strategies. We next identified the counties whose $PM_{2.5}$ levels exceed the alternative combinations of $PM_{2.5}$ standards. We then performed a monitor rollback to adjust the annual $PM_{2.5}$ levels in each county such that they attain the alternative combinations of $PM_{2.5}$ standards. This approach provides us with baseline and rolled-back $PM_{2.5}$ levels that attain this combination of annual and daily $PM_{2.5}$ standards. Within each county exceeding the this combination of $PM_{2.5}$ standards, we estimate the level of all-cause $PM_{2.5}$ mortality risks for adult populations as well as the level of $PM_{2.5}$ mortality risk according to the race and educational attainment of the population.

Our approach to calculating $PM_{2.5}$ mortality risk is generally consistent with the primary analysis with two exceptions: the PM mortality risk coefficients used to quantify impacts and the baseline mortality rates used to calculate education-modified mortality impacts (a detailed discussion of how both the mortality risk coefficients and baseline incidence rates are used to estimate the incidence of $PM_{2.5}$-related deaths may be found in the benefits chapter). Within both this and other analyses of the ACS cohort (see: Krewski et al., 2000), educational attainment has been found to be inversely related to the risk of all-cause mortality. That is, populations with lower levels of education (in particular, < grade 12) are more vulnerable to $PM_{2.5}$-related mortality. Krewski and colleagues note that "…the level of education attainment may likely indicate the effects of complex and multi-factorial socioeconomic processes on mortality…," factors that we would like to account for in this distributional assessment. When estimating PM mortality impacts among populations according to level of education, we applied $PM_{2.5}$ mortality risk coefficients modified by educational attainment: less than grade 12 (Relative Risk (RR) = 1.082, 95% confidence intervals 1.024—1.144 per 10 $\mu g/m^3$ change), grade 12 (RR = 1.072, 95% confidence intervals 1.020—1.127 per 10 $\mu g/m^3$ change), and greater than grade 12 (RR = 1.055, 95% confidence intervals 1.018—1.094 per 10 $\mu g/m^3$ change). The Pope et al. (2002) study, which EPA has frequently relied upon to quantify PM-related mortality, does not provide education-stratified RR estimates. The principal reason we applied risk estimates from the Krewski study was to ensure that the risk coefficients used to estimate all-cause mortality risk and education-modified mortality risk were drawn from a consistent modeling framework and because the use of the Krewski study is consistent with the primary benefits analysis.

The other key difference between this distributional analysis and the main benefits analysis for this rule relates to the baseline mortality rates. As described in Chapter 5 of this RIA, we calculate $PM_{2.5}$-related mortality risk relative to baseline mortality rates in each county. Traditionally, for benefits analysis, we have applied county-level age- and sex-stratified baseline mortality rates when calculating mortality impacts (Abt Associates, 2010). To calculate $PM_{2.5}$ impacts by race, we incorporated race-specific (stratified by White/Black/Asian/Native American) baseline mortality rates. This approach improves our ability to characterize the relationship between race and $PM_{2.5}$-related mortality however, we do not have a differential concentration-response function as we do for education, and as a result, we are not able to capture the full impacts of race in modifying the benefits associated with reductions in $PM_{2.5}$. Table 5.A-1 summarizes the key attributes of the two distributional assessments.

Table 5.A-1. Key Attributes of the Distributional Analyses in this Appendix

	Distributional Analysis	
Input parameter	***Education-modified PM Mortality Risk***	***Race-stratified PM mortality risk***
Effect coefficient	Stratified by education attainment (<12, =12, >12)	All-cause, applied to each population subgroup
Baseline mortality rates	Cause, age and sex stratified	Cause, age, sex **race** and **ethnicity** stratified

The result of this analysis is a distribution of $PM_{2.5}$ mortality risk estimates by county, stratified by each of the two population variables (race and educational attainment). We have less confidence in county-level estimates of mortality than the national or even state estimates. However, the modeling down to the county level can be considered reasonable because the estimates are based on monitored air quality modeling estimates of $PM_{2.5}$, county level baseline mortality rates, and a concentration-response function that is derived from county level data. We next identified the counties projected to exceed the current combination of annual and daily $PM_{2.5}$ standards (15/35) ("baseline") in 2020. The second step of the analysis was to repeat the sequence above by estimating $PM_{2.5}$ mortality risk in counties projected to exceed an illustrative combination of $PM_{2.5}$ standards (12/35) after rolling back monitor values to reach attainment in 2020.

5.A.3 Results

Figures 5.A-1 and 5.A-2 summarize the change in the median level of $PM_{2.5}$ mortality risk among populations stratified by educational attainment and race in non-attaining counties. The percentage of deaths due to $PM_{2.5}$ among populations with less than a grade 12 education is significantly higher than those who have either completed high school or who have attained an education level greater than high school. This finding is consistent with the relative levels of risk coefficients for each population, where we apply a much larger risk coefficient for populations with less than a grade 12 education. The level of risk reduction between the baseline and 12/35 is roughly equal between the three groups.

In Figure 5.A-2, Black and Native American populations are at significantly greater $PM_{2.5}$ mortality risk in the baseline, as compared to other races. White and Asian populations are at lower levels of $PM_{2.5}$ mortality risk. The finding that black populations are at greater $PM_{2.5}$ mortality risk in the baseline may be due both to the elevated baseline mortality risks or greater exposure to $PM_{2.5}$ among this population. After attaining 12/35, populations of all races benefit, though the reduction in PM mortality risk among whites is within rounding error.

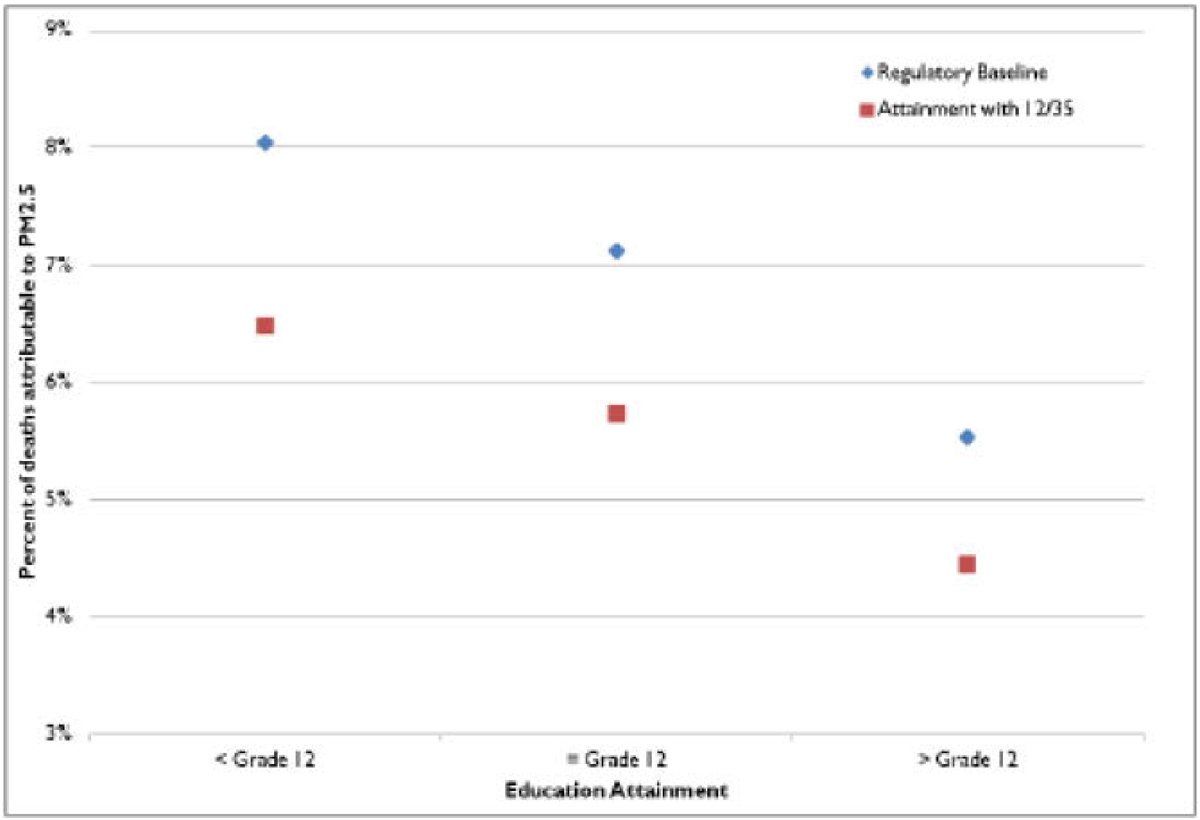

Figure 5.A-1. $PM_{2.5}$ Mortality Risk Modified by Educational Attainment in Counties Projected to Exceed 12/35

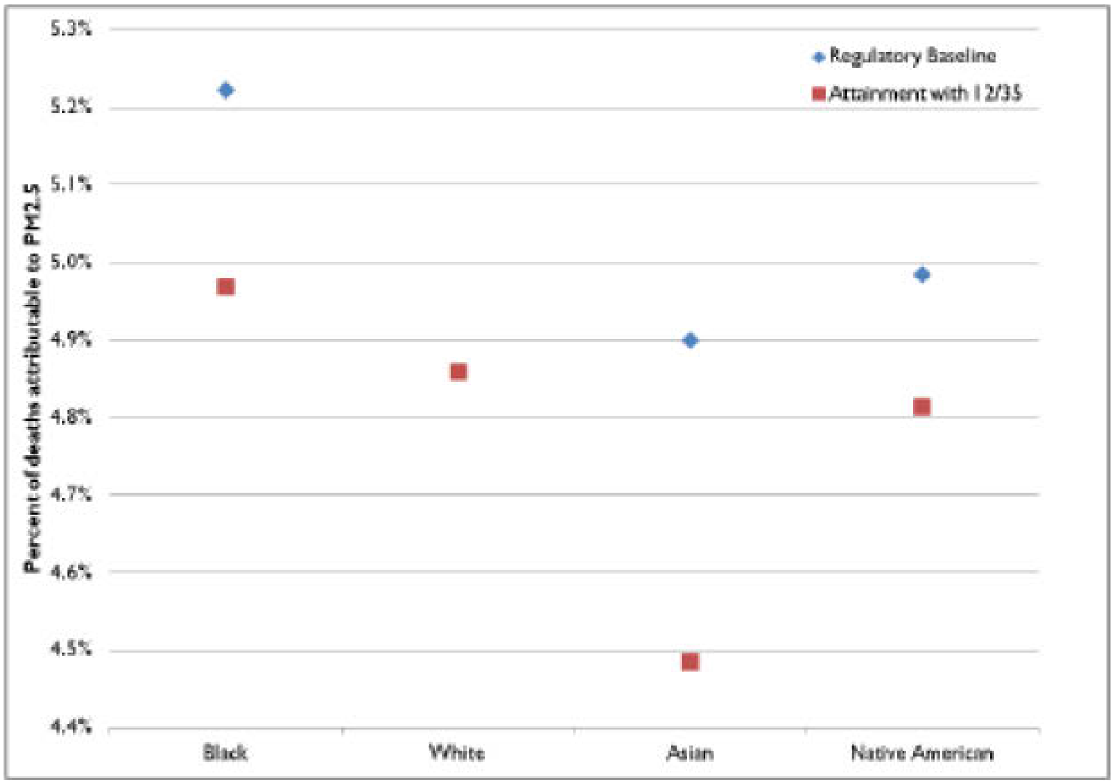

Figure 5.A-2. $PM_{2.5}$ Mortality Risk by Race in Counties Projected to Exceed 12/35

Table 5.A-2. Numerical Values Used for Figures 5.A-1 and 5.A-2 Above[a]

	Scenario, Percent	
Year	**Baseline**	**12/35**
Impacts by education		
< Grade 12	8%	6.5%
= Grade 12	7.1%	5.7%
> Grade 12	5.5%	4.4%
Impacts by race		
Asian	4.9%	4.5%
Black	5.2%	5%
Native American	5%	4.8%
White	4.9%	4.9%

[a] Estimates expressed with a greater number of significant digits to facilitate comparisons among values.

5.A.4 Discussion

This analysis is subject to certain limitations, some of which we note above but are worth repeating here. First, the change in the distribution of $PM_{2.5}$-related mortality risk we estimate here depends is influenced strongly by the simulated attainment strategy. While we performed simple monitor rollbacks to attain a more stringent standard, we describe other approaches above that may maximize human health benefits while also reducing the level of risk inequality. The monitor rollback approach employed here simulates improvements in $PM_{2.5}$ levels in proximity to monitors projected to exceed a tighter PM NAAQS; we would expect an attainment strategy to achieve air quality improvements over a broader geographic area, affecting a greater portion the population than we have reflected here.

Notwithstanding these uncertainties, these results suggest that all populations, irrespective of education attainment or race, living in locations projected to exceed an illustrative annual standard of 12 $\mu g/m^3$ in conjunction with a 24-hour standard of 35 $\mu g/m^3$ in 2020 would experience a reduction in PM-related mortality risk. Certain sub-populations, including those with less than a grade 12 education and Native Americans, area at an elevated risk in the baseline. Attainment of this illustrative standard in 2020 would reduce the level of mortality risk among these sub-populations.

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APPENDIX 5.B
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 modeling 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 benefit results 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 both for physical effects issues (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 an annual standard of 12 $\mu g/m^3$ in conjunction with a 24-hour standard of 35 $\mu g/m^3$ 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.B.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.

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. No animal models have yet been developed to quantify these cumulative effects, nor are there epidemiologic studies bearing on this question. 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 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. 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).

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

[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.

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:

$$RR(t) = ERR \times exp^{-kt} + R_0 \tag{5.B.1}$$

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.B.1) 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.

$$\Delta M = ERR \times t - \int_0^t ERR \times exp^{-kt} dt \tag{5.B.2}$$

Integrating Equation (5.B.2) gives:

$$\Delta M = ERR \times t - \frac{ERR}{k} + \frac{ERR}{k} \times exp^{-kt} \tag{5.B.3}$$

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). Applying combinations of these studies to equation 5.B.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.[4] Table 5.B. 1 provides the time constants for each of these combinations and averages, and Figure 5.B.2 illustrates the exponential decay lag structures.

[3] The relative risk estimate 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] 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.B-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.B-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.B.1 illustrates the cumulative distributions of the cessation lags applied in this appendix.

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.B-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		12/35 Value (billion 2006$)[a,b]	12/35 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	$2.3	N/A
	7% discount rate	$2.1	N/A
No lag	Incidences all occur in the first year		
	3% discount rate	$2.5	10.4%
	7% discount rate	$2.5	22.5%
8-year	Incidences all occur in the 8th year		
	3% discount rate	$2.1	−10.3%
	7% discount rate	$1.6	−23.7%
15-year	Incidences all occur in the 15th year		
	3% discount rate	$1.7	−27.0%
	7% discount rate	$1.0	−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	$2.2	−3.2%
	7% discount rate	$1.9	−6.6%
5-Year Distributed	50% of incidences occur in years 1 and 2 and 50% in years 2 to 5		
	3% discount rate	$2.4	4.9%
	7% discount rate	$2.3	9.4%
Exponential Decay (k=0.45)	Incidences occur at an exponentially declining rate		
	3% discount rate	$2.4	5.0%
	7% discount rate	$2.3	9.9%

[a] Dollar values rounded to two significant digits. The percent difference using effect estimates from Laden et al. would be identical, but the value would be approximately 2.5 times higher.

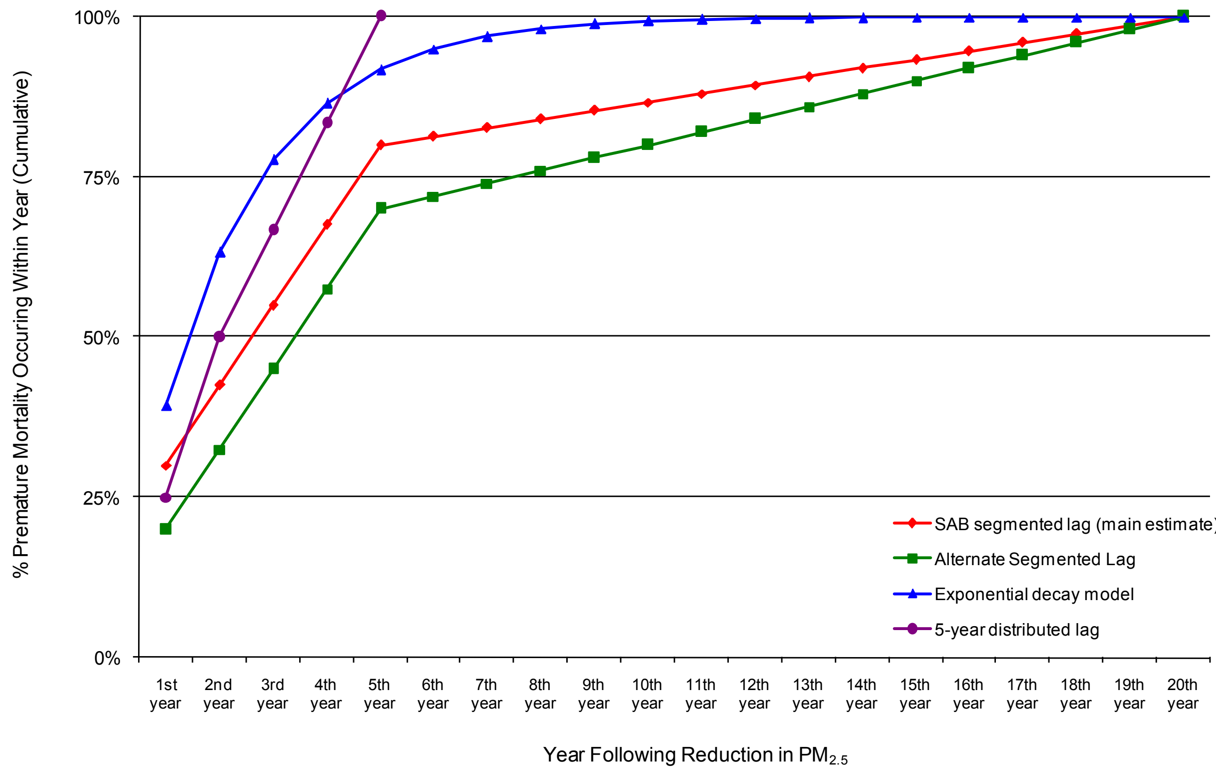

Figure 5.B-1. Alternate Lag Structures for $PM_{2.5}$ Premature Mortality (Cumulative)

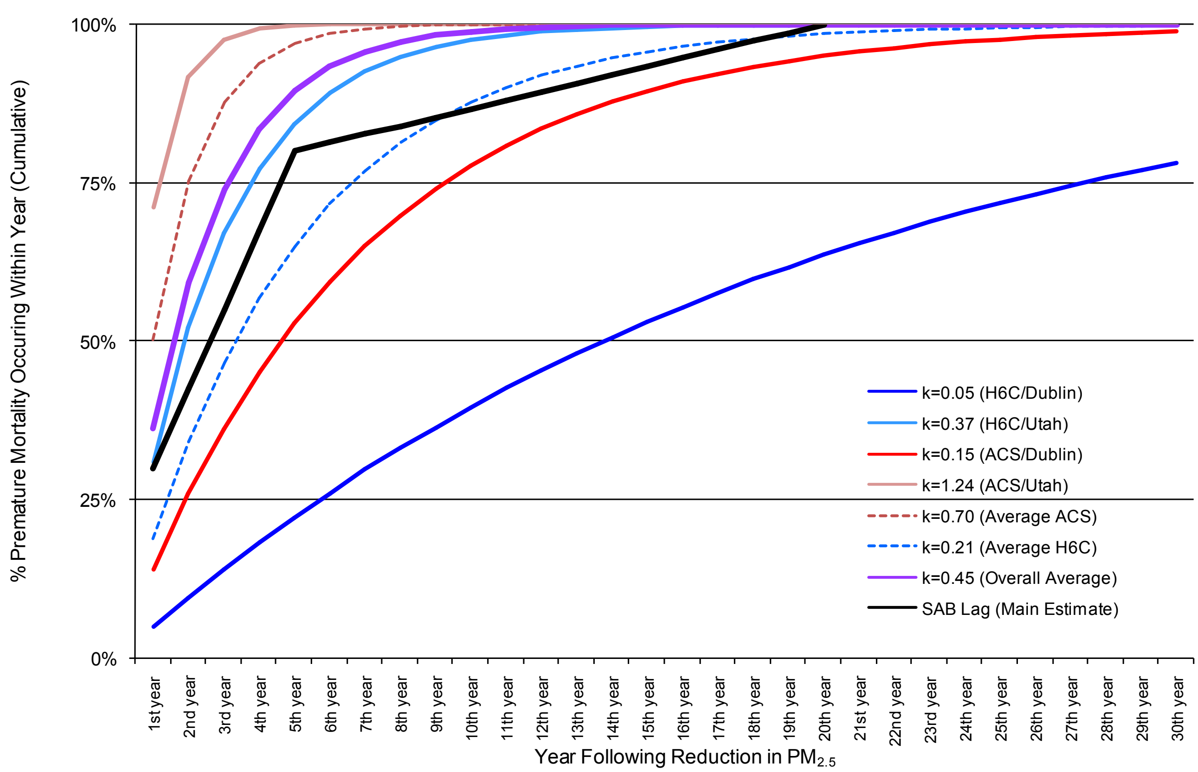

Figure 5.B-2. Exponential Lag Structures for $PM_{2.5}$ Premature Mortality (Cumulative)

5.B.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.B-3 lists the ranges of elasticity values used to calculate the income adjustment factors, while Table 5.B-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.B-5.

Table 5.B-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.B-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.B-5. Sensitivity of Monetized Benefits to Alternative Income Elasticities[a]

	Benefits Incremental to 15/35 Attainment Strategy (Millions of 2006$)	
	12/35	
Benefit Category	Lower Sensitivity Bound	Upper Sensitivity Bound
Minor Health Effect	$18	$20
Premature Mortality[b]	$2,200	$3,300

[a] All estimates rounded to two significant digits.

[b] Using mortality effect estimate from Krewski et al. (2009) and 3% discount rate. Results using Laden et al. (2006) 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 96% to 108% of the main estimate based on the lower and upper sensitivity bounds on the income adjustment factor. The effect on the value of minor health effects is much less pronounced, ranging from 86% to 133% of the main estimate for minor effects.

5.B.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.B-6). While in the benefits chapter we provide a full description of the rationale for treating these endpoints as a sensitivity, 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 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 as a sensitivity because of the absence of newer studies finding a relationship between long-term $PM_{2.5}$ exposure and this endpoint.

To quantify cardiovascular hospital admissions, we apply risk coefficient drawn from Metzger et al. (2004) (RR= 1.033, 95% confidence intervals 1.01–1.056 per 10 µg/m^3 $PM_{2.5}$) and Tolbert et al. (2007) (RR= 1.005, 95% confidence intervals 0.993–1.017 per 10 µg/m^3 $PM_{2.5}$). 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.B-6. Avoided Cases of Cardiovascular Emergency Department Visits, Stroke and Chronic Bronchitis in 2020 (95th percentile confidence intervals)*

Endpoint	***12/35***
Cardiovascular hospital admissions	
Metzger et al. (2004) (ages 0–99)	300 (180–480)
Tolbert et al. (2007) (ages 0–99)	42 (−16–130)
Stroke	
Miller et al. (2007) (ages 50–79)	77 (36–140)
Chronic Bronchitis	
Abbey et al. (1995) (ages 27–99)	220 (99–420)

* All estimates rounded to two significant digits.

5.B.4 New Hospitalization Cost-of-Illness Functions and Median Wage Data

As described in Chapter 5 of this RIA, we updated the cost-of-illness functions for hospitalizations. Specifically, we updated the estimates of hospital charges and lengths of hospital stays were based on discharge statistics provided by the Agency for Healthcare Research and Quality's Healthcare Utilization Project National Inpatient Sample (NIS) database for 2000 (AHRQ, 2000) to 2007 (AHRQ, 2007). In addition, we updated the county-level median wage data reported by the 2007 American Community Survey (ACS) (Abt Associates, 2011). Using cost-of-illness functions for hospital admissions, which include updated charges, length of stay, and median wages, the rounded estimates of total monetized benefits do not change, but the monetary value of respiratory hospital admissions increases 3% and cardiovascular hospital admissions increase 2%. Because the median wages were updated, the valuation also changed the valuation for work loss days. It is important to note that while the national average median daily wage slightly decreased (i.e., approximately 2% in 2000$), the county-level median income increased slightly in the locations where $PM_{2.5}$ levels improved for 12/35. Tables 5.B.7 and 5.B.8 show the previous and current unit values, respectively. Table 5.B.9 shows the sensitivity of the monetized hospitalization benefits to this update.

Table 5.B-7. Unit Values for Hospital Admissions in BenMAP 4.0.51 (Abt Associates, 2011)[a]

End Point	ICD Codes	Age Range *min.*	Age Range *max.*	Mean Hospital Charge (2000$)	Mean Length of Stay (days)	Total Cost of Illness (Unit value in 2000$)
HA, All Cardiovascular	390–429	18	64	$26,654	4.12	$27,119
HA, All Cardiovascular	390–429	65	99	$24,893	4.88	$25,444
HA, All Respiratory	460–519	65	99	$20,667	6.07	$21,351
HA, Asthma	493	0	64	$9,723	3.00	$10,051
HA, Chronic Lung Disease	490–496	18	64	$12,836	3.90	$13,276

[a] National average median daily wage is $112.86 (2000$).

Table 5.B-8. Unit Values for Hospital Admissions in BenMAP 4.0.43 (Abt Associates, 2010)[a]

End Point	ICD Codes	Age Range *min.*	Age Range *max.*	Mean Hospital Charge (2000$)	Mean Length of Stay (days)	Total Cost of Illness (Unit value in 2000$)
HA, All Cardiovascular	390–429	20	64	$22,300	4.15	$22,778
HA, All Cardiovascular	390–429	65	99	$20,607	5.07	$21,191
HA, All Respiratory	460–519	65	99	$17,600	6.88	$18,393
HA, Asthma	493	0	64	$7,448	2.95	$7,788
HA, Chronic Lung Disease	490–496	20	64	$10,194	$5.92	$15,375

[a] National average median daily wage is $115.20 (2000$).

Table 5.B-9. Change in Monetized Hospitalization Benefits for 12/35

Endpoint	2000 AHRQ (millions of 2006$)	2007 AHRQ (millions of 2006$)	Percent Change
Respiratory hospital admissions	$2.3	$2.4	3.4%
Cardiovascular hospital admissions	$3.1	$3.2	2.1%
Work loss days	$6.7	$6.7	0.02%

* All estimates rounded to two significant digits.

5.B.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; Laden et al., 2006), as well as summarized the effect estimates for additional cohort studies. In this appendix, we provide additional information about cohort studies in California.[1] As shown in Table 5.x in the health benefits chapter, a large percentage of the monetized human health benefits associated with the illustrative control strategy to attain the alternative combination of standards are projected to occur in California. Specifically, for an annual $PM_{2.5}$ standard of 12 μg/m^3 in conjunction with retaining the 24-hour standard of 35 μg/m^3, 70% of the total monetized benefits were estimated to occur in California and 98% for an annual $PM_{2.5}$ standard of 13 μg/m^3. 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 (and thus not summarized here) with the exception of the Ostro et al. (2010, 2011) studies, which we describe below. Table 5.B.10 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.

Ostro et al. (2010) characterize the risk of premature death associated with long-term exposure to $PM_{2.5}$ in California among a cohort of about 134,000 current and former female public school professionals (i.e., the California Teacher's Study (CTS)). In this prospective cohort study, Ostro and colleagues estimated long-term PM exposure to several PM constituents, including elemental carbon, organic carbon, sulfates, nitrates, iron, potassium, silicon and zinc. In an erratum, Ostro et al. (2011) modified their approach to assigning $PM_{2.5}$ levels to the cohort populations, noting that they "reanalyzed the CTS data using time-dependent pollution metrics—in which the exposure estimates for everyone remaining alive in the risk set were recalculated at the time of each death—in order to compare their average exposures up to that time with that of the individual who had died. In this way, decedents and survivors comprising the risk set had similar periods of pollution exposure, without subsequent pollution trends influencing the surviving women's exposure estimates." This change in assumption attenuated the hazard ratios significantly, though hazard ratios remained significant for cardiovascular mortality and total $PM_{2.5}$ mass and certain constituents, nitrate and sulfate; no association was observed between all-cause mortality and total $PM_{2.5}$ mass or its constituents. The authors note that these revised results are generally consistent with other long-term PM cohort studies, including the ACS and H6C studies.

[1] 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.B-10. Summary of Effect Estimates From Associated With Change in Long-Term Exposure to $PM_{2.5}$ in Recent Cohort Studies in California

Authors	Cohort	Hazard Ratios per 10 µg/m³ Change in $PM_{2.5}$ (95th percentile confidence interval)		
		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)

[a] Table 3, adjusted for 10 µg/m³ 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.B.10, most of the cohort studies conducted in California report central effect estimates similar to the (nation-wide) all-cause mortality risk estimate we applied from Krewski et al. (2009) and Laden et al. (2006) albeit with wider confidence intervals. A couple cohort studies conducted in California indicate higher risks than the risk estimates we applied.

5.B.6 References

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APPENDIX 5.C
QUALITATIVE ASSESSMENT OF UNCERTAINTY

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 attainment strategies for each of the alternative standards:

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. EPA has applied similar approaches in peer-reviewed analyses of $PM_{2.5}$-related impacts (U.S. EPA, 2010b, 2011a). In addition to the WHO uncertainty framework, we also include an assessment of how each aspect of uncertainty could affect the benefits results, including the direction of potential bias, the magnitude of impact on results, and the degree of confidence in our approach. In Table 5.C-1, 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. Because this approach reflects a new application for regulatory benefits analysis, we request comments on this general approach as well as the specific uncertainty assessments.

5.C.1 Description of Classifications Applied in the Uncertainty Characterization

Uncertainty Characterization Tiers

The WHO framework (2008) defines 4 tiers of uncertainty characterization, which vary depending on the degree of quantification. In Table 5.C-1, we apply these tiers considering the degree of quantification of uncertainty we have conducted in this analysis or that we plan to conduct for the final RIA. Ultimately, the tier decision is professional judgment based on the availability of information.

- **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

Magnitude of Impact

The magnitude of impact is an assessment of how much a plausible alternative assumption or approach could influence the overall monetary benefits. Similar classifications have been included in a previous analyses (U.S. EPA, 2010b, 2011a), but we have revised the category names and the cut-offs here.[1] We note that $PM_{2.5}$-related mortality benefits comprise over 98% of the monetized benefits in this analysis, thus alternative assumptions affecting mortality have the potential to have higher impacts on the total monetized benefits. Including currently omitted categories of benefits would lead to a reduction in the fraction of monetized benefits attributable to lower mortality risk. Ultimately, the magnitude decision is professional judgment based on the experience with various sensitivity analyses.

- **High**—If this uncertainty could influence the total monetized benefits by more than 25%
- **Medium**—If this uncertainty could influence the total monetized benefits by 5% to 25%
- **Low**—If this uncertainty could influence the total monetized benefits by less than 5%

Degree of Confidence in Our Analytic Approach

The degree of confidence is an assessment based on our assessment of the available body of evidence. That is, based on the given available evidence, how certain we are that the selected assumption is the most plausible of the alternatives. Similar classifications have been included in a previous analyses (U.S. EPA, 2010b, 2011a).[2] Ultimately, the degree of confidence is professional judgment based on the volume and consistence of supporting evidence, much of

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[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; "high" if it was are likely to influence the interpretation of risk in the context of the PM NAAQS review.

[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.

which has been evaluated in the PM ISA.

- **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

Table 5.C-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	Degree of Confidence in Our Analytical Approach	Ability to Assess Uncertainty
Uncertainties Associated with $PM_{2.5}$ Concentration Changes				
Projections of future levels of emissions and emissions reductions necessary to attain alternative standards	Both 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.	Medium	Medium	Tier 1 See Chapter 3
Responsiveness of air quality model to changes in precursor emissions from control scenarios	Both	Medium-high	Medium	Tier 1 See Chapter 3
Air quality model chemistry, particularly for formation of ambient nitrate concentrations	Both	Medium	High	Tier 1 See Chapter 3
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 See Chapter 3
Uncertainties Associated with Concentration-Response Functions				
Causal relationship between $PM_{2.5}$ exposure and premature mortality	Overestimate, if no causal relationship	High PM-mortality effects are the largest contributor to the monetized benefits. If the $PM_{2.5}$/mortality relationship were not causal, benefits would be significantly overestimated.	High The PM ISA (U.S. EPA, 2009b), which was twice peer reviewed by CASAC, evaluated the entire body of scientific literature and concluded that the relationship between both short-term and long-term exposure to $PM_{2.5}$ and mortality is causal.	Tier 3 Experts included likelihood of causal relationship, so causality addressed in results derived from $PM_{2.5}$ expert elicitation (Roman et al., 2008).

(continued)

Table 5.C-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	Degree of Confidence in Our Analytical Approach	Ability to Assess Uncertainty
Uncertainties Associated with Concentration-Response Functions (continued)				
Modification of Mortality C-R function by socio-economic status (SES)	Potential underestimate for ACS cohort (Krewski et al., 2009) because of the demographics of that study population. Unknown for H6C cohort (Laden et al., 2006)	Potentially medium-high for ACS cohort Unknown for H6C cohort	Medium We only have mortality risk coefficients modified by educational attainment (Krewski, 2000), not other risk modifiers such as income or race.	Tier 2 Effect modification for educational attainment evaluated in distributional analysis in Appendix 5A.
Exposure misclassification in epidemiology studies	Underestimate (generally) Reducing exposure error can result in stronger associations between pollutants and effect estimates than generally observed in studies having less exposure detail.	Medium-high Recent analyses reported in Krewski et al. (2009) demonstrate the potentially significant effect 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.	High The results from Krewski et al. (2009) and Jerrett et al. (2005) suggest that exposure error underestimates effect estimates (U.S. EPA, 2009b).	Tier 1
Spatial matching of air quality estimates from epidemiology studies to air quality estimates from air quality modeling	Unknown Epidemiology studies often create a composite air quality monitor that is assumed to be representative of an entire urban area to estimate health risks, while benefits are often calculated using air quality modeling conducted at 12 km spatial resolution. This spatial mismatch could introduce uncertainty.	Medium	Medium-Low	Tier 1

(continued)

Table 5.C-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	Degree of Confidence in Our Analytical Approach	Ability to Assess Uncertainty
Uncertainties Associated with Concentration-Response Functions (continued)				
Variation in effect estimates reflecting differences in $PM_{2.5}$ composition (mixtures)	Both	Medium- High Epidemiology studies examining regional differences in $PM_{2.5}$-related health effects have found differences in the magnitude of those effects. While these may be the result of factors other than composition (e.g., different degrees of exposure misclassification), composition remains one potential explanatory factor.	Medium	Tier 1
Differential toxicity of particle components	Unknown We assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. Depending on the toxicity of each PM species reduced, this could over or underestimate benefits.	Medium If the benefits are due to a variety of PM species reduced, the magnitude of this uncertainty is likely to be small. If only one PM species is reduced, this uncertainty may have larger magnitude.	Medium-Low The PM ISA (U.S. EPA, 2009b), which was twice peer reviewed by CASAC, evaluated the entire body of scientific literature and concluded that because there is no clear scientific evidence that would support the development of differential effects estimates by particle type (U.S. EPA, 2009b).	Tier 2 To be assessed in final RIA
Application of C-R relationships only to those subpopulations matching the original study population	Underestimate The C-R functions for several health endpoints were applied only to subgroups of the U.S. population (e.g., adults 30+ for mortality, children 8-12 for acute bronchitis), and thus this may underestimate the whole population benefits of reductions in pollutant exposures.	Low The baseline mortality rate for PM-related health effects is significantly lower in those under the age of 30. Mortality valuation generally dominates monetized benefits.	High Our approach follows recommendations from the NAS (NRC, 2002)	Tier 2 To be assessed in final RIA

(continued)

Table 5.C-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	Degree of Confidence in Our Analytical Approach	Ability to Assess Uncertainty
Uncertainties Associated with Concentration-Response Functions (continued)				
Shape of the C-R functions, particularly at low concentrations	Both If there is a threshold (i.e., a level of exposure below which health effects do not occur), then the relative risk (i.e., steeper slope) estimates would be higher within the range of observed effects.	Medium For $PM_{2.5}$-related long-term mortality, the PM ISA concludes that a log-linear non-threshold model is best supported in the scientific literature (U.S. EPA, 2009b). Although consideration for alternative model forms (Krewski et al., 2009) does suggest that different models can impact effect estimates to a certain extent, generally this appears to be a moderate source of overall uncertainty.	High Our approach follows recommendations from the SAB (U.S. EPA-SAB, 2010a)	Tier 3 Assessed in LML assessment and the results derived from the expert elicitation
Impact of historical exposure on long-term effect estimates	Both Long-term studies of mortality suggest that different time periods of PM exposure can produce significantly different effects estimates, raising the issue of uncertainty in relation to determining which exposure window is most strongly associated with mortality.	Medium The Reanalysis II study (HEI, 2009) which looked at exposure windows (1979-1983 and 1999-2000) for long-term exposure in relation to mortality, did not draw any conclusions as to which window was more strongly associated with mortality. However, the study did suggest that moderately different effects estimates are associated with the different exposure periods (with the more recent period having larger estimates). Overall, the evidence for determining the window over which the mortality effects of long-term pollution exposures occur suggests a latency period of up to five years, with the strongest results observed in the first few years after intervention (PM ISA, section 7.6.4. p. 7-95).	Medium See PM risk assessment (U.S. EPA, 2010b)	Tier 2 To be assessed in final RIA

(continued)

Table 5.C-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	Degree of Confidence in Our Analytical Approach	Ability to Assess Uncertainty
Uncertainties Associated with Concentration-Response Functions (continued)				
Confounding by co-pollutants	Both	Medium For long-term health endpoints, the final ISA states, "Given similar sources for multiple pollutants (e.g., traffic), disentangling the health responses of co-pollutants is a challenge in the study of ambient air pollution." The PM ISA also notes that in some instances, consideration of co-pollutants can have a significant impact on effect estimates. For morbidity, the PM ISA concludes that observed associations are fairly robust to the inclusion of co-pollutants in the predictive models (see PM ISA).	Medium	Tier 1
Confounding by ecologic factors, such as SES or smoking	Both	Medium	Medium-High To minimize confounding, we selected the risk coefficient that controlled for ecologic factors from Krewski et al. (2009).	Tier 1
Exclusion of C-R functions from short-term exposure studies in PM mortality calculations	Underestimate	Medium PM/mortality is the top contributor to the benefits estimate. If short-term functions contribute substantially to the overall PM-related mortality estimate, then the benefits could be underestimated.	Medium 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.	Tier 1

(continued)

Table 5.C-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	Degree of Confidence in Our Analytical Approach	Ability to Assess Uncertainty
Uncertainties Associated with Valuation				
Value-of-a-Statistical-Life (VSL)	Both Some studies suggest that EPA's VSL is too high, while other studies suggest that it is too low. The VSL used by EPA is based on 26 labor market and stated preference studies published between 1974 and 1991.	High Mortality valuation generally dominates monetized benefits.	Medium 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.	Tier 2 Assessed uncertainty in mortality valuation using a Weibull distribution.
Cessation lag structure for long term PM mortality	Underestimate Recent studies (Schwartz, 2008) have shown that the majority of the risk occurs within 2 years of reduced exposure. EPA's current lag structure assumption was provided by the SAB, and it estimates 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 $PM_{2.5}$ (U.S. EPA-SAB, 2004c).	Medium Although the cessation lag does not affect the number of premature deaths attributable to $PM_{2.5}$ exposure, it affects the timing of those deaths and thus the discounted monetized benefits.	Medium The main cessation lag applied was recently confirmed by the SAB (U.S. EPA-SAB, 2010a).	Tier 2 Assessed in sensitivity analysis

(continued)

Table 5.C-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	Degree of Confidence in Our Analytical Approach	Ability to Assess Uncertainty
		Uncertainties Associated with Valuation (continued)		
Morbidity valuation	Underestimate Morbidity benefits such as hospital admissions and heart attacks are calculated using cost-of-illness (COI) estimates, which studies have shown (Alberini and Krupnick, 2000) are generally half as much as willingness-to-pay to avoid the illness. in addition, the quantified morbidity impacts do not reflect physiological responses or sequelae events, such as increased susceptibility for future morbidity.	Low Mortality valuation generally dominates monetized benefits.	Low Although the COI estimates for hospitalizations reflect recent data, other COI estimates such as for AMI have not yet been updated. Nevertheless, even current COI valuation estimates do not capture the full valuation of these morbidity impacts.	Tier 1
Income growth adjustments	Both Income growth increases willingness-to-pay 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).	Medium Adjusting for income growth is consistent with SAB recommendations (U.S. EPA,-SAB, 2000). 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 To be assessed in final RIA

(continued)

Table 5.C-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	Degree of Confidence in Our Analytical Approach	Ability to Assess Uncertainty
Uncertainties Associated with Baseline Incidence and Population				
Uncertainty in projecting baseline incidence rates for mortality	Both Baseline mortality rates are at the county level and projected for 5-year increments for multiple age groups. Due to data suppression for small numbers of specific age/gender/race/ethnicity combinations, many counties have missing baseline mortality rates.	Medium Mortality valuation generally dominates monetized benefits. The county-level baseline mortality rates reflect recent databases (i.e., 2004-2006). Also, the mortality rates projections for future years are internally consistent with population projections in that they reflect changes in mortality patterns as well as population growth.	Medium-High The mortality rate databases (CDC, 2008) are generally considered to have relatively low uncertainty. These projections account for both spatial and temporal changes.	Tier 1
Uncertainty in projecting baseline incidence rates and prevalence rates or morbidity	Both Morbidity baseline incidence is available for year 2000 only (i.e., no projections available).	Low Mortality valuation generally dominates monetized benefits. The magnitude of uncertainty associated with projections of morbidity baseline incidence varies with the health endpoint. 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 one national average.	Low There is no current method for projecting baseline morbidity rates beyond 2000. Asthma prevalence rates reflect recent increases in baseline asthma rates (i.e., 2008).	Tier 1

(continued)

Table 5.C-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	Degree of Confidence in Our Analytical Approach	Ability to Assess Uncertainty
Uncertainties Associated with Baseline Incidence and Population (continued)				
Population estimates and projections	Both 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 projections accuracy can vary by locality.	Medium These projections cannot account for future population migration due to possible catastrophic events.	Tier 1
Uncertainties Associated with Omitted Categories				
Unquantified PM health benefit categories, such as pulmonary function, cerebrovascular events or low birth weight	Underestimate	Medium Mortality valuation generally dominates monetized benefits, but it is possible that some of these omitted categories could be significant, especially for morbidity.	Low Current data and methods are insufficient to develop (and value) national quantitative estimates of the health effects of these pollutants.	Tier 1
Unquantified health benefit categories for components of PM, such as air toxics (organics and metals)	Underestimate	Medium Studies have found air toxics cancer risks to be orders of magnitude lower than overall risks from criteria pollutants. However, air toxics can also be associated with cardiovascular, reproductive, respiratory, developmental, and neurological risks with potentially synergistic effects.	Low Current data and methods are insufficient to develop (and value) national quantitative estimates of the health effects of these pollutants.	Tier 1

5.C.2 References

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CHAPTER 6
WELFARE BENEFITS ANALYSIS APPROACH

6.1 Important Caveats Regarding this Chapter

Due to data limitations for this proposed rule, we were unable to calculate changes in light extinction associated with emission reductions from the illustrative control strategies, which are necessary for calculating the visibility benefits. Instead, this chapter and associated appendix describe in detail the methodology for calculating visibility benefits to encourage comment on the revised approach.

6.2 Synopsis

Emission reductions associated with the illustrative control strategies to attain alternative combinations of the PM NAAQS have numerous documented effects on environmental quality that affect human welfare, 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. 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. Due to data limitations for this proposed rule, we are unable to estimate the recreational visibility and residential visibility benefits associated for alternative standard combinations in 2020 even though we have a complete methodology for estimating benefits for scenarios with light extinction estimates for both the baseline and control scenarios. We intend to apply the approach described in this chapter in the RIA accompanying the final rulemaking, and as such, we solicit comment here. Despite our goal to quantify and monetize as many of the benefits as possible, welfare benefits remain unquantified and nonmonetized in this analysis due to data, methodology, and resource limitations. The monetized value of these unquantified effects is represented by adding an unknown "B" to the aggregate total benefits. These unquantified welfare benefits may be substantial, although the magnitude of these benefits is highly uncertain.

6.3 Introduction to Welfare Benefits Analysis

Emission reductions associated with the illustrative control strategies to attain alternative combinations of the PM NAAQS have numerous documented effects on environmental quality that affect human welfare. We define 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. EPA's Integrated Science Assessments for PM (hereafter, "PM ISA") (U.S. EPA, 2009b) and NO_X/SO_X—Ecological Criteria (U.S. EPA, 2008), as well as the Criteria Document for ozone (U.S. EPA, 2006) 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 nitrogen and sulfur emissions, vegetation effects from ozone exposure, ecological effects from mercury deposition, and climate effects.

These welfare benefits are associated with reductions in emissions of specific pollutants resulting from emissions controls applied to attain the suite of PM standards, 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 emission reductions associated with the illustrative control strategies can be grouped into four categories: directly emitted PM (e.g., metals, organic compounds, dust), reductions of $PM_{2.5}$ precursors (e.g., NOx, SOx, VOCs), other ancillary reductions from illustrative control strategies (e.g., mercury and CO_2), and secondary co-pollutant formation from PM precursors (e.g., ozone from NOx 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 associated with these emission reductions to provide a comprehensive understanding of the likely public impacts of attaining alternative standard level combinations. 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 control strategies to attain the alternative standard level combinations.

Based on previous EPA analyses, we believe the welfare 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 as possible, welfare benefits remain unquantified and nonmonetized in this analysis due to data, methodology, and resource limitations. For the final rulemaking, we anticipate that visibility would be the only welfare benefit category with sufficient data to quantify monetized benefits. Although it is possible to estimate some of the acidification and ozone vegetation benefits, we are limited by the time

Table 6-1. Welfare Effects by Pollutants Potentially Affected by Attainment of the PM NAAQS

	Atmospheric Effects			Atmospheric and Deposition Effects		Deposition Effects			
Pollutant	Visibility Impairment	Vegetation Injury (SO_2)	Vegetation Injury (Ozone)	Materials Damage	Climate	Ecosystem Effects—(Organics & Metals)	Acidification (freshwater)	Nitrogen Enrichment	Mercury Methylation
Direct $PM_{2.5}$	✓			✓	✓	✓			
NOx	✓		✓	✓	✓		✓	✓	
SO_2	✓	✓		✓	✓		✓		✓
VOCs	✓		✓	✓		✓			
PM_{10}	✓			✓	✓				
Hg						✓			✓
CO_2					✓				

✓ = Welfare category affected by this pollutant.

and resources available, and we do not anticipate being able to quantify these benefits in the final rulemaking. The other welfare effects have additional data and methodology limitations that preclude us from monetizing those benefits. Therefore, the total benefits would be larger than we have estimated in this analysis. The monetized value of these unquantified effects is represented by adding an unknown "B," which includes both unmonetized health and welfare effects, to the aggregate total for the cost-benefit comparison. These unquantified benefits may be substantial, although the magnitude of these benefits is highly uncertain. For these categories of welfare benefits that we are unable to quantify in this analysis, we include a qualitative analysis of the anticipated effects in this chapter to characterize the type and potential extent of those benefits. In Table 6-2, we identify the quantified and unquantified welfare benefits.

The remainder of this chapter is organized as follows: Section 6.3 provides the methodology for the visibility benefits analysis. Sections 6.4 through 6.6 provide qualitative benefits for the unquantified benefits categories of materials damage, climate, and ecosystem benefits. References are provided in Section 6.7. Additional information regarding technical details of the visibility benefits analysis is provided in Appendix 6a.

Table 6-2. Quantified and Unquantified Welfare Benefits

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	–[1]	–[1]	Section 6.3
	Visibility in Class I areas in other regions	–	–[4]	Section 6.3
	Visibility in 8 cities	–[1]	–[1]	Section 6.3
	Visibility in other residential areas	–	–[4]	Section 6.3
Reduced climate effects	Global climate impacts from CO_2	–	–	SCC TSD[1]
	Climate impacts from ozone and PM	–	–	Ozone CD, Draft Ozone ISA, PM ISA[2]
	Other climate impacts (e.g., other GHGs, other impacts)	–	–	IPCC[2]
Reduced effects on materials	Household soiling	–	–	PM ISA[2]
	Materials damage (e.g., corrosion, increased wear)	–	–	PM ISA[2]
Reduced effects from PM deposition (metals and organics)	Effects on Individual organisms and ecosystems	–	–	PM ISA[2]
Reduced vegetation and ecosystem effects from exposure to ozone	Visible foliar injury on vegetation	–	–	Ozone CD, Draft Ozone ISA[2]
	Reduced vegetation growth and reproduction	–	–	Ozone CD, Draft Ozone ISA[1]
	Yield and quality of commercial forest products and crops	–	–	Ozone CD, Draft Ozone ISA[1,3]
	Damage to urban ornamental plants	–	–	Ozone CD, Draft Ozone ISA[2]
	Carbon sequestration in terrestrial ecosystems	–	–	Ozone CD, Draft Ozone ISA[2]
	Recreational demand associated with forest aesthetics	–	–	Ozone CD, Draft Ozone ISA[2]
	Other non-use effects			Ozone CD, Draft Ozone ISA[2]
	Ecosystem functions (e.g., water cycling, biogeochemical cycles, net primary productivity, leaf-gas exchange, community composition)	–	–	Ozone CD, Draft Ozone ISA[2]

(continued)

Table 6-2. Quantified and Unquantified Welfare Benefits (continued)

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Improved Environment (continued)				
Reduced effects from acid deposition	Recreational fishing	–	–	NOx SOx ISA[1]
	Tree mortality and decline	–	–	NOx SOx ISA[2]
	Commercial fishing and forestry effects	–	–	NOx SOx ISA[2]
	Recreational demand in terrestrial and aquatic ecosystems	–	–	NOx SOx ISA[2]
	Other non-use effects			NOx SOx ISA[2]
	Ecosystem functions (e.g., biogeochemical cycles)	–	–	NOx SOx ISA[2]
Reduced effects from nutrient enrichment	Species composition and biodiversity in terrestrial and estuarine ecosystems	–	–	NOx SOx ISA[2]
	Coastal eutrophication	–	–	NOx SOx ISA[2]
	Recreational demand in terrestrial and estuarine ecosystems	–	–	NOx SOx ISA[2]
	Other non-use effects			NOx SOx ISA[2]
	Ecosystem functions (e.g., biogeochemical cycles, fire regulation)	–	–	NOx SOx ISA[2]
Reduced vegetation effects from ambient exposure to SO2 and NOx	Injury to vegetation from SO_2 exposure	–	–	NOx SOx ISA[2]
	Injury to vegetation from NO_x exposure	–	–	NOx SOx ISA[2]
Reduced ecosystem effects from exposure to methylmercury (through the role of sulfate in methylation)	Effects on fish, birds, and mammals (e.g., reproductive effects)	–	–	Mercury Study RTC[2,3]
	Commercial, subsistence and recreational fishing	–	–	Mercury Study RTC[2]

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

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

[3] 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.

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

6.4 Visibility Benefits

6.4.1 Visibility and Light Extinction

The illustrative strategies designed to attain alternative standard level combinations would reduce emissions of directly emitted $PM_{2.5}$ as well as precursor emissions such as NO_x and SO_2. These emission reductions would improve the level of visibility throughout the United States because these suspended particles and gases impair visibility by scattering and absorbing light (U.S. EPA, 2009b).[1] 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 benefits associated with improved visibility as a result of emission reductions associated with the alternative $PM_{2.5}$ standard level combinations.

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.

[1] The visibility benefits results shown in this section only reflect the emission reductions associated with attaining the alternative $PM_{2.5}$ primary standards. Visibility benefits results associated with attaining alternative secondary PM NAAQS levels are provided in Chapter 13 of this RIA.

[2] We use the term VAQ to refer to the visibility effects caused solely by air quality conditions, excluding fog.

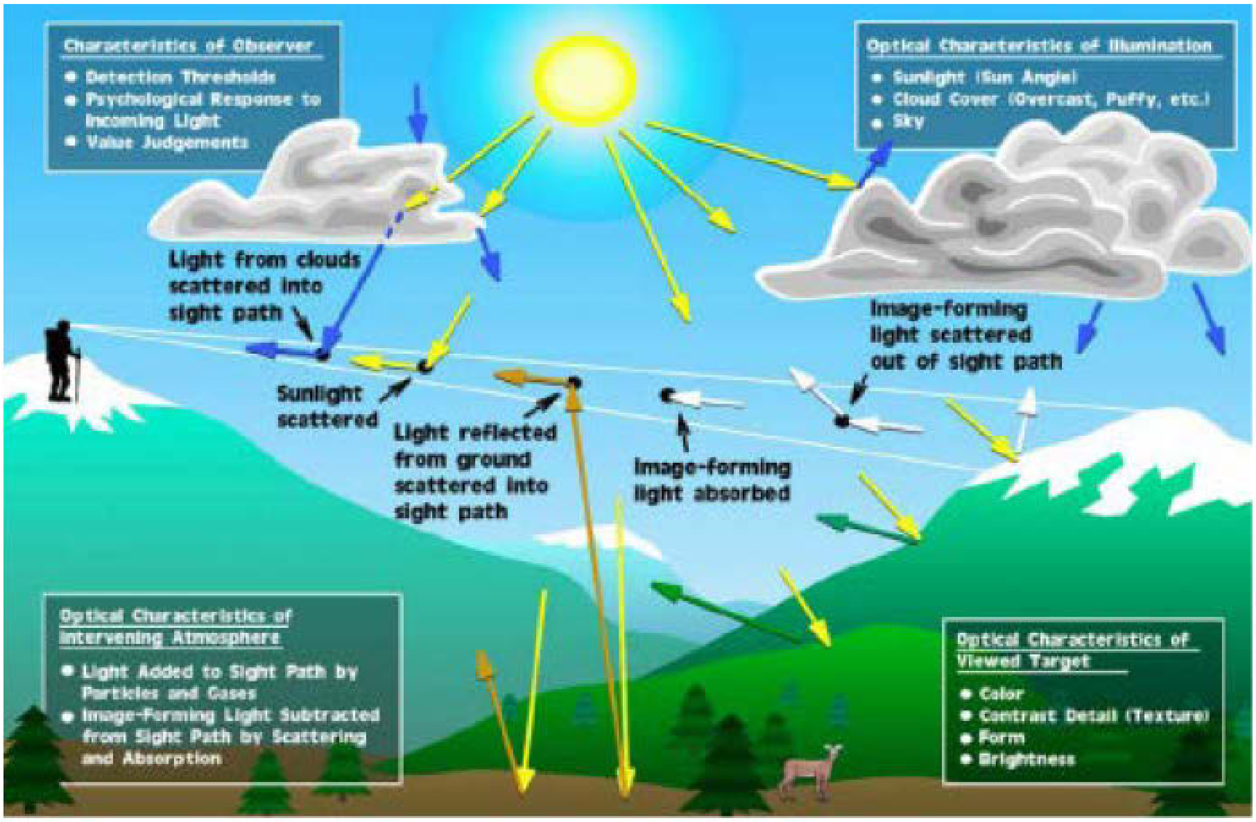

Figure 6-1. Important Factors Involved in Seeing a Scenic Vista (Malm, 1999)

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 benefits analysis 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. The secondary PM NAAQS uses a different averaging time than the benefits analysis (see Chapter 13).

$$Deciviews = 10 * \ln\left(\frac{391}{VR}\right) = 10 * \ln\left(\frac{\beta_{ext}}{10}\right)$$

where VR denotes visual range (in kilometers) and β_{ext} denotes light extinction (in Mm^{-1}).[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 NOx 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] Recent EPA 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

[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).

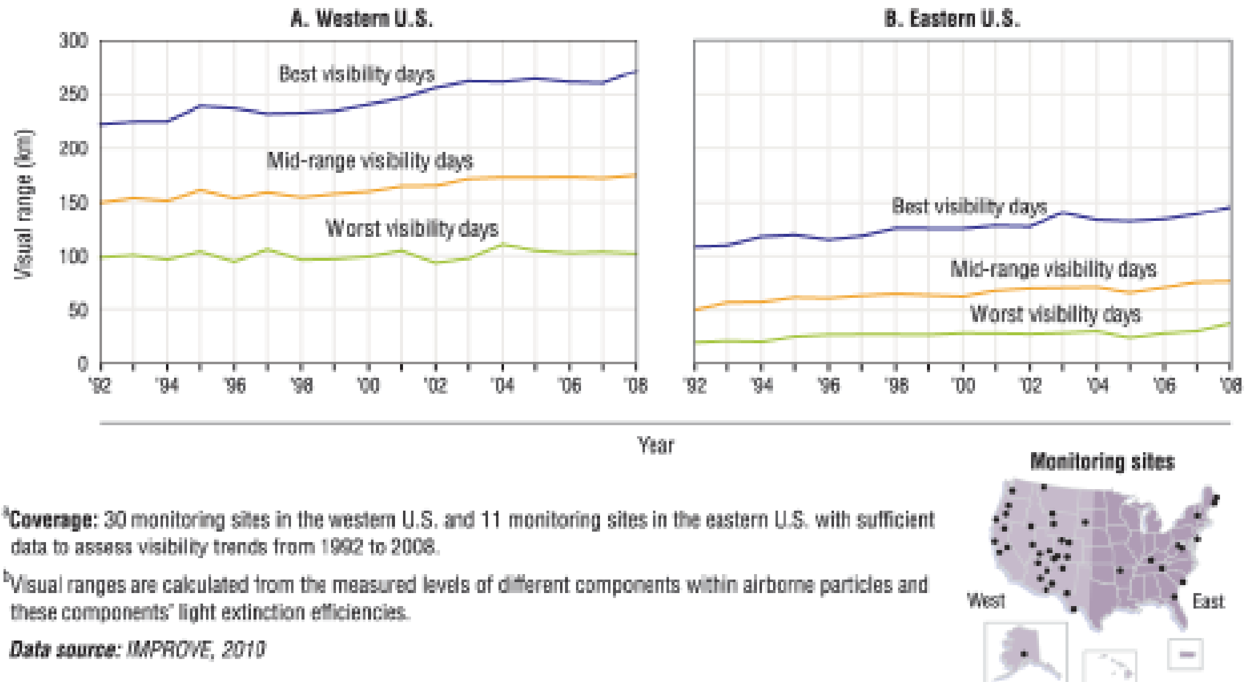

Figure 6-2. Visibility in Selected National Parks and Wilderness Areas in the U.S., 1992–2008[a,b]

(Source: U.S. EPA (2008) updated, IMPROVE (2010))

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 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).

For the final rulemaking, we would 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).[7] The procedure for calculating light extinction

[7] According to the PM ISA, the algorithm performs reasonably well despite its simplicity (U.S. EPA, 2009b).

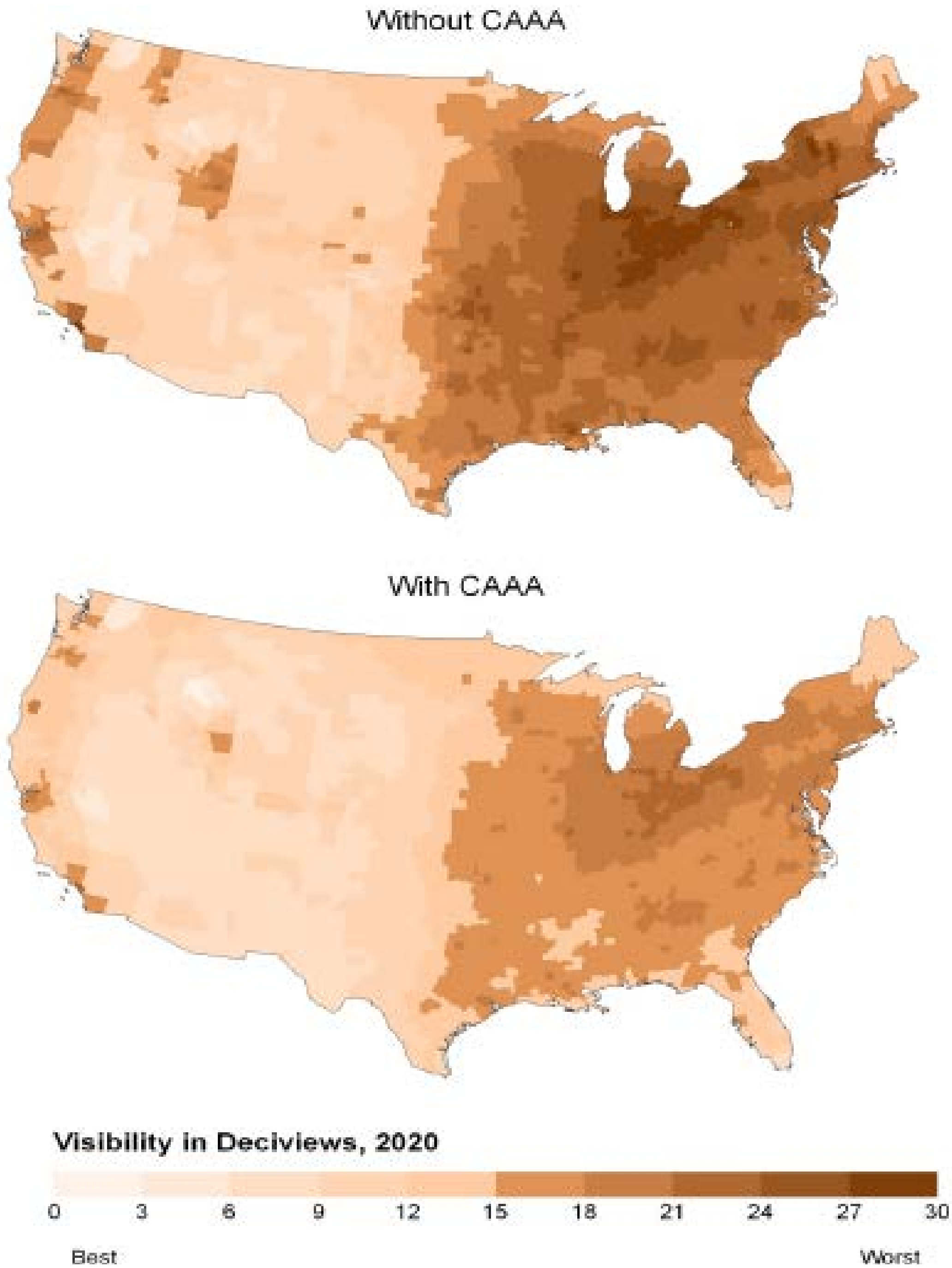

Figure 6-3. Estimated Improvement in Annual Average Visibility Levels Associated with the CAAA Provisions in 2020

Source: U.S. EPA, 2011b[8]

[8] It is important to note that visibility levels shown in these maps were modeled differently than the modeling conducted for this analysis, including coarser grid resolution (i.e., 36 km instead of 12 km). In addition, these maps present annual average visibility levels, which are different than the short-term averages being considered for the secondary standard.

associated with alternative standard level combinations is described in detail in Chapter 3 of this RIA. In addition, Appendix 6a describes how the spatial resolution of the light extinction estimates was then adjusted for the benefits analysis.

It is important to note that the light extinction estimates used in this benefits analysis 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 alternative standard level combinations, we are not able to quantify those impacts. These data gaps result in an underestimate of visibility benefits associated with extreme days. We recognize that recent advice from EPA's Science Advisory Board recommends estimating visibility benefits considering daytime visibility on days with severe impairment (U.S. EPA-SAB, 2010), but the available data and valuation studies do not allow such fine temporal resolution.

While we have made substantial improvements in estimating light extinction nationally in this analysis, 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, we provide sensitivity analyses to show the potential impact of omitting coarse particles from the recreational and residential visibility benefits analysis. For this sensitivity analysis, 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 (Debell et al., 2006). We estimate the sensitivity of impacts on recreational and residential visibility benefits using four levels of coarse particles: no coarse particles, 5 $\mu g/m^3$ nationwide, 15 $\mu g/m^3$ in the Southwest with 5 $\mu g/m^3$ in the rest of the country, and 15 $\mu g/m^3$ in the Southwest with 8 $\mu g/m^3$ in the rest of the country.[9]

In Table 6-3, we also provide a qualitative assessment of how key assumptions in the estimation of light extinction affect the visibility benefits.

[9] We define "Southwest" for this sensitivity analysis to be the states of California, Nevada, Utah, Arizona, New Mexico, Colorado, and Texas.

Table 6-3. Key Assumptions in the Light Extinction Estimates Affecting the Visibility Benefits Analysis[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 5C of this RIA.

6.4.2 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.[10] 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 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 to estimate the residential and recreational visibility benefits

[10] See Section 169(a) of the Clean Air Act.

associated with alternative standard level combinations 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 benefits associated with national regulations that reduce air pollution (Leggett and Neumann, 2004).

In this 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 this analysis attempts to isolate visibility from other effect categories, but the different studies take different approaches (U.S. EPA, 2009b).[11] Because we believe that residual potential for double-counting visibility and health effects is relatively minimal, we do not further adjust the benefits to account for potentially embedded health effects beyond what the studies have already done.

Similarly, it is important to try to distinguish residential visibility from recreational visibility benefits, specifically whether these can these be treated as distinct and additive benefit categories based on the available literature. In this analysis, we assume that residential and recreational visibility 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. However, we believe that the potential for double-counting recreational and residential visibility is minimal for several reasons. First, we only include a subset of areas in the primary estimates of recreational and residential visibility benefits, which overlap in only a few places.[12] Second, a number of the overlapping counties are wilderness areas, which contribute little to the overall monetized 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 exclude the residential visibility benefits that accrue to 10 million residents in Los Angeles County and only include the very small recreational visibility benefits for the wilderness area, we would be substantially biasing

[11] See Leggett and Neumann (2004) for a more detailed discussion of this issue.
[12] As described in detail in Sections 6.3.3 and 6.3.4, we only include a subset of visibility benefits in the primary benefits estimates, while providing the rest of the visibility benefits in sensitivity analyses.

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 analysis. Consistent with the health benefits analysis, the monetized visibility benefits would be adjusted for inflation and income growth. These benefits would be specific to the analysis year, and as population and income increase over time, these benefits can be expected to increase each year for the same incremental change in light extinction.

6.4.3 Recreational Visibility

6.4.3.1 Methodology

The methodology for estimating recreational visibility benefits in this RIA follows a well-established approach that has been used in numerous EPA analyses (U.S. EPA, 2006; U.S. EPA, 2005; U.S. EPA, 2010; U.S. EPA, 1999; U.S. EPA, 2011b). For the purposes of this analysis, recreational visibility benefits apply to Class I areas, such as National Parks and Wilderness Areas.[13] 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 I areas, with the goal of achieving 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 U.S. 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 I areas across the country (U.S. EPA, 2009b).[14] 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.

[13] 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. Mandatory 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. EPA has designated 156 areas as mandatory Class I 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).

[14] The formula used to estimate light extinction from concentrations of PM constituents and relative humidity is referred to as the IMPROVE algorithm.

Figure 6-4. Mandatory Class I Areas in the U.S.

For recreational visibility, EPA has determined that only one existing study provides adequate monetary estimates of the value of changes in recreational visibility: a contingent valuation (CV) survey conducted by Chestnut and Rowe in 1988 (1990a; 1990b). Although there are several other studies in the literature on recreational visibility valuation, they are older and use less robust methods. In EPA's judgment, the Chestnut and Rowe study contains many of the elements of a valid CV study and is sufficiently reliable to serve as the basis for monetary estimates of the benefits of visibility changes in recreational areas.[15] This study serves as an essential input to our estimates of the benefits from improving recreational visibility.

In this analysis, we assume that the household WTP is higher if the Class I 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

[15] In 1999, EPA's SAB stated, "many members of the Council believe that the Chestnut and Rowe study is the best available" study for recreational visibility valuation" (U.S. EPA-SAB, 1999). In July 2010, the SAB stated that the studies were dated, but EPA "used what the Council understands to be the only relevant studies." (U.S. EPA-SAB, 2010)

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 I 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), this analysis calculates the benefits from households for a particular region for a given change in visibility at a particular Class I 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 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 I areas.

To value recreational visibility improvements associated with its rulemakings, EPA developed a valuation WTP equation function based on the baseline of visibility, the magnitude

[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.

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 analysis of the Chestnut and Rowe survey (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 their region takes the following form:

$$WTP(\Delta Q_{ik}) = m - [m^{\rho} + \gamma_{ik} * (Q_{0ik}^{\rho} - Q_{1ik}^{\rho})]^{\frac{1}{\rho}}$$

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 6a of this RIA, this 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, 1997). Using the income elasticity calculated by Chestnut (1997), the 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

[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).

the parks are more numerous, so collective WTP is likely higher).[19] We also adjust the benefits for inflation and growth in real income.

Recreational visibility benefits are 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 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 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.

The primary estimate for recreational visibility only includes benefits for 86 Class I areas in the original study regions (i.e., California, the Southwest, and the Southeast).[20] These benefits reflect the value to households living in the same region as the Class I area as well as values for all households in the United States living outside the state containing the Class I area.

The Chestnut and Rowe study did not measure values for visibility improvement in Class I areas in the Northwest, Northern Rockies, and Rest of U.S. regions.[21] Their study covered 86 of the 156 Class I areas in the United States. We can infer the value of visibility changes in the 70 additional Class I areas by transferring values of visibility changes at Class I areas in the study regions.[22] In order to obtain estimates of WTP for visibility changes for parks in these additional regions, we have to transfer the WTP values from the studied regions. This 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

[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 I areas in the three studied park regions represented 68% of the total visitor days to Class I areas in 2008 (NPS, 2008).

[21] The Northwest region is defined as the states of Washington and Oregon. The Northern Rockies region include 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 I areas represented 32% of the total visitor days to Class I areas in 2008 (NPS, 2008).

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 benefits transfer method used to infer values for visibility changes in Class I 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. Figure 6-5 shows how the visitation rates vary across Class I areas and regions and indicates whether each Class 1 area is located within one of the studied regions.

6.4.3.2 Recreational Visibility Limitations, Caveats, and Uncertainties

This analysis 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 benefits. Each of these inputs may contain uncertainty that would affect the recreational visibility benefits estimates. Though we are unable to quantify the cumulative effect of all of these uncertainties in this analysis, 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 recreational visibility benefits estimate 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.

[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.

[24] See Chapter 4 for more information on model evaluation.

Table 6-4. WTP for Visibility Improvements in Class I 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

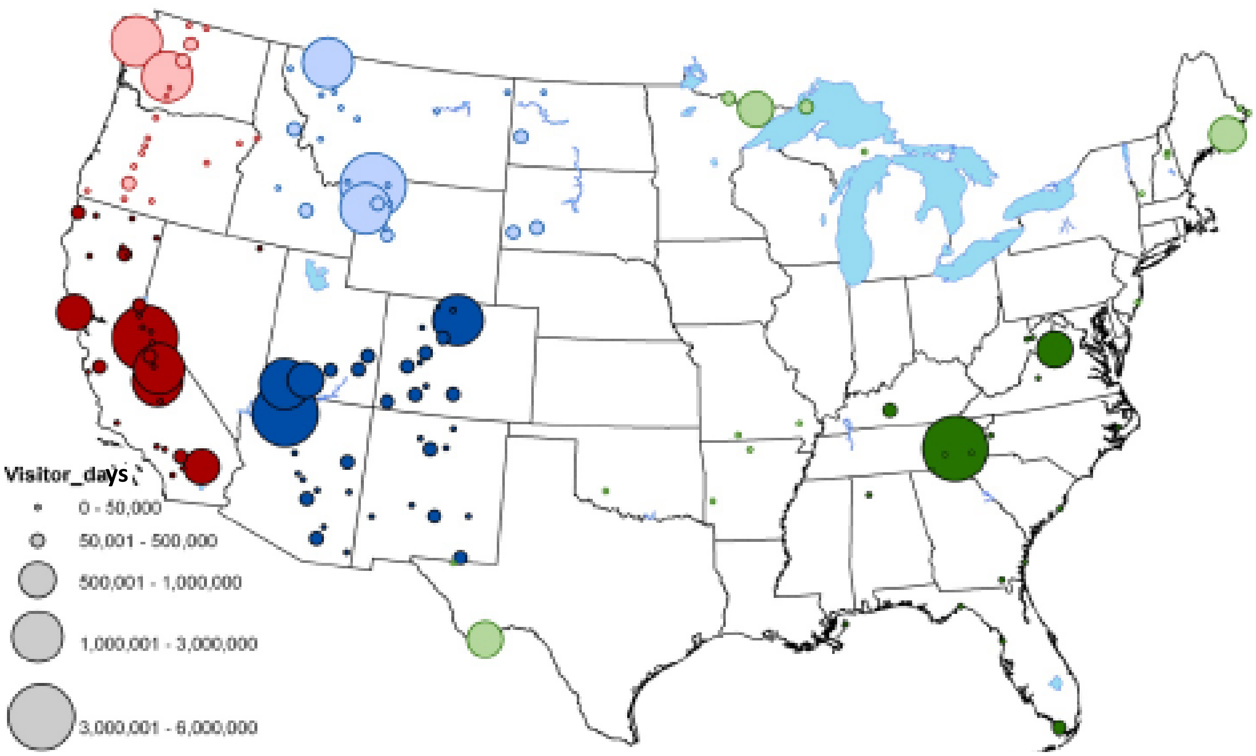

Figure 6-5. Visitation Rates and Park Regions for Class I Areas*

* 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).

Table 6-5. Summary of Key Assumptions in the Recreational Visibility 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 this analysis.	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 5C of this RIA.

Since the valuation of recreational visibility benefits relies upon one study (Chestnut and Rowe, 1990a; 1990b), all of the uncertainties within that study also pertain to this analysis. 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 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 this analysis, 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. For the purpose of the benefits analysis for this rule, 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.

6.4.4 Residential Visibility

6.4.4.1 Methodology

Residential visibility benefits are those that occur from visibility changes in urban, suburban, and rural areas where people live. These 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 I areas. For example, the State of Colorado established a local visibility standard for the Denver metropolitan area in 1990 (Ely et al., 1991). For the purposes of this analysis, 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 EPA to inform the 2006 PM NAAQS review (Abt Associates Inc., 2001). While these studies indicate that visual air quality associated with ambient levels of air pollution in urban areas have been deemed unacceptable, they do not provide sufficient information on which to develop monetized benefits estimates. Specifically, the public perception studies do not provide preferences expressed in dollar values, even though they do provide additional evidence that the benefits associated with improving residential visibility are not zero.

A wide range of published, peer-reviewed literature supports a non-zero value for residential visibility (Brookshire et al., 1982; Rae, 1983; Tolley et al., 1984; Chestnut and Rowe, 1990c; McClelland et al., 1993; Loehman et al., 1994). Furthermore, Chestnut and Rowe (1990c) conclude that residential visibility benefits are likely to be at least as high as recreational visibility 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, 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).[26] 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)[27] and reviewed by the SAB (U. S. EPA-SAB, 2004, 2010).

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[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 from the Brookshire et al. (1979), Loehman et al. (1985) and Tolley et al. (1984) 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:

$$WTP\ (\Delta VR) = \beta^* \ln\left(\frac{VR_1}{VR_0}\right)$$

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.

Total residential visibility benefits within a particular MSA are driven by visibility improvements, population density, and the WTP value applied. Only those people living within

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[28] 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 specific details need to compute visibility benefits using Tolley et al. (1984) were not subsequently published, but the overall work including study and survey design was subject to peer review during study development. (see Leggett et al, 2004 and Patterson et al., 2005) In addition, Tolley et al. subsequently published a book based on this research, which notes in the preface that the methods were critiqued throughout by various external economists (Tolley et al., 1988).

[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).

in an MSA are assumed to receive 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. (1984)	321	$31	$72
Boston	Tolley et al. (1984)	398	$38	$89
Chicago	Tolley et al. (1984)	310	$30	$69
Denver	Tolley et al. (1984)	696	$66	$155
Los Angeles	Brookshire et al. (1979)	94	$9	$21
Mobile	Tolley et al. (1984)	313	$30	$70
San Francisco	Loehman et al. (1985)	989	$94	$220
Washington, DC	Tolley et al. (1984)	614	$59	$137

[a] The table assumes full attainment of the alternative standard level combinations. Because these benefits occur within the analysis year, the monetized benefits are the same for all discount rates. These benefits reflect the WTP for households who live in MSAs.

Similar to recreational visibility 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. 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:

$$WTP(\Delta VR) = m - [m^{\rho} + \theta * (VR_0^{\rho} - VR_1^{\rho})]^{\frac{1}{\rho}}$$

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 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, the sensitivity analysis for residential visibility includes the benefits extrapolated to the 351 additional MSAs.[30] Because there is considerable uncertainty about the validity of this benefit transfer approach, these extrapolated benefits are included in a sensitivity analysis only.

There are many factors that could influence WTP for residential visibility, and these factors vary across urban areas. For the purpose of this analysis, we utilize the benefit transfer approach developed for the Second Prospective 812 analysis, 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

[30] The 351 additional MSAs plus the 8 study area MSAs represent 84% of the total US population in 2020 (U.S. Census).

[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.

despite exceeding the elevation threshold.[32] Figure 6-6 indicates the study cities as well as the assignment of the other MSAs to the study cities.

6.4.4.2 Residential Visibility Limitations, Caveats, and Uncertainties

Similar to recreational visibility benefits, there are many data inputs into the residential visibility benefits that contribute to overall uncertainty. We provide sensitivity analyses to characterize major omissions (i.e., benefits in other MSAs and coarse particles). A summary of the key assumptions including direction and magnitude of bias is provided in Table 6-7.

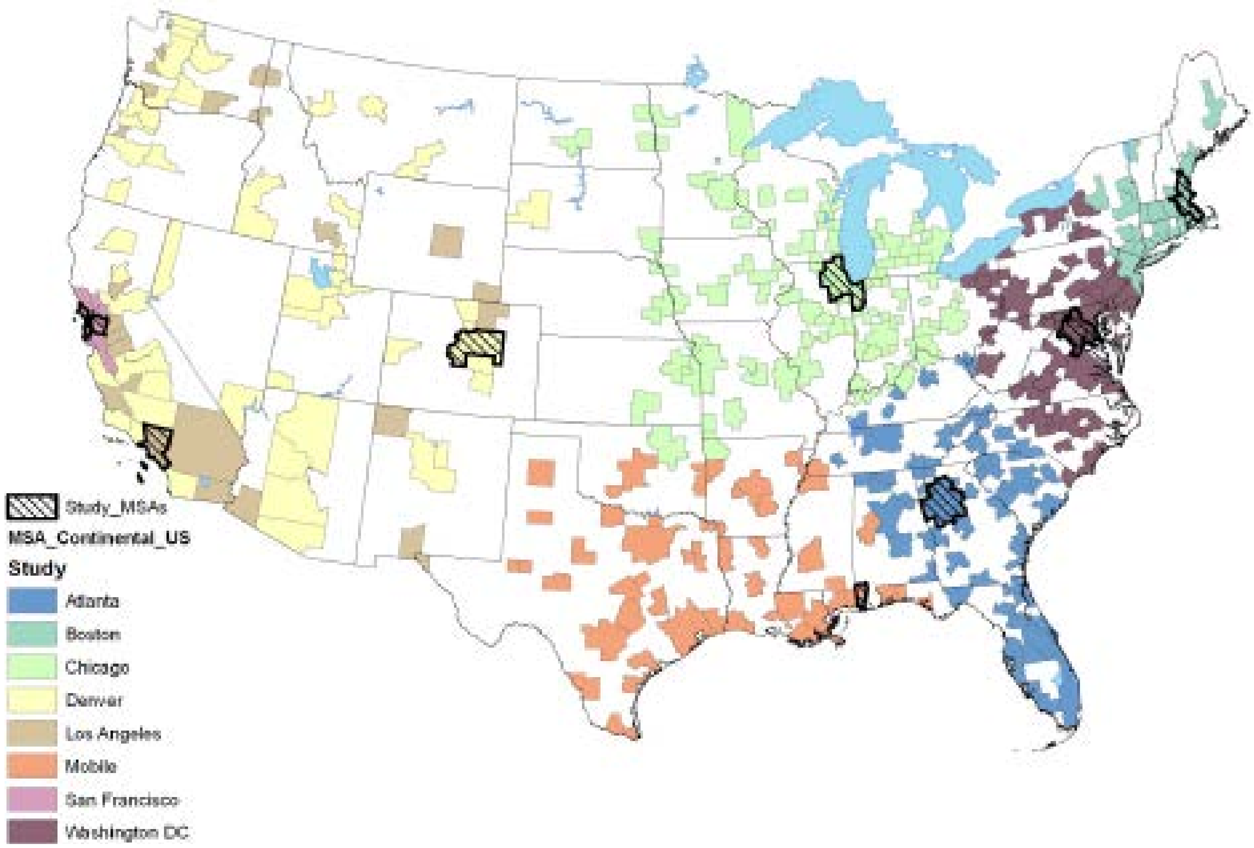

Figure 6-6. Residential Visibility Study City Assignment

[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.

Table 6-7. Summary of Key Assumptions in the Residential Visibility 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 this analysis.	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 5C of this RIA.

The valuation studies relied upon for the residential visibility 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 this analysis, 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.

6.4.5 Discussion of Visibility Benefits

As described in the previous sections of this chapter, the estimation of visibility benefits is complex and suffers from unavoidable limitations. While we are confident that the underlying scientific literature supports a non-zero estimate for visibility benefits attributable to emission reductions, we are less confident in the magnitude of those benefits outside of previously studied locations. While acknowledging these limitations, it is important to emphasize that these valuation studies have withstood intense scrutiny (U.S. EPA-SAB, 2010). To minimize uncertainties related to extrapolation and double counting, we only include a subset of monetized visibility benefits in the primary benefits estimate to correspond with our higher level of confidence in recreational benefits within the study regions and residential benefits within the study cities. Although we are confident that visibility benefits extend beyond these studied areas, we are less confident about the magnitude of those benefits.

In the approach described here, we have revised several aspects of the visibility benefits analysis since previous RIAs, including light extinction estimation methods, visitation data for Class I areas (used in extrapolating benefits), valuation studies for residential visibility benefits, and the benefit transfer technique for residential benefits. Including residential visibility benefits in the primary 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 I 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 benefits, which have not undergone a similar expansion as the health literature. In fact, to our knowledge, no peer-reviewed studies have been published in the past 10 years on visibility valuation that we are able to incorporate into this analysis.[33] When EPA's Scientific Advisory Council reviewed the visibility benefits analysis for the Second Prospective 812 analysis, they also lamented on the need for additional research to improve methods and estimates (U.S. EPA-SAB, 2010). Because of time and resource constraints, performing original

[33] While several studies using hedonic valuation techniques to value air quality have been published in the last 10 years (Kim et al., 2010; Bayer et al., 2009; Anselin and Lozano-Gracia, 2006, 2008; Chay and Greenstone, 2005), these studies do not provide any mechanism to distinguish visibility from health or ecosystem effects.

research for regulatory analyses of specific policy actions is infeasible. Therefore, we actively encourage and solicit new research to address many of the limitations in our analysis. 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.

For these reasons, EPA requests public comment on the approach taken here to quantify the monetary value of changes in recreational and residential visibility. Specifically, we request comment on additional valuation studies, methods for benefit transfer, and methods for characterizing uncertainty.

6.5 Materials Damage 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 NOx although mixtures of NOx, 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 NOx/SOx 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 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 benefits of reduced PM household soiling.

In order to estimate the monetized 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 EPA benefits analyses 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 benefits are significantly smaller than the health benefits associated with reduced exposure to $PM_{2.5}$ and ozone, or even visibility benefits. However, studies of materials damage to historic buildings and outdoor artwork in Sweden (Grosclaude and Soguel, 1994) indicate that these benefits could be an order of magnitude larger than household soiling benefits.

In the absence of quantified 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)

6.6 Climate Benefits

Actions taken by state and local governments to implement the proposed $PM_{2.5}$ standards are likely to have implications for climate change because emission controls 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 proposed standards is quite 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 control 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 proposed standards because of the differences in warming and cooling potential of the difference pollutants.

6.6.1 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 (currently undergoing peer review), 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 benefits of measures to reduce SLCFs including methane, ozone, and black carbon assessing the health, climate, and agricultural 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).

6.6.1.1 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, indirectly, and through snow and ice albedo. Specifically, 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. 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. Finally, when BC is deposited on snow and ice, it darkens the surface and decreases reflectivity, thereby increasing absorption and accelerating melting.

The illustrative control strategies evaluated for this proposal 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 EPA traditionally estimates EC emissions rather than BC and for the purpose of this analysis these measures are essentially equivalent. Emissions reductions discussed below are from the modeled scenarios for each regulatory alterative, and not from the full attainment scenarios. This is because speciated $PM_{2.5}$ data were not available for emissions reductions beyond known controls.

The snow/ice albedo effects from BC deposition have been linked to accelerated snow and ice melting (Wiscombe and Warren, 1980). While many glaciers around the world and Arctic sea ice have receded in recent decades, determining whether this phenomenon is attributable to BC is challenging due to other contributing factors. Emissions north of the 40th parallel latitude are thought to be particularly important for BC's climate related effects in the Arctic (Shindell, 2007; Ramanathan and Carmichael, 2008).

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. National elemental carbon and organic carbon deposition maps are included in Appendix 6B to this report.

6.6.1.2 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.

6.6.1.3 Climate Effects of Ozone

Ozone changes due to this proposed regulation are not estimated for this analysis but may occur due to the NOx 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 discernable 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).

6.6.1.4 SLCFs Summary and Conclusions

Assessing the net climate impact of SLCFs for the illustrative emission control 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 direct emitted $PM_{2.5}$ controls 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.

6.6.2 Climate Effects of Long-Lived Greenhouse Gases

The importance of mitigating long-lived climate gases such as CO_2 has been stressed by the Intergovernmental Panel on Climate Change (IPCC 2007). While addressing short-lived climate forcers may result in more immediate climate benefits in specific areas, long-term policies must deal with long lived GHGs to address long-term climate change. We are unable to quantify the impact of the illustrative control 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.

6.7 Ecosystem 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 EPA science assessments (U.S. EPA, 2006a; U.S. EPA, 2008c; U.S. EPA, 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. 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).

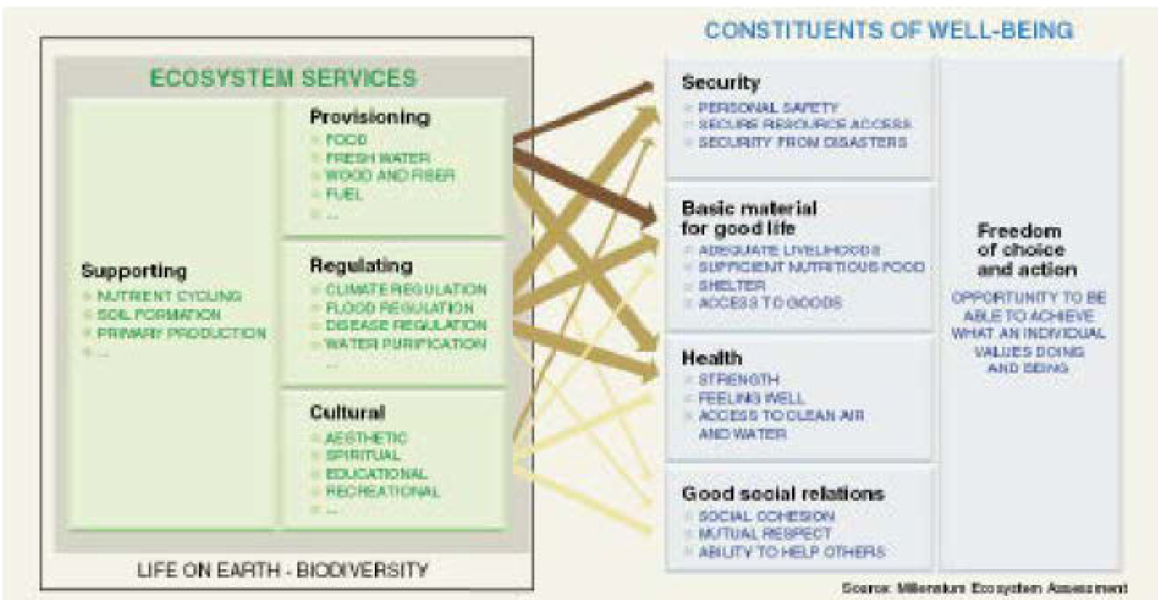

Figure 6-7. Linkages between Categories of Ecosystem Services and Components of Human Well-Being from Millennium Ecosystem Assessment (MEA, 2005)

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.

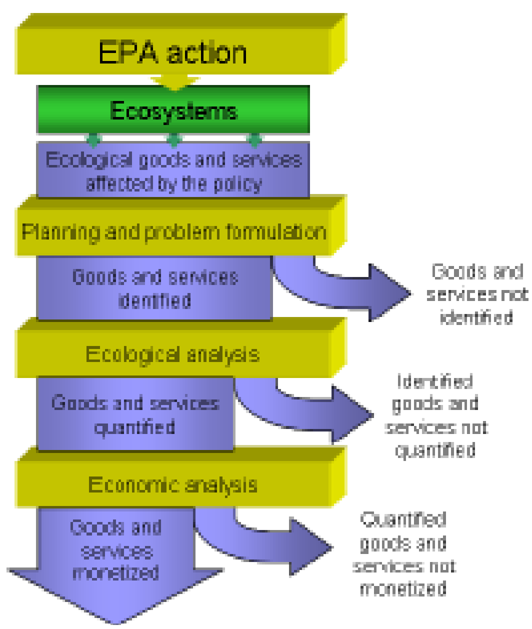

Figure 6-8. Schematic of the Benefits Assessment Process (U.S. EPA, 2006c)

6.7.1 Ecosystem 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 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.

6.7.2 Ecosystem 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.

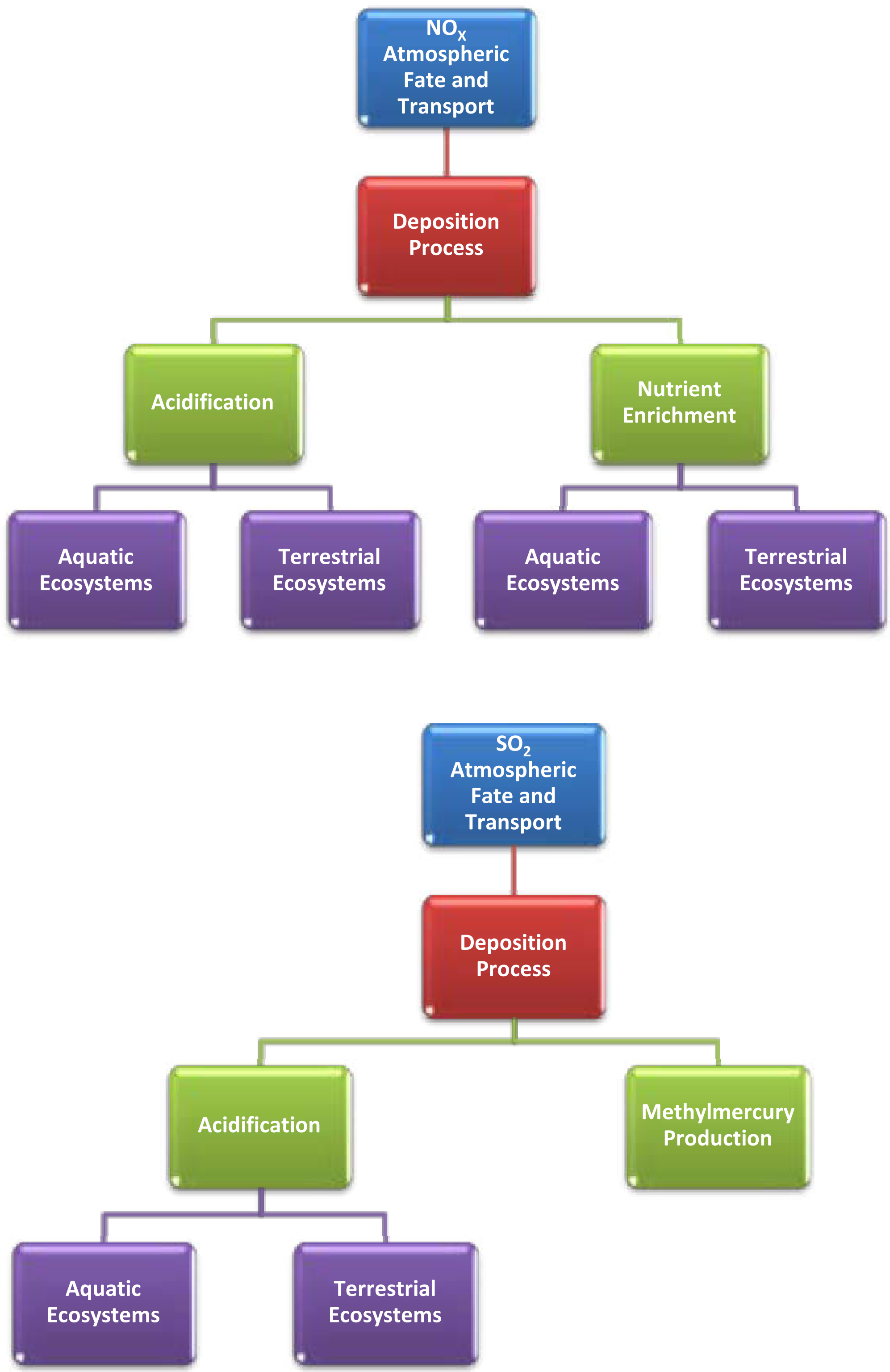

Figure 6-9. Schematics of Ecological Effects of Nitrogen and Sulfur Deposition

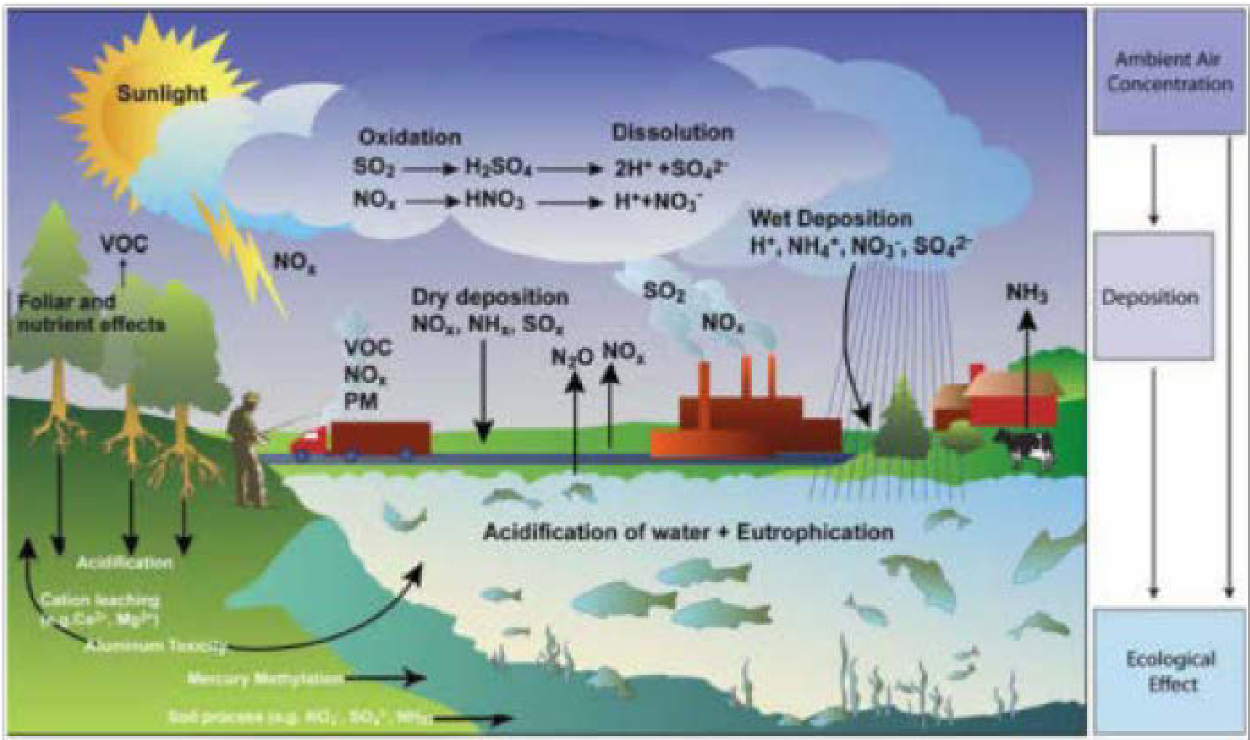

Figure 6-10. Nitrogen and Sulfur Cycling, and Interactions in the Environment

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).

6.7.2.1 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 NOx/SOx—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[34], 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.

[34] 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.

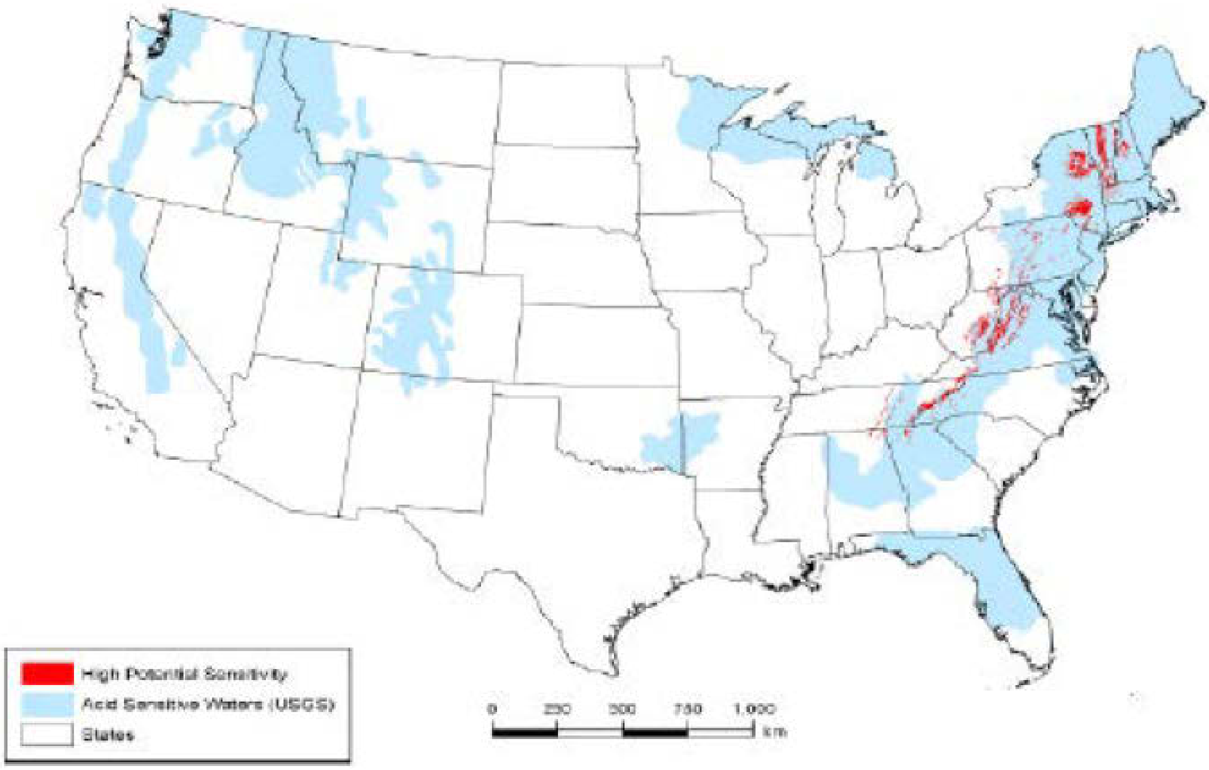

Figure 6-11. Areas Potentially Sensitive to Aquatic Acidification

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.[35] For recreation days, consumer surplus value is most commonly measured using recreation demand, travel cost models.

Another estimate of the overarching ecological 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 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

[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.

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).[36]

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 NOx/SOx—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 (2008c). Figure 6-12 depicts the areas across the U.S. that are potentially sensitive to terrestrial acidification.

[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.

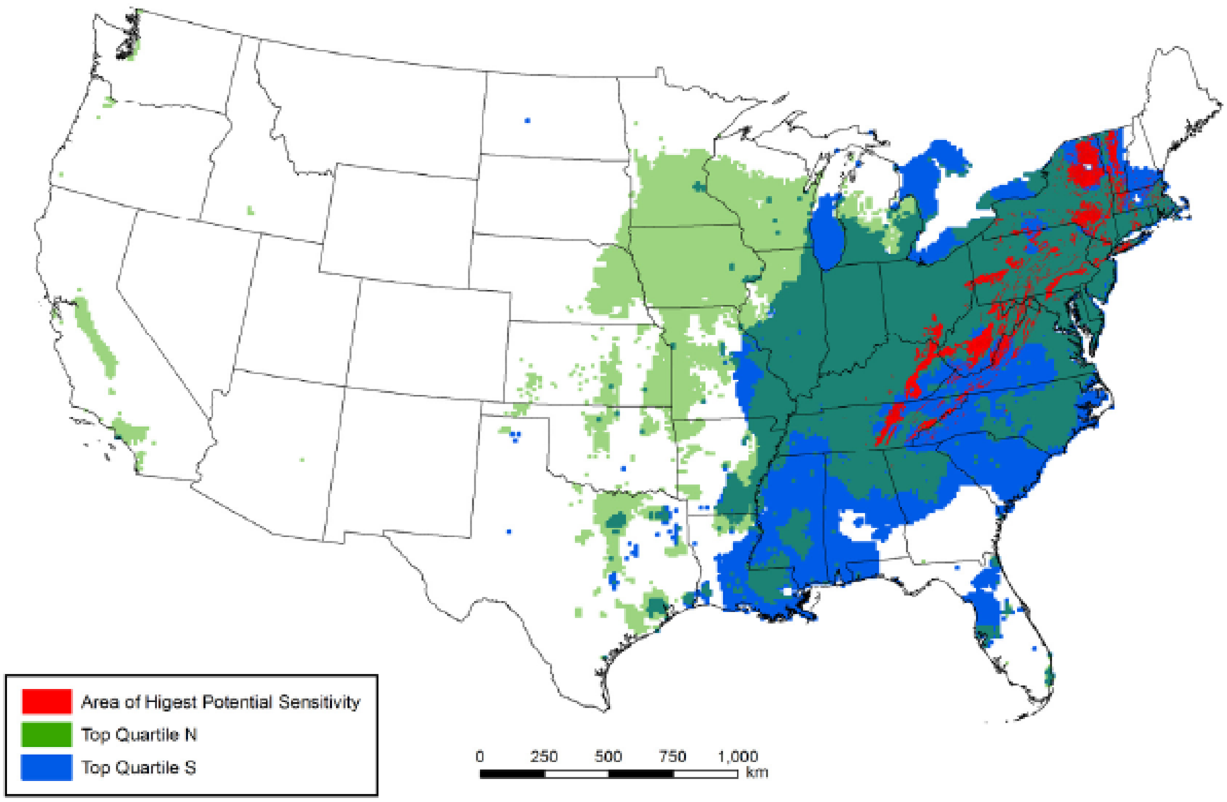

Figure 6-12. Areas Potentially Sensitive to Terrestrial Acidification

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).[37] 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).

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),

[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.

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).[38]

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 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

[38] 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.

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).[39]

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.

6.7.2.2 Ecological Effects from Nitrogen Enrichment

Aquatic Enrichment. The ISA for NOx/SOx—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 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

[39] 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.

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).[40] 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).[41] 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 benefits associated with the alternative standard level combinations 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 Integrated Science Assessment (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 ecosystem conditions and indicators. This long time scale also affects the timing of the ecosystem service changes. The ISA for NOx/SOx—Ecological Criteria

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[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.

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.[42] 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 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.

[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.

Based on analyses in the NOx SOx 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).[43] 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).[44] 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 benefits associated with the alternative standard level combinations 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 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

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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 NOx/SOx—Ecological Criteria concluded that the evidence is sufficient to infer a causal relationship between SO_2 injury to vegetation (U.S. EPA, 2008c).

6.7.2.4 Mercury-Related 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 NOx SOx 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).

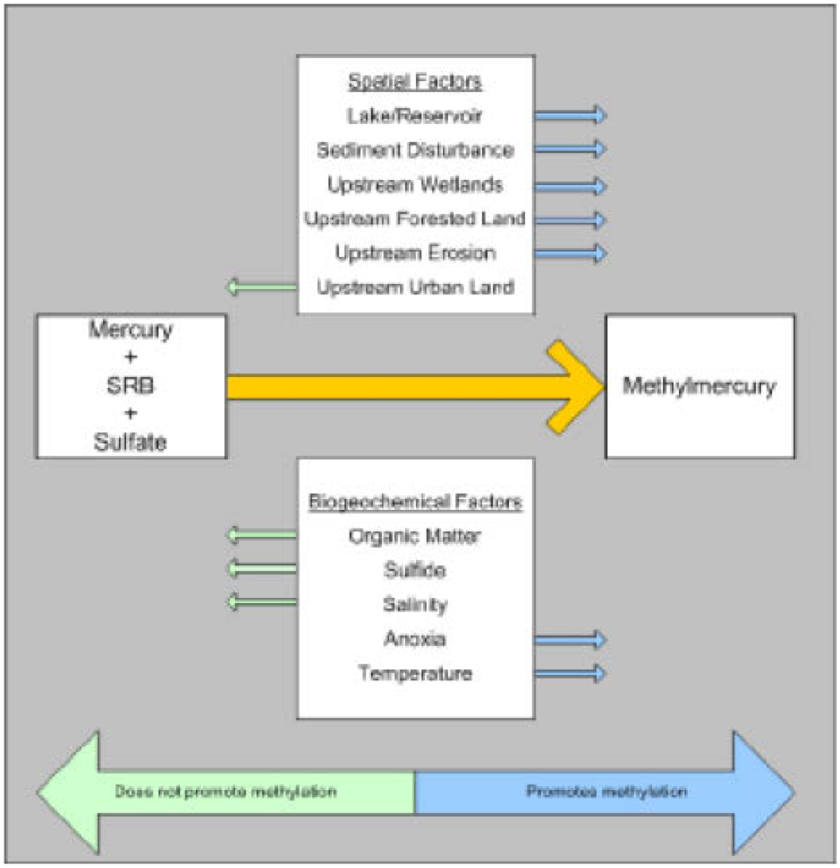


Figure 6-14. Spatial and Biogeochemical Factors Influencing MeHg Production

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).

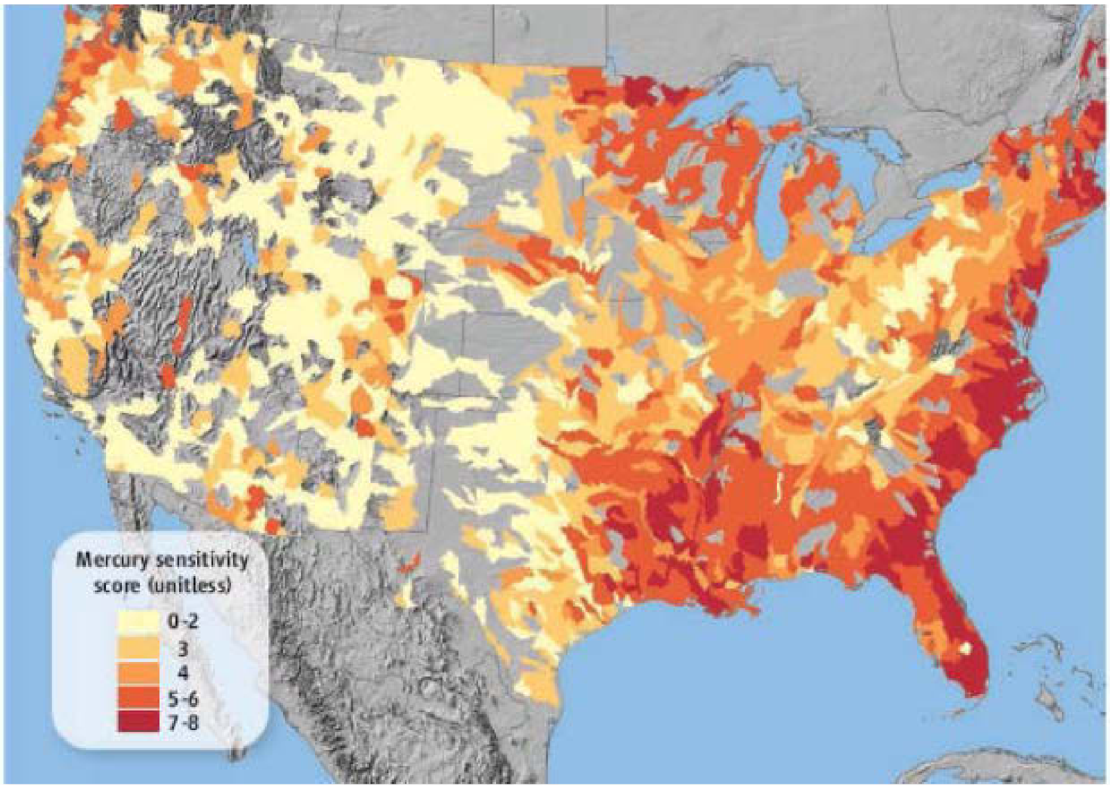


Figure 6-15. Preliminary USGS Map of Mercury Methylation–Sensitive Watersheds

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.

6.7.3 Ecosystem 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.

6.7.3.1 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.

6.7.3.2 Mercury Effects on Birds

In addition to effects on fish, mercury also affects avian species. In previous reports (U.S. EPA (1997); U.S. EPA (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).

6.7.3.3 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).

6.7.3.4 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. EPA is not, however, currently able to quantify or monetize the benefits of reducing mercury exposures affecting provision of ecosystem services.

6.7.4 Vegetation Benefits from Reductions in Ambient Ozone

Control strategies that include emission reductions of NOx 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).

6.7.4.1 Ozone Effects on Forests

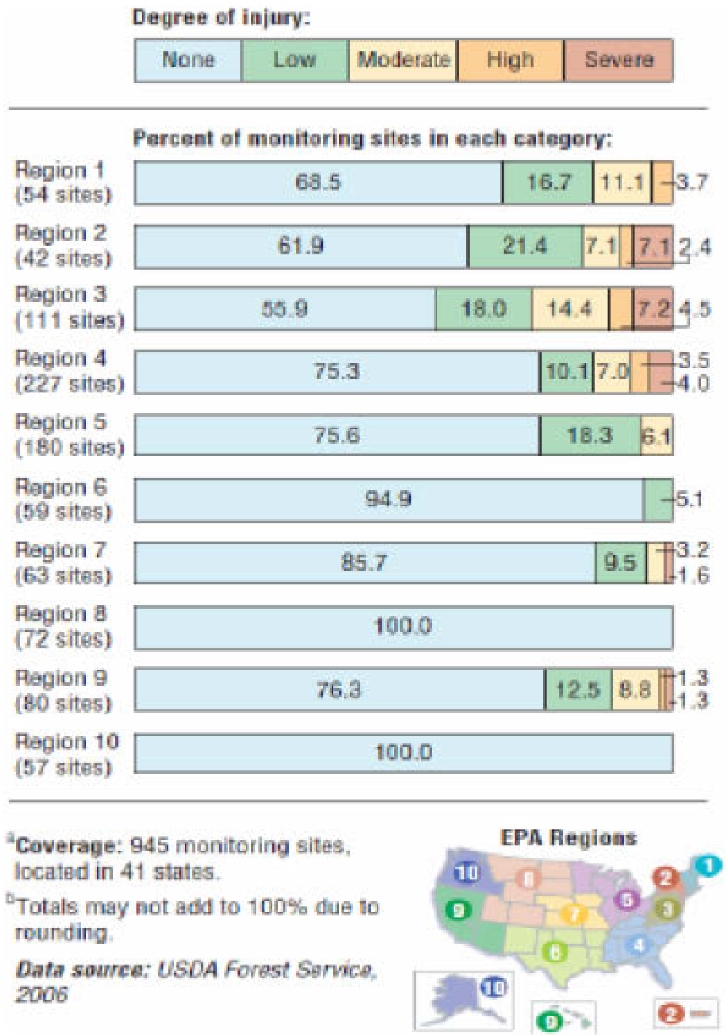
Air pollution can affect the environment and affect ecological systems, leading to changes in the ecological community and influencing the diversity, health, and vigor of individual species (U.S. EPA, 2006a). Ozone has been shown in numerous studies to have a strong effect on the health of many plants, 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.[45] 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

[45] 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]

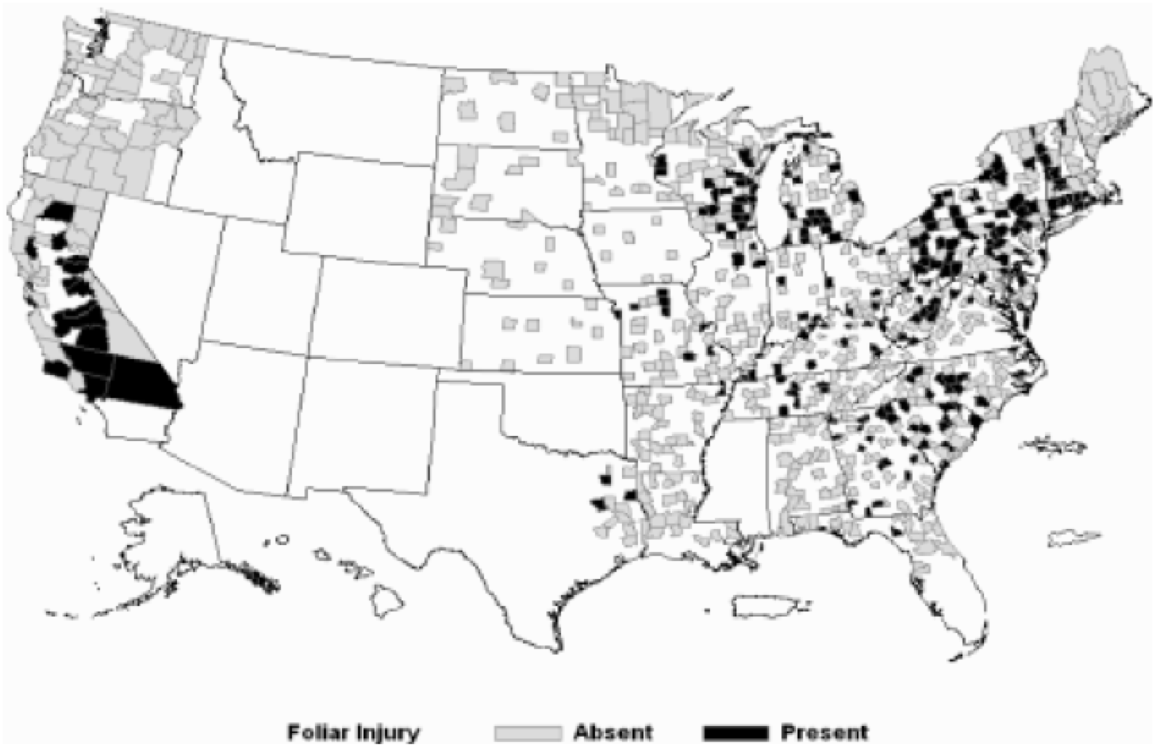


Figure 6-17. Presence and Absence of Visible Foliar Injury, as Measured by U.S. Forest Service, 2002

Source: U.S. EPA, 2007b

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 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).

6.7.4.2 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 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, U.S. 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).[46]

[46] 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.

6.7.4.3 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 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 benefits associated with the alternative standard level combinations due to data, time, and resource limitations.

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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

$$Deciviews = 10 * \ln\left(\frac{391}{VR}\right) = 10 \; * \ln\left(\frac{\beta_{\text{ext}}}{10}\right)$$

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 nth residential area and the i^{th} region of the country is:

$$U_{ni} = (X^{\rho} + \theta Z_n^{\rho} + \sum_{k=1}^{N_i} \gamma_{ik} Q_{ik}^{\rho} + \sum_{j \neq i} \sum_{k=1}^{N_j} \delta_{jk} Q_{jk}^{\rho})^{1/\rho} \quad ,$$

$$\theta > 0, \gamma_{ik} > 0, \forall i,k, \delta_{jk} > 0, \forall j,k, \ \rho \leq 1.$$

[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 $\rho=1$, the marginal WTP curve is horizontal. When $\rho<1$, 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 $\rho = 1$. 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, δ_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 δ_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 $\hat{\gamma}_j$ 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 $\hat{\gamma}_j$ 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 δ_6. We estimate γ_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 the 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

$$WTP_{1k}^{out} = \frac{n_{1k}}{n_1} * WTP_1^{out} \quad .$$

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 kth Class I area in the jth region. We use an analogous procedure to derive δ_{jk} from δ_j* for the kth Class I area in the jth 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:

$$\theta = \frac{(m - WTP)^{\rho} - m^{\rho}}{(Z_0^{\rho} - Z_1^{\rho})} .$$

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[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

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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 alternative, more stringent levels for the $PM_{2.5}$ primary standard analyzed in this RIA. This chapter provides the estimates of the engineering costs of alternative annual standards of 13, 12, and 11 $\mu g/m^3$ in conjunction with retaining the 24-hour standard of 35 $\mu g/m^3$, as well as one of the alternative, more stringent annual standards (11 $\mu g/m^3$) in conjunction with an alternative, more stringent 24-hour standard of 30 $\mu g/m^3$ (referred to as 13/35, 12/35, 11/35, and 11/30). This chapter also presents engineering cost estimates for the control strategies 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 (beyond known controls) tons of emission reductions that are needed to move to full attainment of the alternative standards analyzed; this estimation approach, discussed in Section 7.2.3, is referred to as extrapolated costs.

The engineering costs described in this chapter generally include the costs of purchasing, installing, operating, and maintaining the referenced 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 alternative standards, and EPA anticipates that state and local governments will consider programs that are best suited for local 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.[1] Also, EPA recognizes 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. This analysis focuses on the emission reductions needed for attainment of a range of alternative revised standards, not

[1] See Chapter 4, Section 4.3 for additional discussion of uncertainties associated with predicting technological advancements that may occur between now and 2020.

implementation of a final revised standard. EPA understands that some states will incur costs both designing State Implementation Plans (SIPs) for 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. This analysis does not assume specific control measures that would be required in order to implement these technologies on a regional or local level.

7.2 $PM_{2.5}$ Engineering 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[2] (CoST) to estimate engineering control costs for non-EGU point and area sources.[3] CoST calculates engineering costs using one of three different methods: (1) by multiplying an average annualized cost-per-ton estimate against the total tons of a pollutant reduced to derive a total cost estimate; (2) by calculating cost using an equation that incorporates key plant information; or (3) by using both cost-per-ton and cost equations. 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 parameters used in other non-cost-per-ton methods may not be readily available or broadly representative across sources within the emissions inventory. The costing equations used in CoST require either plant capacity or stack flow to determine annual, capital and/or operating and maintenance (O&M) costs. Capital costs are converted to annual costs using the capital recovery factor (CRF).[4] Where possible, cost calculations 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.

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[2] The Control Strategy Tool recently underwent peer review by an ad hoc panel of experts. Responses to the peer review are currently under development and will be available by final promulgation of this rule.

[3] Area sources are not necessarily non-urban sources.

[4] The capital recovery factor formula is expressed as [r*(1+r)^n/(1+r)^n -1]. Where *r* is the real rate of interest and *n* is the number of time periods. For more information on this cost methodology and the CoST, please refer to the documentation at http://www.epa.gov/ttn/CoST, 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.

Engineering costs will differ based upon quantity of emissions reduced, plant capacity, or stack flow, which can vary by the emissions inventory year. Engineering costs will also differ in nominal terms by the year the costs are calculated for (i.e., 1999$ versus 2006$).[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 incorporated into the CRF. Annualized costs represent an equal stream of yearly costs over the period the control technology is expected to operate. 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. Note that the application of these control strategies results in some, but not all, geographic areas reaching attainment for the alternative $PM_{2.5}$ standards.

Because we obtain control cost data from many sources, we are not always able to obtain consistent data across original data sources.[7] 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 percent discount rate. 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 percent discount rate. For non-EGU point source controls, some disaggregated data is available and we were able to calculate costs at both 3 and 7 percent discount rates for that control cost data. For the 12/35,

[5] The engineering costs will not be any different in real (inflation-adjusted) terms if calculated in 2006 versus other-year dollars, if the other-year dollars are properly adjusted. For this analysis, all costs are reported in real 2006 dollars.

[6] http://epa.gov/ttn/catc/products.html#cccinfo

[7] Data sources can include states and technical studies, which do not typically include the original data source.

11/35, and 11/30 alternative standards approximately 31 percent, 33 percent, and 29 percent, respectively, of known control costs are disaggregated at a level that could be discounted at 3 percent. Because we do not have disaggregated control cost data for any area source controls, total annualized costs are assumed to be calculated using a 7 percent discount rate. See Table 7-1 for a summary of sectors, control costs, and discount rates used.

Table 7-1. Summary of Sectors, Control Costs, and Discount Rates for Known Control Costs (millions of 2006$)[a]

Alternative Standard	Emissions Sector	Known Control Costs (Millions of 2006$)		
		7 Percent Discount Rate [b]	Partial Control Cost at 3 Percent Discount Rate [c]	3 Percent Discount Rate (3 Percent & 7 Percent Discount Rates Combined) [d]
13/35	non-EGU Point Sources	—	—	—
	Area Sources	—	—	—
	Total	—	—	—
12/35	non-EGU Point Sources	$0.098	$0.057	$0.061
	Area Sources	$0.210	—	$0.210
	Total	**$0.31**	**$0.057**	**$0.27**
11/35	non-EGU Point Sources	$25	$16	$24
	Area Sources	$28	—	$28
	Total	**$53**	**$16**	**$52**
11/30	non-EGU Point Sources	$46	$25	$42
	Area Sources	$54	—	$54
	Total	**$100**	**$25**	**$96**

[a] All estimates rounded to two significant figures.

[b] All non-EGU point source costs and all area source costs are included in this column and are assumed to be calculated at a 7 percent discount rate.

[c] This column includes the non-EGU point source costs that we were able to calculate at a 3 percent rate and no area source costs. The non-EGU point source costs calculated at a 3 percent rate are those for which we have disaggregated control cost data.

[d] Our known control costs discounted at a 3 percent rate are a combination of area source costs and non-EGU point source costs discounted at a 7 percent rate only and non-EGU point source costs discounted at 3 percent when disaggregated control cost data is available.

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 discount rates of 3 percent and 7 percent, consistent with the guidance provided in the Office of Management and Budget's (OMB) (2003) Circular A-4. However, it is important to note that it is not possible to estimate both 3 percent and 7 percent discount rates for each individual facility.

In this RIA, non-EGU point sources were the only sources with available data to perform a sensitivity analysis of our annualized control costs to the choice of interest rate. As such, the 3 percent column in Table 7-2 reflects the sum of some non-EGU point sources at a 3 percent discount rate, some non-EGU point sources at a 7 percent discount rate, and the area sources at a 7 percent discount rate. The 3 percent column represents a slight overestimate because some non-EGU point sources and area sources are included using a 7 percent discount rate.[8] With the exception of the 3 percent Total Annualized Cost estimate in Table 7-2, engineering cost estimates presented throughout this and subsequent chapters are based on a 7 percent discount rate.

Table 7-2. Partial Attainment Known Control Costs in 2020 for Alternative Standards Analyzed[a] (millions of 2006$)[a]

Alternative Standard	Region	Known Controls	
		3%[b]	7%
13/35[c]	East	—	—
	West	—	—
	California	—	—
	Total	—	—
12/35	East	$0.061	$0.098
	West	$0.21	$0.21
	California[c]	—	—
	Total	$0.27	$0.31
11/35	East	$46	$48
	West	$3.0	$3.0
	California	$3.0	$3.0
	Total	$52	$53
11/30	East	$46	$48
	West	$31	$33
	California	$19	$19
	Total	$96	$100

[a] Estimates are rounded to two significant figures, as such numbers may not sum down columns.

[8] In these analyses, the discount rates refer to the rate at which capital costs are annualized. A higher discount, or interest, rate results in a larger annualized cost of capital estimate.

[b] Because we obtain control cost data from many sources, we are not always able to obtain consistent data across original data sources. Where disaggregated control cost data is available (i.e., where capital, equipment life value, and O&M costs are explicit) we can calculate costs using a 3 percent discount rate. Therefore the cost estimate provided here is a summation of costs at 3 percent and 7 percent discount rates.

[c] All known controls were applied in the baseline.

The total annualized engineering costs associated with the application of known controls, incremental to the baseline and using a 7 percent discount rate, are approximately $310,000 for an alternative annual standard of 12/35, $53 million for an 11/35 alternative standard, and $100 million for an 11/30 alternative standard. In addition, it is important to note there is no partial attainment known control costs for a 13/35 alternative standard.[9]

7.2.3 Extrapolated Costs

This section presents the methodology and results of the extrapolated engineering cost calculations for attainment of alternative $PM_{2.5}$ standards of 13/35, 12/35, 11/35, and 11/30. All costs presented for the illustrative control strategies are calculated incrementally from the current $PM_{2.5}$ standard of 15/35, 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. *Note that the extrapolated costs don't account for cost differences between reducing additional tons by emissions source sector.*

As mentioned earlier in this chapter, the application of the modeled control strategy was not successful in reaching nationwide attainment 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 $PM_{2.5}$ standards of 13/35, 12/35, 11/35, and 11/30. For each alternative standard and geographic area that cannot reach attainment with known controls, we estimated, in a least-cost way, the additional emission reductions needed for $PM_{2.5}$ and its precursors, NOx and SO_2, to attain the standard. To generate estimates of the costs and benefits of meeting alternative standards, EPA has assumed the application of unspecified future controls that make possible the 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 unspecified future controls: a fixed-cost methodology and a hybrid methodology. The fixed-cost methodology is more straight forward and transparent than the hybrid methodology. In the hybrid methodology, the coefficient for the X^2 term can be difficult to estimate and if it is zero, the functional form

[9] Only one county (Riverside County, CA) exceeded the 13/35 alternative standard. All known controls were used in the baseline analysis for this county; therefore there are no known control costs for this standard. Total costs for 13/35 are represented by extrapolated costs only—see Table 7.3 below.

becomes the same as the fixed-cost methodology's functional form. Additional discussion of the functional form associated with the hybrid methodology is in Appendix 7.A. Both approaches assume that innovative strategies and new control options make possible the emissions reductions needed for attainment by 2020. The fixed-cost methodology uses a \$15,000/ton estimate for each ton of $PM_{2.5}$, SO_2, and NO_x reduced, and the hybrid approach is similar to the hybrid approach used for the 2008 Ozone NAAQS RIA cost analysis. The \$15,000/ton amount is commensurate with that used in the 1997 Ozone Regulatory Impact Analysis and is consistent with what an advisory committee to the Section 812 second prospective analysis on the Clean Air Act Amendments suggested. In Section 7.A.2.1 we conduct sensitivity analysis on the fixed-cost estimate of \$15,000/ton. The fixed-cost methodology was preferred by EPA's Science Advisory Board over two other options, including a marginal-cost-based approach.

> *"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."*

EPA requests comments or suggestions on all aspects of the methodologies for estimating the costs of unspecified future controls to provide illustrative estimates of NAAQS costs, including choice of functional forms of the equations, initial parameter estimates, and the initial fixed-cost estimate of \$15,000/ton.

In Appendix 4.A we include estimates of the relationship between additional emission reductions for each pollutant and air quality improvements. In this chapter we present estimates of the costs for each additional emission reduction for each pollutant and geographic area. The mix of pollutants varies by area, because each area has different amounts of known controls, different additional air quality improvements required, and different amounts of uncontrolled emissions remaining.

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 of known controls through technological improvements or higher control efficiencies. EPA requests comment on the likelihood that new technologies that control direct $PM_{2.5}$ and its precursors will become available between now and 2020.

Because three different pollutants are involved, there are many different combinations of pollutant reductions that would result in the required air quality improvements. Early in our analysis we decided to use the hybrid-cost methodology to estimate the costs of the emission reductions needed from unknown controls. However, the hybrid methodology still has a number of important uncertainties, and its reliability for extrapolating costs has not been evaluated. While we applied the methodology to generate emission reduction estimates needed beyond known controls for the proposed $PM_{2.5}$ standards and several alternatives, the degree of extrapolation for emissions reductions caused us to reconsider applying the hybrid methodology to obtain estimates of extrapolated costs. Consistent with an SAB recommendation, we use the fixed-cost per ton methodology to generate the estimates of extrapolated costs for emission reductions needed from unknown controls. We perform sensitivity analyses using both the alternative fixed cost per ton and the hybrid methodologies in Appendix 7.A

As discussed in Chapter 4, we developed the emission reduction estimates for each alternative standard using the hybrid methodology. As a result, the emissions reductions that form the basis of the primary cost and benefit estimates may include a different mix of $PM_{2.5}$ and SO_2 emissions reductions than may have been identified as least cost had we employed the fixed-cost methodology to develop the emission reduction estimates. Using the fixed-cost methodology, the less expensive pollutant, for air quality improvements, to reduce will be selected until there are no remaining tons to reduce. Using the hybrid methodology, the less expensive pollutant to reduce will be selected until the marginal cost to reduce the next ton exceeds the marginal cost to reduce the next ton of an alternate pollutant. At that point, the methodology chooses a mix of pollutants to achieve the least-cost solution. Since the cost per ton is held constant in the fixed-cost methodology, the least-cost solution would select all available direct $PM_{2.5}$ emissions reductions before selecting SO_2 emissions reductions.[10] Therefore, the hybrid methodology estimates $PM_{2.5}$ emissions reductions lower than or equal

[10] Because the marginal cost equation for each pollutant is expected to be less accurate for the very last portion of a pollutant in an area, and it is unlikely an area would reduce all anthropogenic emissions to zero on one pollutant prior to controlling others, we included the constraint that no more than 90% of the remaining emissions in an area for a given pollutant can be reduced from emission reductions beyond known control measures.

to the fixed-cost methodology and SO_2 emission reductions higher than or equal to the fixed-cost methodology.

Because we used the hybrid methodology in selecting emissions reductions, the total cost estimate is higher than if we had selected needed emissions reductions using the fixed-cost methodology. This is because the total number of tons identified and summed across pollutants under the hybrid methodology may be higher than the total number of tons needed under the fixed-cost methodology. For example, the hybrid methodology may choose to reduce direct $PM_{2.5}$ by 15 tons of and SO_2 by 8 tons, whereas the fixed-cost methodology may choose to reduce direct $PM_{2.5}$ by 20 tons to obtain the same air quality improvement for an area. Applying the fixed cost-per-ton to the total reductions, the hybrid methodology would result in total costs of $345,000 (23 tons * $15,000/ton), and the fixed-cost methodology would result in total costs of $300,000 (20 tons * $15,000/ton).

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. Table 7-3 provides the extrapolated cost estimates using the fixed-cost methodology described above, using a fixed cost-per-ton of $15,000/ton. The extrapolated costs estimate is $69 million dollars (2006$) for the 12/35 alternative standard.

Table 7-3. Fixed Costs by Alternative Standard Analyzed[a] (millions of 2006$)[a]

Alternative Standard	Region	Fixed Cost Methodology
		7%
13/35	East	—
	West	—
	California	$2.9
	Total	**$2.9**
12/35	East	—
	West	$3.3
	California	$65
	Total	**$69**
11/35	East	$1.3
	West	$38
	California	$180
	Total	**$220**
11/30	East	$21
	West	$79
	California	$190
	Total	**$290**

[a] Estimates are rounded to two significant figures.

Of note is the geographic distribution of extrapolated costs. For all of the alternative standards, the above costs indicate that California, as possibly expected, represents a significant portion of the extrapolated costs. For the 11/30, 11/35, 12/35, and 13/35 alternative standards, California represents 65 percent, 82 percent, 94 percent and 100 percent, respectively, of the extrapolated cost estimates.

7.2.4 Total Cost Estimates

Table 7-4 presents a summary of the total national costs of attaining 13/35, 12/35, 11/35, and 11/30 alternative standards in 2020. This summary includes the engineering costs presented above from the known controls analysis, as well as the extrapolated costs. As discussed in Section 7.2.2, costs for known controls for non-EGU point sources where capital cost and equipment life information were available were calculated at a 3 percent discount rate. Extrapolated costs were calculated at a 7 percent discount rate only.

To calculate total cost estimates at a 3 percent discount rate and to include the extrapolated costs in those totals, we added the known control estimates at a 3 percent discount rate to the extrapolated costs at a 7 percent discount rate. To more clearly present the total cost calculations for both approaches, we include column labels in Table 7-4, e.g., A, B, and C.

The costs associated with monitoring, reporting, and record keeping for affected sources are not included in these annualized cost estimates. Based on preliminary estimates prepared for the upcoming $PM_{2.5}$ Implementation Rule Information Collection Request (ICR), EPA believes these costs are minor compared to the control costs.

Table 7-4. Total Costs by Alternative Standard Analyzed (millions of 2006$)[a]

Alternative Standard	Region	Known Controls		Fixed-Cost Methodology	Total Costs Fixed-Cost Methodology	
		3% [b] A	7% B	7% C	3% (A+C)	7% (B+C)
13/35	East	—	—	—	—	—
	West	—	—	—	—	—
	California	—	—	$2.9	$2.9	$2.9
	Total	—	—	$2.9	$2.9	$2.9
12/35	East	$0.061	$0.098	—	$0.061	$0.098
	West	$0.21	$0.21	$3.3	$3.6	$3.6
	California	—	—	$65	$65	$65
	Total	$0.27	$0.31	$69	$69	$69
11/35	East	$46	$48	$1.3	$47	$49
	West	$3.0	$3.0	$38	$41	$41
	California	$3	$3	$180	$180	$180
	Total	$52	$53	$220	$270	$270
11/30	East	$46	$48	$21	$67	$69
	West	$31	$33	$79	$110	$110
	California	$19	$19	$190	$210	$210
	Total	$96	$100	$290	$390	$390

[a] Estimates are rounded to two significant figures, as such numbers may not sum down columns.

[b] Cost Estimates are not available at 3% for all control measures. Therefore the cost estimate provided here is a summation of costs at 3% and 7% discount rates.

7.3 Changes in Regulatory Cost Estimates over Time

Our analyses focus on controls for non-EGU and area sources. Future technology developments in sectors not analyzed here (e.g., EGUs) may be transferrable to non-EGU and area sources, making these sources more viable for achieving future attainment at a lower cost. These same future technology developments may also make the sectors not analyzed here (e.g., EGUs) more viable for achieving future attainment at a lower cost. 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 emission 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.[11] Technological change will affect baseline conditions for our analysis. This change may lead to potential improvements in the efficiency with which firms produce goods and services; for example, firms may use less energy to produce the same quantities of output.

Constantly increasing marginal abatement costs are likely to 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)[12] 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. EPA requests comment on how marginal control costs for specific technology applications may have changed over the past 20 years.

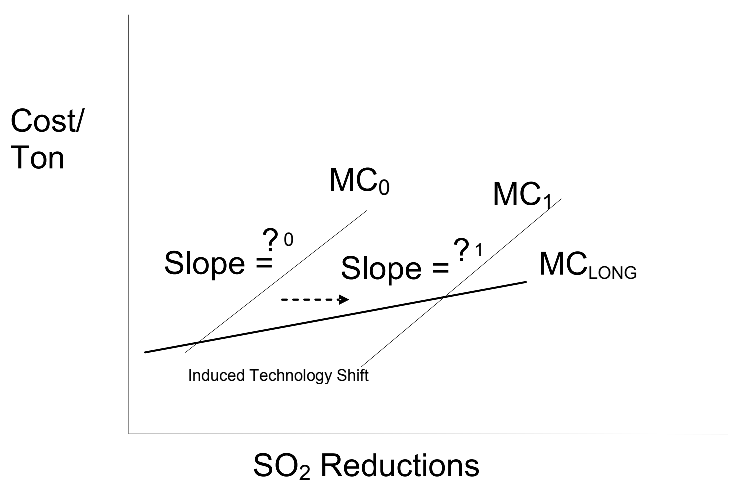

Figure 7-1. Technological Innovation Reflected by Marginal Cost Shift

[11] 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.

[12] 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.

7.3.1 Examples of Technological Advances in Pollution Control

There are numerous examples of low-emission technologies developed and/or commercialized over the past 15 or 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
- 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 2 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.

What is known as “learning by doing” or “learning curve impacts,” which is a concept distinct from technological innovation, have also made it possible to achieve greater emissions reductions than had been feasible earlier, or have reduced the costs of emission control in relation 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 lowering of expected costs for operation of technologies in the future below what they 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 Draft Direct Cost Report for the second EPA Section 812 Prospective Analysis of the Clean Air Act Amendments of 1990.[13] In that report, learning curve adjustments were included for those sectors and technologies for which learning curve data was 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 other words, capital or O&M costs will be reduced by some percentage for every doubling of output for the given sector or technology.

T.P. Wright, in 1936, was the first to characterize the relationship between increased productivity and cumulative production. He analyzed man-hours required to 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).[14] 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 percent, depending on the sector and technology. Learning curve adjustments were prepared in a memo by IEc supplied to US EPA and applied for the mobile source sector (both onroad and nonroad) and for application

[13] 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.

[14] 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.

of various EGU control technologies within the Draft Direct Cost Report.[15] 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 emission reductions being used when output data was unavailable.

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)[16] conducted an analysis of the predicted and actual costs of 28 federal and state rules, including 21 issued by 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 fall within 25 percent (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

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[15] 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.

[16] 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.

overestimated in 14 cases, while they were underestimated in six cases. In the case of 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 examination of eight economic incentive rules, "for those rules that employed economic incentive mechanisms, overestimation of per-unit costs seems to be the norm," the study said. It is worth noting here that the controls applied for this NAAQS do not use an economic incentive mechanism. In addition, Harrington et al. also states that overestimation of total costs can be due to error in the quantity of emission reductions achieved, which would also cause the benefits to be overestimated. A 2010 update to this study by Harrington et al. of RFF showed that EPA and other regulatory agencies tend to overestimate the total costs of regulations; their estimates of the cost per-unit of pollution eliminated by regulations tend to be more accurate, however. Calculations of the total cost of regulation include not only the "unit costs" multiplied by the number of units of pollution avoided, but also estimates of the basic adjustment process and costs of change itself. Of the rules initially examined, 14 projected inflated total costs, while pre-regulation estimates were too low for only three rules. These exaggerated adjustment costs are often attributable to underestimates of the potential that technological change could minimize pollution abatement costs.[17]

Based on the case study results and existing literature, the authors identified technological innovation as one of five explanations of why predicted and actual regulatory cost estimates differ: "Most regulatory cost estimates ignore the possibility of technological innovation … Technical change is, after all, notoriously difficult to forecast … In numerous case studies actual compliance costs are lower than predicted because of unanticipated use of new technology."

It should be noted that many (though not all) of the EPA rules examined by Harrington et al. had compliance dates of several years, which allowed a limited period for technical innovation.

[17] Harrington, W, R Morgenstern, and P Nelson. "How Accurate Are Regulatory Cost Estimates?" *Resources for the Future*, March 5, 2010. Available on the Internet at http://www.rff.org/wv/Documents/HarringtonMorgensternNelson_regulatory%20estimates.pdf.

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 percent lower than originally projected by EPA.[18] As noted in the RIA for the Clean Air Interstate Rule, the ex ante numbers in 1989 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. 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 1990's. 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 7-5. 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 EPA Fuel Control Rules

A 2002 study by two economists with EPA's Office of Transportation and Air Quality[19] examined EPA vehicle and fuels rules and found a general pattern that "all ex ante estimates tended to exceed actual price impacts, with the EPA estimates exceeding actual prices by the smallest amount." The paper notes that cost is not the same as price, but suggests that a comparison nonetheless can be instructive.[20] An example focusing on fuel rules is provided in Table 7-6.

[18] 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.

[19] Anderson, J.F., and Sherwood, T., 2002. "Comparison of EPA and Other Estimates of Mobile Source Rule Costs to Actual Price Changes," Office of Transportation and Air Quality, U.S. Environmental Protection Agency. Technical Paper published by the Society of Automotive Engineers. SAE 2002-01-1980.

[20] The paper notes: "Cost is not the same as price. This simple statement reflects the fact that a lot happens between a producer's determination of manufacturing cost and its decisions about what the market will bear in terms of price change."

Table 7-6. Comparison of Inflation-Adjusted Estimated Costs and Actual Price Changes of EPA Fuel Control Rules[a]

	Inflation-adjusted Cost Estimates (c/gal)				Actual Price Changes (c/gal)
	EPA	DOE	API	Other	
Gasoline					
Phase 2 RVP Control (7.8 RVP—Summer) (1995$)	1.1	1.8		0.5	
Reformulated Gasoline Phase 1 (1997$)	3.1–5.1	3.4–4.1	8.2–14.0	7.4 (CRA)	2.2
Reformulated Gasoline Phase 2 (Summer) (2000$)	4.6–6.8	7.6–10.2	10.8–19.4	12	7.2 (5.1, when corrected to 5yr MTBE price)
30 ppm sulfur gasoline (Tier 2)	1.7–1.9	2.9–3.4	2.6	5.7 (NPRA), 3.1 (AIAM)	N/A
Diesel					
500 ppm sulfur highway diesel fuel (1997$)	1.9–2.4		3.3 (NPRA)	2.2	
15 ppm sulfur highway diesel fuel	4.5	4.2–6.0	6.2	4.2–6.1 (NPRA)	N/A

[a] Anderson, J.F., and Sherwood, T., 2002. "Comparison of EPA and Other Estimates of Mobile Source Rule Costs to Actual Price Changes," Office of Transportation and Air Quality, U.S. Environmental Protection Agency. Technical Paper published by the Society of Automotive Engineers. SAE 2002-01-1980.

Chlorofluorocarbon (*CFC) 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. This 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 percent less than was predicted at the time the 1990 Clean Air Act Amendments were enacted.[21]

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[22] notes, 'since 1991 most new U.S.

[21] 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.

[22] 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.

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). He cites a similar story for HCFRC-141b and 142b, which are currently substituting for CFC-11 in important foam-blowing applications."

Additional examples of decreasing costs of emissions controls include: SCR catalyst costs decreasing from \$11k-\$14k/m^3 in 1998 to \$3.5k-\$5k/m^3 in 2004, and improved low NO_x burners reduced emissions by 50% from 1993–2003 while the associated capital cost dropped from \$25-\$38/kW to \$15/kW.[23] Also, FGD scrubber capital costs have been estimated to have decreased by more than 50 percent from 1976 to 2005, and the O&M costs decreased by more than 50% from 1982 to 2005. Many process improvements contributed to lowering the capital costs, especially improved understanding and control of process chemistry, improved materials of construction, simplified absorber designs, and other factors that improved reliability.[24]

We cannot estimate the precise interplay between EPA regulation and technology improvement, but it is clear that a *priori* cost estimation often results in overestimation of costs because changes in technology (whatever the cause) make less costly control possible.

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 met with existing technology but not 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). Therein, they demonstrate through a careful 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.

Work for EPA by RTI in 2011 studied the relationship between patents and the SO_2 cap-and-trade program. This preliminary, non-peer reviewed study seems to indicate that patents

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[23] ICF Consulting. October 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.

[24] Yeh, Sonia and Rubin, Edward. February 2007. "Incorporating Technological Learning in the Coal Utility Environmental Cost (CUECost) Model: Estimating the Future Cost Trends of SO_2, NOx, and Mercury Control Technologies." Prepared for ARCADIS Geraghty and Miller, Research Triangle Park, NC 27711. Available at http://steps.ucdavis.edu/People/slyeh/syeh-resources/Drft%20Fnl%20Rpt%20Lrng%20for%20CUECost_v3.pdf.

may be related to SO_2 regulatory actions and drops in the long-term SO_2 allowance price. 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 bases its estimates of emission 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, emission 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.

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, as well as permitting costs. ▪ 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

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APPENDIX 7.A
OTHER EXTRAPOLATED COST APPROACHES

7.A.1 Extrapolated Cost Equations

The **hybrid methodology** creates a total cost curve for each pollutant for unknown future controls that might be used in order to move toward 2020 attainment. This approach explicitly estimates the cost of removing emissions for each pollutant for each area, with a higher cost-per-ton in areas needing a higher proportion of unknown controls relative to known modeled controls. For each pollutant in each area, the cost begins with a national constant cost-per-ton for that pollutant and increases as more of that pollutant is selected. The selection is made so that costs are minimized. The incremental improvement in air quality for an unknown control is determined, by pollutant, using an area-by-area ratio of air quality improvement to air quality change multiplied by the emission reductions from unknown controls for each pollutant and area. For each pollutant in an area, the per-unit costs of control increase with each additional unit of that pollutant chosen. The choices are made to minimize total cost.

The hybrid methodology has the advantage of using the information about how significant the needed reductions from unspecified control technologies are relative to the known control measures and matching that with expected increasing per-unit cost for applying controls beyond those modeled. Under this approach, the relative costs of unknown controls in different geographic areas reflect 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. Because no cost data exists for unknown future strategies, 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 approach that uses a range of national cost-per-ton values. 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 the analysis.

7.A.1.1 Theoretical Model for Hybrid Methodology

A model of increasing total costs was developed for each pollutant. 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 a pollutant, and c is zero because there is no cost to imposing no control. The hybrid methodology has a different *a* for each pollutant and geographical area. For a particular geographic area and pollutant *a* is *N/E* where

N = national, initial cost/ton (b from above)

E = by geographic area and by pollutant, is the denominator[1] and represents all emission reductions estimated to be required (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.

The hybrid methodology attempts to consider the varying conditions in each geographic area exceeding the alternative standard. Using the hybrid methodology, we develop marginal control cost curves associated with unknown control costs for each geographic area. Because the rate at which the unknown control costs increase varies across areas, the marginal cost will vary. For example, applying unknown controls to obtain the needed emission reductions in a geographic area with few emissions sources, very few known controls and many emission reductions needed will have a higher marginal cost. Where in another circumstance, applying unknown controls to obtain the needed emission reductions in a geographic area with many emissions sources, many known controls and many emission reductions needed will have a lower marginal cost.

The total cost equations presented below in Section 7.A.1.2 are used with the constraint that

$$Q - R_{SO_2}U_{SO_2} - R_{PM_{2.5}}U_{PM_{2.5}} - R_{NO_x}U_{NO_x} = 0 \tag{7.1}$$

where

U = unknown emission reductions

Q = air quality change needed for area to reach attainment

R = air quality to emissions ratio for area.

This equation above uses the R developed in *Section 3.3.1.3 (Estimating Full Attainment)* for each pollutant and area that is used to estimate the change in air quality for each unit of emissions. In other words, R is a measure of how much the air quality concentration changes

[1] The numerator differs by pollutant and is the national, initial cost-per-ton for unknown controls discussed below, e.g., $17,000/ton for $PM_{2.5}$.

when the emissions change. The least-cost solution is found for each geographic area and for each standard. See Section 4.2.2 for a discussion of the order in which controls were applied for geographic areas and pollutants.

The cost-per-ton estimates were calculated by developing marginal cost curves for all known controls applied for the alternative standards as well as the baseline. We reviewed data on controls in approximately 120 counties with higher $PM_{2.5}$ concentrations. We plotted, in order of increasing marginal cost, the cost-per-ton and cumulative emission reductions associated with these controls for $PM_{2.5}$, SO_2, and NO_x. For $PM_{2.5}$, the data show that approximately 96 percent of potential emission reductions associated with the known controls in those 120 counties could be obtained for $17,000/ton (2006 dollars) or less. For SO_2, the data show that approximately 94 percent of potential emission reductions associated with the known controls in those 120 counties could be obtained for $5,400/ton (2006 dollars) or less. For NO_x, the data show that approximately 92 percent of potential emission reductions associated with the known controls in those 120 counties could be obtained for $5,500/ton (2006 dollars) or less. We conservatively defined a threshold of 90 percent for unknown emission reductions for a particular pollutant in an area after reductions associated with known controls are achieved. As such, the national, initial cost-per-ton for unknown controls for the parameters $N_{PM_{2.5}}$, N_{SO_2}, and N_{NOx} were chosen at $17,000/ton (2006 dollars), $5,400/ton (2006 dollars), and $5,500/ton (2006 dollars) for $PM_{2.5}$, SO_2, and NO_x emission reductions, respectively.

The results for the alternative hybrid methodology described in this Appendix and the fixed-cost approach described in Chapter 7 are presented for comparison in Table 7.A-1.

Table 7.A-1. Extrapolated Costs by Alternative Standard Analyzed (millions of 2006$)[a]

		Extrapolated Costs	
		Fixed-Cost Methodology	**Alternative Hybrid Methodology**
Alternative Standard	**Region**	**7%**	**7%**
13/35	East	—	—
	West	—	—
	California	$2.9	$3.6
	Total	**$2.9**	**$3.6**
12/35	East	—	—
	West	$3.3	$24

	California	$65	$100
	Total	**$69**	**$130**
11/35	East	$1.3	$3
	West	$38	$310
	California	$180	$280
	Total	**$220**	**$590**
11/30	East	$21	$47
	West	$79	$470
	California	$190	$320
	Total	**$290**	**$830**

[a] Estimates are rounded to two significant figures.

The **hybrid model** is presented more formally below. For the simple form of $ax^2 + bx + c$, where x is the tons of a particular pollutant to be reduced in a particular area and a, b, and c are constants.

7.A.1.2 Framework Applied to This Analysis

Total Cost Equation for the three pollutants is

$$Minimize\ \left(\frac{N_{SO_2}}{E_{SO_2}}\right)U_{SO_2}{}^2 + N_{SO_2}U_{SO_2} + c + \sum_{\mathrm{j=1}}^{\mathrm{n}}[\left(\frac{N_{PM_{2.5}}}{E_{PM_{2.5(j)}}}\right)U_{PM_{2.5\,(j)}}{}^2 + N_{PM_{2.5}}U_{PM_{2.5(j)}}] +\ d$$

$$+\left(\frac{N_{NO_x}}{E_{NO_x}}\right)U_{NO_X}{}^2 + N_{NO_X}U_{NO_X} + f +$$

$$subject\ to\ the\ constraint\ \sum_{j=1}^{n} Q_j - R_{SO_2 j}U_{SO_2} -\ R_{PM_{2.5j}}U_{PM_{2.5j}} - R_{NO_{Xj}}U_{NO_X} = 0$$

Where j is jth county up to the total number of nonattainment counties in each defined geographic area

$U_{SO2} \geq 0$

$$\sum_{j=1}^{n} U_{PM_{2.5j}} \geq 0$$

$U_{NOx} \geq 0$

$U_{SO2} \leq 0.9(T_{SO2} - E_{SO2})$

$$\sum_{j=1}^{n} U_{PM_{2.5j}} \leq = 0.9(TPM2.5j\ -\ EPM2.5j)$$

$U_{NOx} \leq 0.9(T_{NOx} - E_{NOx})$

The constraints require the emissions changes to result in the required air quality change. The emissions changes cannot be negative. The unknown emission reductions for a particular pollutant in an area cannot exceed 90% of the remaining emissions after reductions associated with known controls are achieved. Section 7.2.3.1 includes information on the selection of the 90% threshold.

where

N = national constant cost/ton

E = known emission reductions

U = unknown emission reductions

Q = air quality change

R = air quality to emissions ratio

T = total emissions

This optimization can be executed in a number of ways (all resulting in the same answer). For this analysis we employed the data solver add-in (solver) for Microsoft Excel. We ran the solver for each area needing unknown controls for each standard. The detailed spreadsheet will be in the PM NAAQS docket.

7.A.1.3 Cost Minimization Approach

The solver iterates to select how much of each pollutant to choose in a cost-minimizing manner. In each geographic area with one or more counties that requires additional emission reductions to reach attainment, to select a quantity of each pollutant to minimize the costs of the needed emissions reductions, the solver

- looks simultaneously at $PM_{2.5}$ emissions within any counties not meeting the standard and SO_2 and NO_x emissions from within the geographic area.

For example, for a three-county area where two counties are not estimated to meet the alternative standard, the solver picks $PM_{2.5}$ reductions in each of the two counties and then SO_2

and NO_x reductions for the entire geographic area such that both counties would reach attainment in the least cost way.

In Chapter 3, we define ratios for each of the two counties associated with how much air quality concentration improvement would result from 1,000 tons of $PM_{2.5}$ reduction in each respective county. Also in Chapter 3, we define ratios for each of the two counties associated with how much air quality concentration improvement would result from 1,000 tons of SO_2 reduction in the defined geographic area (not just the respective counties). Lastly in Chapter 3, we define ratios for each of the two counties associated with how much air quality concentration improvement would result from 1,000 tons of NO_x reduction in the defined geographic area (not just the respective county). In Chapters 3 and 4, we define and present geographic areas where SO_2 and NO_x emissions contribute to the air quality problems in a close cluster of counties.

7.A.2 Sensitivity Analyses of Extrapolated Cost Approaches

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 Approach

Table 7.A.2 below presents the sensitivity analysis of the fixed-cost approach and includes, by region and alternative standard, the primary cost estimate of $15,000/ton. The Table also includes, by region and alternative standard, cost estimates using $10,000/ton and $20,000/ton. For the 12/35 alternative standard, the total cost estimate associated with unknown control costs ranges from $46 million to $92 million, depending on the fixed-cost-per-ton assumed.

Table 7.A-2. Sensitivity Analysis of Fixed-Cost Approach for Unknown Controls by Alternative Standard Analyzed (millions of 2006$)[a]

		Extrapolated Costs		
		$10,000/ton	$15,000/ton	$20,000/ton
Alternative Standard	Region	7%	7%	7%
13/35	East	—	—	—
	West	—	—	—
	California	$1.9	$2.9	$3.9
	Total	**$1.9**	**$2.9**	**$3.9**
12/35	East	—	—	
	West	$2.2	$3.3	$4.5
	California	$44	$65	$87
	Total	**$46**	**$69**	**$92**
11/35	East	$0.90	$1.3	$1.8
	West	$26	$38	$51
	California	$120	$180	$240
	Total	**$150**	**$220**	**$290**
11/30	East	$14	$21	$28
	West	$53	$79	$110
	California	$130	$190	$250
	Total	**$190**	**$290**	**$390**

[a] Estimates are rounded to two significant figures.

7.A.2.2 Sensitivity Analysis of Alternative Hybrid Approach

Table 7.A.3 below presents the sensitivity analysis of the alternative hybrid approach. To be consistent with the sensitivity analysis of the fixed-cost approach, the table also includes, by region and alternative standard, cost estimates using alternate parameter estimates for the initial cost per ton. For the 12/35 alternative standard, the total cost estimate associated with unknown control costs ranges from $85 million to $170 million.

Table 7.A-3. Sensitivity Analysis of Alternative Hybrid Approach for Unknown Controls by Alternative Standard Analyzed (millions of 2006$)[a]

		Extrapolated Costs		
		30 Percent Lower [b]	Estimate w/Original Cost of Initial Ton [c]	30 Percent Higher [d]
Alternative Standard	**Region**	**7%**	**7%**	**7%**
13/35	East	—	—	—
	West	—	—	—
	California	$2.4	$3.6	$4.7
	Total	**$2.4**	**$3.6**	**$4.7**
12/35	East	—	—	—
	West	$16	$24	$32
	California	$69	$100	$140
	Total	**$85**	**$130**	**$170**
11/35	East	$1.8	$3	$3.6
	West	$210	$310	$420
	California	$190	$280	$370
	Total	**$400**	**$590**	**$790**
11/30	East	$32	$47	$63
	West	$310	$470	$630
	California	$210	$320	$420
	Total	**$550**	**$830**	**$1,100**

[a] Estimates are rounded to two significant figures.

[b] These estimates reflect national, initial cost-per ton estimates for the three parameters that are 30 percent lower.

[c] As discussed in Section 7.A.1.1 above, the national, initial cost-per-ton for unknown controls for the parameters $N_{PM_{2.5}}$, N_{SO_2}, $\boldsymbol{and\ N_{NOx}}$ were chosen at $17,000/ton (2006 dollars), $5,400/ton (2006 dollars), and $5,500/ton (2006 dollars) for $PM_{2.5}$, SO_2, and NO_x emission reductions, respectively. In addition, as discussed in Chapter 7, Section 7.2.3, the hybrid cost methodology is used to estimate the needed emission reductions.

[d] These estimates reflect national, initial cost-per ton estimates for the three parameters that are 30 percent higher.

CHAPTER 8
COMPARISON OF BENEFITS AND COSTS

8.1 Synopsis

This chapter compares estimates of the benefits with economic costs and summarizes the net benefits of alternative standards relative to a baseline that includes recently promulgated national regulations (CSAPR, MATS, and others). We include a discussion of net benefits for the case of full attainment and discuss selected limitations of the analyses.

8.2 Analysis

In the analysis, we estimate the net benefits of the proposed range of annual $PM_{2.5}$ standards of 12/35 to 13/35. For 12/35, net benefits are estimated to be $2.3 billion to $5.9 billion at a 3% discount rate and $2.0 billion to $5.3 billion at a 7% discount rate in 2020 (2006 dollars).[1] For 13/35, net benefits are estimated to be $85 million to $220 million at the 3% discount rate and $76 million to $200 million at the 7% discount rate.

The RIA also analyzes the benefits and costs of two alternative primary $PM_{2.5}$ standards (11/35 and 11/30) that are more stringent than the proposed standard range of 12/35 to 13/35. The EPA estimated the net benefits of the alternative annual $PM_{2.5}$ standard of 11/35 to be $8.9 billion to $23 billion at a 3% discount rate and $8.0 billion to $21 billion at a 7% discount rate in 2020. The EPA estimated the net benefits of the alternative annual $PM_{2.5}$ standard of 11/30 to be $14 billion to $36 billion at a 3% discount rate and $13 billion to $33 billion at a 7% discount rate in 2020. All estimates are in 2006$.[2]

The EPA determined that all counties would meet the 14/35 standard concurrently with meeting the existing 15/35 standard at no additional cost. Consequently, there is no need to present an analysis of 14/35 in this RIA.

We provide these results in Table 8-1. In Table 8-2, we provide the avoided health incidences associated with these standard levels.

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[1] Using a 2010$ year increases estimated costs and benefits by approximately 8%. Because of data limitations, we were unable to discount compliance costs for all sectors at 3%. As a result, the net benefit calculations at 3% were computed by subtracting the monetized benefits at 3% minus the costs at 7%.

[2] Using a 2010 $ year increases estimated costs and benefits by approximately 8%. Because of data limitations, we were unable to discount compliance costs for all sectors at 3%. As a result, the net benefit calculations at 3% were computed by subtracting the monetized benefits at 3% minus the costs at 7%.

8.3 Conclusions of the Analysis

EPA's illustrative analysis has estimated the health and welfare benefits and costs associated with proposed revised 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.

Table 8-1. Total Monetized Benefits, Total Costs, and Net Benefits in 2020 (millions of 2006$[a])—Full Attainment

	Total Costs		Monetized Benefits [b]		Net Benefits [b]	
Alternative Standard	**3% Discount Rate[c]**	**7% Discount Rate**	**3% Discount Rate**	**7% Discount Rate**	**3% Discount Rate[c]**	**7% Discount Rate**
13/35	$2.9	$2.9	$88 to $220	$79 to $200	$85 to $220	$76 to $200
12/35	$69	$69	$2,300 to $5,900	$2,100 to $5,400	$2,300 to $5,900	$2,000 to $5,300
11/35	$270	$270	$9,200 to $23,000	$8,300 to $21,000	$8,900 to $23,000	$8,000 to $21,000
11/30	$390	$390	$14,000 to $36,000	$13,000 to $33,000	$14,000 to $36,000	$13,000 to $33,000

[a] Rounded to two significant figures. Using a 2010$ year increases estimated costs and benefits by approximately 8%.

[b] 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.

[c] Due to data limitations, we were unable to discount compliance costs for all sectors at 3%. As a result, the net benefit calculations at 3% were computed by subtracting the monetized benefits at 3% minus the costs at 7%.

For the lower end of the proposed standard range of 12/35, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 34 to 86 times at a 3% discount rate and 30 to 78 times at a 7% discount rate. For the upper end of the proposed standard range of 13/35, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 30 to 77 times at a 3% discount rate and 27 to 69 times at a 7% discount rate. For the alternative standards, 11/35 and 11/30, the EPA estimates that the benefits of full attainment exceed the costs of full attainment by 34 to 94 times at a 3% discount rate and 30 to 85 times at a 7% discount rate.

Table 8-2. Estimated Number of Avoided $PM_{2.5}$ Health Impacts for Standard Alternatives—Full Attainment[a]

Health Effect	Alternative Combination of Primary $PM_{2.5}$ Standards			
	13/35	12/35	11/35	11/30
Adult Mortality				
Krewski et al. (2009)	11	280	1,100	1,700
Laden et al. (2006) (adult)	27	730	2,900	4,500
Woodruff et al. (1997) (infant)	0	1	3	4
Non-fatal heart attacks (age >18)				
Peters et al. (2001)	11	320	1,300	1,900
Pooled estimate of 4 studies	1	35	140	210
Hospital admissions—respiratory (all ages)	3	98	430	620
Hospital admissions—cardiovascular (age > 18)	3	95	400	580
Emergency department visits for asthma (age < 18)	6	160	730	1,000
Acute bronchitis (age 8–12)	22	540	2,000	3,100
Lower respiratory symptoms (age 7–14)	290	6,900	25,000	39,000
Upper respiratory symptoms (asthmatics age 9–11)	410	9,800	37,000	56,000
Asthma exacerbation (age 6–18)	410	24,000	89,000	140,000
Lost work days (age 18–65)	1,800	44,000	170,000	260,000
Minor restricted-activity days (age 18–65)	11,000	260,000	1,000,000	1,500,000

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

8.4 Caveats and Limitations

EPA acknowledges several important limitations of the primary and secondary analysis. These include:

8.4.1 Benefits Caveats

- $PM_{2.5}$ mortality co-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 twelve estimates 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, 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.

- 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 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.

8.4.2 Control Strategy and Cost Analysis Caveats and Limitations

Control Technology Data

- Technologies applied may not reflect emerging devices that may be available in future years.
- Control efficiency data depend on equipment being well maintained.
- Area source controls assume a constant estimate of emission reductions, despite variability in extent and scale of application.

Control Strategy Development

- States may develop different control strategies than the ones illustrated.
- Data on baseline controls from current SIPs are lacking.
- Timing of control strategies may be different than envisioned in the RIA.
- Controls are applied within the county with the violating monitor. It is possible that additional known controls could be available in a wider geographical area.

- Unknown controls were needed to reach attainment in several counties. Costs associated with these unknown controls were estimated using a fixed cost per ton methodology as well as an extrapolated cost methodology.

- Emissions reductions from mobile sources, EGUs, other $PM_{2.5}$ precursors (i.e., ammonia and VOC), and voluntary programs are not reflected in the analyses.

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 not accounted for in the emission reduction estimates.
- Future technology developments in sectors not analyzed here (e.g., EGUs) may be transferrable to non-EGU and area sources, making these sources more viable for achieving future attainment at a lower cost than the cost presented in this analysis.

Engineering Cost Estimates

- Because of data limitations, we were unable to discount compliance costs for all sectors at 3%.
- Estimates of private compliance cost are used as a proxy for social cost.

Unquantified Costs

- A number of costs remain unquantified, including administration costs of federal and state SIP programs, and transactional costs.

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. 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 proposed rule on small entities, I certify that this action will not have a significant economic impact on a substantial number of small entities. This proposed 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). Please refer to the preamble for additional details.

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 proposed 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 on UMRA in the preamble, 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 on UMRA in the preamble, 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. Please refer to the preamble for additional details on the Executive Order.

9.7 Executive Order 13175: Consultation and Coordination with Indian Tribal Governments

The action does not have tribal implications, as specified in Executive Order 13175 (65 FR 67249, November 9, 2000). 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. The Tribal Authority Rule gives Tribes the opportunity to develop and implement CAA programs such as the PM NAAQS, but it leaves to the discretion of the Tribe whether to develop these programs and which programs, or appropriate elements of a program, they will adopt. Thus, Executive Order 13175 does not apply to this rule.

Although Executive Order 13175 does not apply to this rule, the EPA consulted with tribal officials or other representatives of tribal governments in developing this action. Please refer to the preamble for additional details on the Executive Order.

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 a susceptible 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, IV.B, and IV.C of the preamble. A listing of documents that contain the evaluation of scientific evidence and policy considerations that pertain to children is found in the section on Children's Environmental Health in the Supplementary Information section of the preamble, and a copy of all documents have been placed in the public docket for this action.

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 proposed rulemaking involves technical standards for environmental monitoring and measurement. Specifically, the EPA proposes to retain the indicators for fine ($PM_{2.5}$) and coarse (PM10) particles. The indicator for fine particles is measured using the Reference Method for the Determination of Fine Particulate Matter as $PM_{2.5}$ in the Atmosphere (appendix L to 40 CFR part 50), which is known as the $PM_{2.5}$ 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. The EPA also proposes a separate secondary standard defined in terms of a calculated $PM_{2.5}$ light extinction indicator, which would use $PM_{2.5}$ mass species and relative humidity data to calculate $PM_{2.5}$ light extinction.

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 $PM_{2.5}$ continuous FEMs has resulted in the approval of six approved FEMs. In summary, for measurement of $PM_{2.5}$ 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.

For the proposed distinct secondary standard defined in terms of a calculated $PM_{2.5}$ light extinction indicator, the EPA proposes to use existing monitoring technologies that are already deployed in the CSN and IMPROVE monitoring programs as well as relative humidity data from sensors already deployed at routine weather stations. The sampling and analysis protocols in use in the CSN program are the result of substantial input and recommendations from CASAC both during their initial deployment about ten years ago, and during the more recent transition to carbon sampling that is consistent with IMPROVE protocols (Henderson 2005c). Monitoring agencies also played a strong role in directing the sampling technologies used in the CSN. During the first few years of implementing the CSN there were up to four different sampling approaches used in the network. Over time as monitoring agencies shared their experiences and data with each other, several agencies shifted their network operations to the sampling technology used today. By 2008, the EPA was working closely with all remaining monitoring agencies to transition to the current CSN sampling for ions and elements. All carbon sampling was fully transitioned to the current method by October of 2009 for consistency with the IMPROVE program. Therefore, while the current CSN sampling methods were not developed or adopted by a voluntary consensus standard body, they are the result of harmonizing the network by monitoring agency users and EPA. The CSN network and methods are described in more detail in the Policy Assessment (US EPA, 2011a, Appendix B, section B.1.3).

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 from this proposed rule.

The EPA has identified persons from lower socioeconomic strata as a susceptible 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 preamble. The Agency expects this proposed rule would 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 preamble and copies of all documents have been placed in the public docket for this action.

CHAPTER 10
SECONDARY STANDARDS ANALYSIS

10.1 Introduction

As defined by section 109(b)(2) of the Clean Air Act (CAA), the purpose of a secondary NAAQS standard is to protect public welfare against negative effects of criteria air pollutants, including decreased visibility, climate effects, and damage to ecological systems and building materials. Ambient PM has been associated with visibility impairment in diverse regions across the United States and is considered adverse to the public welfare. EPA is proposing a national visibility standard in conjunction with the Regional Haze Program as a means of achieving appropriate levels of protection against PM-related visibility impairments in urban, non-urban and Federal Class I areas across the country. EPA evaluated two proposed secondary standard levels of 30 deciviews (dv) and 28 dv, along with a more stringent standard level of 25 dv, based on 24-hour average speciated $PM_{2.5}$ measurements and with a 3-year average, 90^{th} percentile form.

10.2 The Secondary NAAQS Standard

The secondary $PM_{2.5}$ NAAQS standard consists of three parts: a level, averaging period, and form. In the *Urban-focused Visibility Assessment* (U.S. EPA, 2010) and the *Policy Assessment for the Review of the PM NAAQS* (U.S. EPA, 2011), several preference studies provide the foundation for the secondary PM NAAQS.[1] 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 EPA to inform the 2006 PM NAAQS review (Abt Associates Inc., 2001). Using the results of these studies, EPA determined that for a majority of individuals, the range of acceptable urban visibility falls between 20 dv and 30 dv based on a 4-hour average indicator (U.S. EPA, 2011). For this analysis, we consider the two proposed standard levels of 30 dv and 28 dv, both averaged over 24 hours, along with a more stringent 24-hour average standard level.[2] While a sub-daily (i.e., 4-hour) averaging period would capture the wide variations in

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[1] 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).

[2] In order to provide generally equivalent protection, the level of a NAAQS based on a 24-hour average indicator should include an adjustment compared to the level that would be applied to a NAAQS based on a daily maximum daylight 4-hour average indicator. Using 15 study sites, EPA staff investigated five approaches to making this adjustment, for 4-hour indicator NAAQS levels of 20, 25, and 30 dv. An approach thought by EPA staff to be more appropriate for further consideration yielded adjusted NAAQS levels of 21, 25, and 28 dv as

visibility conditions that occur over the course of a day, a 24-hour averaging period avoids data quality uncertainties associated with instruments currently available to measure hourly $PM_{2.5}$ mass (U.S. EPA, 2011).

EPA proposes using a 3-year average, 90th percentile form for the standard. Determining attainment using this form requires comparing the level of the standard to the 3-year average of the 90th percentile of the measured indicator. Using a multi-year percentile form for the standard lessens the influence of unusual emissions values and provides a degree of stability for implementation planning (U.S. EPA, 2011).

10.3 Visibility Benefits from PM Reduction[3]

Visibility directly impacts the quality of life in the places where people live, work, and travel (U.S. EPA, 2009). Air pollution, including particulates, contributes to decreased visibility by scattering and absorbing light, which reduces visual range and clarity. $PM_{2.5}$ component species that contribute to decreased visibility include sulfates, nitrates, organic carbon, elemental carbon, and soil (Sisler, 1996). Visibility impairment is expressed in terms of light extinction, measured in inverse megameters (Mm-1), or in terms of the deciview haze index, which is calculated based on total light extinction (Pitchford and Malm, 1994). A change of one dv is believed to be the smallest change in visible air quality perceptible by the human eye (Pitchford and Malm, 1994).

Visibility conditions and sources of visibility impairment vary both by region and season. Humidity increases visibility impairment because some particles, such as ammonium sulfate and ammonium nitrate, absorb water and become larger when relative humidity increases, resulting in increased visibility impairment (U.S. EPA, 2009). The eastern U.S. generally experiences greater visibility impairment due to higher concentrations of particulates and higher average humidity levels. Particulate sulfate is the dominant source of reduced visual air quality in the eastern U.S. (>50% of the particulate light extinction) and an important contributor to visibility impairment elsewhere in the country (>20% of particulate light extinction) (U.S. EPA, 2009). Particulate nitrate contributes to decreased visibility in California and the upper Midwest, particularly during the winter (U.S. EPA, 2009). In all regions, urban particulate concentrations are higher than those in the surrounding non-urban area, but

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the 24-hour $PM_{2.5}$ light extinction indicator levels that are generally equivalent to levels of 20, 25, and 30 dv applied to a daily maximum daylight 4-hour $PM_{2.5}$ light extinction indicator (U.S. EPA, 2011).

[3] Additional discussion of visibility benefits related to attainment of the primary PM NAAQS standard can be found in Chapter 6 of this RIA.

western urban areas show greater differences from the surrounding non-urban areas than do eastern urban areas (U.S. EPA, 2009).

10.4 Baseline Modeling Projection Data (2020)

EPA has proposed to use a calculated indicator of PM-related light extinction to determine whether an area is in attainment for the secondary $PM_{2.5}$ NAAQS standard. The IMPROVE[4] algorithm uses the estimated impact of each PM component species and relative humidity to calculate the amount of PM-related light extinction. In the equation, each PM component species is multiplied by a factor related to its impact on light extinction. Component species affected by the presence of water in the ambient air are also multiplied by a factor representing the relative humidity. These factors are summed to determine the total light extinction caused by PM. To calculate design values for this analysis, EPA is using a modified version of this algorithm, which is explained in more detail in Chapter 3 of this RIA.

To estimate design values for this analysis, we apply the original IMPROVE algorithm to 24-hour, speciated $PM_{2.5}$ concentrations measured at 236 Chemical Speciation Network (CSN) monitors across the country and climatological mean relative humidity data to calculate the estimated light extinction in each location. The 207 counties with CSN monitors are identified in Figure 10-1. We then compare the calculated light extinction from a monitor to the standard level to determine whether the county where the monitor is located is in attainment (U.S. EPA, 2011).

Even before incorporating reductions to attain the current primary standard of 15 $\mu g/m^3$ annual and 35 $\mu g/m^3$ 24-hour (denoted 15/35), no monitors are expected to exceed a secondary standard level of 30 dv and only three monitors are expected to exceed a secondary standard level of 28 dv in 2020. Because all three of these monitors also exceed the 15/35 primary standard,[5] we would expect each would attain a secondary standard of 28 dv when controlled at the primary standard level. Further emission reductions to meet a more stringent primary standard would lead to additional improvement in visibility in all areas. Table 10-1

[4]The Interagency Monitoring of Protected Visual Environment (IMPROVE) program was established in 1985 to aid in the creation of state and federal implementation plans for visibility in Class I areas as required in the 1977 amendments to the CAA.

[5]The monitors that are above 15/35 are monitor id numbers: 60658001 (located in Riverside, CA); 60290014 (located in Kern, CA); and 60990005 (located in Stanislaus, CA). The projected 2020 base case design values for the primary standard for these monitors are 16.30/46.5 $\mu g/m^3$, 14.18/44.0 $\mu g/m^3$, and 10.85/37.0 $\mu g/m^3$, respectively. The projected 2020 base case design values for the secondary standard for these monitors are: 29 dv, 30 dv, and 29 dv, respectively. We believe that the emissions reductions needed to obtain the current primary standard levels of 15/35 will be enough to lower the projected 2020 secondary standard design values to 28 dv.

shows the percentage of monitors projected to exceed 30, 28, or 25 dv in 2020, prior to full attainment of the current primary standard.

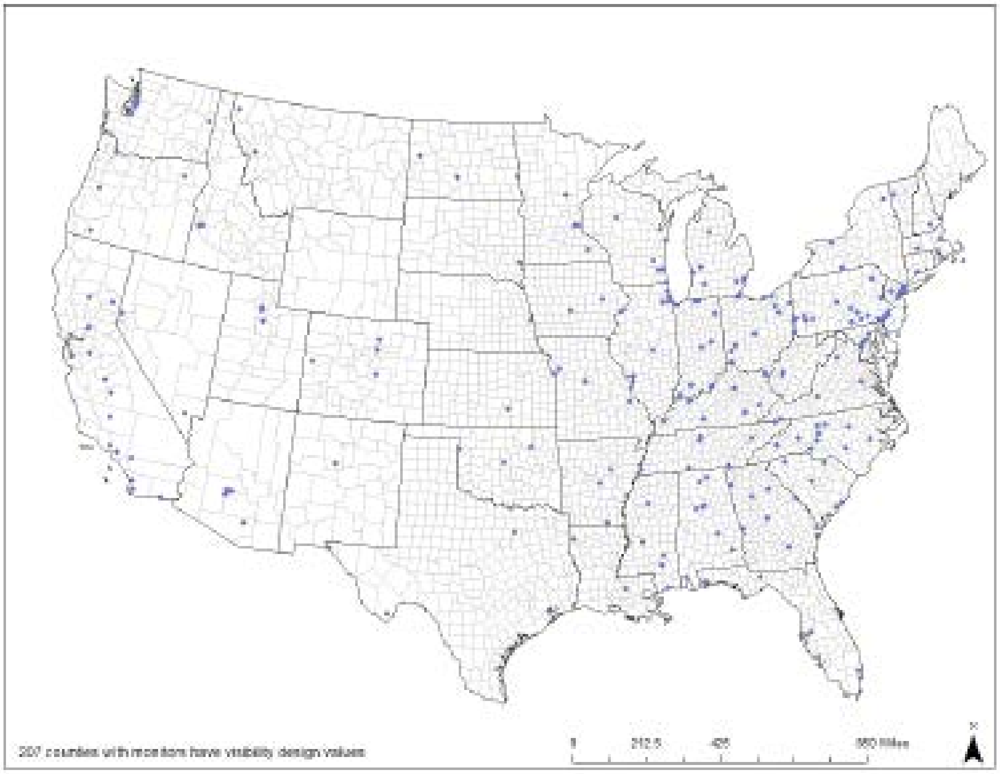

Figure 10-1. Counties with Monitors Included in Analysis

Table 10-1. Percentage of Monitors Projected to Exceed Alternative Secondary Standards in 2020, Prior to Attainment of Primary Standard of 15/35

Level	Number Exceeding Selected Level	% Exceeding Selected Level
30	0	0.0%
28	3	1.3%
25	28	11.9%

Of the 236 monitors for which visibility design values are available, 208 (88%) attain a secondary standard of 25 dv or better in 2020, prior to full attainment of the current primary standard. Figure 10-2 shows the counties that would exceed the secondary standards in this analysis. Visibility design values calculated from data for each monitor location included in this analysis can be found in Appendix 10-A.

Figure 10-2. Design Values in 2020, Prior to Full Attainment of a Primary Standard of 15/35

10.5 Impacts of Attaining a Distinct Secondary Standard

Based on the air quality analysis conducted for the primary $PM_{2.5}$ standard, all monitored areas are estimated to be in attainment with a secondary standard level of either 30 dv or 28 dv in 2020, assuming full attainment of the primary $PM_{2.5}$ standard. For the two optional levels proposed for the secondary standard, no additional costs or benefits will be realized beyond those quantified for the primary $PM_{2.5}$ standard in this RIA.[6]

10.6 Limitations of Analysis

Visibility design values for 2020 were calculated using the CMAQ modeling information and 2004-2006 ambient measurements. To determine the design values for meeting the current primary $PM_{2.5}$ standard and proposed alternative primary standards, we used a

[6] Based on the air quality modeling used in this analysis, EPA does not believe that any county that is in compliance with primary will violate secondary standard. However, different modeling trajectories could potentially lead to cases in which the secondary standard is binding.

methodology, described in Chapter 3, to estimate the small emissions reductions needed from control measures to show attainment and to estimate the costs and benefits of attaining the proposed alternative primary standards. It is not possible to apply this methodology to the visibility design values.[7] As a result, the only analysis available for the proposed alternative secondary standards in 2020 is prior to full attainment of the current primary standard. All monitors analyzed are projected to attain a secondary standard of 30 dv in the 2020 base case. Given the 24-hr design value reductions that were included in simulating attainment of 15/35 in the 2020 base case, it is likely that all monitors will also attain a secondary standard of 28 dv when they attain the current primary standards.[8]

10.7 References

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[7] As described in Chapter 3, we apply a methodology of air quality ratios to estimate the emissions reductions needed to meet the current and proposed alternative levels for the primary standard. While this methodology can estimate how these emissions reductions will affect changes in the future-year annual design value and the corresponding response of the future-year 24-hr design value to changes in the annual design value, it is unable to estimate how each of the $PM_{2.5}$ species will change with these emission reductions. Given that estimating changes in future-year visibility is dependent on the IMPROVE equation and how the $PM_{2.5}$ species are projected to change in time, we are unable to estimate visibility design values for meeting the current and proposed alternative levels for the primary standard.

[8] The projected 2020 base case design values for the secondary standard for the following monitors with id numbers 60658001 (located in Riverside, CA), 60290014 (located in Kern, CA), and 60990005 (located in Stanislaus, CA) are 29 dv, 30 dv, and 29 dv, respectively. The emissions reductions selected for simulating attainment of 15/35 in the 2020 base case resulted in the following reductions in the 24-hr design values for these three monitors: 11.1 $\mu g/m^3$, 21.9 $\mu g/m^3$ and 5.3 $\mu g/m^3$, respectively. Based on the trends presented in Wayland, 2012, we believe that these emissions reductions and 24-hr design value changes for simulating the current primary standard levels of 15/35 will be enough to lower the projected 2020 secondary standard design values for these three monitors to 28 dv or lower.

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APPENDIX 10.A
2017 MODELED DESIGN VALUES BY STATE, COUNTY, AND SITE

Table 10.A-1. 2017 Modeled Design Values by State, County, and Site

State	County	Site	Design Value
Alabama	Barbour	010050002	20
	Jefferson	010730023	26
	Jefferson	010732003	23
	Jefferson	010731009	20
	Madison	010890014	22
	Mobile	010970003	21
	Montgomery	011011002	22
	Morgan	011030011	21
	Russell	011130001	23
Arizona	Maricopa	040130019	20
	Maricopa	040139997	19
	Maricopa	040139998	18
	Maricopa	040137003	15
	Maricopa	040137020	15
	Pima	040191028	14
Arkansas	Ashley	050030005	21
	Pulaski	051190007	22
	White	051450001	20
California	Butte	060070002	24
	Fresno	060190008	28
	Imperial	060250005	23
	Kern	060290014	30
	Los Angeles	060371103	27
	Plumas	060631009	22
	Riverside	060658001	29
	Sacramento	060670010	28

(continued)

Table 10.A-1. 2017 Modeled Design Values by State, County, and Site (continued)

State	County	Site	Design Value
California (continued)	Sacramento	060670006	25
	San Diego	060730003	23
	San Diego	060731002	23
	Santa Clara	060850005	24
	Stanislaus	060990005	29
	Tulare	061072002	28
	Ventura	061112002	23
Colorado	Adams	080010006	19
	El Paso	080410011	16
	Mesa	080770017	19
	Weld	081230008	18
Connecticut	New Haven	090090027	24
D.C.	Washington	110010042	24
	Washington	110010043	24
Delaware	Kent	100010003	23
	New Castle	100032004	25
Florida	Broward	120111002	17
	Escambia	120330004	23
	Hillsborough	120573002	19
	Leon	120730012	21
	Miami-Dade	120861016	17
	Pinellas	121030026	21
Georgia	Bibb	130210007	24
	Chatham	130510017	21
	Clarke	130590001	22
	Coffee	130690002	19
	De Kalb	130890002	22
	Floyd	131150005	22

(continued)

Table 10.A-1. 2017 Modeled Design Values by State, County, and Site (continued)

State	County	Site	Design Value
Georgia (continued)	Muscogee	132150011	22
	Richmond	132450091	23
	Walker	132950002	23
Idaho	Ada	160010010	23
	Canyon	160270004	23
Illinois	Cook	170310057	27
	Cook	170310076	25
	Cook	170314201	25
	Du Page	170434002	27
	Macon	171150013	25
	Madison	171192009	25
Indiana	Allen	180030004	25
	Dubois	180372001	26
	Elkhart	180390003	27
	Henry	180650003	24
	Lake	180890022	27
	Lake	180892004	27
	Marion	180970078	26
	St Joseph	181411008	26
	Vanderburgh	181630012	24
Iowa	Linn	191130037	24
	Polk	191530030	22
	Scott	191630015	25
Kansas	Sedgwick	201730010	21
	Wyandotte	202090021	23
Kentucky	Boyd	210190017	24
	Daviess	210590005	26
	Daviess	210590014	23

(continued)

Table 10.A-1. 2017 Modeled Design Values by State, County, and Site (continued)

State	County	Site	Design Value
Kentucky (continued)	Fayette	210670012	24
	Jefferson	211110043	24
	Jefferson	211110048	24
	Kenton	211170007	23
	Laurel	211250004	22
	McCracken	211451004	23
	Perry	211930003	21
	Warren	212270007	24
Louisiana	Bossier	220150008	20
	East Baton Rouge	220330009	24
Maryland	Anne Arundel	240030019	23
	Baltimore	240053001	25
	Prince Georges	240330030	23
Massachusetts	Hampden	250130008	21
	Suffolk	250250042	22
Michigan	Allegan	260050003	25
	Chippewa	260330901	22
	Kalamazoo	260770008	25
	Kent	260810020	25
	Missaukee	261130001	21
	Monroe	261150005	26
	Washtenaw	261610008	26
	Wayne	261630033	27
	Wayne	261630001	25
Minnesota	Hennepin	270530963	23
	Mille Lacs	270953051	19
	Olmsted	271095008	23
	Ramsey	271230871	22

(continued)

Table 10.A-1. 2017 Modeled Design Values by State, County, and Site (continued)

State	County	Site	Design Value
Mississippi	Forrest	280350004	23
	Grenada	280430001	20
	Harrison	280470008	22
	Hinds	280490018	21
	Jones	280670002	23
Missouri	Clay	290470005	22
	Cooper	290530001	22
	Jefferson	290990012	24
	St Louis City	295100085	24
	Ste Genevieve	291860005	22
Montana	Lincoln	300530018	25
	Missoula	300630031	23
Nebraska	Douglas	310550019	22
Nevada	Clark	320030561	22
	Washoe	320310016	18
New Hampshire	Hillsborough	330110020	23
	Rockingham	330150014	19
New Jersey	Camden	340070003	23
	Middlesex	340230006	21
	Morris	340273001	22
	Union	340390004	25
New Mexico	Bernalillo	350010023	14
New York	Bronx	360050110	26
	Bronx	360050083	25
	Erie	360290005	24
	Essex	360310003	17
	Monroe	360551007	23
	New York	360610062	25

(continued)

Table 10.A-1. 2017 Modeled Design Values by State, County, and Site (continued)

State	County	Site	Design Value
New York (continued)	Queens	360810124	24
	Steuben	361010003	20
North Carolina	Buncombe	370210034	22
	Catawba	370350004	23
	Cumberland	370510009	21
	Davidson	370570002	23
	Forsyth	370670022	23
	Guilford	370810013	22
	Lenoir	371070004	20
	Mecklenburg	371190041	22
	Rowan	371590021	22
	Wake	371830014	22
North Dakota	Burleigh	380150003	18
	Cass	380171004	19
	McKenzie	380530002	15
Ohio	Butler	390171004	24
	Cuyahoga	390350038	26
	Cuyahoga	390350060	25
	Franklin	390490081	25
	Hamilton	390610040	24
	Jefferson	390810017	26
	Lawrence	390870010	25
	Lorain	390930016	25
	Lorain	390933002	21
	Lucas	390950026	26
	Mahoning	390990014	25
	Montgomery	391130031	24
	Stark	391510017	25

(continued)

Table 10.A-1. 2017 Modeled Design Values by State, County, and Site (continued)

State	County	Site	Design Value
Ohio (continued)	Stark	391510020	24
	Summit	391530023	24
Oklahoma	Ellis	400450890	17
	Oklahoma	401091037	20
	Tulsa	401431127	21
Oregon	Jackson	410290133	24
	Lane	410390060	19
	Multnomah	410510246	22
	Union	410610119	19
Pennsylvania	Adams	420010001	23
	Allegheny	420030064	28
	Allegheny	420030008	24
	Centre	420270100	24
	Chester	420290100	25
	Dauphin	420430401	26
	Delaware	420450002	25
	Erie	420490003	23
	Lackawanna	420692006	22
	Lancaster	420710007	27
	Northampton	420950025	24
	Perry	420990301	21
	Philadelphia	421010055	26
	Philadelphia	421010004	25
	Philadelphia	421010136	23
	Washington	421255001	20
	Westmoreland	421290008	23
	York	421330008	25
Rhode Island	Providence	440070022	22

(continued)

Table 10.A-1. 2017 Modeled Design Values by State, County, and Site (continued)

State	County	Site	Design Value
South Carolina	Charleston	450190049	21
	Charleston	450190046	19
	Chesterfield	450250001	20
	Greenville	450450009	23
	Richland	450790019	22
South Dakota	Minnehaha	460990006	21
Tennessee	Davidson	470370023	22
	Hamilton	470654002	23
	Knox	470931020	22
	Lawrence	470990002	20
	Shelby	471570024	22
	Shelby	471570047	22
	Sullivan	471631007	24
	Sumner	471650007	22
Texas	Brewster	480430101	15
	Dallas	481130069	20
	El Paso	481410044	17
	Harris	482011039	22
Utah	Davis	490110004	25
	Salt Lake	490353006	24
	Utah	490494001	24
Vermont	Chittenden	500070012	21
Virginia	Bristol City	515200006	24
	Henrico	510870014	23
	Page	511390004	22
	Roanoke City	517700014	23
Washington	King	530330024	23
	King	530330057	23

(continued)

Table 10.A-1. 2017 Modeled Design Values by State, County, and Site (continued)

State	County	Site	Design Value
Washington (continued)	King	530330032	22
	King	530330048	21
	King	530330080	20
	Pierce	530530029	22
	Spokane	530630016	21
West Virginia	Kanawha	540391005	24
	Kanawha	540390011	22
	Marshall	540511002	22
Wisconsin	Dodge	550270007	24
	Kenosha	550590019	25
	Manitowoc	550710007	24
	Milwaukee	550790026	25
	Taylor	551198001	21
	Waukesha	551330027	25

CHAPTER 11
QUALITATIVE DISCUSSION OF EMPLOYMENT IMPACTS OF AIR QUALITY REGULATIONS

11.1 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 is intended to provide context for considering the potential influence of environmental regulation on growth and job shifts in the U.S. economy. Section 11.2 addresses the particular influence of this proposed rule on employment. Section 11.3 presents a descriptive overview of the peer-reviewed literature relevant to evaluating the effect of air quality regulation on employment. Finally, in Section 11.4, we offer several conclusions.

11.2 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.

Other factors affecting U.S. employment include cyclical, technological, demographic, and economic trends both in the United States and abroad. In this section, we focus on the studies most directly applicable to EPA analyses.

Estimating specific employment impacts from a new NAAQS standard is particularly challenging for two reasons. First, the NAAQS is a target 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

[1] This is the case except to the extent that labor costs are part of total costs in a cost-benefit analysis.

NAAQS target. State and local officials can consider 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 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 11.3 focuses on qualitative insights from currently available peer-reviewed literature on the impact of air quality regulations in general.

11.3 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.

11.3.1 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

[2] See Schmalansee and Stavin (2011)

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).

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 state implementation plans (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.

11.3.2 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) demonstrate that 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 demonstrate, using economic theory, that the overall net effect of environmental regulation on employment, predicted by this model, is ambiguous. 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. First, 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. Second, is the cost effect, which increases the demand for inputs, including labor, as 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 show that, on average across the industries studied, each additional $1 million ($1987) spent on pollution abatement results 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 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.

[3] Note, like Morgenstern, Pizer, and Shih (2002), this study does not estimate the number of jobs created in the environmental protection sector.

11.4 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 (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 [GDP]), 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 economy in a significant way.

11.5 References

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